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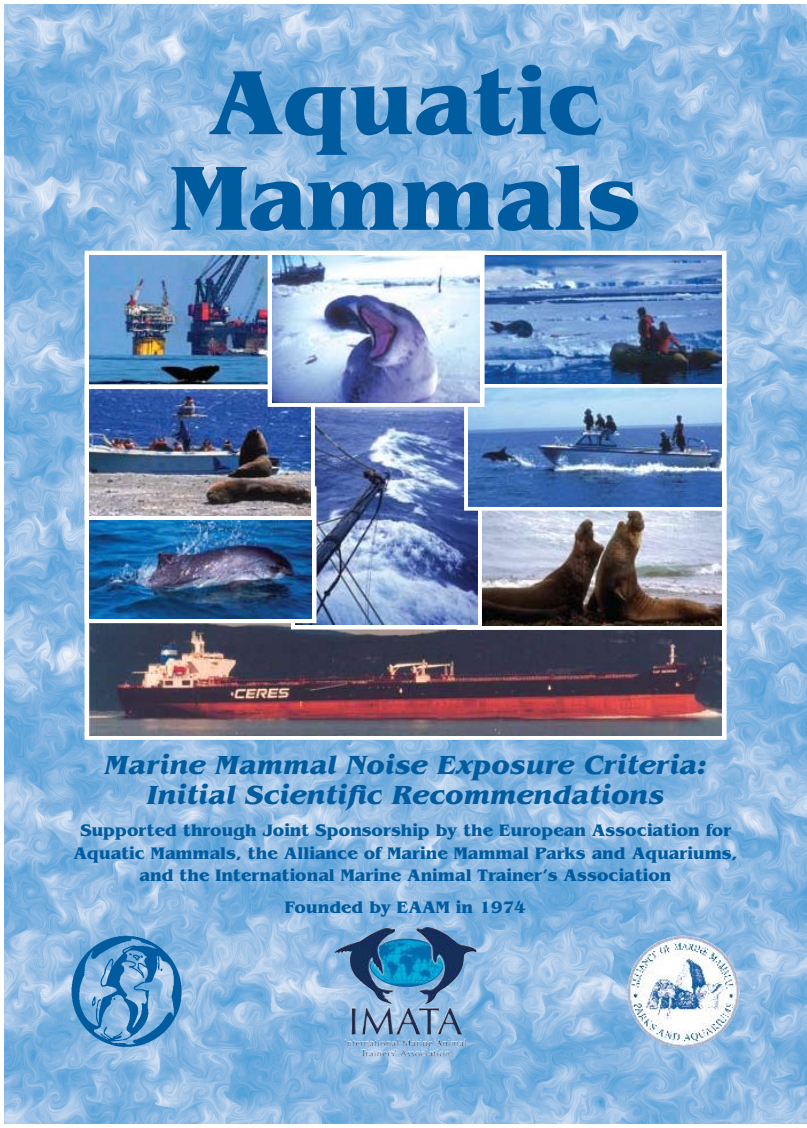

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Acronyms

Acronym	Definition
A-weighting	Frequency-selective weighting for aerial hearing in humans derived from the inverse of the idealized 40-phon equal loudness hearing function across frequencies
ABR	Auditory brainstem response
ADD	Acoustic deterrent device
AEP	Auditory evoked potentials
AHD	Acoustic harassment device
ANSI	American National Standards Institute
ASSR	Auditory steady-state response
ATOC	Acoustic Thermometry of Ocean Climate program
CF	Center frequency
C-weighting	Frequency-selective weighting for aerial hearing in humans derived from the inverse of the idealized 100-phon equal loudness hearing function across frequencies
EFR	Envelope following response
EPA	U.S. Environmental Protection Agency
ES	Explosion simulator
f_{high}	Estimated upper functional hearing limit
f_{low}	Estimated lower functional hearing limit
HESS	High Energy Seismic Survey
HPA	Hypothalamic-pituitary-adrenal axis
IMAPS	Integrated Marine Mammal Monitoring and Protection System
ISO	International Standards Organization
JNCC	U.K. Joint Nature Conservation Committee
L_{eqT}	Equivalent-continuous sound level over period T
L_{leqT}	Impulse equivalent-continuous sound level over period T
LFA	Low Frequency Active (sonar)
M-weighting	Generalized frequency weightings for various groups of marine mammals, allowing for their functional bandwidths and appropriate in characterizing auditory effects of strong sounds
M_{lf}	Frequency weighting for low-frequency cetaceans (mysticetes)
M_{mf}	Frequency weighting for mid-frequency cetaceans (most odontocetes)
M_{hf}	Frequency weighting for high-frequency cetaceans (odontocetes specialized for use of very high frequencies)
M_{pw}	Frequency weighting for pinnipeds, listening in water
M_{pa}	Frequency weighting for pinnipeds, listening in air
MMPA	U.S. Marine Mammal Protection Act
NIHL	Noise-induced hearing loss
NIPTS	Noise-induced permanent threshold shift
NIOSH	U.S. National Institute for Occupational Safety and Health
NMFS	U.S. National Marine Fisheries Service

Acronym	Definition
NOAA	U.S. National Oceanic and Atmospheric Administration
NRC	U.S. National Research Council
NRL	U.S. Naval Research Laboratory
P_{max}	Maximum sound pressure
OBN	Octave-band noise
PCAD	National Research Council's Population Consequences of Acoustic Disturbance Model
PICE	Porpoise incidental catch elimination
PTS	Permanent threshold shift
REFMS	A computer program for predicting shock-wave propagation from underwater explosions
RL	Received level
RMS	Root-mean-square
SEL	Sound exposure level
SL	Source level (received level measured or estimated 1 m from the source)
SLM	Sound level meter
SPL	Sound pressure level
TS	Threshold shift
TTS	Temporary threshold shift
USC	United States Code
VAFB	Vandenberg Air Force Base

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Overview

A group of experts in acoustic research from behavioral, physiological, and physical disciplines was convened over a several year period. The purpose of this panel was to review the expanding literature on marine mammal hearing and on physiological and behavioral responses to anthropogenic sound, and to propose exposure criteria for certain effects. The group employed all available relevant data to predict noise exposure levels above which adverse effects on various groups of marine mammals are expected. Recent advances in these fields and the pressing need for a science-based paradigm to assess the effects of sound exposure were the primary motivations for this effort. Two categories of effects were considered: (1) injury and (2) behavioral disturbance. The proposed criteria for the onset of these effects were further segregated according to the functional hearing capabilities of different marine mammal groups, and according to the different categories and metrics of typical anthropogenic sounds in the ocean. The group achieved many of its objectives but acknowledges certain limitations in the proposed criteria because of scarcity or complete absence of information about some key topics. A major component of these recommendations is a call for specific research on critical topics to reduce uncertainty and improve future exposure criteria for marine mammals. This publication marks the culmination of a long and challenging initial effort, but it also initiates a necessary, iterative process to apply and refine noise exposure criteria for different species of marine mammals.

The process of establishing policy guidelines or regulations for anthropogenic sound exposure (i.e., the application of these exposure criteria) will vary among nations, jurisdictions, and legal/policy settings. Such processes should carefully consider the limitations and caveats given with these proposed criteria in deciding whether sufficient data currently exist to establish simplistic, broad criteria based solely on exposure levels. In many cases, especially for behavioral disturbance, context-specific analyses considering previous studies on species and conditions similar to those in question might, at least for the foreseeable future, be more appropriate than general guidelines.

State of Current Knowledge

The available data on the effects of noise on marine mammals are quite variable in quantity and quality. In many respects, data gaps severely restrict the derivation of scientifically-based noise exposure criteria and, in some cases, explicit threshold criteria for certain effects are not appropriate given the amount and type of data available. Scientific inquiry into acoustic communication among marine mammals extends back more than half a century, but most of the specific data relevant to the proposed criteria have been published within the last two decades. Owing to the mounting public, scientific, and regulatory interest in conservation issues related to acoustics, the available science is progressing rapidly (e.g., see NRC, 2003, 2005).

This paper proposes, for various marine mammal groups and sound types, levels above which there is a scientific basis for expecting that exposure would cause auditory injury to occur. Controlled measurements of hearing and of the effects of underwater and aerial sound in laboratory settings have greatly expanded the ability to assess auditory effects. While understanding of the hearing capacities among all marine mammals remains admittedly rudimentary, there is a fairly detailed understanding of some key aspects of underwater and aerial hearing in a few representative species of odontocetes, pinnipeds, and sirenians, although hearing in mysticetes remains untested. Available data, along with the compelling evidence of similar auditory processes among all mammals, enables some reasonable extrapolations across species for estimating auditory effects, including the exposure levels of probable onset of injury. Recent evidence suggests that exposure of beaked whales to underwater noise may, under certain (generally unknown) conditions, result in non-auditory injury as well (e.g., Fernández et al., 2005). At present, however, there are insufficient data to allow formulation of quantitative criteria for non-auditory injuries.

There are many more published accounts of behavioral responses to noise by marine mammals than of direct auditory or physiological effects. Nevertheless, the available data on behavioral responses do not converge on specific exposure conditions resulting in particular reactions, nor do they point to a common behavioral mechanism. Even data obtained with substantial controls, precision, and standardized metrics indicate high variance both in behavioral responses and in exposure conditions required to elicit a given response. It is clear that behavioral responses are strongly

affected by the context of exposure and by the animal's experience, motivation, and conditioning. This reality, which is generally consistent with patterns of behavior in other mammals (including humans), hampered our efforts to formulate broadly applicable behavioral response criteria for marine mammals based on exposure level alone.

Frequency-Weighting Functions

In humans, hearing processes in a large number of male and female subjects of different ages have been tested to determine a basic audiometric curve, equal-loudness curve, and the levels and exposure durations needed to induce either recoverable hearing loss (called temporary threshold shift or TTS) or permanent threshold shift (PTS). In addition, the manner in which successive exposures to noise contribute to TTS growth has been well-documented in humans (e.g., Kryter, 1994; Ward, 1997). In assessing the effects of noise on humans, either an A- or C-weighted curve is applied to correct the sound-level measurement for the frequency-dependent hearing function of humans. Early on, the panel recognized that similar, frequency-weighted hearing curves were needed for marine mammals; otherwise, extremely low- and high-frequency sound sources that are detected poorly, if at all, might be subject to unrealistic criteria.

One of the major accomplishments in this effort was the derivation of recommended frequency-weighting functions for use in assessing the effects of relatively intense sounds on hearing in some marine mammal groups. It is abundantly clear from measurements of marine mammal hearing in the laboratory, call characteristics, and auditory morphology that there are major differences in auditory capabilities across marine mammal species (e.g., Wartzok & Ketten, 1999). Most previous assessments of acoustic effects either failed to account for differences in functional hearing bandwidth among marine mammal groups or did not recognize that the "nominal" audiogram might be a relatively poor predictor of how the auditory system responds to relatively strong exposures.

The authors delineated five groups of marine mammals based on similarities in their hearing, and they developed a generalized frequency-weighting (called "M-weighting") function for each. The five groups and the associated designators are (1) mysticetes (baleen whales), designated as "low-frequency" cetaceans (M_{lf}); (2) some odontocetes (toothed whales), designated as "mid-frequency" cetaceans (M_{mf}); (3) odontocetes specialized for using high frequencies (i.e., porpoises, river dolphins, and the genera *Kogia* and *Cephalorhynchus*) (M_{hf}); (4) pinnipeds (i.e., seals, sea lions, and walruses) listening in water (M_{pw}); and (5) pinnipeds listening in air (M_{pa}). These criteria do not specifically address sirenians, the sea otter, or the polar bear, in part because of the lack of key data in these species.

The M-weighting functions were defined based on known or estimated auditory sensitivity at different frequencies rather than vocal characteristics per se. Owing to the paucity of relevant data, these auditory functions are intentionally precautionary (wide) and likely overestimate the functional bandwidth for most or all species. Their primary application is in predicting auditory damage rather than levels of detection or behavioral response. Consequently, it is more appropriate to use "flatter" functions than would be obtained by employing a simple inverse-audiogram function.

Exposure Criteria Metrics

To further complicate the derivation of noise exposure criteria, sounds can be described with various acoustic metrics, including sound pressure levels and sound exposure levels. The latter is a measure of received sound energy. Available literature provides a mixture of both measures, but many sound sources have primarily been described in pressure level units. To accommodate these two measures, and to account for all relevant acoustic features that may affect marine mammals, we developed dual criteria for noise exposures in each of the five functional hearing groups, using both sound pressure and sound exposure levels.

Exposure Criteria for Injury

Another area in which we provide substantive conclusions is in the determination of sound exposures believed to cause direct auditory injury to marine mammals. By all accounts, the inner ear is the organ system most directly sensitive to sound exposure and, thus, the most susceptible to sound-derived damage. We define the minimum exposure criterion for injury as the level at which a single exposure is estimated to cause onset of permanent hearing loss (PTS). Data on TTS in marine mammals, and on patterns of TTS growth and its relation to PTS in other mammals, were used to estimate thresholds for injury. Owing to the limited availability of relevant data on TTS and PTS, the extrapolation procedures underlying these estimations are necessarily precautionary.

To account for all of the potentially injurious aspects of exposure, dual criteria for injury were established for each functional marine mammal hearing group based on instantaneous peak pressure (unweighted) and total energy (M-weighted). Exposure criteria for injury are given for two types of sounds, pulse and nonpulse, and for single and multiple exposures. The term *pulse* is used here to describe brief, broadband, atonal, transients (ANSI,

1986; Harris, 1998, Chapter 12), which are characterized by a relatively rapid rise-time to maximum pressure followed by a decay that may include a period of diminishing and oscillating maximal and minimal pressures. Examples of pulses are sounds from explosions, gunshots, sonic booms, seismic airgun pulses, and pile driving strikes. Nonpulse (intermittent or continuous) sounds can be tonal, broadband, or both. They may be of short duration but without the essential properties of pulses (e.g., rapid rise-time). Examples of anthropogenic, oceanic sources producing such sounds include vessels, aircraft, machinery operations such as drilling or wind turbines, and many active sonar systems. As a result of propagation, sounds with the characteristics of a pulse at the source may lose pulsatile characteristics at some (variable) distance and can be characterized as a nonpulse by certain receivers.

Regardless of the anthropogenic sound, if a marine mammal's received exposures exceed the relevant (pulse or nonpulse) criterion, auditory injury (PTS) is assumed to be likely. Chapter 3, "Criteria for Injury," provides details regarding the exposure levels required to cause TTS-onset and the extrapolation of those results to estimate levels above which PTS-onset may occur. For all five functional hearing groups, we propose dual exposure criteria above which auditory injury is likely.

Exposure Criteria for Behavior

One challenge in developing behavioral criteria is to distinguish a significant behavioral response from an insignificant, momentary alteration in behavior. For example, the startle response to a brief, transient event is unlikely to persist long enough to constitute significant disturbance. Even strong behavioral responses to single pulses, other than those that may secondarily result in injury or death (e.g., stampeding), are expected to dissipate rapidly enough as to have limited long-term consequence. Consequently, upon exposure to a single pulse, the onset of significant behavioral disturbance is proposed to occur at the lowest level of noise exposure that has a measurable transient effect on hearing (i.e., TTS-onset). We recognize that this is not a behavioral effect per se, but we use this auditory effect as a *de facto* behavioral threshold until better measures are identified. Lesser exposures to a single pulse are not expected to cause significant disturbance, whereas any compromise, even temporarily, to hearing functions has the potential to affect vital rates through altered behavior.

For other anthropogenic sound types (multiple pulses, nonpulses), we conducted an extensive review of the available literature but were unable to derive explicit and broadly applicable numerical threshold values for delineating behavioral disturbance. We did develop a quantitative scoring paradigm that numerically ranks, as a severity scaling, behavioral responses observed in either field or laboratory conditions. We applied this approach to the appropriate behavioral data for multiple pulses and nonpulses. Some of these data suffer from poor statistical power, limited information on received sound levels and background noise, insufficient measurements of all potentially important contextual variables, and/or insufficient controls. Some such data are analyzed here solely for illustrative purposes. Most behavioral studies suffered from at least some of these problems. Therefore, we do not intend to give uniform scientific credence to all of the cited data, and we expect future studies to give greater attention and rigor to these critical requirements.

This review and scoring process, while not a formal meta-analysis for normalizing and pooling disparate observations, corroborated certain interesting aspects of marine mammal behavioral responses to sound exposure. Foremost was that a behavioral response is determined not only by simple acoustic metrics, such as received level (RL), but also by contextual variables (e.g., laboratory vs field conditions, animal activity at the time of exposure, habituation/sensitization to the sound, etc.). Also important is the presence or absence of acoustic similarities between the anthropogenic sound and biologically relevant natural signals in the animal's environment (e.g., calls of conspecifics, predators, prey). Within certain similar conditions, there appears to be some relationship between the exposure RL and the magnitude of behavioral response. However, in many cases, such relationships clearly do not exist, at least when response data are pooled across multiple species and contexts. This argues for a context-based approach to deriving noise exposure criteria for behavioral responses. That concept, along with our review and scaling of the available observational data, provides a foundation for establishing dose-response relationships for some specific circumstances and a starting point for future analyses when additional data are available.

Conclusions and Research Recommendations

This process has resulted in several significant advances. These include a review and interpretation of the available literature on injury and behavioral data using precautionary extrapolation procedures, derivation of marine mammal frequency-weighting functions, specification of quantitative criteria for auditory injury, and derivation of a "severity scale" for behavioral responses.

The inability to identify broadly applicable, quantitative criteria for behavioral disturbance in response to multiple-pulse and nonpulse sounds is an acknowledged limitation.

Our efforts to derive marine mammal noise exposure criteria clearly illustrate the fact that, at present, research in this field remains limited in many areas. The need for extrapolation procedures and precautionary assumptions points directly to research needs in a variety of areas on a variety of species. In certain conditions, proposed criteria for an entire marine mammal group are based on the most precautionary measurement or observation for a species within that group, despite the fact that, for other species within that group, there are empirical data indicating that higher exposures are required to induce the same effect. We believe it is appropriate to use the most precautionary data in proposing group-wide criteria applicable for species where there are no direct measurements. We also feel it is appropriate on a case-by-case basis to apply the most relevant empirical data (i.e., from the species or genus of concern) in setting the exposure thresholds specified in policy guidelines.

