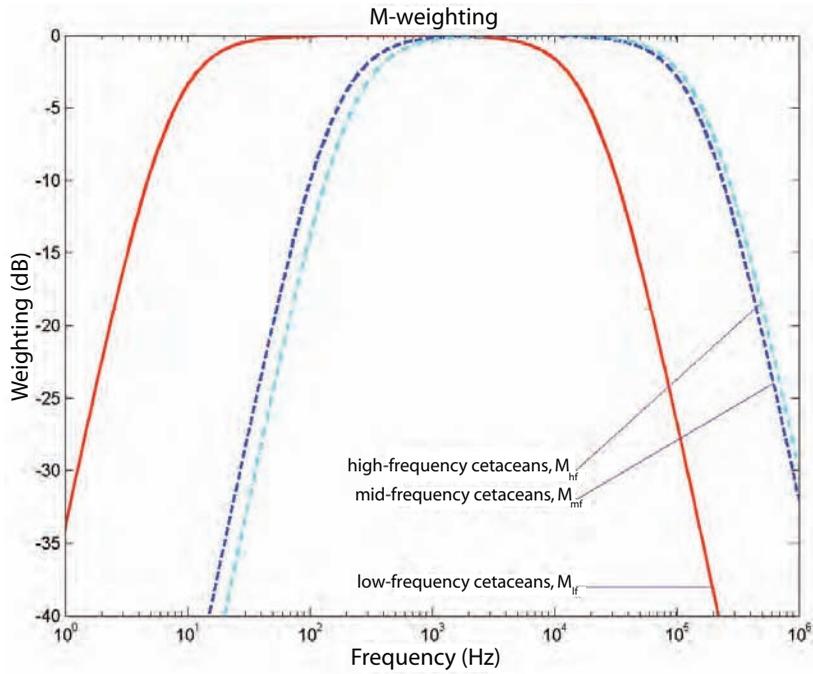
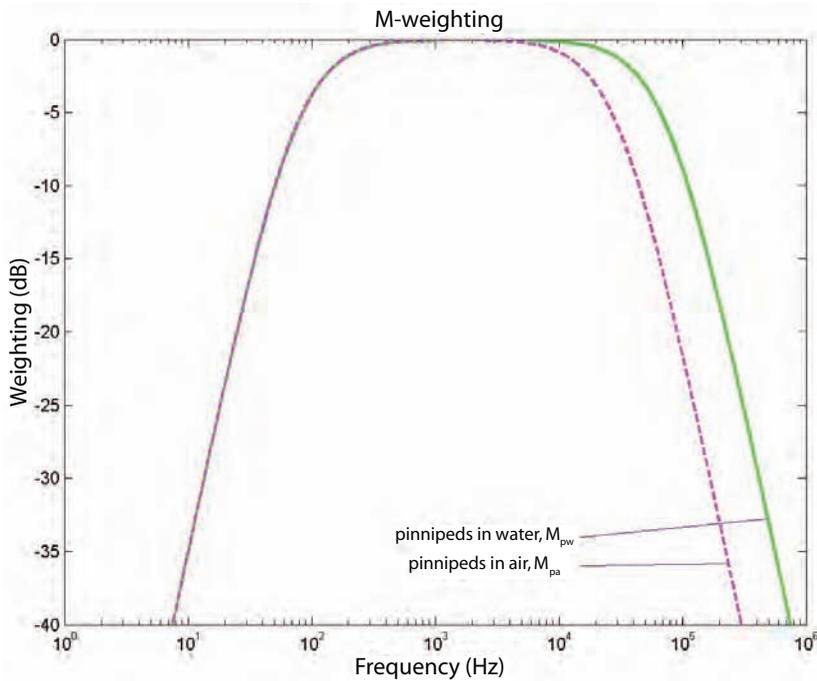


(A)



(B)



**Figure 1.** The M-weighting functions for (A) low-, mid-, and high-frequency cetaceans, as well as for (B) pinnipeds in water and air.

metric(s) for estimating TTS-onset and predicting PTS-onset in humans (ISO, 1990).

This use of SEL is based on the assumption that sounds of equivalent energy will have generally similar effects on the auditory systems of exposed human subjects, even if they differ in SPL, duration, and/or temporal exposure pattern (Kryter, 1970; Nielsen et al., 1986; Yost, 1994; NIOSH, 1998). Under the equal-energy assumption, at exposure levels above TTS-onset, each doubling of sound duration is associated with a 3 dB reduction in the SPL theoretically required to cause the same amount of TTS. This relationship has been used in the derivation of exposure guidelines for humans (e.g., NIOSH, 1998). Numerous authors have questioned the predictive power of using a simplistic total energy approach in all conditions. It fails to account for varying levels and temporal patterns of exposure/recovery, among other factors, and will thus likely overestimate the TTS resulting from a complex noise exposure (Hamernik & Hsueh, 1991; Hamernik et al., 1993, 2002; Ahroon et al., 1993; Ward, 1997; Strasser et al., 2003). A comparative assessment of TTS as a function of exposure level in mammals, fish, and birds suggests that there are direct relationships but that the slopes vary among taxa (Smith et al., 2004). The debate over the validity of the equal energy "rule" of noise exposure remains unresolved, even for humans.

Some limited evidence favoring an SEL approach exists for marine mammals, however. Specifically, an equal-energy relationship for TTS-onset appears to hold reasonably well for certain noise exposure types within several mid-frequency cetacean species (Finneran et al., 2002b, 2005a; see "Effects of Noise on Hearing in Marine Animals: TTS Data" section in Chapter 3). A recent study of in-air TTS in a California sea lion (Kastak et al., 2007) illustrates some conditions in which exposures with identical SEL result in considerably different levels of TTS. Nevertheless, because the very limited marine mammal data agree reasonably well (at least as a first-order approximation) with equal-energy predictions, and predictions based on SEL will be precautionary for intermittent exposures, we regard it as appropriate to apply the SEL metric for certain noise exposure criteria until future research indicates an alternate and more specific course. In certain applications, there is much more scientific justification for use of SEL-based criteria than for previous ad hoc SPL criteria (discussed in the "Historical Perspective" section in Chapter 1). In applications involving auditory effects, SEL-based criteria will likely more reliably distinguish cases where

phenomena of concern (TTS, PTS, etc.) will and will not likely occur.

#### **Levels of Noise Effect: Injury and Behavioral Disturbance**

Direct auditory tissue effects (injury) and behavioral disruption are the two categories of noise effect that are considered in these marine mammal exposure criteria. Chapter 3 summarizes all available data on the effects of noise on marine mammal hearing. It also describes how these data are applied and extrapolated using precautionary measures to predict auditory injury and to derive thresholds and proposed criteria for injury.

In Chapter 4 and Appendices B & C, we summarize the current understanding and available data regarding marine mammal behavioral responses to noise. Chapter 4 includes a quantitative severity scale based generally on the NRC's (2005) Population Consequences of Acoustic Disturbance (PCAD) Model. Chapter 4 also includes a limited and cautious entry of behavioral-response data into a matrix of severity scaling as a function of RL. Currently available data, pooled by functional hearing group, do not support specific numerical criteria for the onset of disturbance. Rather, they indicate the context-specificity of behavioral reactions to noise exposure and point to some general conclusions about response severity in certain, specific conditions.

### 3. Criteria for Injury: TTS and PTS

The criteria for injury for all marine mammal groups and sound types are received levels (frequency-weighted where appropriate) that meet the definition of PTS-onset used here (40 dB-TTS, described below). Criteria were derived from measured or assumed TTS-onset thresholds for each marine mammal group plus TTS growth rate estimates (given below). Available TTS data for two mid-frequency cetacean species and three species of pinnipeds are used as the basis for estimating PTS-onset thresholds in all cetaceans (“cetacean procedure” described below; see “PTS-Onset for Pulses”) and in all pinnipeds (see “PTS-Onset for Nonpulse Sounds”), respectively. The proposed injury criteria are presented by sound type because, for a given sound type, many of the same extrapolation and summation procedures apply across marine mammal hearing groups.

A dual-criterion approach was used for the recommended injury criteria. That is, any received noise exposure that exceeds either a peak pressure or a SEL criterion for injury is assumed to cause tissue injury in an exposed marine mammal. Of the two measures of sound exposure, peak pressures are to be unweighted (i.e., “flat-weighted”), whereas SEL metrics are to be M-weighted for the relevant marine mammal group (Figure 1). In practice, the received noise conditions should be compared to the two exposure criteria for that sound type and functional hearing group, and the more precautionary of the two outcomes accepted.

#### **Effects of Noise on Hearing in Marine Mammals: TTS Data**

Noise exposure criteria for auditory injury ideally should be based on exposures empirically shown to induce PTS-onset; however, no such data presently exist for marine mammals. Instead, PTS-onset must be estimated from TTS-onset measurements and from the rate of TTS growth with increasing exposure levels above the level eliciting TTS-onset. PTS is presumed to be likely if the threshold is reduced by  $\geq 40$  dB (i.e., 40 dB of TTS). We used available marine mammal TTS data and precautionary extrapolation procedures based on terrestrial mammal data (see “Level of Noise Effect” in Chapter 2) to estimate exposures associated with PTS-onset. Existing TTS measurements for marine mammals are reviewed in detail here since they serve as the quantitative foundation for the injury criteria.

To date, TTSs measured in marine mammals have generally been of small magnitude (mostly  $< 10$  dB). The onset of TTS has been defined as being a temporary elevation of a hearing threshold by 6 dB (e.g., Schlundt et al., 2000), although smaller threshold shifts have been demonstrated to be statistically significant with a sufficient number of samples (e.g., Kastak et al., 1999; Finneran et al., 2005a). Normal threshold variability within and between both experimental and control sessions (no noise) does warrant a TTS-onset criterion at a level that is always clearly distinguishable from that of no effect. We considered a 6 dB TTS sufficient to be recognized as an unequivocal deviation and thus a sufficient definition of TTS-onset.

Most of the frequencies used in TTS experiments to date are within the flat portions of the M-weighting functions given here, but not necessarily within the regions of greatest hearing sensitivity. Within the range of best hearing sensitivity for a given individual, detection thresholds are generally similar. Within this band, exposures with the same absolute level but different frequency are thus similar in terms of their effective sensation level. *Sensation level* is the amount (in dB) by which an RL exceeds the threshold RL for that signal type within a prescribed frequency band (Yost, 2000). If two exposures with identical absolute level are both audible, but one is outside the frequency range of best hearing sensitivity, sensation level will be less for the latter exposure, and its potential effects will be diminished. By creating frequency-weighted functions that are flat across virtually the entire functional hearing band, rather than just the region of best sensitivity, we have made another precautionary decision in the absence of underlying data on equal-loudness functions.

Auditory fatigue (i.e., TTS) in mid-frequency cetaceans has been measured after exposure to tones, impulsive sounds, and octave-band noise (OBN). In pinnipeds, it has been measured upon exposure to construction noise and OBN in both air and water.

#### *Cetacean TTS*

The sound exposures that elicit TTS in cetaceans have been measured in two mid-frequency species—bottlenose dolphin and beluga (specific references given below)—with at least limited data

being available for exposures to a single pulse and to nonpulsed sounds ranging from 1-s to ~50-min duration. There are no published TTS data for any other odontocete cetaceans (either mid- or high-frequency) or for any mysticete cetaceans (low-frequency). This review is organized according to the duration of the fatiguing stimulus, with shortest exposures discussed first.

Finneran et al. (2000) exposed two bottlenose dolphins and one beluga to single pulses from an "explosion simulator" (ES). The ES consisted of an array of piezoelectric sound projectors that generated a pressure waveform resembling that from a distant underwater explosion. The pressure waveform was generally similar to waveforms predicted by the Navy REFMS model (Britt et al., 1991). The ES failed to produce realistic energy at frequencies below 1 kHz, however. No substantial (i.e.,  $\geq 6$  dB) threshold shifts were observed in any of the subjects exposed to a single pulse at the highest received exposure levels (peak: 70 kPa [10 psi]; peak-to-peak: 221 dB re: 1  $\mu$ Pa (peak-to-peak); SEL: 179 dB re: 1  $\mu$ Pa<sup>2</sup>-s)].

Finneran et al. (2002b) repeated this experiment using a seismic watergun that produced a single acoustic pulse. Experimental subjects consisted of one beluga and one bottlenose dolphin. Measured TTS<sub>2</sub> was 7 and 6 dB in the beluga at 0.4 and 30 kHz, respectively, after exposure to intense single pulses (peak: 160 kPa [23 psi]; peak-to-peak: 226 dB re: 1  $\mu$ Pa (peak-to-peak); SEL: 186 dB re: 1  $\mu$ Pa<sup>2</sup>-s). Thresholds returned to within  $\pm 2$  dB of the pre-exposure value within 4 min of exposure. No TTS was observed in the bottlenose dolphin at the highest exposure condition (peak: 207 kPa [30 psi]; peak-to-peak: 228 dB re: 1  $\mu$ Pa (peak-to-peak); SEL: 188 dB re: 1  $\mu$ Pa<sup>2</sup>-s). These studies demonstrated that, for very brief pulses, higher sound pressures were required to induce TTS than had been found for longer tones (discussed below).

Schlundt et al. (2000) reported TTS in five bottlenose dolphins and two belugas exposed to 1-s pure tones (nonpulses). This paper also included a re-analysis of TTS data from a technical report by Ridgway et al. (1997). At frequencies of 3 kHz, 10 kHz, and 20 kHz, SPLs necessary to induce TTS-onset were 192 to 201 dB re: 1  $\mu$ Pa (SEL: 192 to 201 dB re: 1  $\mu$ Pa<sup>2</sup>-s). The mean exposure SPL for TTS-onset was 195 dB re: 1  $\mu$ Pa (195 dB re: 1  $\mu$ Pa<sup>2</sup>-s). Note the appropriately different metrics for the nonpulse sources used in this study and those involving pulses. Also note that the SPL and SEL values are identical in this special case because of the 1-s duration fatiguing stimuli. At 0.4 kHz, no subjects exhibited shifts after exposures up to SPL exposures of 193 dB re: 1  $\mu$ Pa (193 dB re: 1  $\mu$ Pa<sup>2</sup>-s). Data at 75 kHz

were inconclusive: one dolphin exhibited a TTS after exposure at 182 dB SPL re: 1  $\mu$ Pa (182 dB re: 1  $\mu$ Pa<sup>2</sup>-s) but not at higher exposure levels. The other dolphin experienced no threshold shift after exposure to maximum SPL levels of 193 dB re: 1  $\mu$ Pa (193 dB re: 1  $\mu$ Pa<sup>2</sup>-s). The shifts occurred most often at frequencies above the fatiguing stimulus.

Finneran et al. (2005a) measured TTS in bottlenose dolphins exposed to 3 kHz tones with durations of 1, 2, 4, and 8 s and at various SPL values. Tests were conducted in a quiet pool in contrast to previous studies in San Diego Bay, where thresholds were masked by broadband noise. Small amounts of TTS (3 to 6 dB) occurred in one dolphin following exposures with SELs of 190 to 204 dB re: 1  $\mu$ Pa<sup>2</sup>-s. These results are consistent with those of Schlundt et al. (2000), indicating that their results had not been significantly affected by the use of masked hearing thresholds in quantifying TTS. In general, the SEL necessary for TTS-onset was relatively consistent across the range of exposure durations, whereas exposure SPL values causing TTS-onset tended to decrease with increasing exposure duration. These results confirmed that, for these testing conditions (bottlenose dolphins exposed to  $\leq 8$ -s tones of variable SPL), TTS magnitude was best correlated with exposure SEL rather than SPL.