Finally, we emphasize that exposure criteria for single individuals and relatively short-term (not chronic) exposure events, as discussed here, are insufficient to describe the cumulative and ecosystem-level effects likely to result from repeated and/or sustained human input of sound into the marine environment and from potential interactions with other stressors. Also, the injury criteria proposed here do not appear to predict what may have been indirect injury from acoustic exposure in several cases where cetaceans of several species mass-stranded following exposure to military sonar.

The extensive research recommendations given here (see Chapter 5) represent our collective view of the concerted effort that will be required over the coming decades. High priority categories of research include (1) continued expansion of knowledge on basic marine mammal hearing capabilities, including sound localization, the detection of realistic sound signals, communication masking, and auditory "scene analysis"; (2) continued expansion of knowledge on baseline marine mammal behavioral patterns; (3) well-controlled, direct measurements (using appropriate, standardized acoustic metrics) of the effects of sound exposure on marine mammal hearing, behavior, and physiology; and (4) risk-assessment studies of the cumulative and synergistic effects of noise and other exposure(s) on individuals and populations.

Understanding and managing the effects of noise on marine life without unjustifiably constraining important human activities in the oceans will continue to be challenging for the foreseeable future. With sustained and focused research in key areas, future scientists will be equipped to make informed improvements to the initial scientific recommendations presented here. These improvements should ideally be integrated into science-based risk assessment models that consider all aspects of sound exposure and other potential stressors on individual marine mammals, populations, and marine ecosystems.

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1. Introduction

Objectives

Recent interest and concern about the effects of anthropogenic noise on marine mammals has triggered considerable new research (e.g., Costa et al., 2003; Fristrup et al., 2003; Finneran et al., 2005a), summaries of available information (Richardson et al., 1995; Wartzok & Ketten, 1999), and recommendations for specific action (NRC, 1994, 2000, 2003, 2005). Systematic, objective, science-based interpretation of the available data is critically needed to inform management agencies charged with mitigating adverse effects of anthropogenic noise on protected species. In response to this need, we use here the full body of scientific data on marine mammal hearing and the effects of noise on hearing and behavior, augmented where appropriate by interpretations of terrestrial mammal (including human) data, to develop proposed exposure criteria that are as comprehensive, defensible, and precise as is currently possible. The scope of these criteria includes injurious and behavioral effects of a single noise exposure event on an individual cetacean (whales, dolphins, and porpoises) or pinniped (seals, sea lions, and walruses).

The recommended noise exposure criteria are science-based, developed without addressing the commercial, societal, or practical ramifications of implementing the conclusions reached here. We intend to mirror the process used in the development of damage risk criteria for humans (see Crocker, 1997). Policy "guidelines" developed for regulatory and societal purposes are based both on scientific evidence (as summarized in this paper for marine mammals) and on other considerations (e.g., economic, practical, social, and ethical) not dealt with here. Thus, on certain points, policy guidelines that are developed separately for the purposes of various jurisdictions, nations, or users of these criteria may differ from the science-based criteria recommended here.

All forms of anthropogenic noise received by marine mammals were considered, whether produced under water or in air, and we adopted a comparative approach, which we regard as essential to any criteria-setting process for nonhuman animals. For most of the ~128 marine mammal species and subspecies (Rice, 1998) considered here, no empirical data were available on nominal hearing characteristics or on the effects of noise on hearing or behavior. Practical, ethical, and legal considerations limit the level of scientific information that is available for deriving criteria applicable to either humans or marine mammals. Consequently, certain assumptions and criteria proposed here were based on information from other mammalian groups, where justified. Where such data present a variety of options, we made intentionally precautionary decisions (i.e., lower proposed exposure levels) to reduce the risk of assuming no effect when one was actually present. The term "precautionary" is used here without reference to any regulatory or policy implication of this word. Scientists would more conventionally use the term "conservative" in this regard rather than the more bureaucratic "precautionary," but in certain complex instances here, the term "conservative" would be potentially ambiguous, depending on the perspective of the reader. When information was limited, extrapolations were made cautiously to minimize the risk of failing to recognize an effect when one actually occurs (Type-II statistical error) as can occur with small sample sizes or imprecise measurements.

Each generalization/extrapolation was identified, all precautionary decisions were noted, and the logic leading to each proposed criterion was specified. Thus, when new data become available, appropriate modifications can be made readily. Studies that are needed to resolve the uncertainties encountered in developing the current criteria are discussed in detail (see Chapter 5, "Research Recommendations"). Realistically, however, the generalization of information between related species will remain essential in many cases for the foreseeable future.

Our intent was to derive recommended noise exposure criteria using the best information currently available, identify weaknesses in the present approach, call for relevant research, and structure the criteria such that future improvements can be incorporated easily. Lack of data limited the proposed noise exposure criteria to individual marine mammals exposed to acute exposure events (such as the passage of one vessel or a series of active sonar transmissions). Also, the proposed criteria are limited to cetaceans and pinnipeds. We expect that noise exposure criteria for other marine mammals (manatees, dugongs, polar bears, and sea otters), as well as other marine taxa, will be developed as additional data become available and are evaluated. In fact, a separate expert panel (S3/

WG92: "Effects of Sound on Fish and Turtles") has been established under the Standards Committee (S3) of the Acoustical Society of America to consider noise exposure criteria for fish and turtles. Additionally, criteria are clearly needed for cumulative effects and for effects at species or even ecosystem levels, but data to support those types of criteria do not currently exist.

The present recommended criteria represent a major step in initiating a lengthy, systematic process to predict and identify acoustic exposure conditions (natural or anthropogenic) associated with various effects on marine mammals. This paper is deliberately structured in a somewhat formulaic and report-like manner so that the logic underlying certain assumptions and extrapolations (as well as the data needed to test and/or strengthen them) is self-evident. We expect there will be an iterative process of improving and expanding the complexity of the exposure criteria, similar to the decades-long development of human noise exposure criteria (see Crocker, 1997). Because of the matrix structure of the proposed criteria, thresholds in specific cells can be updated independently as new information becomes available.

There is an extensive history and diversity of exposure criteria for humans with various kinds of acoustic exposure. A full discussion of these criteria is beyond the scope of this paper, but examples include workplace noise standards (e.g., NIOSH, 1998), standards for the protection of military personnel (U.S. DoD, 1997), and national policy guidelines (e.g., EPA, 1974; BG PPG, 1994). Several additional examples were also considered, whether received under water or in air, in various decisions underlying the marine mammal criteria proposed here. The process of establishing human noise exposure criteria has been difficult and contentious, but establishing noise exposure criteria for marine mammals is considerably more daunting given the diversity of marine mammal species across three orders, the complexity of aerial and underwater acoustic exposures, and profound data limitations.

Historical Perspective

Concerns about potential adverse effects of anthropogenic noise on marine life began in the 1970s (e.g., Payne & Webb, 1971) and expanded in the 1980s. Experiments during the 1980s with seismic airguns indicated that bowhead whales (*Balaena mysticetus*) and gray whales (*Eschrichtius robustus*) exhibited clear, sustained avoidance of operational areas at distances where pulse root-mean-square (RMS) sound pressure levels (SPLs) were 160 to 170 dB re: 1 μPa (Malme et al., 1983, 1984, 1986, 1988; Richardson et al., 1986; Ljungblad et al., 1988). In contrast, early observations of bowhead and gray whales exposed to continuous industrial sounds, such as those associated with drilling operations, suggested 120 dB re: 1 μPa as the approximate threshold for behavioral disturbance of these baleen whales (Malme et al., 1984; Richardson et al., 1990a, 1995 [pp. 286-287]). Significant individual variability was noted in "typical" behavioral responses, however, with some individual whales responding only when very close to sound sources and others reacting at much longer distances (and to lower received sound levels). This variability raises questions as to whether behavioral responses are most appropriately described by the exposure received level (RL) of the stimulus at the animal, the signal-to-ambient noise differential, the rate of change of the signal, or simply to the presence of the human activity as indicated by acoustic cues and/or visual stimuli.

Concern about the effects of acoustic pulses from seismic exploration and continuous sound from other industrial activities resulted in the imposition of mitigation requirements on some industrial activities in certain jurisdictions by the early- to mid-1980s. Subsequent events, such as the Heard Island Feasibility Test in 1991 (Baggeroer & Munk, 1992), the Acoustic Thermometry of Ocean Climate (ATOC) program in the late-1990s (see NRC, 1994, 2000; Au et al., 1997; Costa et al., 2003), and the U.S. Navy's low-frequency active sonar program (e.g., Croll et al., 2001) resulted in popular and governmental interest in setting criteria for safe levels of sound for marine mammal exposure (NRC, 1994, 2000, 2003; Richardson et al., 1995). This interest has expanded with the finding that tactical, mid-frequency, military sonar transmissions are sometimes correlated, in specific conditions, with mass stranding events of (predominantly) several beaked whale species, including Cuvier's (*Ziphius cavirostris*), Blainville's (*Mesoplodon densirostris*), and Gervais' (*Mesoplodon europaeus*) beaked whales (see Evans & England, 2001; Fernández et al., 2005; Cox et al., 2006).

In 1995, the U.S. National Marine Fisheries Service (NMFS) set underwater "do not exceed" criteria for exposure of marine mammals to underwater pulses from seismic airguns. These criteria were 190 dB re: 1 μPa for pinnipeds and most odontocete cetaceans and 180 dB re: 1 μPa for mysticetes and sperm whales (*Physeter macrocephalus*) (and, by inference, for pygmy and dwarf sperm whales [*Kogia* spp.]). These exposure limits were intended as precautionary estimates of exposures below which physical injury would not occur in these taxa. There was no empirical evidence as to whether exposure to higher levels of

pulsed sounds would or would not cause auditory or other injuries. Given the limited data then available, however, it could not be guaranteed that marine mammals exposed to higher levels would not be injured. Further, it was recognized that behavioral disturbance could, and in some cases likely would, occur at lower RLs.

In June 1997, the High Energy Seismic Survey (HESS) team (1999, Appendix 5) convened a panel of experts to assess noise exposure criteria for marine mammals exposed to seismic pulses. The consensus was that, given the best available data at that time, exposure to airgun pulses with RLs above 180 dB re: 1 μPa (averaged over the pulse duration) was "likely to have the potential to cause serious behavioral, physiological, and hearing effects." The panel noted the potential for ± 10 dB variability around the 180 dB re: 1 μPa level, depending on species, and that more information was needed.

The NMFS has continued to use a "do not exceed" exposure criterion of 180 dB re: 1 μPa for mysticetes and (recently) all odontocetes exposed to sequences of pulsed sounds, and a 190 dB re: 1 μPa criterion for pinnipeds exposed to such sounds. Higher thresholds have been used in the U.S. for single pulses such as explosions used in naval vessel-shock trials. Behavioral disturbance criteria for pulsed sounds have typically been set at an SPL value of 160 dB re: 1 μPa, based mainly on the earlier observations of mysticetes reacting to airgun pulses (e.g., Malme et al., 1983, 1984; Richardson et al., 1986). The relevance of the 160 dB re: 1 μPa disturbance criterion for odontocetes and pinnipeds exposed to pulsed sounds is not at all well-established, however. Although these criteria have been applied in various regulatory actions (principally in the U.S.) for more than a decade, they remain controversial, have not been applied consistently in the U.S., and have not been widely accepted elsewhere.

More recently, a considerable body of data has accumulated on the levels at which transient and more prolonged sounds cause the onset of temporary threshold shift (TTS) and various behavioral reactions. Some of these data are not consistent with the aforementioned *de facto* criteria used in recent years in the United States.

One main purpose of this paper is to synthesize and apply all available information to derive proposed objective noise exposure criteria for a large subset of marine mammals. The effect levels considered (injury and significant behavioral disturbance) were generally consistent with the definitions of levels A and B harassment, respectively, of the U.S. Marine Mammal Protection Act (MMPA) of 1972 (16 USC, § 1361); however, many of the behaviors considered at the lower end of our severity scaling paradigm would almost certainly not constitute biologically significant disturbance (or consequently level B harassment under the MMPA). However, our exposure criteria were derived without regard for policy decisions of the U.S. or any nation and should therefore not be assumed to correspond with regulatory categories or definitions of effects. Since harassment definitions under the MMPA are not uniform for all human activities and are subject to change, additional interpretation of the information presented would be required to evaluate effects with regard to this (or any other) statute.

Acoustic Measures and Terminology

This section briefly considers those acoustic measures and terminology that are directly relevant to these marine mammal exposure criteria. More detailed descriptions of some of the terms given in this and other sections, including equations relevant to many of the definitions, are given in Appendix A. Basic acoustic terminology is presented in numerous other sources (e.g., Kinsler et al., 1982; ANSI, 1986, 1994; Richardson et al., 1995; Harris, 1998; NRC, 2003).

Sound is appropriately described as having two components: (1) a pressure component and (2) a particle motion component. Particle motion—the oscillatory displacement, velocity, or acceleration of the actual "particles" of the medium at a particular location—is directional and best described by a 3-dimensional vector. Marine mammal sensitivity to particle motion is poorly understood, but it appears to be functionally limited (Finneran et al., 2002a) in contrast to the sensory capabilities of most or all fish (see Popper et al., 2003). Conversely, as compared to fish, marine mammals generally have greater sensitivity to sound pressure (lower detection thresholds) and much wider functional hearing bandwidths (see Fay, 1988; Richardson et al., 1995; Popper et al., 2003). Consequently, in considering the potential effects of sound on marine mammals, particle motion is rarely discussed. Except for special circumstances (e.g., plane and spherical waves), there is no simple relationship between pressure and particle velocity. The vast majority of studies of hearing in captive marine mammals have been conducted in relatively small enclosed volumes of water, making the plane wave assumption (and *a priori* knowledge of the relationship between pressure and velocity) invalid.

It is important to distinguish between the *source level* (SL), or level measured 1 m from the source, vs the *received level* (RL), which is the level measured at the receiver (usually a marine mammal herein).

The term "intensity" is often used generally with respect to subjective acoustic parameters (i.e., loudness), but it is used here in a strict sense. Sound intensity is normally defined as the time-averaged active intensity (Kinsler et al., 1982; Fahy, 1995); this quantity corresponds to local net transport of sound energy and is related to the product of the sound pressure and the particle velocity component in-phase with the sound pressure. In the majority of laboratory studies, complex sound fields typically create complex, spatially varying relationships between pressure and velocity. In these circumstances, sound intensity cannot be estimated from pressure measurements alone (which assume that pressure and particle velocity are in-phase), and specific measurements of the sound particle velocity (or pressure gradient) are required in order to characterize intensity.

We distinguished two basic sound types: (1) *pulse* and (2) *nonpulse*. Our operational definitions of sound types are given in Chapter 2, "Structure of the Noise Exposure Criteria," and are discussed at greater length in Appendix A. The pulse/nonpulse distinction is important because pulses generally have a different potential to cause physical effects, particularly on hearing (e.g., Ward, 1997).

Peak sound pressure (P_{max}) is the maximum absolute value of the instantaneous sound pressure during a specified time interval and is denoted in units of Pascals (Pa). It is in no sense an averaged pressure. Peak pressure is a useful metric for either pulse or nonpulse sounds, but it is particularly important for characterizing pulses (ANSI, 1986; Harris, 1998, Chapter 12). Peak-to-peak sound pressure is the algebraic difference between the maximum positive and maximum negative instantaneous peak pressure. The mean-squared pressure is the average of the squared pressure over some duration. Sound pressure levels are given as the decibel (dB) measures of the pressure metrics defined above. The RMS SPL is given as dB re: 1 μPa for underwater sound and dB re: 20 μPa for aerial sound. Peak sound pressure levels are denoted hereafter as dB re: 1 μPa (peak) in water and dB re: 20 μPa (peak) in air. Peak-to-peak sound pressure levels are dB re: 1 μPa (peak-to-peak) in water and dB re: 20 μPa (peak-to-peak) in air.

Duration is the length of a sound in seconds. Duration is important because it affects other sound measures, specifically mean-square and/or RMS sound pressure (Madsen, 2005). Because of background noise and reverberation, duration can be difficult to specify precisely, but a functional definition (see Appendix A) is used here.

Sound exposure level (SEL) is a measure of energy. Specifically, it is the dB level of the time integral of the squared-instantaneous sound pressure normalized to a 1-s period. It can be an extremely useful metric for assessing cumulative exposure because it enables sounds of differing duration, sometimes involving multiple exposures, to be compared in terms of total energy. Several methods exist for summing energy over multiple exposures to generate a single exposure "equivalent" value. The relatively straightforward approach used here is described in Appendix A (eq. 5). This summation procedure essentially generates a single exposure "equivalent" value that assumes no recovery of hearing between repeated exposures. As discussed below, recovery functions for marine mammal TTS during and following multiple exposures are still unknown; however, considering nominal TTS recovery functions in terrestrial mammals when exposures occur minutes to hours apart (see Kryter, 1994; Ward, 1997), the above summation procedure would likely overestimate the effect of multiple exposures in many conditions. This summation procedure was intentionally selected as a precautionary measure in the absence of empirical information, although note the temporal conditions given in the "Sound Types" section of Chapter 2. The appropriate units are dB re: 1 μPa^2-s for underwater SEL and dB re: $(20\ \mu Pa)^2$-s for aerial SEL.

Frequency-selective weighting is often employed to measure (as a single number) sound pressure or energy in a specific frequency band of sound, with emphasis or de-emphasis on particular frequencies as a function of the relative sensitivity of a receiver. For aerial hearing in humans, A-weighting is derived from the inverse of the idealized 40-phon equal loudness hearing function across frequencies, standardized to 0 dB at 1 kHz (Harris, 1998). This provides level measures denoted as dB(A). C-weighting is determined from the inverse of the idealized 100-phon equal loudness hearing function (which differs in several regards from the 40-phon function), standardized to 0 dB at 1 kHz (Harris, 1998). This provides level measures denoted as dB(C). In the absence of equal-loudness contours for marine mammals, special frequency-weighting functions based loosely on human C-weighting and general knowledge of functional hearing bandwidth were developed here for functional marine mammal hearing groups (see the "Marine Mammal Functional Hearing Groups" section of Chapter 2).