Schlundt et al. (2006) reported on the growth and recovery of TTS in a bottlenose dolphin exposed to 3 kHz tones with SPLs up to 200 dB re 1  $\mu$ Pa and durations up to 128 s. The maximum exposure SEL was 217 dB re 1  $\mu$ Pa<sup>2</sup>-s, which produced a TTS<sub>4</sub> of ~23 dB. All thresholds recovered to baseline values within 24 h, most within 30 min. The growth of TTS<sub>4</sub> with increasing exposure SEL was ~1 dB TTS per dB SEL for TTS<sub>4</sub> of ~15 to 18 dB.

Finneran et al. (2007b) measured TTS in a bottlenose dolphin after single and multiple exposures to 20 kHz tones. Hearing thresholds were estimated at multiple frequencies (10 to 70 kHz) both behaviorally and electrophysiologically (by measurement of multiple auditory steady-state responses). Three experiments were performed. The first two featured single exposures (20 kHz, 64-s tones at 185 and 186 dB re 1  $\mu$ Pa). The third featured three 20 kHz, 16-s exposures separated by 11 and 12 min, with a mean SPL of 193 dB re 1  $\mu$ Pa (SD = 0.8 dB). Hearing loss was frequency-dependent, with the largest TTS occurring at 30 kHz, less at 40, and then 20 kHz, and little or no TTS at other measured frequencies. AEP threshold shifts reached 40 to 45 dB and were always larger than behavioral shifts, which were 19 to 33 dB. Complete recovery required up to 5 d, with the recovery rate at 20 kHz being ~2 dB/doubling

of time and the rate at 30 and 40 kHz ~5 to 6 dB/doubling of time.

Nachtigall et al. (2003) measured TTS (*ca.* 20 min after noise cessation) in a bottlenose dolphin and found an average 11 dB shift following a 30-min net exposure to OBN with a 7.5 kHz center frequency (CF) (max SPL: 179 dB re: 1  $\mu$ Pa; SEL: ~212 to 214 dB re: 1  $\mu$ Pa<sup>2</sup>-s). The net exposure time was calculated as the total experimental time minus the time required for the subject to surface to breathe. Exposure during breathing periods was measured and factored into the SEL measurement. No TTS was observed after exposure to the same OBN at maximum SPL values of 165 and 171 dB re: 1  $\mu$ Pa (SEL: ~198 to 200 dB re: 1  $\mu$ Pa<sup>2</sup>-s and 204 to 206 dB re: 1  $\mu$ Pa<sup>2</sup>-s, respectively).

Using AEP methods, Nachtigall et al. (2004) found TTSs of *ca.* 4 to 8 dB following nearly 50-min exposures to OBN with a CF of 7.5 kHz (max SPL: 160 dB re: 1  $\mu$ Pa; SEL: ~193 to 195 dB re: 1  $\mu$ Pa<sup>2</sup>-s). The difference in results between the two Nachtigall et al. studies (slightly lower TTS after exposure to much lower exposure energy) was attributed to measuring TTS at a shorter interval after the exposure ended (5 vs ~20 min), and thus allowing less opportunity for hearing recovery. Further, Nachtigall et al. (2004) repeatedly measured hearing until recovery had occurred. TTS recovery was shown to occur within minutes or tens of minutes, depending on the amount of the threshold shift. Generally, the recovery rate was 1.5 dB of recovery per doubling of time and was consistent in both studies (Nachtigall et al., 2003, 2004).

The National Research Council (NRC) (1994) identified the need to know whether marine mammals experience greatest TTS at a frequency ½-octave above the frequency of exposure when exposed to loud tones as has been shown in terrestrial mammals. Nachtigall et al. (2004) observed an average threshold shift of 4 dB at 8 kHz but 8 dB shift at 16 kHz following the exposure to OBN centered at 7.5 kHz as described above. A similar upward frequency shift also has been observed by Schlundt et al. (2000) and Finneran et al. (2007b) for mid-frequency cetaceans. These findings provide “strong evidence for fundamental similarities in cochlear micromechanics in marine and land mammals” (NRC, 1994, p. 51) and further justify the judicious extrapolation of TTS data within marine mammal functional hearing groups and from terrestrial to marine mammals.

The above results provide empirical measures of exposure conditions associated with TTS-onset in mid-frequency cetaceans exposed to single pulses and nonpulses. Combined, these data demonstrate that, as compared with the exposure levels necessary to elicit TTS when exposure duration is short,

lower SPLs (but similar SEL values) are required to induce TTS when exposure duration is longer. These findings are generally consistent with measurements in humans and terrestrial mammals (Kryter, 1970; Harris, 1998; NIOSH, 1998) and support the use of SEL to approximate the auditory effects of variable exposure level/duration conditions. Although there are certain (possibly many) conditions under which an explicit “equal-energy rule” may fail to adequately describe the auditory effects of variable and/or intermittent noise exposure, the combined cetacean TTS data presented above generally support the use of SEL as a first-order approximation, at least until additional data are available.

For cetaceans, published TTS data are limited to the bottlenose dolphin and beluga (Finneran et al., 2000, 2002b, 2005a; Schlundt et al., 2000; Nachtigall et al., 2003, 2004). Where data exist for both species, we use the more precautionary result (usually for beluga) to represent TTS-onset for all mid-frequency cetaceans. No published data exist on auditory effects of noise in either low- or high-frequency cetaceans (an area of needed research as discussed in Chapter 5); therefore, data from mid-frequency cetaceans are used as surrogates for these two other groups (**cetacean procedure**). [We are aware of some very recent TTS measurements for an individual harbor porpoise exposed to single pulses (Lucke et al., 2007a) but lack sufficient details regarding methodology and data analysis to directly consider those data quantitatively.]

Low-frequency cetaceans (mysticetes), based on their auditory anatomy (Wartzok & Ketten, 1999) and ambient noise levels in the frequency ranges they use (Clark & Ellison, 2004), almost certainly have poorer absolute sensitivity (i.e., higher thresholds) across much of their hearing range than do the mid-frequency species (but see earlier discussion). Mid-frequency cetaceans experience TTS-onset at relatively high levels compared with their absolute hearing sensitivity at similar frequencies (i.e., high sensation levels), although it is not known that this is similarly characteristic of low-frequency cetaceans. Our use of TTS data from mid-frequency cetaceans as a surrogate for low-frequency cetaceans presumes that the two groups have similar auditory mechanisms and are not radically different in relative sensitivity to fatiguing noise, and that relative differences in absolute sensitivity between the two groups are generally as expected.

For high-frequency species, data from mid-frequency cetaceans are currently used as a surrogate in the absence of available group-specific data. Aside from their extended upper-frequency hearing, high-frequency cetaceans appear to be

generally similar in auditory anatomy and hearing capabilities to mid-frequency species, though there are some general differences between the groups in sound production. Based on available information and our extrapolation procedures, slightly lower estimates of TTS-onset may be warranted for high-frequency cetaceans exposed to very high-frequency sounds ( $\geq 100$  kHz). [Also, preliminary measurements of TTS in a harbor porpoise exposed to a single airgun pulse (Lucke et al., 2007a) suggest that this species may experience TTS-onset at levels lower than would be suggested by extrapolating from mid-frequency cetaceans. Those results, if confirmed, may provide a more empirical basis for estimating TTS-onset in high-frequency cetaceans and deriving group-specific injury criteria.]

#### *Pinniped TTS (Under Water)*

Sound exposures that elicit TTS in pinnipeds under water have been measured in individual subjects of three pinniped species (harbor seal, California sea lion, and northern elephant seal). Available data involved exposures to either broadband or octave-band nonpulse noise over durations ranging from ~12 min to several hours, plus limited data on exposure to underwater pulses. Interestingly, there were consistent among-species differences in the exposure conditions that elicited TTS under water. For the conditions tested, the harbor seal experienced TTS at lower exposure levels than did the California sea lion or northern elephant seal. There are no underwater TTS data for any other pinniped species.

The following review first considers exposure to nonpulses, organized chronologically, followed by a brief discussion of the lone study on exposure to pulses. All but one of the studies (Finneran et al., 2003) came from one laboratory and from the same individual test subjects. Kastak & Schusterman (1996) reported a TTS of ~8 dB (measured under water at 100 Hz) in a harbor seal following exposure to broadband airborne, nonpulse noise from nearby construction. Under controlled conditions, Kastak et al. (1999) measured TTS of *ca.* 4 to 5 dB in a harbor seal, California sea lion, and northern elephant seal following 20- to 22-min exposure to underwater OBN centered at frequencies from 100 Hz to 2 kHz. Exposures were normalized to octave-band levels 60 to 75 dB above each subject's hearing threshold (i.e., 60 to 75 dB sensation level) to present similar effective exposure conditions to each of the three subjects. Because of this approach, absolute exposure values (in terms of both SPL and SEL) were quite variable depending on subject and test frequency.

Subsequently, Kastak et al. (2005) made TTS measurements on the same subjects using 2.5

kHz OBN, higher sensation levels (up to 95 dB), and longer exposure durations (up to 50-min net exposure). These data largely corroborate previous findings concerning TTS-onset in these pinnipeds. They also support sensation level as a relevant metric for normalizing exposures with similar durations across species having different absolute hearing capabilities. Comparative analyses of the combined underwater pinniped data (Kastak et al., 2005) indicated that, in the harbor seal, a TTS of *ca.* 6 dB occurred with 25-min exposure to 2.5 kHz OBN with SPL of 152 dB re: 1  $\mu$ Pa (SEL: 183 dB re: 1  $\mu$ Pa<sup>2</sup>-s). Under the same test conditions, a California sea lion showed TTS-onset at 174 dB re: 1  $\mu$ Pa (SEL: 206 dB re: 1  $\mu$ Pa<sup>2</sup>-s), and a northern elephant seal experienced TTS-onset at 172 dB re: 1  $\mu$ Pa (SEL: 204 dB re: 1  $\mu$ Pa<sup>2</sup>-s).

Data on underwater TTS-onset in pinnipeds exposed to pulses are limited to a single study. Finneran et al. (2003) exposed two California sea lions to single underwater pulses from an arc-gap transducer. They found no measurable TTS following exposures up to 183 dB re: 1  $\mu$ Pa (peak-to-peak) (SEL: 163 dB re: 1  $\mu$ Pa<sup>2</sup>-s). Based on the Kastak et al. (2005) measurements using nonpulse sounds, the absence of TTS for the sea lions following such exposures is generally not surprising.

#### *Pinniped TTS (In Air)*

Auditory fatigue has been measured following exposure of pinnipeds to single pulses of in-air sound and to nonpulse noise.

Bowles et al. (unpub. data) measured TTS-onset for harbor seals exposed to simulated sonic booms at peak SPLs of 143 dB re: 20  $\mu$ Pa (peak) (SEL: 129 dB re: [20  $\mu$ Pa]<sup>2</sup>-s). Higher exposure levels were required to induce TTS-onset in both California sea lions and northern elephant seals in the same test setting, consistent with the results for nonpulse sound both under water and in air.

Auditory fatigue to airborne sound has also been measured in the same three species of pinnipeds after exposure to nonpulse noise, specifically 2.5 kHz CF OBN for 25 min (Kastak et al., 2004a). The harbor seal experienced *ca.* 6 dB of TTS at 99 dB re: 20  $\mu$ Pa (SEL: 131 dB re: [20  $\mu$ Pa]<sup>2</sup>-s). Onset of TTS was identified in the California sea lion at 122 dB re: 20  $\mu$ Pa (SEL: 154 dB re: [20  $\mu$ Pa]<sup>2</sup>-s). The northern elephant seal experienced TTS-onset at 121 dB re: 20  $\mu$ Pa (SEL: 163 dB re: [20  $\mu$ Pa]<sup>2</sup>-s). The subjects in these tests were the same individuals tested in water (Southall et al., 2001; Kastak et al., 2005).

Kastak et al. (2007) measured TTS-onset and growth functions for the same California sea lion exposed to a wider range of noise conditions. A total of 192 exposure sequences were conducted

with OBN (centered at 2.5 kHz) at levels 94 to 133 dB re: 20  $\mu$ Pa and durations 1.5 to 50 min net exposure duration. In these more intense noise exposures, TTS magnitudes up to 30 dB were measured at the 2.5 KHz test frequency. Full recovery was observed following all exposures; this occurred rapidly (likely within tens of minutes) for small shifts but took as long as 3 d in the case of the largest TTS. The estimated SEL value coinciding with TTS-onset across these varied exposure conditions was 159 dB re: (20  $\mu$ Pa)<sup>2</sup>-s with a TTS growth function of  $\sim$ 2.5 dB TTS/dB noise. For TTS exceeding 20 dB, a recovery rate of  $\sim$ 2.6 dB/doubling of time was calculated. These results generally agree with those of Kastak et al. (2004a) but provide a larger data set, across a wider range of exposure conditions with which to derive an empirical TTS-growth function. They also support the conclusion that patterns of TTS growth and recovery are generally similar to those of terrestrial mammals and that sensation level for the particular species and medium (water or air) is the appropriate metric for comparing the effects of underwater and aerial noise exposure.

#### **Injury from Noise Exposure: PTS-Onset Calculation**

As discussed in Chapter 1, PTS is an irreversible elevation of the hearing threshold (i.e., a reduction in sensitivity) at a specific frequency (Yost, 2000). This permanent change following intense noise exposure results from damage or death of inner or outer cochlear hair cells. It is often followed by retrograde neuronal losses and persistent chemical and metabolic cochlear abnormalities (Saunders et al., 1991; Ward, 1997; Yost, 2000).

Noise-induced PTS represents tissue injury, but TTS does not. Although TTS involves reduced hearing sensitivity following exposure, it results primarily from the fatigue (as opposed to loss) of cochlear hair cells and supporting structures and is, by definition, reversible (Nordmann et al., 2000). Many mammals, including some pinnipeds (Kastak et al., 1999, 2005) and cetaceans (e.g., Schlundt et al., 2000; Nachtigall et al., 2004), demonstrate full recovery even after repeated TTS. Since TTS represents a temporary change in sensitivity without permanent damage to sensory cells or support structures, it is not considered to represent tissue injury (Ward, 1997). Instead, the onset of tissue injury from noise exposure is considered here as PTS-onset.

PTS as a function of age (*presbycusis*; discussed in Chapter 1) generally appears to be a normal process of aging in mammals (including humans and marine mammals), but no specific allowance for this is included in our proposed exposure criteria.

Data that would be needed to support alternate criteria allowing for presbycusis are lacking. Our approach, which uses TTS data from subjects presumed to have “normal” hearing as the starting point for estimating PTS-onset, is precautionary. It is expected to overestimate damaging effects for those individuals with diminished absolute hearing sensitivity and/or functional bandwidth prior to the exposure.