Other measures of noise interference with critical functions in humans, including the Articulation Index (French & Steinberg, 1947) and the more recent Speech Interference Level (see Beranek & Ver, 1992), focused on the perception of speech and effects of noise. Consequently, exposure criteria geared toward speech perception (e.g., Beranek, 1989) focus on a frequency

bandwidth narrower than the audible bandwidth. For a detailed discussion of speech intelligibility and noise impacts, see Chapter 6 in Kryter (1994). It is clear that the perception of conspecific vocal signals in marine mammals is critically important in various life history functions (discussed below; see Wartzok & Ketten, 1999) and that interference with these functions may have particularly negative consequences.

The hypothesis that vocalizations coincide with the range of hearing is based on an adaptive argument that vocal energy should be selected to lie within the range of hearing for maximum efficiency of communication. However, several lines of evidence suggest that other adaptive pressures may shape the vocal range. First, vocal anatomy may produce energy at other frequencies as a byproduct of producing sound within the hearing range. If there is no pressure to eliminate these frequencies, they can be expected to persist. An example is the ultrasonic components of hummingbird song, which lie well outside the range of bird hearing (Pytte et al., 2004). Second, to promote long-range transmission, the vocal range may be adapted to produce greater energy at the low end of the range than would be expected based on the auditory threshold function (Larom et al., 1997). Greater relative energy at low frequencies is also seen in a number of primate species as a byproduct of producing the formant structure of their calls (Fitch & Hauser, 1995). Finally, animals may produce sounds with disproportionate low-frequency information to signal greater size, potentially targeting predators rather than conspecifics (Fitch, 1999; Matrosova et al., 2007). Thus, a number of selective forces can drive the development of an emphasis on low-frequency energy in vocalizations not matched by the shape of the auditory threshold function. While vocal range can be expected to correlate with hearing range to some degree, giving a rough indication of the frequency range of hearing, it cannot be used to estimate either the shape of the auditory threshold function or to assign upper and lower frequency limits.

We lack sufficient empirical data on whether vocal frequency range sufficiently predicts all frequencies that are biologically significant, however.

Certain marine mammal responses to anthropogenic sounds, such as the sometimes strong reactions by beaked whales to mid-frequency sonar, would not be expected if only sounds within the bandwidth of vocal output were important in predicting a behavioral response. Hence, our precautionary frequency-weighting approach assumes that the full audible band is relevant. As additional data become available on both hearing capabilities (specifically, equal-loudness contours) and behavioral responses to natural (including conspecific) and anthropogenic sounds, a more refined means of frequency-weighting than the intentionally precautionary (broad) M-weighting functions may be recommended.

Kurtosis is a statistical measure of a probability distribution often applied to describe the shape of the amplitude distribution (Hamernik & Hsueh, 1991; Lei et al., 1994; Hamernik et al., 2003). In some regards, it appears to be a highly relevant metric in that impulsive sound with high negative kurtosis, rapid onset, and high instantaneous peak-pressure may be particularly injurious to some mammals (Hamernik et al., 2003).

Sound Production and Use in Marine Mammals

As a general statement, all studied marine mammals can produce sounds in various important contexts. They use sound in social interactions as well as to forage, to orient, and to respond to predators. Interference with these functions, through the various effects of noise on hearing and/or behavior identified below, thus has the potential to interfere with vital rates identified by the NRC (2005) as particularly significant effects of exposure.

The noise exposure criteria given here are focused on current knowledge of hearing and the effects of noise on hearing and/or behavior in marine mammals. Thus, a detailed discussion and review of the expansive literature on the production and the uses of sound is beyond the scope of this paper; interested readers are referred to the many reviews of marine mammal acoustic signals (e.g., Schusterman, 1981; Watkins & Wartzok, 1985; Au, 1993; Richardson et al., 1995; Wartzok & Ketten, 1999; Clark & Ellison, 2004). Because of the extreme importance of detecting conspecific social signals in marine mammal life history functions, however, a brief and very general discussion of sound output characteristics in the major marine mammal groups is given here.

The large whales (mysticete cetaceans, as described below) generally produce low-frequency sounds in the tens of Hz to the several kHz band, with a few signals extending above 10 kHz. These sounds appear to serve predominantly social functions, including reproduction and maintaining contact, but they may also play some role in spatial orientation.

The dolphins and porpoises (odontocete cetaceans, also described below) produce sounds across some of the widest frequency bands that have been observed in animals. Their social sounds are generally in the range audible to humans, from a few hundreds of Hz to several tens of kHz, but specialized clicks used in biosonar (echolocation)

systems for prey detection and navigation extend well above 100 kHz.

Pinnipeds (seals, sea lions, and walruses) also produce a diversity of sounds, though generally over a lower and more restricted bandwidth (generally from 100 Hz to several tens of kHz). Their sounds are used primarily in critical social and reproductive interactions. Pinnipeds spend time both at sea and on land, however, and thus produce sounds in both water and air.

Because sound production in marine mammals is integral to so many important behaviors, interference with these communicative functions is considered to be particularly adverse (see severity scaling described in Chapter 4, "Criteria for Behavioral Disturbance"). As discussed in Chapter 5, considerable additional research is needed to identify conditions in which anthropogenic noise exposure interferes with acoustic communication as well as ways in which marine mammals cope with masking noise to overcome interference in detecting real-world signals in complex, 3-dimensional marine environments.

Responses to Sound

Animals exposed to either natural or anthropogenic sound may experience physical and psychological effects, ranging in magnitude from none to severe. This brief discussion considers the range of potential impacts, which depend on spatial relationships between a sound source and the animal receiver; sensitivity of the receiver; received exposure level, duration, and duty cycle; and many other factors (see also Richardson et al., 1995).

The same acoustic source may have radically different effects depending on operational and environmental variables, and on the physiological, sensory, and psychological characteristics of exposed animals. It is important to note that these animal variables may differ (greatly in some cases) among individuals of a species and even within individuals depending on various factors (e.g., sex, age, previous history of exposure, season, and animal activity). Responses elicited can depend both on the context (feeding, mating, migrating, etc.) in which an individual is ensonified and on a host of experiential variables (see Wartzok et al., 2004). Consequently, certain effects may be poorly described with simple measures such as SPL alone, and may only be predictable when additional variables are considered. We considered all known factors in developing the noise exposure criteria proposed here, but data limitations precluded the derivation of explicit exposure criteria for all of the effects discussed below.

Audibility

When a sound can be perceived amidst background noise, it is considered to be audible. Audibility can differ from detectability in that a receiving system may detect a signal at some level even when it is incapable of meaningful perception. Audibility is determined by the characteristics of received sound, characteristics of the receiving system, and background noise conditions (either external or internal). Audition (hearing) is a well-developed and primary sensory modality for most, if not all, marine vertebrates (Schusterman, 1981; Tyack, 1998; Fay & Popper, 2000). It involves coding, processing, integrating, and responding to sound in a variety of ways, some not outwardly evident (Yost, 2000). Like other animals, marine mammals have multiple sound-reception pathways and rely on signal processing at multiple levels integrated within the cochlea and nervous system to optimize perception.

Marine mammal hearing capabilities are quantified in live subjects using behavioral audiometry and/or electrophysiological techniques (e.g., Schusterman, 1981; Au, 1993; Kastak & Schusterman, 1998; Wartzok & Ketten, 1999; Nachtigall et al., 2000, 2007; Finneran & Houser, 2006; André & Nachtigall, 2007; Supin & Popov, 2007). For species not studied with *in vivo* audiometry, some auditory characteristics can be estimated based on sound production frequencies; on observations of sound characteristics that either do or do not elicit behavioral responses in untrained animals (e.g., Richardson et al., 1995; Erbe, 2002); or on auditory morphology, including biomechanical properties of the basilar membrane and other characteristics (Wartzok & Ketten, 1999).

Behavioral audiograms are obtained from captive, trained animals using standard psychometric testing procedures. With appropriate controls and sufficient training, behavioral data are presently considered to most accurately represent hearing capabilities of a test subject. Behavioral audiometric studies are time-consuming, however, and the results depend on the training and attention of subjects as well as the background noise conditions in captive settings. Because marine mammals are large and difficult to maintain, behavioral audiograms representing an entire species are typically based on a few individuals (often one animal). Additionally, subjects are generally obtained opportunistically (e.g., individuals rehabilitated after stranding) rather than by random sampling of individuals from wild populations. This may provide a somewhat biased representation of "normal" hearing for the species if rehabilitated animals have compromised hearing capabilities (see André et al., 2007). Individual differences in hearing sensitivity among subjects,

and methodological differences among investigators, can lead to improper conclusions when nominal species audiograms are based on data from a single animal (e.g., compare Hall & Johnson, 1972, with Szymanski et al., 1999). Hearing sensitivity has been measured using behavioral methods in fewer than 20 of the ~128 cetacean and pinniped species (based on the taxonomy of Rice, 1998).

Electrophysiological audiometry involves measuring small electrical voltages (auditory evoked potentials [AEPs]) produced by neural activity when the auditory system is stimulated by sound. With this technique, neural responses are typically averaged while many relatively short duration signals are presented. This technique is comparatively fast and less sensitive to factors such as subject experience and reproductive, behavioral, or motivational states that affect behavioral audiometry. Whereas behavioral audiograms can only be made with trained, captive animals, AEP measures of sound detection can also be made with untrained individuals that are stranded, temporarily restrained, or in rehabilitation (see Cook et al., 2006; André et al., 2007; Delory et al., 2007; Taylor et al., 2007).

AEP and behavioral techniques measure different features of the auditory system and may generate somewhat different measured results. Relevant comparisons of AEP and behavioral audiograms are limited and are the subject of ongoing scientific investigation. Besides the need to obtain both types of data on the same individuals, there are complications due to differences in the types of test stimuli used by different researchers, problems in estimating the true RL at the relevant sensory organ(s), and the difficulty of determining absolute signal amplitudes that barely elicit neural responses. Even so, Yuen et al. (2005), Finneran et al. (2007b), and Schlundt et al. (2007) demonstrated that, with carefully calibrated and repeated measurements, the two procedures can produce comparable detection thresholds in at least a few cetacean species.

An auditory threshold, estimated by either behavioral or electrophysiological responses, is the level of the quietest sound audible in a specified percent of trials. An auditory threshold is not an invariant critical value above which a sound is always heard and below which it is never heard. Instead, it is a sound level at which there is an explicit signal detection probability (often 50%; determined *a priori*). This probability depends on a number of intrinsic factors (Green & Swets, 1974; Egan, 1975; McMillan & Creelman, 1991). In all species tested thus far, the hearing response in relation to frequency is a generally U-shaped curve with a frequency range of best sensitivity (lowest hearing thresholds) and frequencies both below and above this range where sensitivity is relatively poor (higher threshold values). Species differ in absolute sensitivity and functional frequency bandwidth (see Fay, 1988; Richardson et al., 1995), such that identical sounds may be perceived radically differently by individuals of different species. Individual differences within species have also been demonstrated in some terrestrial species (see Fay, 1988) and, to a lesser extent, in marine mammals as well (see Houser & Finneran, 2006b, for the most definitive example of this). Sounds whose levels barely exceed background noise levels may be detectable but may or may not elicit changes in individual behavior. Ideally, "absolute" or unmasked hearing thresholds should be measured in low background noise conditions such as anechoic testing enclosures. While this is standard practice in human audiometry, very few of the marine mammal hearing data obtained to date have been measured in such conditions. Limited recent data obtained with pinnipeds tested in a hemi-anechoic testing chamber in air (described in Kastak et al., 2005) suggest that masking from environmental noise in testing enclosures may have significantly affected measurements of "absolute" hearing; thresholds in a harbor seal (*Phoca vitulina*) were in fact ≥ 30 dB lower in very low background noise conditions (Holt et al., 2001).

While the above concepts and studies are essential in understanding general hearing capabilities (e.g., functional bandwidth, range of best hearing sensitivity) of marine mammals, animals in the "real world" rarely listen for simple acoustic signals from point sources and do not live in a noise-controlled environment. Rather, they are presented with spatially complex and time-varying streams of acoustic information in often noisy environments. Measurements using simple sound stimuli have indicated that marine mammals are generally quite adept at localizing acoustic sources in laboratory conditions (Møhl, 1964; Gentry, 1967; Terhune, 1974; Moore & Au, 1975; Renaud & Popper, 1975; Holt et al., 2004, 2005). Many of the behavioral observations discussed in Chapter 4 (and in Appendices B & C) indicated relatively precise orientation behaviors to sound sources (or sound localization) in the field as well. Limited laboratory data are also available regarding how marine mammals detect relatively simple stimuli over background masking noise (discussed below). A more complex perceptual matter related to localization and detection over masking noise is the manner in which vertebrates process complex information to perceive the acoustic (or auditory) scene—that is, gain useful information from

the suite of sounds around them in the real world (e.g., Fay & Popper, 2000).

Bregman (1990) considered how the human auditory system constructs a perceptual acoustic image of the surrounding environment and events occurring in that environment. He posits that, as in visual perception, hearing systems are organized in such a manner that related acoustic events (such as the frequency structure of a harmonic signal or a repeated signal from the same source in a 3-dimensional space) are grouped perceptually in a meaningful way. According to the process of *auditory scene analysis*, the auditory system sorts-out related elements of a complex natural acoustic environment into those arising from different sound sources. Furthermore, previous experience can have powerful effects on the processing and interpretation of sounds. This too is similar to psychological processes underlying visual perception in which the range to an object may be inferred from knowledge of an object's general size and physical appearance.

Presuming such capabilities occur in marine vertebrates, which is logical given the importance of sound to marine mammals, it seems likely that they could perceive range and the general nature (e.g., movement) of sound sources. *Acoustic stream segregation*, the identification of relatively simple stimuli from different, overlapping patterns, has been demonstrated in several bird and bat species (MacDougall-Shackleton et al., 1998; Moss & Surlykke, 2001). Neither acoustic stream segregation nor auditory scene analysis has yet been investigated in marine mammals (but see Madsen et al., 2005a). Each of these processes, along with more data on sound localization, may be relevant in the continued development of appropriate marine mammal noise exposure criteria (see the "Marine Mammal Functional Hearing Groups" section of Chapter 5, for research recommendations).

Auditory Masking

Noise may partially or entirely reduce the audibility of signals, a process known as *auditory masking*. The extent of interference depends on the spectral, temporal, and spatial relationships between signals and masking noise, in addition to other factors. Human auditory systems perform frequency-based assessment (similar to Fourier analysis) on incoming signals such that, for most exposure levels, significant masking of tonal signals is almost exclusively by noise in a narrow band (called the critical band) of similar frequencies (Wegel & Lane, 1924; Fletcher, 1940; Greenwood, 1961). With increasing masker level, however, there is an asymmetrical spread in the masking effect such that detection of frequencies above those of the masking stimulus is more significantly impeded (see Buus, 1997; Yost, 2000).

Because of common biomechanical cochlear properties across taxa (Echteler et al., 1994), masking is expected to follow similar principles in other mammals (including marine mammals). The structure and function of the outer and middle ear differ profoundly between terrestrial and marine mammals (Wartzok & Ketten, 1999); however, the characteristics of auditory masking are strikingly similar among nonspecialized mammals in general (Fay, 1988; Echteler et al., 1994), including marine mammals tested in air and in water (Turnbull & Terhune, 1990; Southall et al., 2000, 2003). Similarities in morphology and mammalian cochlear functional dynamics (as revealed by masking studies) suggest that auditory data from terrestrial mammals may be reliably used in some situations where marine mammal data are lacking. Data on auditory masking in marine mammals are not presented in detail here because they are not directly used in formulating the recommended noise exposure criteria (but see Southall et al., 2000, 2003, for reviews).

Auditory Threshold Shift

Animals exposed to sufficiently intense sound exhibit an increased hearing threshold (i.e., poorer sensitivity) for some period of time following exposure; this is called a *noise-induced threshold shift* (TS). Factors that influence the amount of TS include the amplitude, duration, frequency content, temporal pattern, and energy distribution of noise exposure. The magnitude of TS normally decreases over time following cessation of the noise exposure. The amount of TS just after exposure is called the initial TS.

If TS eventually returns to zero (i.e., the threshold returns to the pre-exposure value), it is called TTS. The following physiological mechanisms are thought to play some role in inducing TTS, also referred to as auditory fatigue: effects on sensory hair cells in the inner ear that reduce their sensitivity, modification of the chemical environment within sensory cells, residual middle-ear muscular activity, displacement of certain inner ear membranes, increased blood flow, and post-stimulatory reduction in both efferent and sensory neural output (Kryter, 1994; Ward, 1997). Where these effects result in TTS rather than a permanent change in hearing sensitivity, they are within the nominal bounds of physiological variability and tolerance and do not represent physical injury (Ward, 1997). Recovery of nominal hearing function may occur quickly, and the amount of TTS measured depends on the time elapsed since the cessation of noise exposure; subscripts are used to indicate the time in minutes after exposure. For

example, TTS_2 means TTS measured 2 min after exposure cessation.

If TS does not return to zero after a relatively long interval (on the order of weeks), the residual TS is called a noise-induced permanent threshold shift (PTS). The distinction between PTS and TTS depends on whether there is a complete recovery of TS following noise exposure. PTS is considered to be auditory injury. Some of the apparent causes of PTS in mammals are severe extensions of effects underlying TTS (e.g., irreparable damage to the sensory hair cells). Others involve different mechanisms, such as exceeding the elastic limits of certain tissues and membranes in the middle and inner ears and resultant changes in the chemical composition of inner ear fluids (Ward, 1997; Yost, 2000). The relationship between TTS and PTS depends on a highly complex suite of variables concerning the study subject and the exposure. This relationship remains poorly understood, even for humans and small terrestrial mammals in which this topic has been investigated intensively (see Kryter, 1994; Yost, 2000).

In addition to the potential for discrete, intense sounds to result in TTS or PTS, chronic sound exposure, common in industrialized societies, can result in noise-induced PTS in humans as they age (see Kryter, 1994). Reduced hearing sensitivity as a simple function of development and aging (*presbycusis*) has been demonstrated in both children (Roche et al., 1978) and adults (e.g., Brant & Fozard, 1990). In the long-term, noise-induced hearing loss and presbycusis appear to result in a progressive PTS that is a complex, nonlinear process and particularly affects high-frequency hearing. Limited research in cetaceans and pinnipeds has revealed patterns of presbycusis that are similar to those observed in humans (Ridgway & Carder, 1997; Brill et al., 2001; Schusterman et al., 2002; Houser & Finneran, 2006b; Reichmuth et al., 2007), further underscoring certain general similarities in auditory processes across mammals.

PTS and TTS data from humans and nonhuman terrestrial mammals were used to develop safe exposure guidelines for human work environments (e.g., NIOSH, 1998). For marine mammals, recent data are available regarding sounds that cause modest TTS (generally < 20 dB decrease in sensitivity) in a few species of odontocetes and pinnipeds. No data exist on exposures that would cause PTS in these taxa, however (see Chapter 2 for detailed discussions). Consequently, the only current option for estimating exposure conditions that would cause PTS-onset in marine mammals is to use the available marine mammal TTS data combined with data from terrestrial mammals on TTS growth rates with increasing acoustic exposure (see the "Criteria for Injury: TTS and PTS" section of Chapter 3).

Behavioral Reactions to Sound

Behavioral responses to sound are highly variable and context-specific (see Wartzok et al., 2004, for a discussion). Some sounds that are audible to animals may elicit no overt behavioral response. This is most common when the sound does not greatly exceed the minimum detectable level and is not increasing or fluctuating (Richardson et al., 1995). Inability to detect an overt response does not necessarily mean that there is no subtle behavioral (or other) effect, however.

When observable reactions do occur, they may include orientation or attraction to a sound source; increased alertness; modification of characteristics of their own sounds; cessation of feeding or social interaction; alteration of movement/diving behavior; temporary or permanent habitat abandonment; and, in severe cases, panic, flight, stampede, or stranding, sometimes resulting in injury or death (e.g., Richardson et al., 1995; Evans & England, 2001; Gordon et al., 2004; Scheifele et al., 2005; Cox et al., 2006; Nowacek et al., 2007). Minor or temporary behavioral effects are often simply evidence that an animal has heard a sound and may not indicate lasting consequence for exposed individuals. For the purposes of setting criteria, the effects of greatest concern are those that may negatively impact reproduction or survival. Ultimately, it is the biological relevance of the reaction in terms of vital parameters that must be determined. In proposing noise exposure criteria, one must clearly and explicitly differentiate trivial effects from those with the potential to affect vital rates. However, it has proven to be exceedingly challenging to distinguish among and rank the various effects and to establish a generally accepted definition of biologically meaningful behavioral disturbance (see NRC, 2005).

Except for naïve individuals, behavioral responses depend critically on the principles of *habituation* and *sensitization*. An animal's exposure history with a particular sound affects whether it is subsequently less likely (habituation) or more likely (sensitization) to respond to a stimulus such as sound exposure. The processes of habituation and sensitization do not necessarily require an association with a particular adverse or benign outcome. Rather, individuals may be innately predisposed to respond to certain stimuli in certain ways. These responses may interact with the processes of habituation and sensitization for subsequent exposure. Where associative learning occurs, individuals link a particular exposure with a known outcome (positive, negative, or neutral) and use that information in guiding

future decisions on whether and how to respond to similar stimuli. The relationship between these two categories of learning (non-associative and associative) can be highly complex, particularly for experienced individuals (see Deecke et al., 2002).

Many contextual variables may be powerful contributors to an animal's perception of and reaction to the acoustic scene. These include the perception of source proximity (nearness), relative movement (encroachment or retreat), and general novelty or familiarity, all of which may affect the type and magnitude of the resulting behavioral response(s). In terms of proximity, the presence of high-frequency components in a sound and the lack of reverberation, both of which are indicative of proximity, may be more relevant acoustic cues of spatial relationship than simply exposure level alone (see P. Miller, 2002). If a source is perceived to be approaching, the response is often stronger. In addition, the activity of the individual and its fidelity to a current location often affect the response.

Thus, in addition to source characteristics, other factors that may be critical in determining behavioral effects include past experience, situational variables, receiver auditory systems, and the extent to which the sound resembles familiar benign or noxious stimuli (e.g., Irvine et al., 1981; NRC, 2005). Animals that fail to exhibit general avoidance when exposed to a certain sound source may still detect the sound but are either habituated to exposure or may display less dramatic behavioral responses (e.g., altering vocal behavior, modifying orientation/movement patterns).

The magnitude of a given behavioral response may not be a direct function of exposure levels or even of the animal's experiential history. If the sound triggers an anti-predator response in the subject (e.g., Irvine et al., 1981; Finley et al., 1990), the response magnitude may reflect the individual's underlying physiological condition, the relative costs in fitness of failing to respond, the availability of alternative refuges, and other factors specific to predator defense (Gill & Sutherland, 2000; Frid & Dill, 2002; Beale & Monaghan, 2004).

For all these reasons, behavioral responses to anthropogenic sounds are highly variable. Meaningful interpretation of behavioral response data (and biologically relevant conservation decisions) must consider not only the relative magnitude and apparent severity of behavioral reactions to human disturbance but also the relevant acoustic, contextual, and ecological variables. In many cases, specific acoustic features of the sound and contextual variables (e.g., proximity, subject experience and motivation, duration, or recurrence of exposure) may be of considerably greater relevance to the behavioral response than simple acoustic variables such as exposure RL. For example, if an anthropogenic sound is perceived as indicating the presence of a predator, it is likely to trigger a strong defensive reaction at relatively low RLs. On the other hand, sounds that resemble conspecific signals may be ignored or induce approach or avoidance, depending upon the context. Further, typically neutral sounds may cause increasing annoyance reactions (such as avoidance) as a function of exposure level. This makes it difficult or impossible to justify basing broad, objective determinations of impact thresholds on RL alone. This is the primary reason why this paper does not propose explicit behavioral disturbance criteria levels for certain sound types. Rather, we collated available data relating acoustic exposure to the severity of observed behavioral response in a form that allows a variety of relationships to be estimated (Chapter 4). When research allows the separation of annoyance from cases where an animal interprets sounds as signals from predators, prey, or conspecifics, it may become possible to classify signals and predict responses more precisely.

Non-Auditory Effects

The auditory system appears to include the organs most susceptible to noise exposure, at least in humans (e.g., Ward, 1997). The limited data on captive marine mammals exposed to various kinds of noise support a similar conclusion, suggesting that TTS-onset occurs at levels which may be below those required for direct non-auditory physiological trauma (but see discussion of deep-diving species below). Noise exposure does have the potential to induce a range of direct or indirect physiological effects on non-auditory structures. These may interact with or cause certain behavioral or auditory effects, or they may occur entirely in the absence of those effects.

Noise exposure may affect the vestibular and neurosensory systems. For instance, in humans, dizziness and vertigo can result from exposure to high levels of noise, a condition known as *nystagmus* (see Oosterveld et al., 1982; Ward, 1997; Halmagyi et al., 2005). Little is known about vestibular functions in marine mammals. There are significant differences in vestibular structures in some marine mammal species compared to most land mammals (Wartzok & Ketten, 1998; Ketten, 2000). In cetaceans in particular, the vestibular components are sufficiently reduced and have such low neural representation that the principal function may be essentially to provide limited gravitational and linear acceleration cues. Pinnipeds by contrast have a well-developed,

more conventional vestibular apparatus that likely provides multiple sensory cues similar to those of most land mammals. Both pinnipeds and cetaceans retain the direct coupling through the vestibule of the vestibular and auditory systems; therefore, it is possible, albeit not known, that marine mammals may be subject to noise-induced effects on vestibular function as has been shown in land mammals and humans. Responses to underwater sound exposures in human divers and other immersed land mammals suggest that vestibular effects are produced from intense underwater sound at some lower frequencies (Steevens et al., 1997). Theoretical effects on the human vestibular system as well as other organs (e.g., lungs) from underwater sound exposures also have been explored through models (Cudahy & Ellison, 2002); however, there are no comparable measurements or models for marine mammals at this point from which to estimate such effects. Data are clearly needed for all major marine mammal taxa to more fully assess potential impacts on non-auditory systems.

Relatively low-level physiological responses include changes in cardiac rate (*bradycardia* or *tachycardia*) and respiratory patterns, which may lead to changes in metabolism. Stress reactions in humans and other vertebrates include various physiological changes to pulmonary, cardiac, metabolic, neuro-endocrine, immune, and reproductive functions (e.g., Hales, 1973; Lee, 1992; Vrijkotte et al., 2000). Studies of noise-induced stress in marine mammals are very limited, but endocrine secretions of glucocorticoids and altered cardiovascular function have been documented in odontocetes exposed to high-level sound (Romano et al., 2004; *cf.* Thomas et al., 1990c). Noise exposure also often leads to changes in surfacing-respiration-dive cycles of cetaceans (e.g., Richardson & Malme, 1993), which may have various physiological effects. Assuming that effects in marine and terrestrial mammals are similar, intermediate physiological responses to stressors (including noise) may accompany avoidance or aggressive behaviors and include single auditory startle responses, the initiation and sustenance of the catecholamine response, and physiological preparation for fight or flight. The most severe physiological responses would include multiple or repeated auditory startle responses, triggering of the hypothalamic-pituitary-adrenal (HPA) axis and associated elevated blood glucocorticoid level, substantially altered metabolism or energy reserves, lowered immune response, diminished reproductive effort, and potential tissue trauma (e.g., Sapolsky et al., 2000). [The issue of stress responses to noise exposure has been discussed recently by Wright et al. (in press).]

Sound at certain frequencies can cause an air-filled space to vibrate at its resonant frequency (acoustic resonance), which may increase the likelihood of mechanical trauma in the adjacent or surrounding tissue. The resonant frequencies of most marine mammal lungs are below the operating frequencies of many anthropogenic sound sources (Finneran, 2003). Further, biological tissues are heavily damped, estimated tissue displacement at resonant frequencies is predicted to be exceedingly small, and lung tissue damage is generally uncommon in acoustic-related marine mammal stranding events. For these reasons, specialists do not regard lung resonance as a likely significant non-auditory effect for marine mammals exposed to anthropogenic noise sources that operate above 100 Hz (U.S. Department of Commerce, 2002). This conclusion might not apply to lower-frequency sources that operate at a particular frequency for a significant duration.

The non-auditory effect now being most actively discussed in marine mammalogy is nitrogen gas bubble growth, resulting in effects similar to decompression sickness in humans. Jepson et al. (2003) and Fernández et al. (2004, 2005) hypothesized that lesions (gas and fat emboli) observed in individual beaked whales found stranded after military sonar exercises were somehow caused by *in vivo* nitrogen bubble formation. Osteonecrosis in sperm whales has further been suggested as a chronic result of nitrogen bubble formation (Moore & Early, 2004).

To date, the gas bubble hypothesis remains untested, and the acoustic causative mechanism for formation of emboli, if any, is unknown. Theoretically, bubble precursors in supersaturated, homogenized tissue may incrementally enlarge during the successive passage of compression and rarefaction portions of acoustic waves that exceed static pressure (rectified diffusion; Crum & Mao, 1996). Alternatively, a single acoustic exposure could activate bubble precursors, allowing them to grow by gradual expansion into bubbles in nitrogen-supersaturated tissue (static diffusion; see Potter, 2004). The diving patterns of some marine mammals increase gas-tissue saturation and potentially could increase the susceptibility of noise-exposed animals to bubble growth via either mechanism (Ridgway & Howard, 1979; Houser et al., 2001b). Nitrogen supersaturation levels for deep-diving species of interest, including beaked whales, are based on theoretical models, however (Houser et al., 2001b). No unequivocal support for either pathway presently exists.

The evidence for bubble formation as a causal mechanism between certain types of acoustic exposure and stranding events remains equivocal. At a minimum, scientific disagreement and/or

complete lack of information exists regarding the following important points: (1) received acoustic exposure conditions for animals involved in stranding events; (2) pathological interpretation of observed lesions in stranded marine mammals (Fernández et al., 2004; Piantadosi & Thalmann, 2004); (3) acoustic exposure conditions required to induce such physiological trauma directly; (4) whether noise exposure may cause behavioral reactions (e.g., atypical diving behavior) that secondarily induce bubble formation and tissue damage (Jepson et al., 2003; Fernández et al., 2005; Zimmer & Tyack, 2007); and (5) the extent that *post mortem* artifacts introduced by decomposition before sampling, handling, freezing, or necropsy procedures affect interpretation of observed lesions. Tests of the gas bubble hypothesis may yield data pertinent to future marine mammal noise exposure criteria, but too little is currently known to establish explicit exposure criteria for this proposed mechanism.

Courtesy: A. Friedlander

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2. Structure of the Noise Exposure Criteria

When *de facto* noise exposure guidelines are used by management agencies, they generally are based on a small number of categories of marine mammals and sound types. Though it would be convenient to have a single exposure criterion for all species and sound sources, such a simplified approach is not supported by available science. However, some categorization of species and sources is warranted based on current information. The many anthropogenic sound sources used in marine environments can be categorized based on certain acoustic and operational features. Similarly, there is great diversity in hearing and in the biological effects of noise among marine mammals, but current knowledge supports some functional and/or phylogenetic groupings.

It is also neither possible nor desirable to derive distinct exposure criteria for every species and sound source. Important generalizations across taxa would be missed even if resources and time were adequate to study each species and exposure condition. Further, it is impractical to apply numerous, species-specific criteria when predicting and/or attempting to mitigate effects.

A standard scientific approach in such situations is to categorize animals based on functional characteristics and sound sources based on physical similarities, and to summarize the information in a matrix format. We subdivide cetaceans and pinnipeds into five functional hearing categories based on the frequencies they hear. Other methods of categorization are, of course, possible. For instance, Verboom (2002) relied heavily on direct measurements of noise impacts on hearing to quantify the effects of noise exposure on marine mammals. Some of his proposed criteria are comparable with those presented here. The present effort makes broader use of laboratory and field behavioral and audiometric data, additional recent data, and extrapolations from terrestrial mammals not used by Verboom. We divide sound sources into three types according to acoustic characteristics defined at the source. Note that at a distance, a sound may have significantly different features; categorizing sounds based on source characteristics is a precautionary and pragmatic approach (as is described in the next section). The justifications for and assumptions underlying our categorization of functional hearing groups and sound types are described here. The number of subdivisions in future noise exposure criteria will likely increase as more supporting data are acquired.

The format of the recommended marine mammal noise exposure criteria is thus a matrix of 15 "cells" that systematically considers three sound types (see next section) and five functional marine mammal hearing groups (see the "Marine Mammal Functional Hearing Groups" section of this chapter). Within each of those 15 cells, we consider two general acoustic metrics (see the "Exposure Criteria Metrics" section) and two levels of exposure effect ("Levels of Noise Effect: Injury and Behavorial Disturbance" section of this chapter). Sixty possible criteria result (i.e., 3 sound types × 5 marine mammal groups × 2 metrics × 2 impact levels), although fewer than 60 are reported due to data limitations. Whereas sound types are defined by source features, criteria values represent levels received by individual marine mammals.

Sound Types

Three sound types are used: (1) a single pulse, (2) multiple pulses, and (3) nonpulses. The separation between pulses and nonpulses is supported by data on auditory fatigue and acoustic trauma in terrestrial mammals (e.g., Dunn et al., 1991; Hamernik et al., 1993) and is generally consistent with the sound types distinguished for damage risk criteria in humans (e.g., U.S. DoD, 1997; NIOSH, 1998).

Pulses and nonpulses are distinguished by numerous definitions and mathematical distinctions (e.g., Burdic, 1984). The empirical distinction used here is based on a measurement procedure using several temporal weightings. Various exponential time-weighting functions applied in measuring pulse and nonpulse sounds may yield different measured received levels (RLs) (see Harris, 1998). Most sound level meters (SLM) provide options for applying either a "slow" or "fast" time constant (1,000 or 125 ms, respectively) for measuring nonpulses or an impulse time constant (35 ms) appropriate for measuring pulses. For a sound pulse, the slow or fast SLM settings result in lower sound pressure level (SPL) measurements than those obtained using the impulse setting. Each of these time constants is selected based on properties of the human auditory system. These may be at least generally relevant for other mammalian auditory systems, although further empirical data on temporal resolution in marine mammals are needed (see Chapter 5, "Research Recommendations").

Harris (1998) proposed a measurement-based distinction of pulses and nonpulses that is adopted here in defining sound types. Specifically, a ≥ 3-dB difference in measurements between continuous and impulse SLM settings indicates that a sound is a pulse; a < 3-dB difference indicates that a sound is a nonpulse. We note the interim nature of this distinction for underwater signals and the need for an explicit distinction and measurement standard such as exists for aerial signals (ANSI, 1986).

Harris's (1998) definitions assumed use of A-weighting as do most human-oriented definitions of acoustical measurements; however, different frequency-weighting functions should be used for various animal taxa (as discussed below). Leaving that question aside temporarily, it is instructive to compare the impulse equivalent-continuous sound level (L_{IeqT}) for a sound that increases in level with the corresponding equivalent-continuous level (L_{eqT}). Here, L_{IeqT} has an impulse integration time of 35 ms and L_{eqT}, defined as sound exposure divided by T, is expressed as a level. As an example, suppose that a source is examined over a 2-s period (T = 2 s). The highest L_{AIeq2s} ("A" here denotes A-weighting) during this period is 75.2 dB, and the highest L_{ALeq2s} is 65.1 dB. The difference of 10.1 dB is greater than the 3-dB criterion given by Harris (1998); therefore, the sound is considered to be a pulse.