Data on the effects of noise on terrestrial mammals can be useful in considering the effects on marine mammals in certain conditions (as discussed in Chapter 1) because of similarities in morphology and functional dynamics among mammalian cochleae. Under that premise, it is assumed that a noise exposure capable of inducing 40 dB of TTS will cause PTS-onset in marine mammals. Based on available data for terrestrial mammals, this assumption is likely somewhat precautionary as there is often complete recovery from TTS of this magnitude or greater. Such precaution is appropriate, however, because the precise relationship between TTS and PTS is not fully understood, even for humans and small terrestrial mammals despite hundreds of studies (see Kryter, 1994; Ward, 1997). For marine mammals, this presumably complex relationship is unknown, and likely will remain so. The available marine mammal TTS data provide a basis for establishing a maximum allowable amount of TTS up to which PTS is unlikely, however, and for concluding that PTS is increasingly likely to occur above this point. In using TTS data to estimate the exposure that will cause PTS-onset, our approach is to acknowledge scientific uncertainty and to err on the side of overestimating the possibility of PTS (i.e., on the side of underestimating the exposure required to cause PTS-onset).

In humans, when TTS<sub>2</sub> magnitude for a single exposure exceeds *ca.* 40 dB, the likelihood of PTS begins to increase substantially (Kryter et al., 1966; Kryter, 1994). Threshold shifts greater than 40 dB have been demonstrated to be fully recoverable after some period of time in some terrestrial mammal species (human: Ward, 1959; Ahroon et al., 1996; chinchilla: Miller et al., 1971; Mongolian gerbil [*Meriones unguiculatus*]: Boettcher, 1993). Generally, however, TTS exceeding 40 dB requires a longer recovery time than smaller shifts, suggesting a higher probability of irreversible damage (Ward, 1970) and possibly different underlying mechanisms (Kryter, 1994; Nordman et al., 2000).

Our derivation of proposed injury criteria for marine mammals begins with measured or estimated noise exposure conditions associated with TTS-onset in cetaceans and pinnipeds. Procedures for estimating PTS-onset, assumed to occur in

conditions causing 40 dB of TTS, were derived by combining (1) measured or estimated TTS-onset levels in marine mammals and (2) the estimated “growth” of TTS in certain terrestrial mammals exposed to increasing noise levels. The general PTS-onset procedures differ according to sound type (pulses and nonpulses), the extent of available information, and required extrapolation. To estimate exposure conditions that will result in PTS-onset, SEL and SPL were considered separately.

#### *PTS-Onset for Pulses*

Henderson & Hamernik (1986) reported that in chinchillas exposed to pulses up to a certain level, for each dB of added exposure above that which caused TTS-onset, a further TTS of about 0.5 dB resulted. For the highest exposure levels, as much as 3 dB of additional TTS was found per additional dB of noise. Thus, in extrapolating TTS growth functions from terrestrial to marine mammals, a precautionary approach is justified such as using a slope nearer the upper extreme of this range to estimate the growth of TTS with exposure level.

When dealing with pulsed sound, to estimate SEL exposures coincident with PTS-onset, we assume a slope of 2.3 dB TTS/dB noise. This is relatively precautionary in relation to the data by Henderson & Hamernik (1986) on chinchillas. This slope translates to an injury criterion (for pulses) that is 15 dB above the SEL of exposures causing TTS-onset (defined above as 6 dB TTS). That is, PTS-onset (40 dB TTS) is expected to occur on exposure to an M-weighted SEL 15 dB above that associated with TTS-onset ( $[40 \text{ dB TTS} - 6 \text{ dB TTS}] / [2.3 \text{ dB TTS/dB noise exposure}] \approx 15 \text{ dB noise exposure above TTS-onset}$ ).

In terms of sound pressure, TTS-onset thresholds in marine mammals, particularly cetaceans, are quite high (see above). The predicted PTS-onset values would be very high (perhaps unrealistically so as they would approach the cavitation limit of water) if the aforementioned 15 dB difference between TTS-onset and PTS-onset were assumed. Consequently, an additional precautionary measure was applied by arbitrarily assuming that the pressure difference between TTS-onset and PTS-onset for pulses might be just 6 dB. This results in a TTS “growth” relationship of 6 dB TTS/dB noise (i.e.,  $[40 \text{ dB TTS} - 6 \text{ dB TTS}] / [6 \text{ dB TTS/dB noise exposure}] \approx 6 \text{ dB noise exposure above TTS-onset}$ ). That is an extremely conservative slope function given that it is double the highest rate found in chinchillas by Henderson & Hamernik (1986). This 6 dB of added exposure, above the exposure eliciting TTS-onset, essentially establishes a proposed (unweighted) peak-pressure ceiling value for all sound types.

#### *PTS-Onset for Nonpulse Sounds*

The peak pressure values assumed to be associated with onset of injury (PTS-onset) are numerically equivalent for nonpulse and pulse sounds. Among other considerations, this allows for the possibility that isolated pulses could be embedded within the predominantly nonpulse sound.

To estimate the SEL value that would cause PTS-onset for nonpulse sounds, we used the following procedure. In humans, each added dB of nonpulse noise exposure above TTS-onset results in up to 1.6 dB of additional TTS (Ward et al., 1958, 1959). Assuming this relationship applies to marine mammals, ~20 dB of additional noise exposure above that causing TTS-onset is required to induce PTS-onset (i.e.,  $[40 \text{ dB TTS} - 6 \text{ dB TTS}] / [1.6 \text{ dB TTS/dB noise exposure}] = 21.3 \text{ dB of additional noise exposure}$ ). We rounded this down to a slightly more precautionary value of 20 dB of additional noise exposure above TTS-onset. Consequently, to estimate PTS-onset and derive the SEL injury criteria for nonpulses, we add 20 dB to the M-weighted SEL values estimated to cause TTS-onset. The lone exception to this approach is for pinnipeds in air (discussed below) where a more precautionary TTS growth rate was used based on a relatively large empirical data set (Kastak et al., 2007).

#### **Criteria for Injury from a Single Pulse**

As per the “PTS-Onset Calculation” section of this chapter, the recommended criteria for injury from exposure to a single pulse, expressed in terms of peak pressure, are TTS-onset levels plus 6 dB of additional exposure. In terms of SEL, the recommended criteria are TTS-onset levels plus 15 dB of additional exposure.

For all cetaceans exposed to pulses, the data of Finneran et al. (2002b) were used as the basis for estimating exposures that would lead to TTS-onset (and, consequently, PTS-onset). They estimated that, in a beluga exposed to a single pulse, TTS-onset occurred with unweighted peak levels of 224 dB re: 1  $\mu\text{Pa}$  (peak) and 186 dB re: 1  $\mu\text{Pa}^2\text{-s}$ . The latter is equivalent to a weighted ( $M_{\text{mf}}$ ) SEL exposure of 183 dB re: 1  $\mu\text{Pa}^2\text{-s}$  as some of the energy in the pulse was at low frequencies to which the beluga is less sensitive. Adding 6 dB to the former (224 dB) values, the pressure criterion for injury for mid-frequency cetaceans is therefore 230 dB re: 1  $\mu\text{Pa}$  (peak) (Table 3, Cell 4). Adding 15 dB to the latter (183 dB) value, the M-weighted SEL injury criterion is 198 dB re: 1  $\mu\text{Pa}^2\text{-s}$  (Table 3, Cell 4). These results are assumed to apply (see cetacean procedure, p. 439) to low- and perhaps high-frequency cetaceans (Table 3, Cells 1 & 7, respectively) as well as to

mid-frequency cetaceans. These injury criteria, expressed in SEL, are slightly more precautionary than, but generally consistent with, Ketten's 1998 prediction (pers. comm.) that 30% of individual cetaceans exposed to pulses with an SEL of 205 dB re: 1  $\mu\text{Pa}^2\text{-s}$  would experience PTS.

For pinnipeds in water, there are no empirical data concerning the levels of single pulses that would lead to TTS-onset. At least for the California sea lion, the required exposure is expected to be greater than 183 dB re: 1  $\mu\text{Pa}$  (peak) and 163 dB re: 1  $\mu\text{Pa}^2\text{-s}$  because Finneran et al. (2003) found no TTS in two California sea lions following such exposures. In the absence of specific data on the level of a sound pulse that would cause TTS-onset for pinnipeds in water, we used a three-step process to estimate this value:

- (1) We began with the Finneran et al. (2002b) data on TTS-onset from single pulse exposures in a mid-frequency cetacean. TTS-onset occurred with a peak pressure of 224 dB re: 1  $\mu\text{Pa}$  (peak) and  $M_{\text{mr}}$ -weighted SEL of 183 dB re: 1  $\mu\text{Pa}^2\text{-s}$ .
- (2) We assumed that the known pinniped-to-cetacean difference in TTS-onset upon exposure to nonpulse sounds would also apply (in a relative sense) to pulses. Specifically, with nonpulse sounds, harbor seals experience TTS-onset at *ca.* 12 dB lower RLs than do belugas (i.e., 183 vs 195 dB re: 1  $\mu\text{Pa}^2\text{-s}$ ; Kastak et al., 1999,

2005; Southall et al., 2001; Schusterman et al., 2003 vs Finneran et al., 2000, 2005a; Schlundt et al., 2000; Nachtigall et al., 2003, 2004). Assuming that this difference for nonpulse sounds exists for pulses as well, TTS-onset in pinnipeds exposed to single underwater pulses is estimated to occur at a peak pressure of 212 dB re: 1  $\mu\text{Pa}$  (peak) and/or an SEL exposure of 171 dB re: 1  $\mu\text{Pa}^2\text{-s}$ . Each of these metrics is 12 dB less than the comparable value for mid-frequency cetaceans (see Finneran et al., 2002b, and above).

- (3) As per the "PTS-onset Procedure" (discussed earlier), we added 6 dB to the former (212 dB) value to derive the recommended injury pressure criterion of 218 dB re: 1  $\mu\text{Pa}$  (peak) (unweighted) for pinnipeds in water exposed to a single pulse. Similarly, we added 15 dB to the latter value (171 dB) to derive the recommended M-weighted SEL injury criterion of 186 dB re: 1  $\mu\text{Pa}^2\text{-s}$  (Table 3, Cell 10). These proposed criteria are likely precautionary because the harbor seal is the most sensitive pinniped species tested to date, based on results from a single individual (Kastak et al., 1999, 2005).

For pinnipeds in air exposed to a single sound pulse, the proposed criteria for injury were based on measurements by Bowles et al. (unpub. data), which indicated that TTS-onset in harbor

**Table 3.** Proposed injury criteria for individual marine mammals exposed to "discrete" noise events (either single or multiple exposures within a 24-h period; see Chapter 2)

Marine mammal group	Sound type		
	Single pulses	Multiple pulses	Nonpulses
Low-frequency cetaceans	Cell 1	Cell 2	Cell 3
Sound pressure level	230 dB re: 1 $\mu\text{Pa}$ (peak) (flat)	230 dB re: 1 $\mu\text{Pa}$ (peak) (flat)	230 dB re: 1 $\mu\text{Pa}$ (peak) (flat)
Sound exposure level	198 dB re: 1 $\mu\text{Pa}^2\text{-s}$ ( $M_{\text{lr}}$ )	198 dB re: 1 $\mu\text{Pa}^2\text{-s}$ ( $M_{\text{lr}}$ )	215 dB re: 1 $\mu\text{Pa}^2\text{-s}$ ( $M_{\text{lr}}$ )
Mid-frequency cetaceans	Cell 4	Cell 5	Cell 6
Sound pressure level	230 dB re: 1 $\mu\text{Pa}$ (peak) (flat)	230 dB re: 1 $\mu\text{Pa}$ (peak) (flat)	230 dB re: 1 $\mu\text{Pa}$ (peak) (flat)
Sound exposure level	198 dB re: 1 $\mu\text{Pa}^2\text{-s}$ ( $M_{\text{mr}}$ )	198 dB re: 1 $\mu\text{Pa}^2\text{-s}$ ( $M_{\text{mr}}$ )	215 dB re: 1 $\mu\text{Pa}^2\text{-s}$ ( $M_{\text{mr}}$ )
High-frequency cetaceans	Cell 7	Cell 8	Cell 9
Sound pressure level	230 dB re: 1 $\mu\text{Pa}$ (peak) (flat)	230 dB re: 1 $\mu\text{Pa}$ (peak) (flat)	230 dB re: 1 $\mu\text{Pa}$ (peak) (flat)
Sound exposure level	198 dB re: 1 $\mu\text{Pa}^2\text{-s}$ ( $M_{\text{hr}}$ )	198 dB re: 1 $\mu\text{Pa}^2\text{-s}$ ( $M_{\text{hr}}$ )	215 dB re: 1 $\mu\text{Pa}^2\text{-s}$ ( $M_{\text{hr}}$ )
Pinnipeds (in water)	Cell 10	Cell 11	Cell 12
Sound pressure level	218 dB re: 1 $\mu\text{Pa}$ (peak) (flat)	218 dB re: 1 $\mu\text{Pa}$ (peak) (flat)	218 dB re: 1 $\mu\text{Pa}$ (peak) (flat)
Sound exposure level	186 dB re: 1 $\mu\text{Pa}^2\text{-s}$ ( $M_{\text{pw}}$ )	186 dB re: 1 $\mu\text{Pa}^2\text{-s}$ ( $M_{\text{pw}}$ )	203 dB re: 1 $\mu\text{Pa}^2\text{-s}$ ( $M_{\text{pw}}$ )
Pinnipeds (in air)	Cell 13	Cell 14	Cell 15
Sound pressure level	149 dB re: 20 $\mu\text{Pa}$ (peak) (flat)	149 dB re: 20 $\mu\text{Pa}$ (peak) (flat)	149 dB re: 20 $\mu\text{Pa}$ (peak) (flat)
Sound exposure level	144 dB re: (20 $\mu\text{Pa}$ ) <sup>2</sup> -s ( $M_{\text{pa}}$ )	144 dB re: (20 $\mu\text{Pa}$ ) <sup>2</sup> -s ( $M_{\text{pa}}$ )	144.5 dB re: (20 $\mu\text{Pa}$ ) <sup>2</sup> -s ( $M_{\text{pa}}$ )

**Note:** All criteria in the "Sound pressure level" lines are based on the peak pressure known or assumed to elicit TTS-onset, plus 6 dB. Criteria in the "Sound exposure level" lines are based on the SEL eliciting TTS-onset plus (1) 15 dB for any type of marine mammal exposed to single or multiple pulses, (2) 20 dB for cetaceans or pinnipeds in water exposed to nonpulses, or (3) 13.5 dB for pinnipeds in air exposed to nonpulses. See text for details and derivation.

seals occurs following exposure to 143 dB re: 20  $\mu$ Pa (peak) and 129 dB re: (20  $\mu$ Pa)<sup>2</sup>-s. As for underwater exposures to nonpulse sounds (Kastak et al., 1999, 2005), higher exposure levels were required to induce TTS in California sea lions and northern elephant seals. Consequently, using harbor seal TTS data to establish injury criteria for exposure to a single aerial pulse in pinnipeds is likely a precautionary approximation. Based on these estimates of peak pressure and SEL associated with TTS-onset, plus 6 dB and 15 dB, respectively, to estimate PTS-onset, the injury criteria for pinnipeds exposed to a single aerial pulse are 149 dB re: 20  $\mu$ Pa (peak) (unweighted) and 144 dB re: (20  $\mu$ Pa)<sup>2</sup>-s, M-weighted (Table 3, Cell 13).