The distinction between pulses and nonpulses is not always clear in practice. For instance, certain signals (e.g., acoustic deterrent and harassment devices) have characteristics of both pulses and nonpulses. Also, certain sound sources (e.g., seismic airguns and pile driving) may produce pulses at the source but, through various propagation effects, may meet the nonpulse definition at greater distances (e.g., Greene & Richardson, 1988). This means that a given sound source might be subject to different exposure criteria, depending on the distance to the receiver and intervening propagation variables. While this is certainly realistic for many real-world exposures, measurements at the animal are often not practical. Changes in sound characteristics with distance generally result in exposures becoming less physiologically damaging with increasing distance because sharp transient peaks become less prominent. Therefore, these criteria use a precautionary approach and classify sound types based on acoustic characteristics at the source. Additional empirical measurements are needed to advance our understanding of sound type classification as a function of source, range, and environmental variables. We emphasize that the use of source parameters to classify sound types does not negate our decision to recommend exposure criterion levels relative to RLs at the animal.

Treating pulses and nonpulses as discrete sound types is justified by data on mammals in general and several cetacean species in particular (Dunn et al., 1991; Hamernik et al., 1993; also see the "Effects of Noise on Hearing in Marine Mammals TTS Data" section in Chapter 3). Mammalian hearing is most readily damaged by transient sounds with rapid rise-time, high peak pressures, and sustained duration relative to rise-time (for humans: Thiery & Meyer-Bisch, 1988; for chinchillas [*Chinchilla lanigera*], Dunn et al., 1991). Consistent with these results, those odontocetes tested thus far have been shown to experience TTS-onset at lower respective exposure levels if the sound is a pulse rather than a nonpulse (Finneran et al., 2002b, 2005a).

Mammals are also apparently at greater risk from rapidly repeated transients and those with high impulse amplitude *kurtosis* (Erdreich, 1986). Hamernik et al. (1993, 2003) argued that the distinction between exposures with relatively high and low "peakedness" is to some extent an oversimplification. Highly variable threshold shifts can result from exposures of variable peakedness but comparable overall levels, depending on a host of factors. Hamernik et al. (1993, 2003) also noted that peak pressure levels sufficient to exceed mechanical limits of the cochlea, and thus more likely to induce acoustic trauma, tend to be more typical of pulses than nonpulses.

The present criteria also categorize sound types based on repetition. For mammals, single and multiple noise exposures at various levels and durations generally differ in their potential to induce auditory fatigue or trauma. This results principally from the temporal interaction between exposure and recovery periods (e.g., Kryter, 1994) and differences in received total acoustic energy. Further, multiple exposures may increase the likelihood of behavioral responses because of increased probability of detection and the (generally) greater biological significance of continued exposure as opposed to a single, transient event (although see discussion of habituation in the "Responses to Sound" section of Chapter 1).

Single exposures are considered here as discrete acoustic events in which received sound levels exceed ambient noise in at least some portion of the frequency band of functional marine mammal hearing once in a 24-h period; multi-path receptions of a single exposure are not considered multiple exposures. Multiple exposures are considered to be acoustic events causing RLs to exceed ambient noise within the functional bandwidth more than once, with an intervening quiet period not exceeding 24 h. If the exposure event is interrupted, even briefly (other than as a result of the animal's own action—e.g., breaching), it is considered a multiple exposure.

Exposures should be categorized as either pulsed or nonpulsed sounds as described above. Single and multiple exposures to either pulse or nonpulse sounds (or both) are possible. Examples of single pulses and single nonpulses are sounds from a single firing of an airgun or a single vessel passage, respectively.

Multiple pulse or multiple nonpulse sounds are more difficult to delineate, given the diversity and complexity of sound sources. A series exclusively consisting of two or more nonpulses would clearly be a multiple nonpulse exposure (e.g., multiple vessel passages). A multiple pulse exposure would similarly be described as a series exclusively containing pulses (e.g., repeated pile strikes) or a combination of pulses and nonpulses (e.g., the combined vessel noise and airgun transmissions of a seismic vessel). One justification for treating combined pulses and nonpulses as pulses is that the proposed exposure criteria for injury are more precautionary (lower) in the case of pulses than for nonpulses. Specific consideration should be given, on a case-by-case basis, as to whether such a distinction would necessarily be the more precautionary. For instance, if a compound exposure included relatively high-level nonpulses as well as relatively low-level pulses, the more appropriate and protective distinction might be to classify it as a nonpulse exposure.

The proposed exposure criteria for injury from single and multiple exposures to both sound types are numerically identical (Chapter 3). This is another precautionary decision, arising from the fact that no marine mammal data were available regarding the effects of inter-exposure interval on recovery from auditory effects (e.g., TTS). A summation procedure is applied to quantify the fatiguing effects of multiple exposures with an equivalent SEL value (Chapter 1; also Appendix A, eq. 5). The SEL metric takes account of the pressure waveform and duration of either single or multiple sound events; it represents cumulative received energy. This approach effectively negates the need for numerically different injury criteria for single and multiple exposures at the expense of neglecting assumed, but as-yet poorly understood recovery phenomena during intervals between exposures. This is a precautionary approach, pending availability of data on acoustic recovery by marine mammals during intervals between exposures.

When considering behavioral responses, single and multiple nonpulse exposures are considered as a single category. Insufficient information exists to assess the use of SEL as a relevant metric in the context of marine mammal behavioral disturbance for anything other than a single pulse exposure. Future noise exposure criteria for behavioral disturbance may distinguish SPL and SEL exposure criteria for additional conditions, but for most sound types (the exception being single pulses), the available data are best assessed in relation to SPL (discussed in detail in Chapter 4). Consequently, the structure of the exposure criteria matrix includes a categorical distinction between single and multiple pulses given that numerical SEL thresholds are recommended for a single pulse, but not for multiple pulses. No such distinction is made for nonpulses where the available data do not (at least currently) support differential behavioral criteria for single vs multiple exposures.

Thus, the current state of scientific knowledge regarding mammalian hearing and various noise impacts supports three distinct sound types as relevant for marine mammal noise exposure criteria: (1) single pulse, (2) multiple pulses, and (3) nonpulses. Examples of sound sources belonging in each of these categories (based on characteristics of the sound emitted at the source) are given in Table 1. A simplistic measurement procedure using source characteristics (the 3-dB distinction based on Harris, 1998, described above) is used here to distinguish a pulse from a nonpulse, while the simple definitions above distinguish single and multiple exposures.

Table 1. Sound types, acoustic characteristics, and selected examples of anthropogenic sound sources; note sound types are based on characteristics measured at the source. In certain conditions, sounds classified as pulses at the source may lack these characteristics for distant receivers.

Sound type	Acoustic characteristics (at source)	Examples
Single pulse	Single acoustic event; > 3-dB difference between received level using impulse vs equivalent continuous time constant	Single explosion; sonic boom; single airgun, watergun, pile strike, or sparker pulse; single ping of certain sonars, depth sounders, and pingers
Multiple pulses	Multiple discrete acoustic events within 24 h; > 3-dB difference between received level using impulse vs equivalent continuous time constant	Serial explosions; sequential airgun, watergun, pile strikes, or sparker pulses; certain active sonar (IMAPS); some depth sounder signals
Nonpulses	Single or multiple discrete acoustic events within 24 h; < 3-dB difference between received level using impulse vs equivalent continuous time constant	Vessel/aircraft passes; drilling; many construction or other industrial operations; certain sonar systems (LFA, tactical mid-frequency); acoustic harassment/deterrent devices; acoustic tomography sources (ATOC); some depth sounder signals

Marine Mammal Functional Hearing Groups

Species of cetaceans and pinnipeds were assigned to one of five functional hearing groups based on behavioral psychophysics, evoked potential audiometry, auditory morphology, and (for pinnipeds) the medium in which they listen. Cetaceans and pinnipeds are broadly separable based on phylogenetic and functional differences (Reynolds & Rommel, 1999). Cetaceans were further subdivided according to differences in their measured or estimated hearing characteristics and not necessarily according to their phylogeny (as in Wartzok & Ketten, 1999). Pinnipeds are considered a single group, but as amphibious mammals, their hearing differs in air and in water (Kastak & Schusterman, 1998); separate criteria were required for each medium. The taxa in each functional hearing group (based on Rice, 1998) are given in Table 2.

Marine Mammal Hearing

All marine mammals evolved from terrestrial, air-adapted ancestors (Domning et al., 1982; Barnes et al., 1985) and, at least in part, retain the nominal mammalian tripartite peripheral auditory system (i.e., external auditory meatus, air-filled middle ear, and spiral-shaped cochlea). Most of the mechanisms of mammalian hearing are also conserved such as the basic lever structure of the ossicles and the tonotopic organization of the hair cells along the inner ear's basilar membrane.

However, marine mammal auditory systems differ in having some adaptations that seem to be related to pressure, hydrodynamics, and sound reception in water (see Wartzok & Ketten, 1999). For instance, the pinna has been reduced or eliminated in most species, owing to hydrodynamic adaptations. Tissue modifications may enable the reduction or elimination of gas spaces in the middle ear of some marine mammals. Consequently, bone conduction, rather than the conventional ossicular chain, may be an additional (or primary) sound transmission path to the cochlea (e.g., Repenning, 1972; Au, 1993). There are important differences in these adaptations within and between marine mammal taxa.

Knowledge of marine mammal hearing varies widely among groups, but for most species it is quite limited compared to knowledge of terrestrial mammal hearing. Because of the sheer size, limited and disproportionate availability in captive

Table 2. Functional marine mammal hearing groups, auditory bandwidth (estimated lower to upper frequency hearing cut-off), genera represented in each group, and group-specific (M) frequency-weightings

Functional hearing group	Estimated auditory bandwidth	Genera represented (Number species/subspecies)	Frequency-weighting network
Low-frequency cetaceans	7 Hz to 22 kHz	*Balaena, Caperea, Eschrichtius, Megaptera, Balaenoptera* (13 species/subspecies)	M_{lf} (lf: low-frequency cetacean)
Mid-frequency cetaceans	150 Hz to 160 kHz	*Steno, Sousa, Sotalia, Tursiops, Stenella, Delphinus, Lagenodelphis, Lagenorhynchus, Lissodelphis, Grampus, Peponocephala, Feresa, Pseudorca, Orcinus, Globicephala, Orcaella, Physeter, Delphinapterus, Monodon, Ziphius, Berardius, Tasmacetus, Hyperoodon, Mesoplodon* (57 species/subspecies)	M_{mf} (mf: mid-frequency cetaceans)
High-frequency cetaceans	200 Hz to 180 kHz	*Phocoena, Neophocaena, Phocoenoides, Platanista, Inia, Kogia, Lipotes, Pontoporia, Cephalorhynchus* (20 species/subspecies)	M_{hf} (hf: high-frequency cetaceans)
Pinnipeds in water	75 Hz to 75 kHz	*Arctocephalus, Callorhinus, Zalophus, Eumetopias, Neophoca, Phocarctos, Otaria, Erignathus, Phoca, Pusa, Halichoerus, Histriophoca, Pagophilus, Cystophora, Monachus, Mirounga, Leptonychotes, Ommatophoca, Lobodon, Hydrurga,* and *Odobenus* (41 species/subspecies)	M_{pw} (pw: pinnipeds in water)
Pinnipeds in air	75 Hz to 30 kHz	Same species as pinnipeds in water (41 species/subspecies)	M_{pa} (pa: pinnipeds in air)

settings, and, for many species and jurisdictions, the protected status of marine mammals, there are limitations in obtaining hearing data for many species. Behavioral or electrophysiological audiograms exist for fewer than 20 marine mammal species (of ~128 species and subspecies; Rice, 1998). By combining these data with comparative anatomy, modeling, and response measured in ear tissues from species that are difficult to study, however, it is possible to describe the frequency sensitivity and critical adaptations for underwater hearing in each of the five functional hearing groups of marine mammals considered here.

Low-frequency cetaceans consist of 13 species and subspecies of mysticete (baleen) whales in five genera (based on Rice, 1998; see Table 2). No direct measurements of hearing exist for these animals, and theories regarding their sensory capabilities are consequently speculative (for a detailed assessment by species using the limited available information, see Erbe, 2002). They are too large to maintain in the laboratory for psychophysical testing. The limited evoked potential measurements on animals of this size have not yet yielded hearing thresholds (Ridgway & Carder, 2001), but technological advances may soon enable evoked potential audiometry on relatively small and/or young mysticetes. In these species, hearing sensitivity has been estimated from behavioral responses (or lack thereof) to sounds at various frequencies, vocalization frequencies they use most, body size, ambient noise levels at the frequencies they use most, and cochlear morphometry (Richardson et al., 1995; Wartzok & Ketten, 1999; Houser et al., 2001a; Erbe, 2002; Clark & Ellison, 2004). Until better information is available regarding the relationship between auditory sensitivity and marine environmental noise, the sensitivity of mysticetes cannot be easily inferred from the acoustic environment.

The combined information strongly suggests that mysticetes are likely most sensitive to sound from perhaps tens of Hz to ~10 kHz. However, recent data indicated that humpback whales (*Megaptera novaeangliae*) produce some signals with harmonics extending above 24 kHz (Au et al., 2006). These harmonics have considerably lower levels than occur at lower frequencies, and their presence does not necessarily indicate they are audible to the whales. Nonetheless, some high-frequency energy is present. [Additionally, some recent anatomical modeling work by Ketten et al. (2007) suggested that some mysticetes may have functional hearing capabilities at frequencies as high as 30 kHz.] While we do not include these recent results at this time, we note their presence and the possibility that the upper frequency limit of the M-weighting function for mysticetes may need to be revisited based on emerging knowledge. At present, we estimate the lower and upper frequencies for functional hearing in mysticetes, collectively, to be 7 Hz and 22 kHz (Ketten et al., 2007).

Mid- and high-frequency cetaceans are all odontocetes (toothed whales). Unlike the mysticetes, all odontocete cetaceans appear to have highly advanced echolocation (biosonar) systems that use intermediate to very high frequencies (tens of kHz to 100+ kHz: see Au, 1993; Richardson et al., 1995; Wartzok & Ketten, 1999). They also produce social sounds in a lower-frequency band, including generally low to intermediate frequencies (1 kHz to tens of kHz). Consequently, their functional hearing would be expected to cover a wider absolute frequency range than is assumed for mysticetes or has been demonstrated for pinnipeds (discussed below). This has been experimentally confirmed in the odontocete species whose hearing has been measured (discussed below); however, their best hearing sensitivity typically occurs at or near the frequency where echolocation signals are strongest. Based on the differential characteristics of echolocation signals in two groups of odontocetes (see Au, 1993) and on the hearing data described below, odontocetes were divided into mid- and high-frequency functional groups (as seen generally in Wartzok & Ketten, 1999).

Mid-frequency cetaceans include 32 species and subspecies of "dolphins," six species of larger toothed whales, and 19 species of beaked and bottlenose whales (see Table 2). "Functional" hearing in this group was estimated to occur over a wide range of low to very high frequencies. Based on the combined available data, mid-frequency species are estimated to have lower and upper frequency "limits" of nominal hearing at approximately 150 Hz and 160 kHz, respectively. As for the other hearing groups, there is variability within and among species, intense signals below and above the stated bounds may be weakly detectable, and there is a progressive rather than instantaneous reduction in hearing sensitivity near these limits. Mid-frequency cetaceans generally do not appear well-adapted to detect or to discriminate signals outside this frequency band, however. The scarcity (and variability) of empirical data precludes a finer subdivision of this relatively diverse and large group of marine mammals, though it is acknowledged that some mid-frequency species likely have a narrower functional hearing band than the range given above.

Behavioral hearing data are available for the following mid-frequency cetacean species: bottlenose dolphin (*Tursiops truncatus*: Johnson, 1967; Ljungblad et al., 1982; Finneran et al., 2005a), beluga (*Delphinapterus leucas*: White

et al., 1978; Awbrey et al., 1988; Johnson, 1992; Ridgway et al., 2001; Finneran et al., 2005b), killer whale (*Orcinus orca*: Hall & Johnson, 1972; Szymanski et al., 1999), false killer whale (*Pseudorca crassidens*: Thomas et al., 1988, 1990a; Au et al., 1997), Risso's dolphin (*Grampus griseus*: Nachtigall et al., 1995; Au et al., 1997); and Pacific white-sided dolphin (*Lagenorhynchus obliquidens*: Tremel et al., 1998).

Audiograms derived using auditory evoked potential (AEP) methodology (Supin et al., 2001) have been obtained for a number of cetacean species. Specific AEP techniques, which involve measuring electrophysiological responses to sound, include those measuring transient evoked responses, such as the auditory brainstem response (ABR) or mid-latency response, and those measuring steady-state evoked responses such as the envelope following response (EFR) or auditory steady-state response (ASSR). Mid-frequency cetacean species tested include the bottlenose dolphin (Bullock et al., 1968; Seeley et al., 1976; Popov & Supin, 1990; Houser & Finneran, 2006b; Finneran et al., 2007a; Hernandez et al., 2007; Popov et al., 2007), killer whale (Szymanski et al., 1999), beluga (Popov & Supin, 1990; Klishin et al., 2000), common dolphin (*Delphinus delphis*: Popov & Klishin, 1998), Risso's dolphin (Dolphin, 2000; Nachtigall et al., 2005, 2007), tucuxi dolphin (*Sotalia fluviatilis*: Popov & Supin, 1990), striped dolphin (*Stenella coeruleoalba*: Kastelein et al., 2003), Pacific white-sided dolphin (Au et al., 2007), false killer whale (Supin et al., 2003), and Gervais' beaked whale (Cook et al., 2006). Additionally, Yuen et al. (2005) conducted a comparative study of behavioral and AEP thresholds for the false killer whale, and Finneran & Houser (2006), Houser & Finneran (2006a), and Finneran et al. (2007b) have compared behavioral and AEP thresholds in multiple bottlenose dolphins.

The high-frequency cetaceans include eight species and subspecies of true porpoises, six species and subspecies of river dolphins plus the franciscana, *Kogia*, and four species of cephalorhynchids (see Table 2). "Functional" hearing in this group was estimated to occur between 200 Hz and 180 kHz. Behavioral audiograms are available for the following high-frequency cetacean species: harbor porpoise (*Phocoena phocoena*: Andersen, 1970; Kastelein et al., 2002a), Chinese river dolphin (*Lipotes vexillifer*: Wang et al., 1992), and Amazon river dolphin (*Inia geoffrensis*: Jacobs & Hall, 1972). Audiograms using AEP methodology have been obtained for three species: harbor porpoise (Popov et al., 1986, 2006; Beedholm & Miller, 2007; Lucke et al., 2007b); finless porpoise (*Neophocaena phocaenoides*: Popov et al., 2006); and Amazon river dolphin (Popov & Supin, 1990).