### Criteria for Injury from Multiple Pulses

For all marine mammal groups, the recommended criteria for exposure to multiple pulses, expressed in both SPL and SEL units, were numerically identical to the criteria for a single pulse. Any exposure in a series that exceeds the peak pressure criterion would be considered potentially injurious. In addition, the cumulative SEL for multiple exposures should be calculated using the summation technique described in Chapter 1 (Appendix A, eq. 5). The resulting SEL value for multiple pulses is then compared to the SEL injury criterion for a single pulse in the same functional hearing group. As for the single pulse criteria, peak pressures are unweighted (i.e., “flat-weighted”), but SEL should be weighted by the appropriate M-weighting function (Figure 1).

For cetaceans, the proposed criteria for injury by multiple pulses are therefore 230 dB re: 1  $\mu$ Pa (peak) and, following summation, 198 dB re: 1  $\mu$ Pa<sup>2</sup>-s in terms of SEL (Table 3, Cells 2, 5 & 8). As for single pulses, this approach is considered precautionary for mid- and low-frequency species, but some caution is warranted in applying it to high-frequency species (*cf.* Lucke et al., 2007a).

Following the same logic, the proposed injury pressure criterion for pinnipeds in water exposed to multiple pulses is 218 dB re: 1  $\mu$ Pa (peak) and the injury SEL criterion is 186 dB re: 1  $\mu$ Pa<sup>2</sup>-s (Table 3, Cell 11). For pinnipeds in air, the proposed injury pressure criterion for multiple pulses is 149 dB re: 20  $\mu$ Pa (peak) and the injury SEL criterion is 144 dB re: (20  $\mu$ Pa)<sup>2</sup>-s (Table 3, Cell 14).

### Criteria for Injury from Nonpulses

SPL and SEL appear to be appropriate metrics for quantifying exposure to nonpulse sounds. But because SPL measures involve averaging over some duration, they may not adequately quantify

high peak pressure transients embedded within exposures of longer duration but lower-pressure magnitude. There are related limitations with SEL in that temporal integration is involved.

To account for the potentially damaging aspects of high-pressure transients embedded within nonpulse exposures, a precautionary approach was taken, and the same peak pressure criterion for injury proposed for single pulses is also recommended as the criterion for multiple pulses in all functional hearing groups. Thus, if any component of a nonpulse exposure (unweighted) exceeds the peak pressure criterion, injury is assumed to occur. We expect that only rarely will the injury pressure criterion for nonpulse sound be exceeded if the injury SEL criterion is not exceeded (i.e., the SEL criterion will be the effective criterion in most exposure conditions).

For nonpulsed sounds, the recommended SEL criteria for injury (PTS-onset) are M-weighted exposures 20 dB higher than those required for TTS-onset (see “PTS-Onset Calculation: Nonpulses”). Injury SEL criteria for multiple nonpulses are numerically identical to those for single nonpulses for all hearing groups. We make no distinction between single and multiple nonpulses except that the cumulative SEL for multiple exposures is calculated as described in Chapter 1 and Appendix A, eq. 5.

For all cetaceans exposed to nonpulses, the recommended pressure criterion for injury is 230 dB re: 1  $\mu$ Pa (peak) (Table 3, Cells 3, 6, & 9), the same criterion as for single pulses in these functional hearing groups. Injury SEL criteria are based on TTS data for mid-frequency species and extrapolated to the other cetacean groups (see cetacean procedure, p. 439). The SEL criterion for nonpulse injury in cetaceans is calculated to be an M-weighted exposure of 215 dB re: 1  $\mu$ Pa<sup>2</sup>-s (Table 3, Cells 3, 6 & 9). This is based on 195 dB re: 1  $\mu$ Pa<sup>2</sup>-s as an estimate of TTS-onset in mid-frequency cetaceans (Finneran et al., 2002b, 2005a; Schlundt et al., 2000; Nachtigall et al., 2003, 2004) plus 20 dB to estimate PTS-onset. Applying this approach to low-frequency cetaceans is considered precautionary, but some caution may be warranted in extrapolating to high-frequency cetaceans (*cf.* single-pulse data of Lucke et al., 2007a).

We note that special injury criteria, different from those shown in Cell 6 of Table 3, are likely needed for exposure of beaked whale species to nonpulses. Under certain conditions, beaked whales of several species (primarily Cuvier’s, Blainville’s, and Gervais’ beaked whales) have stranded in the presence of sound signals from tactical mid-frequency military sonars (Frantzis, 1998; Evans & England, 2001; Fernández et al., 2005; Cox et al., 2006). There have been other

incidents (e.g., NMFS, 2005; Hohn et al., 2006) where marine mammal strandings or other anomalous events involving other marine mammal species have occurred in association with mid-frequency sonar operations. They are, however, much more ambiguous, difficult to interpret, and appear fundamentally different than the specific beaked whale events. Little is known about the exposure levels, or about the positions or reactions of other marine mammals in the areas during mid-frequency sonar training operations. The most extreme, ultimate response of some beaked whales in specific conditions (stranding and subsequent death) does not appear to be typical of other marine mammals.

Sound fields resulting from sonar operations have been modeled in several of the above cases (e.g., the 1996 event in Greece and the 2000 event in the Bahamas), and it is possible to at least roughly bound the estimated exposures for some of the individuals that stranded (D'Spain et al., 2006). While the specific exposure levels will never be quantitatively known, it does appear likely that the exposures for some of the beaked whales that stranded were below the criteria for tissue injury proposed above.

Consequently, the general injury criteria do not seem sufficiently precautionary for beaked whales exposed to some nonpulse sounds under certain conditions. Empirical data to support discrete, science-based injury criteria specific to beaked whales exposed to tactical, mid-frequency, military sonar are lacking, however. Regulatory agencies should consider adopting provisional injury criteria for beaked whales exposed to active, mid-frequency, military sonars that are lower (in terms of RL) than the criteria used for mid-frequency cetaceans and nonpulse sources generally. Of foremost importance, specific studies are needed to better define the mechanism of injury in these apparently sensitive species (see Chapter 5).

For pinnipeds in water, the recommended pressure criterion for injury from exposure to nonpulse sounds is the same value as applied to pulses: an unweighted value of 218 dB re: 1  $\mu$ Pa (peak) (Table 3, Cell 12). To derive the associated SEL criterion, we began with the measured nonpulse exposure eliciting TTS-onset in a harbor seal, 183 dB re: 1  $\mu$ Pa<sup>2</sup>-s (Kastak et al., 1999, 2005). This is likely a precautionary choice because SEL values ~10 to 20 dB higher were required to induce TTS-onset in a California sea lion and a northern elephant seal. We assume that 20 dB of additional noise exposure will elicit PTS-onset (see "Effects of Noise on Hearing" section of this chapter), resulting in an  $M_{pw}$ -weighted SEL criterion of 203 dB re: 1  $\mu$ Pa<sup>2</sup>-s for pinnipeds exposed to nonpulse sound in water (Table 3, Cell 12).

For pinnipeds in air exposed to nonpulse sound, the injury pressure criterion is a flat-weighted value of 149 dB re: 20  $\mu$ Pa (peak) (Table 3, Cell 15), consistent with that for pulses. The SEL criterion is based on occurrence of TTS-onset in a harbor seal exposed in air to 131 dB re: (20  $\mu$ Pa)<sup>2</sup>-s (Kastak et al., 2004a). In estimating the exposure that would cause PTS-onset, we use empirical measurements of TTS growth as a function of exposure SEL in a California sea lion. Kastak et al. (2007) found a TTS growth rate of 2.5 dB TTS/dB noise based on nearly 200 exposure sequences involving variable exposure level and duration conditions. This growth rate implies a 13.5 dB difference between TTS- and PTS-onset as opposed to the 20 dB value used for marine mammals in water. When the 13.5 dB figure is added to the TTS-onset value for harbor seals (131 dB re: [20  $\mu$ Pa]<sup>2</sup>-s), we obtain a proposed  $M_{pw}$ -weighted SEL criterion of 144.5 dB re: (20  $\mu$ Pa)<sup>2</sup>-s for pinnipeds in air (Table 3, Cell 15).

The use for all pinnipeds of harbor seal TTS data combined with the sea lion growth function would be an exceedingly precautionary procedure. This PTS-onset estimate is considerably below the TTS-onset estimates for both the northern elephant seal (163 dB re: [20  $\mu$ Pa]<sup>2</sup>-s; Kastak et al., 2004a) and the California sea lion (159 dB re: [20  $\mu$ Pa]<sup>2</sup>-s; Kastak et al., 2007). Applying the TTS growth function of 2.5 dB TTS/dB noise from Kastak et al. (2007) to these TTS-onset estimates would yield PTS-onset values of 172.5 and 176.5 dB re: (20  $\mu$ Pa)<sup>2</sup>-s for the California sea lion and northern elephant seal, respectively. As noted in the "Overview," where specific data are available for the species or genus of concern, it is appropriate for criteria to be based on those data rather than the generalized criteria that are recommended for the overall group of marine mammals.

## 4. Criteria for Behavioral Disturbance

Behavioral reactions to acoustic exposure are generally more variable, context-dependent, and less predictable than effects of noise exposure on hearing or physiology. Animals detecting one kind of signal may simply orient to hear it, whereas they might panic and flee for many hours upon hearing a different sound, potentially even one that is quieter, but with some particular significance to the animal. The conservation of cochlear properties across mammals justifies judicious application of auditory data from terrestrial mammals where data on marine mammals are missing. However, the context-specificity of behavioral responses in animals generally makes extrapolation of behavioral data inappropriate. Assessing the severity of behavioral disturbance must consequently rely more on empirical studies with carefully controlled acoustic, contextual, and response variables than on extrapolations based on shared phylogeny or morphology.

Considerable research has been conducted to describe the behavioral responses of marine mammals to various sound sources. Fortunately, at least limited data are available on behavioral responses by each of the five functional marine mammal groups to each sound type considered here. As evident in the extensive literature review summarized below and described in detail in Appendices B & C, however, very few studies involving sufficient controls and measurements exist. In addition, the influence of experience with the experimental stimulus or similar sounds has usually been unknown.

To assess and quantify adverse behavioral effects of noise exposure, a metric for the impact such changes might have on critical biological parameters such as growth, survival, and reproduction is needed. Behavioral disturbances that affect these vital rates have been identified as particularly important in assessing the significance of noise exposure (NRC, 2005). Unfortunately, as Wartzok et al. (2004) pointed out, no such metric is currently available, and it is likely to take decades of research to provide the analytical framework and empirical results needed to create such a metric, if one in fact is ultimately even viable.

In humans, a common and useful means of estimating behavioral disturbance from noise exposure is to ask individuals to rate or describe the degree to which various sounds are bothersome. Subjective perception of noise "annoyance" has

been quantified (e.g., Schultz, 1978; Angerer et al., 1991) and used to develop dose-response relationships for noise exposure in human community noise applications (see Kryter, 1994, Chapter 10). Practical issues (e.g., difficulties in training nonverbal species to provide interpretable responses and questions about the applicability of captive data to free-ranging animals) have prevented this or similar approaches from being applied to marine mammals. Instead, most efforts have focused on analyses of observable reactions to known noise exposure.

For most free-ranging marine mammals, behavioral responses are often difficult to observe. Also, precise measurements of received noise exposure and other relevant variables (e.g., movement of source, presence of high-frequency harmonics indicating relative proximity, and prior experience of exposed individuals) can be difficult to obtain. Only a subset of disturbance studies have estimated received sound levels, and only a very small number have actually measured RLs at the subject. Further, exposures are often complicated by multiple contextual covariants such as the presence of vessels and/or humans close to subjects either for observation or to deploy playback sources (e.g., Frankel & Clark, 1998). Interpretation of the observed results is highly limited by uncertainty as to what does and does not constitute a meaningful response. Also, most behavioral-response studies have concentrated on short-term and localized behavioral changes whose relevance to individual well-being and fitness, let alone population parameters, is likely to be low.

A further complication is that observations from laboratory and field settings cannot be directly equated. Laboratory studies are usually precise in quantifying exposures and responses. The exposure conditions very rarely approximate those in the field, however, and measured behavior may have little or no relevance to the ways in which unconstrained, untrained wild animals respond. Conversely, field measurements may address responses of free-ranging mammals to a specific sound source but often lack adequate controls and precision in quantifying acoustic exposures and responses. Clearly, there is a need for a framework to integrate laboratory and field data, despite the challenges in constructing that framework.

Another difficult issue concerns the appropriate noise exposure metric for assessing behavioral

reactions. Most bioacousticians recommend reporting several different measures of acoustic exposure, such as SPL and SEL (as in Blackwell et al., 2004a, 2004b). Of the many studies that report source SPL, relatively few specify whether RMS, peak, peak-to-peak, or other sound pressure measurements were made. Additionally, relatively few papers provide sufficient relevant information about sound transmission loss in the study area. A small number of papers report estimates or direct measurements of received SPL, but very few report SEL. The appropriate measure for predicting probability of a behavioral response is likely to vary depending upon the behavioral context. For example, if an animal interprets a sound as indicating the presence of a predator, a short faint signal may evoke as strong a response as a longer, strong sound. But if an animal is responding to a context-neutral stimulus that is merely annoying, the probability of response may well scale with duration and level of exposure.

It is difficult to define the SEL for individual animals in the wild exposed to a specific sound source. Ideally, received SEL over the animal's full duration of exposure would be measured (Madsen et al., 2005a). We expect that the probability and severity of some kinds of response will vary with duration as well as level of exposure; for those situations, an SEL metric may be most appropriate. However, the most practical way to look for consistent patterns of response as a function of RL and duration, given the current state of science, is to evaluate how different animals respond to similar sound sources used in similar contexts. For example, the relationship between acoustic exposure and animal responses is likely to be quite different for mammals exposed to sounds from a slow-moving seismic survey vessel operating in a given habitat for many weeks as compared with a torpedo transmitting directional high-frequency sonar pings as it transits an area once at many tens of knots. Similarly, an acoustic harassment device placed in a habitat for years is likely to evoke a different severity of response than would several short pulses at a comparable SPL. Until more controlled studies become available with calibrated measurements of RLs and ambient noise measurements (including signal-to-noise ratio), the best way to predict likely effects will be a common-sense approach that assesses available data from situations similar to the situation of concern.

Considering all of these limitations and the nature of the available data, as a practical matter, we use SPL as the acoustic metric for the behavioral analyses given below. Where necessary and appropriate, simple assumptions regarding transmission loss were applied to predict RLs. This

was done only for studies that provided sufficient information on source and environmental characteristics. Our approach does not presume that SPL is necessarily the acoustic metric best correlated with behavioral changes (significant or otherwise). In particular, SPL fails to account for the duration of exposure whereas this is captured using SEL. SPL is the metric that has most often been measured or estimated during disturbance studies, however. Thus, it is currently the best metric with which to assess the available behavioral response data. Future studies should report the full range of standard acoustic measurements appropriate to the sound source in question and should also include measurements of background noise levels in order to assess signal-to-noise ratios. These additional data should eventually clarify which exposure metrics best predict different kinds of behavioral responses and which are most appropriate for use in policy guidelines applicable to different types of noise exposures.