The pinnipeds include 16 species and subspecies of sea lions and fur seals (otariids), 23 species and subspecies of true seals (phocids), and two subspecies of walrus (odobenids). Pinnipeds produce a wide range of social signals, most occurring at relatively low frequencies. They lack the highly-specialized active biosonar systems of odontocete cetaceans, possibly as a result of their amphibious lifestyle (see Schusterman et al., 2000). Because of this aspect of their life history, pinnipeds communicate acoustically in air and water, have significantly different hearing capabilities in the two media, and may be subject to both aerial and underwater noise exposure (Schusterman, 1981; Kastak & Schusterman, 1998, 1999). These differences necessitate separate noise exposure criteria for pinnipeds in each medium.

For pinnipeds in water, behavioral measures of hearing are available for the northern fur seal (*Callorhinus ursinus*: Moore & Schusterman, 1987; Babushina et al., 1991), California sea lion (*Zalophus californianus*: Schusterman et al., 1972; Moore & Schusterman, 1987; Kastak & Schusterman, 1998, 2002; Southall et al., 2004), northern elephant seal (*Mirounga angustirostris*: Kastak & Schusterman, 1998, 1999; Southall et al., 2004), Hawaiian monk seal (*Monachus schauinslandi*: Thomas et al., 1990b), harp seal (*Pagophilus groenlandicus*: Terhune & Ronald, 1972), ringed seal (*Phoca hispida*: Terhune & Ronald, 1975), harbor seal (Møhl, 1967, 1968; Terhune & Turnbull, 1995; Kastak & Schusterman, 1995, 1998; Southall et al., 2004), and walrus (*Odobenus rosmarus*: Kastelein et al., 2002b). Ridgway & Joyce (1975) measured the gray seal's (*Halichoerus grypus*) underwater hearing using evoked potential audiometry.

For pinnipeds in air, behavioral measures of hearing are available for the northern fur seal (Moore & Schusterman, 1987; Babushina et al., 1991), California sea lion (Schusterman, 1974; Kastak & Schusterman, 1998; Kastak et al., 2004b), northern elephant seal (Kastak & Schusterman, 1998, 1999; Kastak et al., 2004b), harp seal (Terhune & Ronald, 1971), and harbor seal (Møhl, 1968; Kastak & Schusterman, 1998; Kastak et al., 2004b). Aerial hearing in pinnipeds has also been measured using evoked potential audiometry in the gray seal (Ridgway & Joyce, 1975), California sea lion (Bullock et al., 1971; Ridgway & Joyce, 1975; Mulsow & Reichmuth, 2007; Reichmuth et al., 2007), harbor seal (Thorson et al., 1998; Wolski et al., 2003; Mulsow & Reichmuth, 2007; Reichmuth et al., 2007), and northern elephant seal (Houser et al., 2007; Mulsow & Reichmuth, 2007; Reichmuth et al., 2007).

The combined results of these studies indicate that pinnipeds are sensitive to a broader range of sound frequencies in water than in air. The data further suggest differences in the functional hearing range among otariids, phocids, and odobenids, especially under water (Kastak & Schusterman, 1998; Kastelein et al., 2002b). For these proposed noise exposure criteria, however, pinnipeds are considered a single functional hearing group because the data are too limited, both in terms of absolute hearing data and TTS measurements (see "The Effects of Noise on Hearing in Marine Mammals: TTS Data" section in Chapter 3), to support finer subdivisions. We estimate that pinnipeds have "functional" underwater hearing between 75 Hz and 75 kHz and "functional" aerial hearing between 75 Hz and 30 kHz. These ranges are essentially based on data for phocid seals, which have the broadest auditory bandwidths of the pinnipeds. This approach results in a precautionary functional bandwidth for estimating frequency-weighting functions (below) and noise impacts on pinnipeds.

In summary, based on current knowledge of functional hearing in marine mammals, five distinct, functional hearing categories were defined: (1) low-frequency cetaceans (i.e., mysticetes), (2) mid-frequency cetaceans (i.e., most odontocetes), (3) high-frequency cetaceans (i.e., porpoises, river dolphins, pygmy sperm whale, and *Cephalorhynchus*), (4) pinnipeds in water, and (5) pinnipeds in air. The genera in each group, and the estimated lower and upper frequency hearing "limits," are shown in Table 2. Because the five functional hearing groups of marine mammals differ in hearing bandwidth, each may be affected differently by identical noise exposures. Therefore, frequency-weighting functions are required to develop marine mammal noise exposure criteria.

Frequency-Weighting Functions

As a general statement, animals do not hear equally well at all frequencies within their functional hearing range. Frequency weighting is a method of quantitatively compensating for the differential frequency response of sensory systems. Generalized frequency-weighting functions were derived for each functional hearing group of marine mammals using principles from human frequency-weighting paradigms, with adjustments for the different hearing bandwidths of the various marine mammal groups.

For humans, substantial improvement in dose-response models is obtained by filtering noise through equal-loudness functions, particularly the 40-phon, equal-loudness function ("A-weighting") and the 100-phon function ("C-weighting"). These frequency-weighting functions take into account both the frequency bandwidth of human hearing and loudness perception. For use as frequency filters, the functions are inverted; normalized to 0 dB in the frequency range of best hearing (specifically at 1,000 Hz for humans); and idealized for implementation in hearing aids, sound level meters, and other measurement devices.

At minimum, metrics used for animals should eliminate inaudible frequencies both below and above the range of functional hearing. The "absolute" auditory threshold function (audiogram) has been suggested as a frequency-weighting function for marine species exposed to underwater sound (e.g., Malme et al., 1989; Thorson et al., 1998; Heathershaw et al. 2001; Nedwell et al., 2007) as well as for terrestrial animals (Delaney et al., 1999; Bjork et al., 2000). However, the auditory threshold function does not characterize the flattening of equal-loudness perception with the increasing stimulus level that has been demonstrated in humans (Fletcher & Munson, 1933). Acoustic injury would only be expected to occur at levels far above the detection threshold—that is, levels for which the flattening effect would be expected. Consequently, it is unclear how useful or appropriate the auditory threshold function is in deriving frequency-weighting filters in marine mammals for which psychophysical equal-loudness measurements are generally unavailable (although see preliminary measurements by Ridgway & Carder, 2000). Further, the limited TTS data for cetaceans exposed to tones at different frequencies (discussed below) suggest that an audiogram-based frequency-weighting function would produce too much filtering at lower frequencies (i.e., the weighting function for hearing effects should be flatter than the inverted audiogram procedure would indicate).

Therefore, a precautionary procedure was used to derive frequency-specific, marine mammal weighting functions. Each was based on an algorithm that requires only the estimated (as ~80 dB above best hearing sensitivity) lower and upper frequencies of functional hearing as given in the above description of each marine mammal group and in Table 2. The resulting functions were designed to reasonably represent the bandwidth where acoustic exposures can have auditory effects and were designed to be most accurate for describing the adverse effects of high-amplitude noise where loudness functions are expected to flatten significantly. The weighting functions (designated "M" for marine mammal) are analogous to the C-weighting function for humans, which is commonly used in measuring high-amplitude sounds. In the general absence of empirical data, however, the upper and lower frequency roll-offs of the

M-weighting functions are symmetrical, whereas C-weighting admits more energy at the lower than at the upper frequency limits (ANSI, 2001).

The M-weighting functions assume a logarithmic reduction in auditory sensitivity outside of the range of best hearing sensitivity, with the function being 6 dB down from peak sensitivity at the lower and upper frequency "limits." Auditory detection thresholds at these "limits" (see above discussion of lower and upper frequency "cut-offs") can be ≥ 80 dB higher (less sensitive) than those at the frequencies of best hearing sensitivity. Consequently, these frequency filters are much "flatter" than audiograms and probably quite precautionary even considering the expected flattening of equal-loudness contours at high exposure levels. The M-weighting functions are also precautionary in that regions of best hearing sensitivity for most species are likely considerably narrower than the M-weighting functions (designed for the overall marine mammal group) would suggest. The general expression for M-weighting (M[f]), using the estimated lower and upper "functional" hearing limits (f_{low} and f_{high}) for each of the five functional marine mammal hearing groups, is given in Appendix A (eq. 7 & 8). These frequency-weighting functions are identified in Table 2, and each is depicted graphically in Figure 1.

The M-weighting functions de-emphasize frequencies that are near the lower and upper frequency ends of the estimated hearing range as indicated by negative relative values (Figure 1). This de-emphasis is appropriate because, to have a given auditory effect, sound at these frequencies must have higher absolute amplitude than sound in the region of best hearing sensitivity. As a corollary, sound at a given level will have less effect if it is near or (especially) beyond the lower or upper bounds of the functional hearing range than if it is well within that frequency range. It is important to note the incremental nature of the frequency-weighting functions, which approximate the gradual reduction in auditory effect at frequencies outside the range of greatest sensitivity.

Use of such M-frequency-weighting functions is superior to flat weighting across all frequencies because it accounts for known or estimated differences in the frequency response characteristics for each functional hearing group. At least in the context of injury criteria, it is superior to frequency-weighting via the inverse-audiogram method as it takes into account the expected "flattening" of equal-loudness curves at the high exposure levels where TTS and PTS are expected. It is also superior to a "boxcar-type" step function because it more closely approximates the gradual roll-off of sensitivity below and above the range of optimum sensitivity. Furthermore, each of the recommended "shallow" frequency-weighting functions includes, within its relatively flat portion, the full audible range for each species for which auditory data are available. In other words, none of the species included within each functional hearing group has been shown or is expected to have any portion of its best hearing sensitivity outside the flat portion of the relevant frequency-weighting function. Thus, the functions are quite precautionary, which is appropriate given that data are limited or lacking for most species.

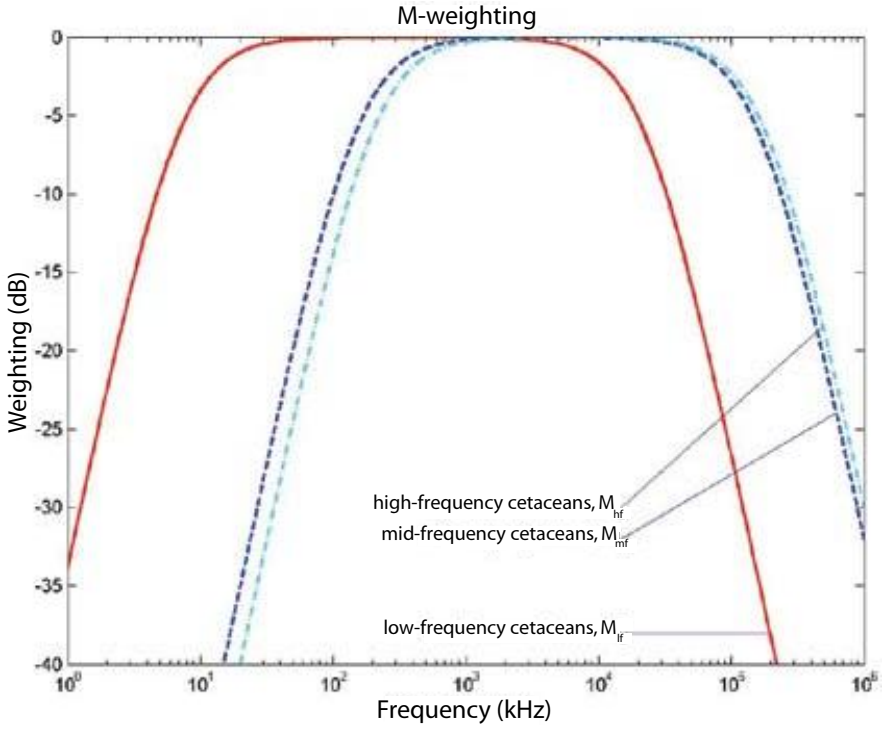
Exposure Criteria Metrics

Many acoustic metrics (e.g., RMS or peak SPL, SEL, kurtosis) could be considered in relation to noise impacts on animals. It is impossible to predict unequivocally which one is best associated with the likelihood of injury or significant behavioral disturbance across all taxa because of species differences and the fact that real-world sound exposures contain many widely differing temporal patterns and pressure signatures. To account for such differences and to allow for current scientific understanding of tissue injury from noise exposure, the proposed injury criteria incorporate a dual-criteria approach based on both peak pressure and energy. For an exposed individual, whichever criterion is exceeded first (i.e., the more precautionary of the two measures) is used as the operative injury criterion. Similarly, a dual-criterion approach (peak sound pressure and energy) is also proposed for behavioral disturbance from a single pulse.

The pressure criteria for injury are defined as those peak SPLs above which tissue injury is predicted to occur, irrespective of exposure duration. Any single exposure at or above this peak pressure is considered to cause tissue injury, regardless of the SPL or SEL of the entire exposure. For each marine mammal group, the recommended pressure-based injury criteria are the same for all sound types and are based on the criterion for a single pulse. This is a precautionary procedure; pressure criteria based on TTS data for nonpulses would yield much higher estimates of the exposure necessary for PTS-onset. By proposing, for all cases, pressure criteria appropriate to a single pulse, we protect against the possibility that, for some sound sources, one or more intense pulses may occasionally be embedded in nonpulse sounds.

For exposures lacking intense peak pressure components, available data indicate that measurements integrating instantaneous pressure squared over the duration of sound exposure are well correlated with the probability of TTS-onset and tissue injury. Consequently, for exposures other than those containing intense peak pressure transients, SEL is the (or at least one of the) appropriate

(A)



(B)

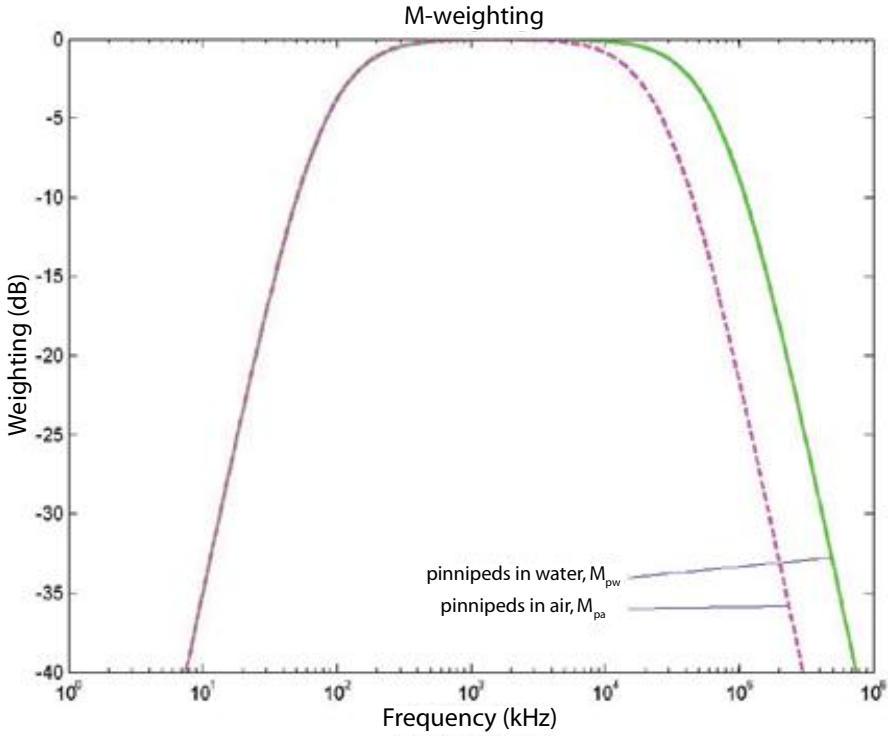


Figure 1. The M-weighting functions for (A) low-, mid-, and high-frequency cetaceans, as well as for (B) pinnipeds in water and air.

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.

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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 ≥ 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., ≥ 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 μPa (peak-to-peak); SEL: 179 dB re: 1 μPa^2-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_2 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 μPa (peak-to-peak); SEL: 186 dB re: 1 μPa^2-s). Thresholds returned to within ± 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 μPa (peak-to-peak); SEL: 188 dB re: 1 μPa^2-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 μPa (SEL: 192 to 201 dB re: 1 μPa^2-s). The mean exposure SPL for TTS-onset was 195 dB re: 1 μPa (195 dB re: 1 μPa^2-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 μPa (193 dB re: 1 μPa^2-s). Data at 75 kHz were inconclusive: one dolphin exhibited a TTS after exposure at 182 dB SPL re: 1 μPa (182 dB re: 1 μPa^2-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 μPa (193 dB re: 1 μPa^2-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 μPa^2-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 ≤ 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 μPa and durations up to 128 s. The maximum exposure SEL was 217 dB re 1 μPa^2-s, which produced a TTS_4 of ~23 dB. All thresholds recovered to baseline values within 24 h, most within 30 min. The growth of TTS_4 with increasing exposure SEL was ~1 dB TTS per dB SEL for TTS_4 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 μPa). The third featured three 20 kHz, 16-s exposures separated by 11 and 12 min, with a mean SPL of 193 dB re 1 μPa (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 μPa; SEL: ~212 to 214 dB re: 1 μPa^2-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 μPa (SEL: ~198 to 200 dB re: 1 μPa^2-s and 204 to 206 dB re: 1 μPa^2-s, respectively).