Beyond the discussion of which metric is most appropriate to quantify the exposure level of a sound, it is recognized that many other variables affect the nature and extent of responses to a particular stimulus. Wartzok et al. (2004) discussed in detail the highly variable response of belugas exposed to similar sounds in different locations—for example, Frost et al. (1984) vs Finley et al. (1990). In those cases, it appears that the context (recent experience of the belugas with the sound stimulus, their current activity, and their motivation to remain or leave) was much more significant in governing their behavioral responses. Similarly, reactions of bowhead whales to seismic airgun sounds depend on whether the whales are feeding (Richardson et al., 1986; Miller et al., 2005) vs migrating (Richardson et al., 1999). Reactions of bowheads and other cetaceans to boats depend on whether the boats are moving or stationary, and on the relative movement of the boat and the whale (see Richardson et al., 1995; Wartzok et al., 2004). In these and some other cases, simple metrics of exposure (without considering context) will not reliably predict the type and severity of behavioral response(s). Our analyses here, which use exposure SPL alone, are admittedly rudimentary and limited by the fact that—for most species and situations—current data do not support a more sophisticated approach.

Another key consideration involves differentiating brief, minor, biologically unimportant reactions from profound, sustained, and/or biologically meaningful responses related to growth, survival, and reproduction. The biological relevance of a behavioral response to noise exposure may depend in part on how long it persists. Many mammals perform vital functions (e.g.,

feeding, resting, traveling, socializing) on a diel cycle. Repeated or sustained disruption of these functions is more likely to have a demonstrable effect on vital rates than a single, brief disturbance episode. The NRC (2005) argued that, although the duration of behaviors likely to affect vital rates is believed to be particularly significant, current scientific knowledge is insufficient to support an analytical treatment of biological significance and ad hoc criteria are needed in the interim. Here, substantive behavioral reactions to noise exposure (such as disruption of critical life functions, displacement, or avoidance of important habitat) are considered more likely to be significant if they last more than one diel period, or recur on subsequent days. Consequently, a reaction lasting less than 24 h and not recurring on subsequent days is not regarded as particularly severe unless it could directly affect survival or reproduction.

In the absence of an overarching means of quantifying the biological significance of an effect, we had to adopt a more descriptive method of assessing the range of possible responses and the severity of behavioral response. To do this, we took two different approaches. For the unusual case of exposure to a single pulse, where the exposure is very brief and responses are usually brief as well, a procedure for determining recommended criteria is identified and applied. For all other conditions, an ordinal and subjective response severity scaling was developed and applied to those data on marine mammal behavioral responses for which estimates of received SPL were available. These analyses were limited to peer-reviewed literature (published or in press) and peer-reviewed technical reports, with some exceptions on a case-by-case basis.

The severity scale was designed to provide some analytical basis for assessing biological significance, but it had to be rooted in the kinds of descriptions provided in the available scientific literature. Our current understanding of the influences of contextual variables on behavioral responses in free-ranging marine mammals is very limited. The analyses presented here should be considered with these cautions and caveats in mind. Our goal was to review the relevant scientific literature, tally behavioral effects by the type of acoustic exposure for each category of marine mammal and sound type, and draw what conclusions were appropriate based on the information available.

The general procedures for determining behavioral response exposure criteria for a single pulse, and for conducting the severity analyses of individual behavioral responses vs received SPL, are discussed in the next section. Subsequent sections discuss the exposure criterion levels for single

pulses and summarize the literature considered in the severity scaling analyses for multiple pulses and nonpulse sources. More detailed discussions of this literature are given in Appendix B for multiple pulses and Appendix C for nonpulse sources.

### **Behavioral Response Data Analysis Procedures: Disturbance Criteria and Severity Scaling**

#### *Single Pulse*

Due to the transient nature of a single pulse, the most severe behavioral reactions will usually be temporary responses, such as startle, rather than prolonged effects, such as modified habitat utilization. A transient behavioral response to a single pulse is unlikely to result in demonstrable effects on individual growth, survival, or reproduction. Consequently, for the unique condition of a single pulse, an auditory effect is used as a *de facto* disturbance criterion. It is assumed that significant behavioral disturbance might occur if noise exposure is sufficient to have a measurable transient effect on hearing (i.e., TTS-onset). Although TTS is not a behavioral effect per se, this approach is used because any compromise, even temporarily, to hearing functions has the potential to affect vital rates by interfering with essential communication and/or detection capabilities. This approach is expected to be precautionary because TTS at onset levels is unlikely to last a full diel cycle or to have serious biological consequences during the time TTS persists. Because this approach is based on an auditory phenomenon, the exposure criteria can reasonably be developed for entire functional hearing groups (as in the injury criteria) rather than on a species-by-species basis. The extrapolation procedures used to estimate TTS-onset for single pulse exposures for each hearing group are described in Chapter 3 (see the "Injury from Noise Exposure: PTS-Onset Calculation" section).

A dual-criterion approach (using both SPL [peak] and SEL) was used to determine behavioral criteria for a single pulse exposure. Consistent with the injury criteria, which also were based on auditory fatigue data, RLs that exceed the criterion for either metric are considered to have greater potential to elicit a biologically significant behavioral response. Proposed criteria for exposure to a single pulse for each functional hearing group are given in the next section. These criteria are the TTS-onset thresholds discussed in Chapter 3.

An exception was made in any case where behavioral data indicate that a single pulse exposure may elicit a sustained and potentially significant response when the RL is below that required for TTS-onset. This can apply to hauled-out pinnipeds, which sometimes stampede from a beach

upon exposure to a sonic boom and may not return for many hours (e.g., Holst et al., 2005a, 2005b). In cases where such behavioral responses may result in the injury or death of pups or other individuals, exposure levels should be considered in the context of injury criteria. Conversely, if available behavioral data indicate that the response threshold for exposure to a single pulse is above the level required for TTS-onset, then the TTS-onset level is retained as the behavioral criterion as a further precautionary procedure.

#### *Multiple Pulses and Nonpulses*

For all other sound types than single pulses, we expect that significant behavioral effects will occur more commonly at levels below those involved in temporary or permanent losses of hearing sensitivity. This argues against basing threshold criteria exclusively on TTS and indicates the need for a paradigm to predict the probability of significant behavioral response as a function of noise exposure. However, because of the extreme degree of group, species, and individual variability in behavioral responses in various contexts and conditions, it is less appropriate to extrapolate behavioral effects as opposed to auditory responses. The available data on marine mammal behavioral responses to multiple pulse and nonpulse sounds are simply too variable and context-specific to justify proposing single disturbance criteria for broad categories of taxa and of sounds.

This should not, however, lead to the conclusion that there are insufficient data to conduct a systematic assessment of the likelihood that certain sound exposures will induce behavioral effects of variable seriousness in marine mammals. On the contrary, this field has seen many and accelerating strides in characterizing how certain kinds of sounds can affect marine mammal behavior. Quantification of the severity or significance of these effects will continue to be challenging. However, based on the NRC (2005) model described above in which behavioral reactions with a greater potential to affect vital rates are of particular concern, a simplistic scaling paradigm in which to consider the available data appears to provide the most justifiable way forward at present.

First, we developed an ordinal ranking of behavioral response severity (see Table 4). The intent of this scaling was to delineate those behaviors that are relatively minor and/or brief (scores 0-3); those with higher potential to affect foraging, reproduction, or survival (scores 4-6); and those considered likely to affect these vital rates (scores 7-9). This is an admittedly simplistic way of scaling the strikingly complex and poorly understood behavioral patterns of marine mammals in real-world conditions. It does provide a

rudimentary framework for assessing the relative biological importance of behavioral responses and is likely a closer approximation of reality than previous step-function thresholds (as discussed in the “Historical Perspectives” section of Chapter 1). This approach emphasizes that “disturbance” is a graduated, rather than a “yes-or-no,” phenomenon and that some noise-induced changes in behavior are more significant than others. We expect that future studies involving multivariate analysis of multiple behavioral response variables, multiple measures of acoustic exposure, and multiple contextual variables will provide a foundation for more sophisticated interpretations.

Second, we reviewed available research and observations for each of the five marine mammal functional hearing groups exposed to either multiple pulse or nonpulse sounds (i.e., Cells 2, 3, 5, 6, 8, 9, 11, 12, 14 & 15 in our matrix of sound type by animal group). We considered measurements of behavioral response both in the field and in the laboratory according to the behavioral severity scale. Studies with insufficient information on exposures and/or responses were considered but not included in the severity analysis. Where individual (and/or groups considered as an “individual”; see below) behavioral responses and associated received sound levels were reported, the observations were assigned the appropriate behavioral “score” from Table 4 and the case was included in a severity scoring table for the relevant matrix cell. One dimension in this type of table was the behavioral score (defined in Table 4); the other dimension was the received SPL within 10-dB ranges. Where multiple responses were reported for the same individual and/or group in a study (or where it was possible that this had been done—pseudoreplication), appropriate fractions of a single observation were assigned to relevant cells in the scoring table. As a result, there are fractional responses for some individual and/or group responses in the tabular severity-scaling forms. For example, a single behavioral observation for one individual was weighted as equivalent to ten observations for another individual by assigning each observation (some potentially in different RL/severity score bins) of the second individual a relative weight of 0.1.

Many observations of marine mammals involve multiple individuals because many species occur in large social groups and are followed as a group. In this case, if one individual responds to a sound, the other group members may respond to the response as opposed to the sound. In such observations, the full group was considered to represent an “individual” (i.e., the group became the unit of analysis). As a precautionary approach, the most severe response by any individual observed

**Table 4.** Severity scale for ranking observed behavioral responses of free-ranging marine mammals and laboratory subjects to various types of anthropogenic sound

Response score <sup>1</sup>	Corresponding behaviors (Free-ranging subjects) <sup>2</sup>	Corresponding behaviors (Laboratory subjects) <sup>2</sup>
0	- No observable response	- No observable response
1	- Brief orientation response (investigation/visual orientation)	- No observable response
2	- Moderate or multiple orientation behaviors - Brief or minor cessation/modification of vocal behavior - Brief or minor change in respiration rates	- No observable negative response; may approach sounds as a novel object
3	- Prolonged orientation behavior - Individual alert behavior - Minor changes in locomotion speed, direction, and/or dive profile but no avoidance of sound source - Moderate change in respiration rate - Minor cessation or modification of vocal behavior (duration < duration of source operation), including the Lombard Effect	- Minor changes in response to trained behaviors (e.g., delay in stationing, extended inter-trial intervals)
4	- Moderate changes in locomotion speed, direction, and/or dive profile but no avoidance of sound source - Brief, minor shift in group distribution - Moderate cessation or modification of vocal behavior (duration ≈ duration of source operation)	- Moderate changes in response to trained behaviors (e.g., reluctance to return to station, long inter-trial intervals)
5	- Extensive or prolonged changes in locomotion speed, direction, and/or dive profile but no avoidance of sound source - Moderate shift in group distribution - Change in inter-animal distance and/or group size (aggregation or separation) - Prolonged cessation or modification of vocal behavior (duration > duration of source operation)	- Severe and sustained changes in trained behaviors (e.g., breaking away from station during experimental sessions)
6	- Minor or moderate individual and/or group avoidance of sound source - Brief or minor separation of females and dependent offspring - Aggressive behavior related to noise exposure (e.g., tail/flipper slapping, fluke display, jaw clapping/gnashing teeth, abrupt directed movement, bubble clouds) - Extended cessation or modification of vocal behavior - Visible startle response - Brief cessation of reproductive behavior	- Refusal to initiate trained tasks
7	- Extensive or prolonged aggressive behavior - Moderate separation of females and dependent offspring - Clear anti-predator response - Severe and/or sustained avoidance of sound source - Moderate cessation of reproductive behavior	- Avoidance of experimental situation or retreat to refuge area (≤ duration of experiment) - Threatening or attacking the sound source
8	- Obvious aversion and/or progressive sensitization - Prolonged or significant separation of females and dependent offspring with disruption of acoustic reunion mechanisms - Long-term avoidance of area (> source operation) - Prolonged cessation of reproductive behavior	- Avoidance of or sensitization to experimental situation or retreat to refuge area (> duration of experiment)
9	- Outright panic, flight, stampede, attack of conspecifics, or stranding events - Avoidance behavior related to predator detection	- Total avoidance of sound exposure area and refusal to perform trained behaviors for greater than a day

<sup>1</sup>Ordinal scores of behavioral response severity are not necessarily equivalent for free-ranging vs laboratory conditions.

<sup>2</sup>Any single response results in the corresponding score (i.e., all group members and behavioral responses need not be observed). If multiple responses are observed, the one with the highest score is used for analysis.

within a group was used as the ranking for the whole group.

A specific category of behavioral studies was one in which marine mammal distributions were measured around a sound source during quiet and active periods. The available data typically involve comparisons of the distribution of animals before exposure (“control” or “reference”) vs during exposure (“experimental”); the difference in distribution of the group was the behavioral response. Using this method, and given equivalent range measurements for control and experimental observations, “phantom” RLs for mammals detected during control periods (RLs that would have existed if in fact the source was active) can be calculated and compared to actual RLs during experimental conditions. In this way, the percentage of avoidance responses by individuals during the exposure was then calculated.

For the studies used in this analysis, noise exposure (including source and RL, frequency, duration, duty cycle, and other factors) was either directly reported or was reasonably estimated using simple sound propagation models deemed appropriate for the sources and operational environment. Because of the general lack of precision in many studies and the difficulties in pooling the results from disparate studies here, we pooled individual exposure SPL into 10-dB bins.

Our analysis of the available behavioral response studies presents raw, individual observations of reactions to multiple pulses and non-pulses as a function of exposure RL. The basic output of this procedure is a series of tables, one for each combination of the five marine mammal functional hearing groups and these two sound types (multiple pulses and nonpulses). The overall tally within each cell represents the number of individuals and/or independent group behavioral responses with estimated and/or measured RL in the specified 10-dB category.

This analysis is intended to provide some foundation for judging the degree to which available data suggest the existence of dose-response relationships between noise exposure and marine mammal behavior. An example of such a dose-response function is the Schultz (1978) curve used to predict growth of human annoyance with increasing noise level. The reader should note, however, that the substantial, acknowledged caveats and limitations of the current approach, particularly those related to contextual variables other than simply exposure level. Any application of the severity analyses given below should carefully consider the nature of the available information regarding sound source, species, sex/age class, sound-propagation environment, and especially the overall context of exposure relative to that

shown in the studies reviewed here. The results from prior behavioral studies in which these variables are fairly similar to those in the anticipated exposure situation will very likely be the most relevant. Information from those studies should be most strongly weighted in assessing the likelihood of significant behavioral disturbance.

### Criteria for Behavioral Disturbance: Single Pulse

For all cetaceans exposed to single pulses, the criteria were based on the Finneran et al. (2002b) results for TTS-onset in a beluga exposed to a single pulse. The unweighted peak sound pressure values of 224 dB re: 1  $\mu$ Pa (peak) and weighted ( $M_{mf}$ ) SEL values of 183 dB re: 1  $\mu$ Pa<sup>2</sup>-s are recommended as “behavioral” disturbance criteria for mid-frequency cetaceans (Table 5, Cell 4). By extrapolation (see cetacean procedure, Chapter 3, p. 439), the same values were also proposed for low- and high-frequency cetaceans (Table 5, Cells 1 & 7, respectively). The only difference in the application of these criteria to the three cetacean groups is the influence of the respective frequency-weighting functions for SEL criteria ( $M_{lf}$  and  $M_{mf}$  vs  $M_{mf}$ ).

For pinnipeds exposed to single pulses in water, the proposed “behavioral” disturbance criteria are also the estimated TTS-onset values. For pinnipeds as a whole, these are 212 dB re: 1  $\mu$ Pa (peak) and weighted ( $M_{pw}$ ) SEL of 171 dB re: 1  $\mu$ Pa<sup>2</sup>-s (Table 5, Cell 10) as discussed in Chapter 3.

For pinnipeds in air, the proposed behavioral criteria are based on the strong responses (stamping behavior that could injure some individuals or separate mothers from pups) of some species, especially harbor seals, to sonic booms from aircraft and missile launches in certain conditions (Berg et al., 2001, 2002; Holst et al., 2005a, 2005b). No responses resulting in injury were observed in these specific studies, but the behavioral responses were, in some cases, among those that would be considered relatively severe in regards to vital rates. It was therefore determined appropriate to use results from these studies rather than TTS-based thresholds for behavioral response criteria. The proposed criteria are 109 dB re: 20  $\mu$ Pa (peak) and frequency-weighted ( $M_{pw}$ ) SEL of 100 dB re: (20  $\mu$ Pa)<sup>2</sup>-s (Table 5, Cell 13). These levels are substantially below TTS-onset values. They are also probably quite precautionary as behavioral response criteria for the group as a whole, especially for species other than harbor seals where higher exposures were not observed to induce strong (or in some cases any) responses.

**Table 5.** Proposed behavioral response criteria for individual marine mammals exposed to various sound types; specific threshold levels are proposed for single pulses. See the referenced text sections and tables for severity scale analyses of behavioral responses to multiple pulses and nonpulses.

Marine mammal group	Sound type		
	Single pulses	Multiple pulses	Nonpulses
Low-frequency cetaceans	Cell 1	Cell 2 <sup>1</sup>	Cell 3 <sup>6</sup>
Sound pressure level	224 dB re: 1 $\mu$ Pa (peak) (flat)	Tables 6 & 7	Tables 14 & 15
Sound exposure level	183 dB re: 1 $\mu$ Pa <sup>2</sup> -s ( $M_{lr}$ )	Not applicable	Not applicable
Mid-frequency cetaceans	Cell 4	Cell 5 <sup>2</sup>	Cell 6 <sup>7</sup>
Sound pressure level	224 dB re: 1 $\mu$ Pa (peak) (flat)	Tables 8 & 9	Tables 16 & 17
Sound exposure level	183 dB re: 1 $\mu$ Pa <sup>2</sup> -s ( $M_{mr}$ )	Not applicable	Not applicable
High-frequency cetaceans	Cell 7	Cell 8 <sup>3</sup>	Cell 9 <sup>8</sup>
Sound pressure level	224 dB re: 1 $\mu$ Pa (peak) (flat)	[Tables 18 & 19]	Tables 18 & 19
Sound exposure level	183 dB re: 1 $\mu$ Pa <sup>2</sup> -s ( $M_{hr}$ )	Not applicable	Not applicable
Pinnipeds (in water)	Cell 10	Cell 11 <sup>4</sup>	Cell 12 <sup>9</sup>
Sound pressure level	212 dB re: 1 $\mu$ Pa (peak) (flat)	Tables 10 & 11	Tables 20 & 21
Sound exposure level	171 dB re: 1 $\mu$ Pa <sup>2</sup> -s ( $M_{pw}$ )	Not applicable	Not applicable
Pinnipeds (in air)	Cell 13	Cell 14 <sup>5</sup>	Cell 15 <sup>10</sup>
Sound pressure level	109 dB re: 20 $\mu$ Pa (peak) (flat)	Tables 12 & 13	Tables 22 & 23
Sound exposure level	100 dB re: (20 $\mu$ Pa) <sup>2</sup> -s ( $M_{pa}$ )	Not applicable	Not applicable

<sup>1</sup> “Low-Frequency Cetaceans/Multiple Pulses (Cell 2)” section

<sup>2</sup> “Mid-Frequency Cetaceans/Multiple Pulses (Cell 5)” section

<sup>3</sup> “High-Frequency Cetaceans/Multiple Pulses (Cell 8)” section

<sup>4</sup> “Pinnipeds in Water/Multiple Pulses (Cell 11)” section

<sup>5</sup> “Pinnipeds in Air/Multiple Pulses (Cell 14)” section

<sup>6</sup> “Low-Frequency Cetaceans/Nonpulses (Cell 3)” section

<sup>7</sup> “Mid-Frequency Cetaceans/Nonpulses (Cell 6)” section

<sup>8</sup> “High-Frequency Cetaceans/Nonpulses (Cell 9)” section

<sup>9</sup> “Pinnipeds in Water/Nonpulses (Cell 12)” section

<sup>10</sup> “Pinnipeds in Air/Nonpulses (Cell 15)” section

### Behavioral Response Severity Scaling: Multiple Pulses

*Low-Frequency Cetaceans/Multiple Pulses (Cell 2)*  
Numerous field observations have been made of low-frequency cetaceans reacting to multiple pulses either incidentally during ongoing operations or intentionally during experiments. A moderate number of species and experimental conditions have been considered, but the sources have usually been seismic airgun arrays. Some of the studies focused on migrating whales seen from fixed observation platforms or in/near migratory corridors. This approach minimizes pseudoreplication without the need for identifying individuals because individuals are unlikely to pass observers more than once.

Table 6 summarizes the methods used to obtain acoustic measurements and observations of behavioral or distributional responses (see Appendix B for more details). As in most cells, a number of reported observations were not scored or reported here due to lack of some key information and, in some cases, difficulties in accounting for various

contextual variables. A few of these “excluded” studies are listed at the bottom of Table 6. Table 7 shows the results of the severity scaling analyses of individual and/or group responses, considering the studies deemed to contain sufficient data on exposure conditions and behavioral responses. For migrating bowhead whales, the onset of significant behavioral disturbance from multiple pulses occurred at RLs (RMS over pulse duration) around 120 dB re: 1  $\mu$ Pa (Richardson et al., 1999). For all other low-frequency cetaceans (including bowhead whales not engaged in migration), this onset was at RLs around 140 to 160 dB re: 1  $\mu$ Pa (Malme et al., 1983, 1984; Richardson et al., 1986; Ljungblad et al., 1988; Todd et al., 1996; McCauley et al., 1998, 2000) or perhaps higher (Miller et al., 2005). There is essentially no overlap in the RLs associated with onset of behavioral responses by members of these two groups based on the information currently available.

*Mid-Frequency Cetaceans/Multiple Pulses (Cell 5)*  
A limited number of behavioral observations have been made of mid-frequency cetaceans exposed to

**Table 6.** Summary of behavioral responses by different species of low-frequency cetaceans exposed to multiple pulses (Cell 2) by type of sound source, available acoustic metrics, description of behavioral response (by individual and/or group), and a summary of corresponding severity score(s); where airgun array volume is stated, this is the total volume for all operating airguns in the array; 1 L = 61 in<sup>3</sup>. Specific severity scores for each study are given in Table 7, and more details are given in Appendix B. Exposure RLs are given in dB SPL, which are RMS sound pressure levels (dB re: 1 µPa) over the duration of a pulse.

Study	Reference number (for Table 7)	Subject species	Sound source	Type of acoustic measurements	Type of individual and/or group behavioral responses	Study included in severity scale	Summary of severity scale analysis (see Table 7)
Malmé et al. (1983)	1	Gray whales	Single airgun (1.64 L) & 20-gun 65.5-L airgun array	Calibrated RL measurements made <i>in situ</i> near areas of exposure	Land-based observations of individuals/groups; movement and respiration patterns during and without airguns	Yes	Exposure RLs 140-180 dB SPL; response severity scores: 0, 1, 3, 5 & 6
Malmé et al. (1984)	2	Gray whales	Single airgun and airgun array	Calibrated RL measurements made <i>in situ</i> near areas of exposure	Land-based observations of individuals/groups; movement and respiration patterns during and without airguns	Yes	Exposure RLs 140-180 dB SPL; response severity scores: 0, 1, 3, 5 & 6
Richardson et al. (1986)	3	Bowhead whales (feeding)	Single airgun (0.66 L) and 30-gun 47-L array	Calibrated RL measurements made <i>in situ</i> near areas of exposure	Aerial observations of individuals/groups; movement and respiration patterns during and without airguns	Yes	Exposure RLs 140-180 dB SPL; response severity scores: 0, 1, 3, 5 & 6
Ljungblad et al. (1988)	4	Bowhead whales (feeding)	Single airgun (1.3 L) or 18- to 20-gun array	Calibrated RL measurements made <i>in situ</i> near areas of exposure	Aerial observations of individuals/groups; movement patterns and behavioral responses during and without airguns	Yes	Exposure RLs 140-180 dB SPL; response severity score: 6
Todd et al. (1996)	5	Humpback whales	Explosions	Limited to nominal measurements of explosives used (not measured on or near subjects)	Visual observations of whale behavior before and during use of explosives	Yes	Exposure RLs 150-160 dB SPL; response severity score: 3
McCauley et al. (1998)	6	Humpback whales (migrating)	Single airgun (0.33 L) and several arrays	Calibrated RL measurements made <i>in situ</i> near areas of exposure	Aerial and boat-based observations of individuals/groups; movement and behavioral patterns during and without airguns	Yes	Exposure RLs 150-170 dB SPL; response severity scores: 6 & 7
Richardson et al. (1999)	7	Bowhead whales (migrating)	Airgun array (6 to 16 guns; 9 to 25 L)	Calibrated RL measurements made <i>in situ</i> near areas of exposure	Aerial surveys of distribution of individuals/groups	Yes	Exposure RLs 110-140 dB SPL; response severity scores: 0, 1, 5 & 6
McCauley et al. (2000)	8	Humpback whales (socializing)	Single airgun (0.33 L)	Calibrated RL measurements made <i>in situ</i> near areas of exposure	Boat-based observations of individuals/groups; movement and behavioral patterns during and without airguns	Yes	Exposure RLs 140-180 dB SPL; response severity score: 6
Miller et al. (2005)	9	Bowhead whales (feeding)	Airgun array (24 guns; 36.9 L)	Calibrated RL measurements made <i>in situ</i> near areas of exposure	Vessel-based observations of individuals; aerial surveys of distribution; movement/diving patterns and behavioral responses during and without airguns	Yes	Exposure RLs 140-180 dB SPL; response severity scores: 0 & 6
Reeves et al. (1984)	Not included	Bowhead whales (migrating)	Seismic airgun array	Insufficient data for this analysis	Insufficient data for this analysis	No	N/A
Malmé et al. (1985)	Not included	Humpback whales	Seismic airgun array	Insufficient data for this analysis	Insufficient data for this analysis	No	N/A
Malmé et al. (1986, 1988)	Not included	Gray whales	Single airgun and airgun array	Insufficient data for this analysis	Insufficient data for this analysis	No	N/A
Koski & Johnson (1987)	Not included	Bowhead whales (migrating)	Seismic airgun array	Insufficient data for this analysis	Insufficient data for this analysis	No	N/A

**Table 7.** Number (in **bold**) of low-frequency cetaceans (individuals and/or groups) reported as having behavioral responses to multiple pulse noise; responses were categorized into 10-dB RL bins, ranked by severity of the behavioral response (see Table 4 for severity scaling), and combined with other observations having the same RL/severity score. A summary of the individual studies included in this table is given in the “Low-Frequency Cetaceans/Multiple Pulses (Cell 2)” section of this chapter. Parenthetical subscripts indicate the reference reporting the observations as listed in Table 6.

Response score	Received RMS sound pressure level (dB re: 1 $\mu$ Pa)											
	80 to < 90	90 to < 100	100 to < 110	110 to < 120	120 to < 130	130 to < 140	140 to < 150	150 to < 160	160 to < 170	170 to < 180	180 to < 190	190 to < 200
9												
8												
7								<b>1.0</b> (6)				
6				<b>9.5</b> (7)	<b>47.4</b> (7)	<b>2.2</b> (7)	<b>3.4</b> (4, 6, 8)	<b>5.8</b> (1, 2, 3, 6)	<b>4.5</b> (1, 2, 3, 4, 6)	<b>8.3</b> (1, 2, 4, 8, 9)		
5					<b>1.0</b> (7)		<b>1.0</b> (4)	<b>1.0</b> (1, 2)				
4												
3									<b>1.0</b> (1, 2)	<b>1.0</b> (1, 2)		
2												
1				<b>5.0</b> (7)	<b>6.0</b> (7)	<b>1.0</b> (7)	<b>2.5</b> (1, 2, 3)	<b>3.0</b> (5)				
0				<b>59.8</b> (7)	<b>17.7</b> (7)	<b>1.1</b> (7, 9)	<b>0.1</b> (9)	<b>0.6</b> (3, 9)	<b>6.8</b> (1, 2, 3, 9)	<b>6.3</b> (1, 2, 9)		

multiple pulses. Field observations have involved sperm whales and a few other odontocete species exposed to seismic airguns and explosives. Laboratory investigations have considered behavioral responses to various kinds of multiple pulse sources. Again, some observations were excluded due to lack of relevant information. Four studies of individual mid-frequency cetacean responses to multiple pulse exposures contained sufficient acoustic and behavioral information for inclusion in this analysis. These include field observations of free-ranging sperm whales and belugas studied by Madsen & Møhl (2000), Madsen et al. (2002), and Miller et al. (2005), as well as laboratory observations of captive false killer whales by Akamatsu et al. (1993). The information from these studies is summarized in Table 8 and discussed in detail in Appendix B; the companion severity scaling analysis is shown in Table 9.

The combined data for mid-frequency cetaceans exposed to multiple pulses do not indicate a clear tendency for increasing probability and severity of response with increasing RL. In certain conditions, multiple pulses at relatively low RLs (~80 to 90 dB re: 1  $\mu$ Pa) temporarily silence individual vocal behavior for one species (sperm whales). In other cases with slightly different stimuli, RLs in the 120 to 180 dB re: 1  $\mu$ Pa range failed to elicit observable reaction from a significant percentage of individuals either in the field or in the laboratory.

#### *High-Frequency Cetaceans/Multiple Pulses (Cell 8)*

Based on our source type distinction (see Chapter 2), virtually all sources of transient sound used in quantitative behavioral studies of high-frequency cetaceans—for example, acoustic harassment devices (AHDs) and acoustic deterrent devices (ADDs)—would be characterized as nonpulse sounds. While individual elements produced by some of these sources could be characterized as pulses, and sequences of them as multiple pulses, they are generally emitted in such rapid fashion that some mammalian auditory systems likely perceive them as nonpulses. Further, some AHDs and ADDs, and most other sources used in behavioral studies with high-frequency cetaceans, lack the characteristics of pulses such as extremely fast rise-time, correspondingly broad frequency bandwidth, and high kurtosis. Due to uncertainty over the extent to which some of these signals may be perceived and the overarching paucity of data, it is not possible to present any data on behavioral responses of high-frequency cetaceans as a function of received levels of multiple pulses. Available data for nonpulse sounds are considered below (see the “High-Frequency Cetaceans/Nonpulses [Cell 9]” section). We note the need for empirical behavioral research in these animals using sound sources (such as airgun or pile-driving stimuli) unequivocally classified as multiple pulses (see Chapter 5).