Using AEP methods, Nachtigall et al. (2004) found TTS_5 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 μPa; SEL: ~193 to 195 dB re: 1 μPa^2-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 (≥ 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 μPa (SEL: 183 dB re: 1 μPa^2-s). Under the same test conditions, a California sea lion showed TTS-onset at 174 dB re: 1 μPa (SEL: 206 dB re: 1 μPa^2-s), and a northern elephant seal experienced TTS-onset at 172 dB re: 1 μPa (SEL: 204 dB re: 1 μPa^2-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 μPa (peak-to-peak) (SEL: 163 dB re: 1 μPa^2-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 μPa (peak) (SEL: 129 dB re: $[20\ \mu Pa]^2$-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 μPa (SEL: 131 dB re: $[20\ \mu Pa]^2$-s). Onset of TTS was identified in the California sea lion at 122 dB re: 20 μPa (SEL: 154 dB re: $[20\ \mu Pa]^2$-s). The northern elephant seal experienced TTS-onset at 121 dB re: 20 μPa (SEL: 163 dB re: $[20\ \mu Pa]^2$-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 μPa 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: $(20\ \mu Pa)^2$-s with a TTS growth function of ~2.5 dB TTS/dB noise. For TTS exceeding 20 dB, a recovery rate of ~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_2 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 dB TTS – 6 dB TTS] / [2.3 dB TTS/dB noise exposure] ≈ 15 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 dB TTS – 6 dB TTS] / [6 dB TTS/dB noise exposure] ≈ 6 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 dB TTS – 6 dB TTS] / [1.6 dB TTS/dB noise exposure] = 21.3 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 µPa (peak) and 186 dB re: 1 µPa^2-s. The latter is equivalent to a weighted (M_{mf}) SEL exposure of 183 dB re: 1 µPa^2-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 µPa (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 µPa^2-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 μPa²-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 μPa (peak) and 163 dB re: 1 μPa²-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 μPa (peak) and M_{mf}-weighted SEL of 183 dB re: 1 μPa²-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 μPa²-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 μPa (peak) and/or an SEL exposure of 171 dB re: 1 μPa²-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 μPa (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 μPa²-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 μPa (peak) (flat)	230 dB re: 1 μPa (peak) (flat)	230 dB re: 1 μPa (peak) (flat)
Sound exposure level	198 dB re: 1 μPa²-s (M_{lf})	198 dB re: 1 μPa²-s (M_{lf})	215 dB re: 1 μPa²-s (M_{lf})
Mid-frequency cetaceans	Cell 4	Cell 5	Cell 6
Sound pressure level	230 dB re: 1 μPa (peak) (flat)	230 dB re: 1 μPa (peak) (flat)	230 dB re: 1 μPa (peak) (flat)
Sound exposure level	198 dB re: 1 μPa²-s (M_{mf})	198 dB re: 1 μPa²-s (M_{mf})	215 dB re: 1 μPa²-s (M_{mf})
High-frequency cetaceans	Cell 7	Cell 8	Cell 9
Sound pressure level	230 dB re: 1 μPa (peak) (flat)	230 dB re: 1 μPa (peak) (flat)	230 dB re: 1 μPa (peak) (flat)
Sound exposure level	198 dB re: 1 μPa²-s (M_{hf})	198 dB re: 1 μPa²-s (M_{hf})	215 dB re: 1 μPa²-s (M_{hf})
Pinnipeds (in water)	Cell 10	Cell 11	Cell 12
Sound pressure level	218 dB re: 1 μPa (peak) (flat)	218 dB re: 1 μPa (peak) (flat)	218 dB re: 1 μPa (peak) (flat)
Sound exposure level	186 dB re: 1 μPa²-s (M_{pw})	186 dB re: 1 μPa²-s (M_{pw})	203 dB re: 1 μPa²-s (M_{pw})
Pinnipeds (in air)	Cell 13	Cell 14	Cell 15
Sound pressure level	149 dB re: 20 μPa (peak) (flat)	149 dB re: 20 μPa (peak) (flat)	149 dB re: 20 μPa (peak) (flat)
Sound exposure level	144 dB re: (20 μPa)²-s (M_{pa})	144 dB re: (20 μPa)²-s (M_{pa})	144.5 dB re: (20 μPa)²-s (M_{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 μPa (peak) and 129 dB re: (20 μPa)^2-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: 20μPa(peak)(unweighted)and144dBre:(20μPa) ^{2}-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 μPa (peak) and, following summation, 198 dB re: 1 μPa^2-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 μPa (peak) and the injury SEL criterion is 186 dB re: 1 μPa^2-s (Table 3, Cell 11). For pinnipeds in air, the proposed injury pressure criterion for multiple pulses is 149 dB re: 20 μPa (peak) and the injury SEL criterion is 144 dB re: (20 μPa)^2-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 μPa (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 μPa^2-s (Table 3, Cells 3, 6 & 9). This is based on 195 dB re: 1 μPa^2-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 μPa (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 μPa^2-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 μPa^2-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 μPa (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: $(20\ \mu Pa)^2$-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: $[20\ \mu Pa]^2$-s), we obtain a proposed M_{pa}-weighted SEL criterion of 144.5 dB re: $(20\ \mu Pa)^2$-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: $[20\ \mu Pa]^2$-s; Kastak et al., 2004a) and the California sea lion (159 dB re: $[20\ \mu Pa]^2$-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: $(20\ \mu Pa)^2$-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.

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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[1]	Corresponding behaviors (Free-ranging subjects)[2]	Corresponding behaviors (Laboratory subjects)[2]
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

[1]Ordinal scores of behavioral response severity are not necessarily equivalent for free-ranging vs laboratory conditions.
[2]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 nonpulses 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 μPa (peak) and weighted (M_{mf}) SEL values of 183 dB re: 1 μPa^2-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_{hf} 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 μPa (peak) and weighted (M_{pw}) SEL of 171 dB re: 1 μPa^2-s (Table 5, Cell 10) as discussed in Chapter 3.

For pinnipeds in air, the proposed behavioral criteria are based on the strong responses (stampeding 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 μPa (peak) and frequency-weighted (M_{pa}) SEL of 100 dB re: (20 μPa)^2-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.

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

[1] "Low-Frequency Cetaceans/Multiple Pulses (Cell 2)" section
[2] "Mid-Frequency Cetaceans/Multiple Pulses (Cell 5)" section
[3] "High-Frequency Cetaceans/Multiple Pulses (Cell 8)" section
[4] "Pinnipeds in Water/Multiple Pulses (Cell 11)" section
[5] "Pinnipeds in Air/Multiple Pulses (Cell 14)" section
[6] "Low-Frequency Cetaceans/Nonpulses (Cell 3)" section
[7] "Mid-Frequency Cetaceans/Nonpulses (Cell 6)" section
[8] "High-Frequency Cetaceans/Nonpulses (Cell 9)" section
[9] "Pinnipeds in Water/Nonpulses (Cell 12)" section
[10] "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 μPa (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 μPa (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^3. 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)
Malme et al. (1983)	1	Gray whales	Single airgun (1.64 L) & 20-gun 65.5-L airgun array	Calibrated RL measurements made *in situ* 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
Malme et al. (1984)	2	Gray whales	Single airgun and airgun array	Calibrated RL measurements made *in situ* 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 *in situ* 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 & 6
Ljungblad et al. (1988)	4	Bowhead whales (feeding)	Single airgun (1.3 L) or 18- to 20-gun array	Calibrated RL measurements made *in situ* 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 *in situ* 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 *in situ* 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 *in situ* 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 *in situ* 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
Malme et al. (1985)	Not included	Humpback whales	Seismic airgun array	Insufficient data for this analysis	Insufficient data for this analysis	No	N/A
Malme 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	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.

	Received RMS sound pressure level (dB re: 1 μPa)											
Response score	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								**1.0** (6)				
6				**9.5** (7)	**47.4** (7)	**2.2** (7)	**3.4** (4, 6, 8)	**5.8** (1, 2, 3, 6)	**4.5** (1, 2, 3, 4, 6)	**8.3** (1, 2, 4, 8, 9)		
5					**1.0** (7)		**1.0** (4)	**1.0** (1, 2)				
4												
3									**1.0** (1, 2)	**1.0** (1, 2)		
2												
1				**5.0** (7)	**6.0** (7)	**1.0** (7)	**2.5** (1, 2, 3)	**3.0** (5)				
0				**59.8** (7)	**17.7** (7)	**1.1** (7, 9)	**0.1** (9)	**0.6** (3, 9)	**6.8** (1, 2, 3, 9)	**6.3** (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 μPa) 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 μPa 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 *in situ* 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 *in situ* 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 *in situ* 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 *in situ* 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.

	Received RMS sound pressure level (dB re: 1 μPa)											
Response score	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					**0.17** (3)	**0.17** (3)	**0.17** (3)			**1.3** (4)		
5												
4												
3												
2												
1												
0			**0.25** (3)	**0.25** (3)	**3.0** (2)	**4.0** (2)				**6.7** (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 μPa range (RMS values over the pulse duration) generally have limited potential to induce avoidance behavior in pinnipeds. RLs exceeding 190 dB re: 1 μPa 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 μPa (RMS) pulse from an airgun is typically *ca*. 175 dB re: 1 μPa^2-s. That exceeds the estimated TTS threshold for the closely related harbor seal (171 dB re: 1 μPa^2-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 μPa.

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 μPa) 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 *in situ* 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 *in situ* 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 *in situ* 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.

	Received RMS sound pressure level (dB re: 1 μPa)											
Response score	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									**1.7** (1)	**2.1** (1)		**45.4** (1)
5												
4												
3												
2												
1						**0.3** (2)						
0								**0.7** (2)	**5.3** (1)	**30.3** (1, 3)	**0.3** (3)	**9.9** (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 μPa and an increasing probability of avoidance and other behavioral effects in the 120 to 160 dB re: 1 μPa 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 μPa, while others failed to exhibit such responses for exposure RLs from 120 to 150 dB re: 1 μPa. 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 μPa 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 *in situ* 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.

	Received RMS sound pressure level (dB re: 20 μPa)						
Response score	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		**0.125** (1)					
0		**0.625** (1)	**0.25** (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 μPa), 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 μPa 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 μPa 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
Malme et al. (1983, 1984)	2	Gray whales (migrating)	Playbacks of drilling and machinery noise	RLs measured *in situ* 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
Malme et al. (1986)	3	Gray whales (feeding)	Playbacks of drilling noise	RLs measured *in situ* 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 *in situ* 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 *in situ* 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 *in situ* 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 wore 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.

	Received RMS sound pressure level (dB re: 1 µPa)											
Response score	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						**2.5** (10)	**1.5** (10)					
6		**4.9** (2)	**7.4** (1, 2, 4)	**16.2** (1, 2, 3, 5)	**13.6** (2, 5)	**4.2** (1, 2)	**0.8** (2)					
5												
4				**3.0** (5, 7)	**1.0** (7)		**1.0** (7)					
3			**1,117** (9)	**0.27** (6)								
2			**0.5** (7)	**4.0** (7)	**5.0** (7)	**2.0** (7)	**1.0** (7)					
1												
0	**1.1** (2)	**82.6** (2, 3, 4)	**33.9** (1, 2, 3, 4)	**7.08** (2, 4, 6, 10)	**7.2** (4, 10)	**1.45** (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 µPa) 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 µPa 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 μPa).

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 *in situ* 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 *in situ* 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 *in situ* 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 *in situ* 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

Table 16 (continued)

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)
Monteiro-Neto et al. (2004)	10	Tucuxi (river dolphins)	Dukane® Netmark ADDs	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 110-120 dB SPL; severity score: 6
Morisaka et al. (2005)	11	Indo-Pacific dolphins	Vessel noise and presence	Calibrated RL measurements made *in situ* near areas of exposure	Passive acoustic monitoring of individual vocal output during vessel approaches	Yes	Exposure RLs 120-130 dB SPL; severity score: 5
Nachtigall et al. (2003)	12	Bottlenose dolphins (captive)	Nonpulse noise (bands)	Calibrated RL measurements *in situ* within test enclosure	Visual observations of behavioral responses within experimental lab context	Yes	Exposure RLs 170-180 dB SPL; severity score: 6
Finneran & Schlundt (2004)	13	Bottlenose dolphins (captive)	Nonpulse noise (tones)	Calibrated RL measurements made *in situ* within test enclosure	Visual observations of behavioral responses within experimental context in laboratory conditions	Yes	Exposure RLs 180-200 dB SPL; severity scores: 0 & 8
Rendell & Gordon (1999)	Not included	Long-finned pilot whales	Active military sonar	Insufficient data for this analysis	Passive acoustic measurements of whistle rates	No	N/A
Chilvers & Corkeron (2001)	Not included	Bottlenose dolphins	Vessel noise and presence	Insufficient data for this analysis	Visual observations of individual foraging behavior	No	N/A
Bordino et al. (2002)	Not included	Franciscana dolphins	ADDs	Insufficient data for this analysis	Insufficient data for this analysis	No	N/A
Williams et al. (2002)	Not included	Killer whales	Vessel noise and presence	Acoustic measurements of source levels but no estimates of RL	Visual observations of movement and diving behavior	No	N/A
Cox et al. (2003)	Not included	Bottlenose dolphins	ADDs	Insufficient data for this analysis	Visual observations of movement and diving behavior	No	N/A
Hastie et al. (2003)	Not included	Bottlenose dolphins	Vessel noise and presence	Insufficient data for this analysis	Visual observations of movement and diving behavior	No	N/A
Lusseau (2003)	Not included	Bottlenose dolphins	Vessel noise and presence	Insufficient data for this analysis	Visual observations of movement behavior	No	N/A
Foote et al. (2004)	Not included	Killer whales	General increase in vessels	Insufficient data for this analysis	Insufficient data on individual exposures/ responses for this analysis	No	N/A

Table 17. Number (in **bold**) of mid-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 "Mid-Frequency Cetaceans/Nonpulses (Cell 6)" section of this chapter. Parenthetical subscripts indicate the reference reporting the observations as listed in Table 16.

	Received RMS sound pressure level (dB re: 1 μPa)											
Response score	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		**1.0** (3)	**7.0** (3)		**5.0** (2)		**1.0** (7)				**5.0** (13)	**1.5** (13)
7												
6				**3.0** (2, 10)	**1.0** (2)				**1.0** (9)	**6.0** (12)		
5					**1.0** (11)							
4				**1.0** (4)	**2.0** (4)							
3	**5.0** (1)		**4.0** (3, 5)	**134** (4, 6)	**1.0** (4)							
2				**15.0** (2, 3, 8)								
1			**1.0** (4)	**1.0** (2, 3)	**1.0** (2, 4)							
0			**8.0** (3, 4)	**2.0** (2, 4)	**1.0** (2, 4)		**1.0** (2)				**3.0** (13)	**1.5** (13)

Courtesy: A. Friedlander

Table 18. Summary of behavioral responses of high-frequency cetaceans exposed to nonpulses (Cell 9) 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 19 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 19)	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 19)
Culik et al. (2001)	1	Harbor porpoises (wild)	PICE pinger	RL estimates based on source and environmental characteristics	Visual observations of individual and group movements and behavioral patterns before and following deployment	Yes	Exposure RLs 80-120 dB SPL; response severity scores: 0 & 6
Olesiuk et al. (2002)	2	Harbor porpoises (wild)	Airmar® AHDs	RL estimates based on source and environmental characteristics	Visual observations of individual and group movements and behavioral patterns before and following deployment	Yes	Exposure RLs 140-160 dB SPL; response severity score: 6
Johnston (2002)	3	Harbor porpoises (wild)	Airmar® AHDs	RL estimates based on source and environmental characteristics	Visual observations of individual and group movements and behavioral patterns before and following deployment	Yes	Exposure RLs 120-130 dB SPL; response severity scores: 0 & 6
Kastelein et al. (1997)	4	Harbor porpoises (captive)	Various non-pulse sounds (laboratory)	Calibrated RL measurements made *in situ* within test enclosure	Visual observations of movement, respiration, and behavior in laboratory conditions	Yes	Exposure RLs 80-120 dB SPL; response severity scores: 0, 4 & 6
Kastelein et al. (2000)	5	Harbor porpoises (captive)	Various non-pulse sounds (laboratory)	Calibrated RL measurements made *in situ* within test enclosure	Visual observations of movement, respiration, and behavior in laboratory conditions	Yes	Exposure RLs 90-120 dB SPL; response severity scores: 0 & 6
Kastelein et al. (2005)	6	Harbor porpoises (captive)	Various nonpulse sounds (laboratory)	Calibrated RL measurements made *in situ* within test enclosure	Visual observations of movement, respiration, and behavior in laboratory conditions	Yes	Exposure RLs 90-120 dB SPL; response severity scores: 0 & 6
Kastelein et al. (2006a)	7	Harbor porpoises (captive)	Various non-pulse sounds (laboratory)	Calibrated RL measurements made *in situ* within test enclosure	Visual observations of movement, respiration, and behavior in laboratory conditions	Yes	Exposure RLs 100-120 dB SPL; response severity scores: 0 & 6

Table 18 (continued)

Study	Reference number (for Table 19)	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 19)
Kraus et al. (1997)	Not included	Harbor porpoises (wild)	Dukane® pingers	Insufficient data for this analysis	Measurements of by-catch rates in commercial fisheries	No	N/A
Taylor et al. (1997)	Not included	Harbor porpoises (wild)	General nonpulse sounds	Review analysis	Review analysis	No	N/A
Johnston & Woodley (1998)	Not included	Harbor porpoises (wild)	Various AHDs	Insufficient data for this analysis	Visual observations of "exclusion" zones	No	N/A
Cox et al. (2001)	Not included	Harbor porpoises (wild)	Various ADDs	Insufficient data for this analysis	Visual observations of "exclusion" zones	No	N/A
Kastelein et al. (2001)	Not included (same subjects as 2000 study)	Harbor porpoises (captive)	Various non-pulse sounds (laboratory)	Calibrated RL measurements made *in situ* near areas of exposure	Aerial observations of individuals; movement and respiration patterns during and without airguns	No	N/A
Barlow & Cameron (2003)	Not included	Harbor porpoises (wild)	Various ADDs	Insufficient data for this analysis	Measurements of by-catch rates in commercial fisheries	No	N/A
Koschinski et al. (2003)	Not included	Harbor porpoises (wild)	Simulated wind turbine noise	Calibrated source-level measurements made but insufficient data on RL	Visual monitoring of general distribution patterns	No	N/A

Table 19. Number (in **bold**) of high-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 "High-Frequency Cetaceans/Nonpulses (Cell 9)" section of this chapter. Parenthetical subscripts indicate the reference reporting the observations as listed in Table 18.