**Table 8.** Summary of behavioral responses by different species of mid-frequency cetaceans exposed to multiple pulses (Cell 5) by type of sound source, available acoustic metrics, description of behavioral response (by individual and/or group), and a summary of corresponding severity score(s); specific severity scores for each study are given in Table 9 and more details are given in Appendix B. Exposure RLs are given in dB SPL, which are RMS sound pressure levels (dB re: 1 µPa) over the duration of a pulse.

Study	Reference number (for Table 9)	Subject species	Sound source	Type of acoustic measurements	Type of individual and/or group behavioral responses	Study included in severity scale	Summary of severity scale analysis (see Table 9)
Madsen & Møhl (2000)	1	Sperm whales	Small explosives (several per day)	Calibrated RL measurements made <i>in situ</i> near areas of exposure	Vessel-based observations of individuals, including visual detection and passive acoustic monitoring of vocalizations	Yes	Exposure RLs 170-180 dB SPL; response severity score: 0
Madsen et al. (2002)	2	Sperm whales	Airgun array (distant)	Calibrated RL measurements made <i>in situ</i> near areas of exposure	Vessel-based observations of individuals, including visual detection and passive acoustic monitoring of vocalizations	Yes	Exposure RLs 120-140 dB SPL; response severity score: 0
Miller et al. (2005)	3	Beluga	Airgun array (24 guns; 36.9 L)	Calibrated RL measurements made <i>in situ</i> near areas of exposure	Vessel-based observations of individuals; aerial surveys of distribution; movement/diving patterns and behavioral responses during and without airguns	Yes	Exposure RLs 100-150 dB SPL; response severity scores: 0 & 6
Akamatsu et al. (1993)	4	False killer whales (captive)	Numerous sounds, including pulse sequences	Calibrated RL measurements made <i>in situ</i> within experimental enclosure	Visual observations of behavioral responses within experimental context in laboratory conditions	Yes	Exposure RLs 170-180 dB SPL; response severity scores: 0 & 6
André et al. (1997)	Not included	Sperm whales	Natural and artificial pulses (repeated)	Insufficient data for this analysis	Vessel-based observations of individuals; diving patterns and vocal behavior	No	N/A
Stone (2003)	Not included	Several mid-freq. cetacean species	Seismic airgun arrays (various)	Insufficient data for this analysis	Vessel-based observations of individuals; sightings data and avoidance behavior	No	N/A

**Table 9.** Number (in **bold**) of mid-frequency cetaceans (individuals and/or groups) reported as having behavioral responses to multiple pulse noise; responses were categorized into 10-dB RL bins, ranked by severity of the behavioral response (see Table 4 for severity scaling), and combined with other observations having the same RL/severity score. A summary of the individual studies included in this table is given in the “Mid-Frequency Cetaceans/Multiple Pulses (Cell 5)” section of this chapter. Parenthetical subscripts indicate the reference reporting the observations as listed in Table 8.

Response score	Received RMS sound pressure level (dB re: 1 $\mu$ Pa)											
	80 to < 90	90 to < 100	100 to < 110	110 to < 120	120 to < 130	130 to < 140	140 to < 150	150 to < 160	160 to < 170	170 to < 180	180 to < 190	190 to < 200+
9												
8												
7												
6					<b>0.17</b> (3)	<b>0.17</b> (3)	<b>0.17</b> (3)				<b>1.3</b> (4)	
5												
4												
3												
2												
1												
0			<b>0.25</b> (3)	<b>0.25</b> (3)	<b>3.0</b> (2)	<b>4.0</b> (2)					<b>6.7</b> (1,4)	

#### *Pinnipeds in Water/Multiple Pulses (Cell 11)*

Information on behavioral reactions of pinnipeds in water to multiple pulses involves exposures to small explosives used in fisheries interactions, impact pile driving, and seismic surveys. Several studies lacked matched data on acoustic exposures and behavioral responses by individuals. As a result, the quantitative information on reactions of pinnipeds in water to multiple pulses is very limited (see Table 10). The severity scaling analysis for individual behavioral responses for Cell 11 is given in Table 11.

Our general finding is that, based on the limited data on pinnipeds in water exposed to multiple pulses, exposures in the ~150 to 180 dB re: 1  $\mu$ Pa range (RMS values over the pulse duration) generally have limited potential to induce avoidance behavior in pinnipeds. RLs exceeding 190 dB re: 1  $\mu$ Pa are likely to elicit responses, at least in some ringed seals (Harris et al., 2001; Blackwell et al., 2004b; Miller et al., 2005). Note that the SEL associated with a single 190 dB re: 1  $\mu$ Pa (RMS) pulse from an airgun is typically *ca.* 175 dB re: 1  $\mu$ Pa<sup>2</sup>-s. That exceeds the estimated TTS threshold for the closely related harbor seal (171 dB re: 1  $\mu$ Pa<sup>2</sup>-s; see Chapter 3). Thus, in the case of ringed seals exposed to sequences of airgun pulses from an approaching seismic vessel, most animals may show little avoidance unless the RL is high enough for mild TTS to be likely.

#### *Pinnipeds in Air/Multiple Pulses (Cell 14)*

How multiple pulses produced in air affect pinnipeds was among the least well-documented of the conditions we considered. Most of the available

data on responses to pulses were from single pulse events (e.g., rocket launches) over populations of pinnipeds exposed to such signals repeatedly (e.g., Thorson et al., 1998, 1999, 2000a, 2000b; Berg et al., 2001, 2002, 2004). These events do not occur frequently enough for the exposures to be considered multiple pulses, and many of them contained nonpulse as well as pulse exposures. They are discussed in some detail in Appendix B (as well as in Appendix C when nonpulses are involved). Appendix B also discusses several other studies potentially relevant to Cell 14 but ultimately not used in this analysis. Consequently, the quantitative information analyzed for reactions of pinnipeds in air exposed to multiple pulses (see Tables 12 & 13) focused on the aerial data by Blackwell et al. (2004b). These extremely limited data suggest very minor, if any, observable behavioral responses by pinnipeds exposed to airborne pulses with RLs 60 to 80 dB re: 20  $\mu$ Pa.

#### **Behavioral Response Severity Scaling: Nonpulses**

##### *Low-Frequency Cetaceans/Nonpulses (Cell 3)*

While there are clearly major areas of uncertainty remaining, there has been relatively extensive behavioral observation of low-frequency cetaceans exposed to nonpulse sources. As summarized in Table 14 (and discussed in greater detail in Appendix C), these field observations involve the majority of low-frequency cetacean species exposed to a wide range of industrial, active sonar, and tomographic research active sources (Baker et al., 1982; Malme et al., 1983, 1984, 1986;

**Table 10.** Summary of behavioral responses by different species of pinnipeds in water exposed to multiple pulses (Cell 11) by type of sound source, available acoustic metrics, description of behavioral response (by individual and/or group), and a summary of corresponding severity score(s); specific severity scores for each study are given in Table 11 and more details are given in Appendix B. Exposure RLs are given in dB SPL, which are RMS sound pressure levels (dB re: 1  $\mu$ Pa) over the duration of a pulse.

Study	Reference number (for Table 11)	Subject species	Sound source	Type of acoustic measurements	Type of individual and/or group behavioral responses	Study included in severity scale	Summary of severity scale analysis (see Table 11)
Harris et al. (2001)	1	Ringed (mainly), bearded, and spotted seals	Single airgun and 11-gun, 21.6-L array	RLs measured <i>in situ</i> near individuals observed in defined spatial zones	Vessel-based observations of individuals within specified zones over a limited range	Yes	Exposure RLs 160-200 dB SPL; response severity scores: 0 & 6
Blackwell et al. (2004b)	2	Ringed seals	Pipe-driving sounds (construction)	RLs measured <i>in situ</i> near individuals observed (detailed measurements, including peak pressure, RMS, SEL, and duration)	Land-based observations of individuals; movement and response patterns during pipe-driving (note that construction activities had been underway for a considerable period before observations)	Yes	Exposure RLs 150-160 dB SPL; response severity scores: 0 & 1
Miller et al. (2005)	3	Ringed and bearded seals	Airgun array (24 airguns; 36.9 L)	Calibrated RL measurements made <i>in situ</i> near areas of exposure	Vessel-based observations of individuals; movement patterns and behavioral responses during and without airguns	Yes	Exposure RLs 170-200 dB SPL; response severity score: 0
Shaughnessy et al. (1981)	Not included	California sea lions	Seal bombs (small explosives)	Insufficient data for this analysis	Visual observations of individual responses over multiple exposures	No	N/A
Mate & Harvey (1987)	Not included	California sea lions	Seal bombs (small explosives)	Insufficient data for this analysis	Visual observations of individual responses over multiple exposures	No	N/A
Moulton et al. (2003, 2005)	Not included	Ringed seals		Calibrated measurements made in the area of exposure	Complicated by simultaneous exposure to pulse and nonpulse sources	No	N/A

**Table 11.** Number (in **bold**) of pinnipeds in water (individuals and/or groups) reported as having behavioral responses to multiple pulse noise. Responses were categorized into 10-dB RL bins, ranked by severity of the behavioral response (see Table 4 for severity scaling), and combined with other observations having the same RL/severity score; a summary of the individual studies included in this table is given in the “Pinnipeds in Water/Multiple Pulses (Cell 11)” section of this chapter. Parenthetical subscripts indicate the reference reporting the observations as listed in Table 10.

Response score	Received RMS sound pressure level (dB re: 1 $\mu$ Pa)											
	80 to < 90	90 to < 100	100 to < 110	110 to < 120	120 to < 130	130 to < 140	140 to < 150	150 to < 160	160 to < 170	170 to < 180	180 to < 190	190 to < 200
9												
8												
7												
6									<b>1.7</b> (1)	<b>2.1</b> (1)		<b>45.4</b> (1)
5												
4												
3												
2												
1						<b>0.3</b> (2)						
0								<b>0.7</b> (2)	<b>5.3</b> (1)	<b>30.3</b> (1, 3)	<b>0.3</b> (3)	<b>9.9</b> (1, 3)

Richardson et al., 1990b; McCauley et al., 1996; Biassoni et al., 2000; Croll et al., 2001; Palka & Hammond, 2001; Nowacek et al., 2004).

The combined information generally indicates no (or very limited) responses at RLs 90 to 120 dB re: 1  $\mu$ Pa and an increasing probability of avoidance and other behavioral effects in the 120 to 160 dB re: 1  $\mu$ Pa range (severity scaling: Table 15). However, these data also indicated considerable variability in RLs associated with behavioral responses. Contextual variables (e.g., source proximity, novelty, operational features) appear to have been at least as important as exposure level in predicting response type and magnitude.

#### *Mid-Frequency Cetaceans/Nonpulses (Cell 6)*

A relatively large number of mid-frequency cetaceans have been observed in the field and in the laboratory responding to nonpulse sounds, including vessels and watercraft (LGL & Greeneridge, 1986; Gordon et al., 1992; Palka & Hammond, 2001; Buckstaff, 2004; Morisaka et al., 2005), pulsed pingers and AHDs/ADDs (Watkins & Schevill, 1975; Morton & Symonds, 2002; Monteiro-Neto et al., 2004), industrial activities (Awbrey & Stewart, 1983; Richardson et al., 1990b), mid-frequency active sonar (NRL, 2004a, 2004b; NMFS, 2005), and tones or bands of noise in laboratory conditions (Nachtigall et al., 2003; Finneran & Schlundt, 2004). Summary information on these studies is given in Table 16; detailed descriptions are given in Appendix C. As in other conditions, a number of potentially relevant field studies are not included in the severity scaling analysis due to lack of sufficiently detailed information.

An additional challenge in interpreting many of the field data for this condition is isolating the effect of RL from the effects of mere source presence (as possibly indicated by visual stimuli or other aspects of acoustic exposure such as the presence of high-frequency components) and other contextual variables. For this reason, several studies were considered but not integrated into the analysis. The laboratory observations are of captive cetaceans exposed to precisely controlled and known noise exposures in the context of hearing and TTS experiments. However, the relevance of behavioral reactions of trained, food-reinforced captive animals exposed to noise to the reactions of free-ranging marine mammals is debatable. This is discussed in greater detail in Appendix C.

The combined field and laboratory data for mid-frequency cetaceans exposed to nonpulse sounds do not lead to a clear conclusion about RLs coincident with various behavioral responses (see severity scaling, Table 17). In some settings, individuals in the field showed behavioral responses with high severity scores to exposures from 90 to 120 dB re: 1  $\mu$ Pa, while others failed to exhibit such responses for exposure RLs from 120 to 150 dB re: 1  $\mu$ Pa. Contextual variables other than exposure RL, and probable species differences, are the likely reasons for this variability in response. Context may also explain why there is great disparity in results from field and laboratory conditions—exposures in captive settings generally exceeded 170 dB re: 1  $\mu$ Pa before inducing behavioral responses.

**Table 12.** Summary of behavioral responses by different species of pinnipeds in air exposed to multiple pulses (Cell 14) by type of sound source, available acoustic metrics, description of behavioral response (by individual and/or group), and a summary of corresponding severity score(s); specific severity scores for each study are given in Table 13 and more details are given in Appendix B. Exposure RLs are given in dB SPL, which are RMS sound pressure levels (dB re: 20 µPa) over the duration of a pulse.

Study	Reference number (for Table 13)	Subject species	Sound source	Type of acoustic measurements	Type of individual and/or group behavioral responses	Study included in severity scale	Summary of severity scale analysis (see Table 13)
Blackwell et al. (2004b)	1	Ringed seals	Pipe-driving sounds (construction)	RLs measured <i>in situ</i> near individuals observed (detailed measurements, including peak pressure, RMS, SEL, and duration)	Land-based observations of individuals; movement and response patterns during pipe-driving (note that construction activities had been underway for a considerable period before observations)	Yes	Exposure RLs 60-80 dB SPL re: 20 µPa; response severity scores: 0 & 1
Perry et al. (2002)	Not included	Harbor and gray seals	Repeated sonic booms	Measured sound overpressure levels on breeding beaches, but not RLs at positions of exposed animals	Land-based observations of animal presence, behavior, and heart rate (note long history of sonic booms in the area)	No	N/A

**Table 13.** Number (in **bold**) of pinnipeds in air (individuals and/or groups) reported as having behavioral responses to multiple pulse noise; responses were categorized into 10-dB RL bins, ranked by severity of the behavioral response (see Table 4 for severity scaling), and combined with other observations having the same RL/severity score. A summary of the individual studies included in this table is given in the “Pinnipeds in Air/Multiple Pulses (Cell 14)” section of this chapter. Parenthetical subscripts indicate the reference reporting the observations as listed in Table 12.

Response score	Received RMS sound pressure level (dB re: 20 $\mu$ Pa)						
	50 to < 60	60 to < 70	70 to < 80	80 to < 90	90 to < 100	100 to < 110	110 to < 120
9							
8							
7							
6							
5							
4							
3							
2							
1		<b>0.125</b> (1)					
0		<b>0.625</b> (1)	<b>0.25</b> (1)				

#### *High-Frequency Cetaceans/Nonpulses (Cell 9)*

Numerous controlled studies have been conducted on the behavioral reactions of high-frequency cetaceans to various nonpulse sound sources both in the field (Culik et al., 2001; Olesiuk et al., 2002; Johnston, 2002) and in laboratory settings (Kastelein et al., 1997, 2000, 2005, 2006a). However, only one high-frequency species (harbor porpoise) has been extensively studied and that species provided all the available data on behavioral response magnitude vs received exposure conditions. The original studies were attempts to reduce harbor porpoise bycatch by attaching warning pingers to fishing gear. More recent studies consider whether AHDs and ADDs also exclude harbor porpoises from critical habitat areas, and whether these devices affect harbor porpoise behavior in controlled laboratory conditions.

The combined wild and captive animal data (summarized in Table 18 and discussed in detail in Appendix C) clearly support the observation that harbor porpoises are quite sensitive to a wide range of human sounds at very low exposure RLs (~90 to 120 dB re: 1  $\mu$ Pa), at least for initial exposures. This observation is also evident in the severity scaling analysis for Cell 9 (Table 19). All recorded exposures exceeding 140 dB re: 1  $\mu$ Pa induced profound and sustained avoidance behavior in wild harbor porpoises. Whether this apparently high degree of behavioral sensitivity to anthropogenic acoustic sources extends to other high-frequency cetacean species (or nonpulse sources other than AHDs and ADDs) is unknown. Given the lack of information to the contrary, however, such a relationship should be assumed as a precautionary measure.

Habituation to sound exposure was noted in some but not all studies. Strong initial reactions of high-frequency cetaceans at relatively low levels may in some conditions wane with repeated exposure and subject experience.

#### *Pinnipeds in Water/Nonpulses (Cell 12)*

The effects of nonpulse exposures on pinnipeds in water are poorly understood. Studies for which enough information was available for analysis include field exposures of harbor seals to AHDs (Jacobs & Terhune, 2002) and exposure of translocated freely diving northern elephant seals to a research tomography source (Costa et al., 2003), as well as responses of captive harbor seals to underwater data communication sources (Kastelein et al., 2006b). These limited available data (see Table 20 & Appendix C) suggested that exposures between ~90 and 140 dB re: 1  $\mu$ Pa generally do not appear to induce strong behavioral responses in pinnipeds exposed to nonpulse sounds in water; no data exist regarding exposures at higher levels. The severity scale results for Cell 12 are given in Table 21.

It is important to note that among these studies of pinnipeds responding to nonpulse exposures in water, there are some apparent differences in responses between field and laboratory conditions. Specifically, in this case, captive subjects responded more strongly at lower levels than did animals in the field. Again, contextual issues are the likely cause of this difference. Captive subjects in the Kastelein et al. (2006b) study were not reinforced with food for remaining in noise fields, in contrast to the laboratory studies for mid-frequency cetaceans described above. Subjects in the

**Table 14.** Summary of behavioral responses by different species of low-frequency cetaceans exposed to nonpulses (Cell 3) by type of sound source, available acoustic metrics, description of behavioral response (by individual and/or group), and a summary of corresponding severity score(s); specific severity scores for each study are given in Table 15 and more details are given in Appendix C. Exposure RLs are given in dB SPL, which are RMS sound pressure levels (dB re: 1 µPa).

Study	Reference number (for Table 15)	Subject species	Sound source	Type of acoustic measurements	Type of individual and/or group behavioral responses	Study included in severity scale	Summary of severity scale analysis (see Table 15)
Baker et al. (1982)	1	Humpback whales	Vessel noise and presence	Individual RLs not reported but vessels identical to previous measurements	Vessel-based observations of individual movement and behavioral patterns around vessels	Yes	Exposure RLs 100-140 dB SPL; severity scores: 0 & 6
Malmé et al. (1983, 1984)	2	Gray whales (migrating)	Playbacks of drilling and machinery noise	RLs measured <i>in situ</i> near individuals observed	Shore-based observations of individual movement and behavioral patterns around simulated drilling operations/platforms	Yes	Exposure RLs 90-150 dB SPL; severity scores: 0 & 6
Malmé et al. (1986)	3	Gray whales (feeding)	Playbacks of drilling noise	RLs measured <i>in situ</i> near individuals observed	Vessel-based observations of individual movement and behavioral patterns before and during playbacks	Yes	Exposure RLs 100-120 dB SPL; severity scores: 0 & 6
Richardson et al. (1990b)	4	Bowhead whales (migrating)	Drilling noise playbacks	Detailed and calibrated source and transmission loss measurements allowed good RL estimates	Visual observations of individual movement and behavioral patterns before, during, and after exposure to drilling sounds	Yes	Exposure RLs 100-140 dB SPL; severity scores: 0 & 6
McCauley et al. (1996)	5	Humpback whales	Vessel noise and presence	RLs measured <i>in situ</i> near individuals observed	Visual observations of individual movement and behavioral patterns during vessel approaches	Yes	Exposure RLs 110-130 dB SPL; severity score: 6
Frankel & Clark (1998)	6	Humpback whales	Low-frequency M-sequence playback	Calibrated RL measurements made <i>in situ</i> near areas of exposure	Visual observations of individual movement and behavioral patterns before, during, and after playbacks	Yes	Exposure RLs 120-130 dB SPL; severity score: 6
Biassoni et al. (2000); Miller et al. (2000)	7	Humpback whales	Low-frequency sonar playback	Calibrated RL measurements made <i>in situ</i> near areas of exposure	Visual observations of individual movement and behavioral patterns before, during, and after playbacks	Yes	Exposure RLs 110-160 dB SPL; severity scores: 2 & 4
Croll et al. (2001)	8	Blue and fin whales (feeding)	Low-frequency sonar playback	Calibrated RL measurements and modeling for area of exposure	Individual responses not reported but a general observation of feeding behavior with/without sonar	Yes	Exposure RLs 140-150 dB SPL; severity score: 0

Table 14 (continued)

Study	Reference number (for Table 15)	Subject species	Sound source	Type of acoustic measurements	Type of individual and/or group behavioral responses	Study included in severity scale	Summary of severity scale analysis (see Table 15)
Palka & Hammond (2001)	9	Minke whales	Vessel noise and presence	RL estimates based on source and environmental characteristics	Visual observations of individual and group movements and behavioral patterns during vessel approaches	Yes	Exposure RLS 110-120 dB SPL; severity score: 3
Nowacek et al. (2004)	10	Right whales	Playbacks of several nonpulses	Subjects were calibrated tags that measured RL and behavior/movement	Detailed measurements of vocal and physical reactions of animals before, during, and after playbacks	Yes	Exposure RLS 120-150 dB SPL; severity scores: 0 & 7
Dahlheim (1987)	Not included	Gray whales	Playbacks of nonpulses	Insufficient data for this analysis	Visual and acoustic observations of individuals	No	N/A
Borggaard et al. (1999)	Not included	Various cetaceans	Industrial noise	Some RL measurements and modeling in area	Insufficient data on individual responses for this analysis	No	N/A
Frankel & Clark (2000)	Not included	Humpback whales	ATOC source	Some RL measurements and modeling in area	Insufficient data on individual responses for this analysis	No	N/A
Schick & Urban (2000)	Not included	Bowhead whales	Drillships	Insufficient data for this analysis	Visual observations of individuals around rigs	No	N/A
Frankel & Clark (2002)	Not included	Humpback whales	ATOC source	Some RL measurements and modeling in area	Insufficient data on individual responses for this analysis	No	N/A
Jahoda et al. (2003)	Not included	Fin whales	Vessel noise and presence	Insufficient data for this analysis	Visual observations of individuals during approaches	No	N/A
Mobley (2005)	Not included	Humpback whales	ATOC source	Some RL measurements and modeling in area	Insufficient data on individual responses for this analysis	No	N/A

**Table 15.** Number (in **bold**) of low-frequency cetaceans (individuals and/or groups) reported as having behavioral responses to nonpulses; responses were categorized into 10-dB RL bins, ranked by severity of the behavioral response (see Table 4 for severity scaling), and combined with other observations having the same RL/severity score. A summary of the individual studies included in this table is given in the “Low-Frequency Cetaceans/Nonpulses (Cell 3)” section of this chapter. Parenthetical subscripts indicate the reference reporting the observations as listed in Table 14.

Response score	Received RMS sound pressure level (dB re: 1 $\mu$ Pa)											
	80 to < 90	90 to < 100	100 to < 110	110 to < 120	120 to < 130	130 to < 140	140 to < 150	150 to < 160	160 to < 170	170 to < 180	180 to < 190	190 to < 200
9												
8												
7							<b>2.5</b> (10)	<b>1.5</b> (10)				
6		<b>4.9</b> (2)	<b>7.4</b> (1, 2, 4)	<b>16.2</b> (1, 2, 3, 5)	<b>13.6</b> (2, 5)	<b>4.2</b> (1, 2)	<b>0.8</b> (2)					
5												
4				<b>3.0</b> (5, 7)	<b>1.0</b> (7)		<b>1.0</b> (7)					
3			<b>1,117</b> (9)	<b>0.27</b> (6)								
2			<b>0.5</b> (7)	<b>4.0</b> (7)	<b>5.0</b> (7)	<b>2.0</b> (7)	<b>1.0</b> (7)					
1												
0	<b>1.1</b> (2)	<b>82.6</b> (2, 3, 4)	<b>33.9</b> (1, 2, 3, 4)	<b>7.08</b> (2, 4, 6, 10)	<b>7.2</b> (4, 10)	<b>1.45</b> (2, 8, 10)						

field may have been more tolerant of exposures because of motivation to return to a safe location (Costa et al., 2003) or motivation to approach enclosures holding prey items (Jacobs & Terhune, 2002).

#### *Pinnipeds in Air/Nonpulses (Cell 15)*

There has been considerable effort to study the effects of aerial nonpulse sounds on pinniped behavior, primarily involving rocket launches, aircraft overflights, powerboat approaches, and construction noise. Unfortunately, as discussed in Appendix C, many of the studies are difficult to interpret in terms of exposure RL and individual or group behavioral responses. In many cases, it was difficult or impossible to discern whether the reported behavioral response was induced by the noise from a specific operation or some correlated variable such as its visual presence. For these reasons, most of the observational studies of behavioral disturbance were not appropriate for scoring behavioral responses relative to exposure RL. However, a number of the technical reports and analyses of rocket launches are relevant for this cell and contain sufficiently detailed information regarding estimated RLs. These observations are, however, complicated by the fact that all studies were conducted in the same general area with subjects likely habituated to the presence of launch noise. Further, in many cases, exposures contained both a nonpulse component and a pulse component (described below). Only

those observations (Thorson et al., 1999, 2000b; Berg et al., 2002) for which there was clearly just nonpulse exposure were considered in the severity scaling analyses for this condition.

The limitations of these and other potentially applicable studies resulted in a very limited data set for use in this analysis (see summary in Table 22 and severity scaling analysis in Table 23). As a general statement from the available information, pinnipeds exposed to intense (~110 to 120 dB re: 20  $\mu$ Pa) nonpulse sounds tended to leave haulout areas and seek refuge temporarily (minutes to a few hours) in the water, whereas pinnipeds exposed to distant launches at RLs ~60 to 70 dB re: 20  $\mu$ Pa tended to ignore the noise. It is difficult to assess the relevance of either of these observations to naïve individuals, however, given the repeated exposure of study colonies to such noise events and the potential that observed individuals were habituated. Due to the limitations of available data, it is not currently possible to make any further general characterizations regarding this condition.

**Table 16.** Summary of behavioral responses by different species of mid-frequency cetaceans exposed to nonpulses (Cell 6) by type of sound source, available acoustic metrics, description of behavioral response (by individual and/or group), and a summary of corresponding severity score(s); specific severity scores for each study are given in Table 17 and more details are given in Appendix C. Exposure RLS are given in dB SPL, which are RMS sound pressure levels (dB re: 1  $\mu$ Pa).

Study	Reference number (for Table 17)	Subject species	Sound source	Type of acoustic measurements	Type of individual and/or group behavioral responses	Study included in severity scale	Summary of severity scale analysis (see Table 17)
Watkins & Schevill (1975)	1	Sperm whales	Pingers	RLs measured <i>in situ</i> near individuals observed	Passive acoustic monitoring of vocal output of individuals during exposure	Yes	Exposure RLS 80-90 dB SPL; severity score: 3
Awbrey & Stewart (1983)	2	Belugas	Playbacks of drilling sounds	RL estimates based on source and environmental characteristics	Visual observations of individual and group movements and behavioral patterns during exposure and control trials	Yes	Exposure RLS 110-150 dB SPL; severity scores: 0, 1, 2 & 6
LGL & Greeneridge (1986)	3	Belugas and narwhals	Ship and ice-breaking noise	Calibrated RL measurements made <i>in situ</i> near areas of exposure	Ice-based and aerial observations of groups of animals; movement and behavioral patterns before, during, and after ice-breaking	Yes	Exposure RLS 90-120 dB SPL; severity scores: 0, 1, 2, 3 & 8
Richardson et al. (1990b)	4	Belugas	Playbacks of drilling sounds	RL estimates based on source and environmental characteristics plus sonobuoy data	Ice-based and aerial observations of individual and group movements and behavior during exposure and control trials	Yes	Exposure RLS 100-130 dB SPL; severity scores: 0, 1, 3 & 4
Gordon et al. (1992)	5	Sperm whales	Vessel noise and presence	Calibrated RL measurements made <i>in situ</i> near areas of exposure	Vessel-based observations and passive acoustic monitoring of individuals; movement patterns and behavioral responses	Yes	Exposure RLS 110-120 dB SPL; severity score: 3
Palka & Hammond (2001)	6	White-sided and white-beaked dolphins	Vessel noise and presence	RL estimates based on source and environmental characteristics	Visual observations of individual and group movements and behavioral patterns during vessel approaches	Yes	Exposure RLS 110-120 dB SPL; severity score: 3
Morton & Symonds (2002)	7	Killer whales	Various AHDs	RL estimates based on source and environmental characteristics	Census data for individual and group sightings used to estimate "exclusion" zones	Yes	Exposure RLS 140-150 dB SPL; severity score: 8
Buckstaff (2004)	8	Bottlenose dolphins	Vessel noise and presence (approaches)	Calibrated RL measurements made <i>in situ</i> near areas of exposure	Passive acoustic monitoring of individual vocal output during vessel approaches	Yes	Exposure RLS 110-120 dB SPL; severity score: 2
NRL (2004a, 2004b); NMFS (2005)	9	Killer whales	Mid-frequency active military sonar	Some calibrated RL measurements and RL estimates from modeling source and environmental characteristics	Visual observations of individual and group movements and behavioral patterns before, during, and after incidental exposure	Yes	Exposure RLS 160-170 dB SPL; severity score: 6