	Received RMS sound pressure level (dB re: 1 µPa)											
Response score	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	**0.3** (4)	**0.3** (4)	**0.9** (1, 2, 4, 5, 6, 7)	**3.3** (1, 2, 4, 5, 6, 7)	**1.0** (3, 7)		**52.1** (2)	**9.3** (2)	**4.6** (2)			
5												
4			**0.1** (4)	**0.1** (4)								
3												
2												
1												
0	**12.8** (1, 5)	**23.1** (1, 2, 5, 6)	**0.4** (4, 7)	**0.1** (7)	**0.3** (3)							

Courtesy: A. Friedlander

Table 20. Summary of behavioral responses by different species of pinnipeds in water exposed to nonpulses (Cell 12) 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 21 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 21)	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 21)
Jacobs & Terhune (2002)	1	Harbor seals	Airmar® dB plus II AHD	RLs measured *in situ* in areas where individuals observed	Visual observations of individuals and groups of seals; movement and behavioral patterns during and without AHDs	Yes	Exposure RLs 120-130 dB SPL; response severity score: 0
Costa et al. (2003)	2	Elephant seals	ATOC (see Appendix B)	RLs measured using calibrated archival tags *in situ* on individuals during exposure	Archival tags placed on animals resulted in detailed quantitative measures of individual diving behavior, responses, and exposure RLs in well-characterized contexts	Yes	Exposure RLs 110-140 dB SPL; response severity scores: 0, 3 & 4
Kastelein et al. (2006b)	3	Harbor seals	Various non-pulse sounds used in underwater data communications	Calibrated RL measurements made *in situ* within experimental enclosure	Individual subject positions and the mean number of surfacing behaviors during control and exposure intervals	Yes	Exposure RLs 80-110 dB SPL; response severity scores: 0 & 6
Frost & Lowry (1988)	Not included	Ringed seals	Underwater drilling sounds	Insufficient data for this analysis	Insufficient data for this analysis	No	N/A
Richardson et al. (1990b, 1991)	Not included	Ringed and bearded seals	Underwater drilling sounds	Insufficient data for this analysis	Insufficient data for this analysis	No	N/A
Norberg & Bain (1994)	Not included	California sea lions	Cascade Applied Sciences® AHDs	Calibrated acoustic measurements taken around arrays of the devices	Insufficient data on individual responses for this analysis	No	N/A
Norberg (2000)	Not included	California sea lions	Airmar® dB plus II AHD	Insufficient data for this analysis	Some behavioral measurements but insufficient data on individual responses as a function of RL	No	N/A
Yurk (2000)	Not included	Harbor seals	AHD	Insufficient data for this analysis	Insufficient data for this analysis	No	N/A
Koschinski et al. (2003)	Not included	Harbor seals	Simulated wind turbine noise	RLs measured *in situ* in areas where individuals observed	Insufficient data on individual responses for this analysis	No	N/A
Moulton et al. (2003)	Not included	Ringed seals	Construction noise	Insufficient data for this analysis	Insufficient data on individual responses for this analysis	No	N/A

Table 21. Number (in **bold**) of pinnipeds in water (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 "Pinnipeds in Water/Nonpulses (Cell 12)" section of this chapter. Parenthetical subscripts indicate the reference reporting the observations as listed in Table 20.

	Received RMS sound pressure level (dB re: 1 µPa)											
Response score	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			**1.0** (3)									
5												
4					**1.0** (2)	**5.0** (2)						
3					**1.0** (2)	**2.0** (2)						
2												
1												
0	**1.0** (3)	**1.0** (3)		**1.0** (2)	**5.0** (1, 2)							

Courtesy: A. Friedlander

Table 22. Summary of behavioral responses by different species of pinnipeds in air exposed to nonpulses (Cell 15) 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 23 and more details are given in Appendix C. Exposure RLs are given in dB SPL, which are RMS sound pressure levels (dB re: 20 μPa).

Study	Reference number (for Table 23)	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 23)
Thorson et al. (1999)	1	Harbor seals, northern elephant seals, California sea lions, and northern fur seals	Athena 2 IKONOS-1 missile launch	RLs measured *in situ* in and around breeding rookeries	Visual observations of movement and behavior of individuals in breeding rookeries before, during, and after rocket launches	Yes	Exposure RLs 110-120 dB SPL; response severity score: 6
Thorson et al. (2000b)	2	Harbor seals, northern elephant seals, and California sea lions	Titan IV B-28 missile launch	RLs measured *in situ* in and around breeding rookeries	Visual observations of movement and behavior of individuals in breeding rookeries before, during, and after rocket launches	Yes	Exposure RLs 60-70 and 110-120 dB SPL; response severity scores: 0 & 6
Berg et al. (2002)	3	Harbor seals	Titan IV B-34 missile launch	RLs measured *in situ* in and around breeding rookeries	Visual observations of movement and behavior of individuals in breeding rookeries before, during, and after rocket launches	Yes	Exposure RLs 110-120 dB SPL; response severity score: 6
Allen et al. (1984)	Not included	Harbor seals	Aerial vessel noise and presence	Insufficient data for this analysis	Insufficient data for this analysis	No	N/A
Gentry et al. (1990)	Not included	Northern fur seals	Underground explosions and quarrying operations	Insufficient data for this analysis	Insufficient data for this analysis	No	N/A
Suryan & Harvey (1998)	Not included	Harbor seals	Aerial vessel noise and presence	Insufficient data for this analysis	Insufficient data for this analysis	No	N/A
Thorson et al. (1998)	Not included	Harbor seals	Titan IV A-18 missile launch	RLs measured *in situ* in and around breeding rookeries	Insufficient data for this analysis	No	N/A
Born et al. (1999)	Not included	Ringed seals	Aircraft noise and presence	Insufficient data for this analysis	Insufficient data for this analysis	No	N/A
Thorson et al. (2000a)	Not included	Harbor seals	Titan II G-13 missile launch	RLs measured *in situ* in and around breeding rookeries	Insufficient data for this analysis	No	N/A
Berg et al. (2001)	Not included	California sea lions and northern elephant seals	Delta II EO-1 missile launch	RLs measured *in situ* in and around breeding rookeries	Insufficient data for this analysis	No	N/A
Moulton et al. (2002)	Not included	Ringed seals	Industrial equipment noise & presence	Insufficient data for this analysis	Insufficient data for this analysis	No	N/A
Holst et al. (2005a, 2005b)	Not included	Harbor seals, California sea lions, and northern elephant seals	Small- and mid-sized missile launches	RLs measured near observed pinnipeds, including peak, RMS, SEL, and duration	Visual observations of animal presence and distribution before launches and behavior during and following launches	No	N/A

Table 23. Number (in **bold**) of pinnipeds in air (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 "Pinnipeds in Air/Nonpulses (Cell 15)" section in this chapter. Parenthetical subscripts indicate the reference reporting the observations as listed in Table 22.

	Received RMS sound pressure level (dB re: 20 µPa)						
Response score	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							**1.0** (1, 2, 3)
5							
4							
3							
2							
1							
0		**1.0** (2)					

Courtesy: Peter M. Scheifele

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5. Research Recommendations

The marine mammal noise exposure criteria proposed here represent a synthesis and precautionary application of current scientific information. Clearly, the reliance on extrapolation procedures, extreme data gaps and limitations in many areas, and precautionary assumptions throughout point to the need for targeted research to fill specific gaps in support of subsequent criteria. We consider the current noise exposure criteria to be merely an initial step in an iterative process to understand and better predict the effects of noise on marine mammal hearing and behavior.

Research recommendations are given below in several broad categories relevant to improving marine mammal noise exposure criteria. No prioritization is implied in the ordering of these areas or research topics within them, however, and this is by no means an exhaustive list. We present, in abbreviated form, what we regard as critical, targeted research needs to improve future iterations of these exposure criteria. Some of the most important research recommendations are summarized in Table 24; each is discussed in more detail in the relevant section below. Many of these research recommendations are similar to recommendations made previously (NRC, 1994, 2000, 2003, 2005; Richardson et al., 1995). Although there has been progress in the last decade, much important work remains to be done.

Measurements of Anthropogenic Sound Sources and Ambient Noise

Comprehensive and systematic measurements are needed of all relevant anthropogenic sound sources that have a reasonable likelihood of adversely affecting marine mammal hearing or behavior. Empirical measures of sound fields enable more accurate estimation of RLs using propagation models and validate the selection of different propagation models as appropriate. Such studies must report the full range of relevant standard acoustic measurements and should include detailed information about equipment calibration and/or propagation modeling methods used (e.g., Goold & Fish, 1998; Wales & Heitmeyer, 2002; Blackwell et al., 2004a). Measurements are also needed to describe conditions where sounds classified as pulses at the source transition to non-pulse exposures. To measure *in situ* exposures from specific sound sources, archival acoustic tags should be deployed on free-ranging marine mammals and/or platforms near the animals in controlled exposure conditions.

If future noise exposure criteria are to consider the important matters of auditory masking, cumulative exposure effects on individuals, and ecosystem effects (discussed below), additional data are needed concerning ambient ocean noise on various spatial and temporal scales. These data should be used to determine how ambient noise "budgets" vary as a function of natural and human activities. These data will need to be integrated with expanded information on marine mammal abundance and distribution. The NRC (2003) recommended that a systematic effort be made to obtain passive acoustic data, including average (steady-state) ambient noise from 1 Hz to 200 kHz, and including transient human sources not identified in classical ambient noise measurements. We concur and call for wide-ranging acoustic measurements designed to test explicit hypotheses about spatial and temporal variability in marine ambient noise.

Marine Mammal Auditory Processes

"Absolute" Hearing Data

Future iterations of these criteria will be significantly improved by increased knowledge of hearing sensitivity derived from behavioral and electrophysiological measurements and anatomical models. The most pressing needs are for data on deep-diving cetaceans such as beaked whales and on low-frequency specialists (mysticetes). Better information on inter-species differences is also needed to validate the functional hearing groups used here or alternatively to identify other relevant subdivisions (e.g., phocid vs otariid pinnipeds or potential partitioning of mid-frequency cetaceans). The number of individuals tested should be increased in all species, with the possible exception of the bottlenose dolphin, in order to better understand individual differences within species. Hearing sensitivity across the full functional hearing range should be measured, where possible, rather than just those frequencies contained within the communication signals of species being investigated.

Improvements are needed in both electrophysiological and behavioral testing methods to increase the number of individuals of each species that can be tested, and to distinguish absolute

Table 24. Research recommendations in various subject areas needed to enhance future marine mammal noise exposure criteria (as discussed in Chapter 5)

Research topic	General description	Critical information needs
Acoustic measurements of relevant sound sources	Detailed measurements needed of source levels, frequency content, and radiated sound fields around intense and/or chronic noise sources.	Comprehensive, calibrated measurements of the properties of human-generated sound sources, including frequency-dependent propagation and received characteristics in different environments.
Ambient noise measurements	Systematic measurements of underwater ambient noise are needed to quantify how human activities are affecting the acoustic environment.	Comprehensive, calibrated measurements of ambient noise, including spectral, temporal, and directional aspects, in different oceanic environments; ambient noise "budgets" indicating relative contribution of natural and anthropogenic sources and trends over time.
"Absolute" hearing measurements	Audiometric data are needed to determine functional bandwidth, species and individual differences, dynamic hearing ranges, and detection thresholds for realistic biological stimuli.	Carefully controlled behavioral and electrophysiological measurements of hearing sensitivity vs frequency for more individuals and species, particularly for high-priority species, such as beaked whales and mysticetes. Also, detection thresholds for complex biological signals.
Auditory scene analysis	Measurements to determine the sophisticated perceptual and processing capabilities of marine mammals that enable them to detect and localize sources in complex, 3-D environments.	Measurements of stream segregation, spatial perception, multidimensional source localization, frequency discrimination, temporal resolution, and feedback mechanisms between sound production and hearing systems.
Marine mammal behavioral responses to sound exposure	Measurements of behavioral reactions to various sound types are needed, including all relevant acoustic, contextual, and response variables.	Carefully constructed observational and exposure experiments that consider not only RL but also source range, motion, signal-to-noise ratio, and detailed information on receivers, including baseline behavior, prior experience with the sound, and responses during exposure.
Effects of sound exposure on marine mammal hearing: masking, TTS, and PTS	Continued effort is needed on the simultaneous and residual physiological effects of noise exposure on marine mammal hearing.	Masked hearing thresholds for simple stimuli in more species and individuals, as well as complex biological signals and realistic maskers; allowance for directional effects; comparative data on TTS-onset and growth in a greater number of species and individuals for nonpulse and pulsed anthropogenic sources; recovery functions after exposures and between repeated exposures.
Effects of sound exposure on marine mammal non-auditory systems	Physiological measurements are needed for both acute and chronic sound exposure conditions to investigate effects on non-auditory systems.	Various baseline and exposure-condition measurements, including nitrogen saturation levels; bubble nuclei; the formation of hemorrhages, emboli, and/or lesions; stress hormones; and cardiovascular responses to acute and chronic noise exposure.
Particularly sensitive species: beaked whales	Baseline and exposure data on these poorly understood taxa to assess their apparent sensitivity to certain anthropogenic sound sources.	Various studies, including measurements and modeling related to (1) hearing sensitivity, (2) diving and vocalization parameters, (3) tissue properties, (4) gas/fat emboli formation and significance, (5) advanced detection capabilities for localizing and tracking them, and (6) behavioral reactions to various anthropogenic and natural sound sources.

from masked thresholds. Auditory evoked potential (AEP) techniques should continue to be improved and standardized for pinnipeds and small cetaceans. Researchers should continue to develop procedures applicable to stranded individuals of species generally not represented in captive settings, particularly for species that may be especially sensitive to certain types of acoustic exposure. The massive body size of mysticetes may require that AEP studies begin using smaller species (e.g., minke whale) that may be stranded, trapped in tidal fishing enclosures (weirs), or temporarily available in a holding facility. Behavioral audiometric methods, which investigate the effect of the overall perceptual and cognitive system on detection, should also continue to be employed and improved, particularly those that increase the speed with which results are obtained without sacrificing precision of measurements.

Additionally, behavioral methods should be developed to measure hearing characteristics that require a subjective judgment of perception such as evaluation of equal loudness between two acoustic stimuli. Equal-loudness hearing contours for marine mammals are needed to refine the broad frequency-weighting networks derived here.

A final consideration is that behavioral audiometric research should eventually move beyond the use of relatively simple artificial stimuli (e.g., pure tones, noise bands, broadband clicks, tone pips). Such stimuli can be precisely controlled and can be used to clearly indicate which acoustic feature triggers the response in the whole animal or its auditory system. Animals in nature, however, rarely encounter such sounds. While some biological signals consist of combinations of tonal elements, most are exceedingly complex. Marine mammal detection thresholds for complex, biologically relevant stimuli may be poorly predicted by experiments using simple artificial stimuli. Humans, for example, are particularly adept at identifying speech-like sounds in noise (Yost, 2000). Animals are expected to be similarly sensitive to important natural sounds. To base future noise criteria on more relevant audiometric data, research is needed on detection thresholds for biologically meaningful sounds, such as vocalizations of conspecifics, prey, and predators, and sounds needed for active or passive acoustic navigation. Such measurements will further be useful in investigating the potential active space (detection range in three dimensions) for acoustic communication (e.g., Brenowitz, 1982; Janik, 2000; Au et al., 2004) and the effects of anthropogenic sound on the active space. Field studies using biologically relevant sounds would be more relevant to real-world communication and masking than studies involving simple, artificial test stimuli.

Auditory Scene Analysis

While baseline hearing information is clearly needed, urgently in some cases, more advanced, comprehensive, and innovative measurements are also needed that provide insight into the ways in which animals use their auditory sense to derive detailed information about their surrounding environment. For future iterations of noise exposure criteria to consider multiple stimuli and cumulative effects, additional data will be needed on sound localization in three-dimensional auditory space, frequency discrimination, temporal resolution, and, specifically, detection of biological signals in complex sound fields.

Several studies of terrestrial animals (MacDougall-Shackleton et al., 1998; Moss & Surlykke, 2001) have investigated how subjects process multiple acoustic stimuli that are simultaneously present but differ in acoustic signature either temporally or spatially. The acoustic scene concept, owing largely to the work of Bregman (1990), has the potential to play a major role in the development and progression of acoustic exposure criteria. Bregman draws powerful analogies between modalities of perception, including the fundamental ways in which higher processing systems associate common elements of complex stimuli in highly cluttered perceptual environments.

One analogy that Bregman (1990) makes with regard to the innate power of visual scene analysis is the ability of the visual processing portion of the human brain to estimate object size without regard to distance. The implication is that the reverse is true as well—if the size of something is known, its distance can be inferred from visual appearance. Extending this ability to animals that rely on underwater hearing to determine distance, similar perceptual processes may occur. If so, mammals may determine range by using various effects of the propagation medium on sound transmission (e.g., presence of structured multi-path signal spreading, frequency dependent multi-path losses, and absorption effects in particular; Ellison & Weixel, 1994). Further, both loudness modulation and source movement relative to the receiver provide significant clues as to the distance and general nature of the sound source. If one considers sound to play a role in the life of marine wildlife similar to that of sight in terrestrial animals, then context clues such as tempo, encroachment, and proximity must take on a powerful role in determining an animal's response to any given sound. These hypotheses need to be studied in marine mammals.

Behavioral Responses of Marine Mammals to Sound

There is an urgent need for better and more extensive data on behavioral responses to sound, including measurement of the specific acoustic features of exposures and consideration of previous experience with the sound and all relevant contextual variables. The current behavioral exposure criteria are quite limited in several ways. Insufficient data exist to support criteria other than those based on SPL alone, and this metric fails to account for the duration of exposure beyond the separation of pulse from nonpulse sounds. Also, there is much variability in responses among species of the same functional hearing group and also within species.

Because of the poorly understood modifying influences of numerous variables, behavioral responses usually cannot be predicted *a priori* with much confidence given present information. In addition, the biological significance of any observed behavioral response is even more difficult to assess (NRC, 2005).

Research is needed to quantify behavioral reactions of a greater number of free-ranging marine mammal species to specifically controlled or well-characterized exposures from different human sound sources. The most direct way to obtain these kinds of extremely detailed data would be to attach acoustic dosimeter tags to individuals and directly measure noise exposure, behavioral response, and physiological changes, if any. It is essential that future research investigates responses in contexts as similar as possible to those of interest. Responses of both naïve and previously exposed individuals should be studied and distinguished to the greatest extent possible.

Further, such experiments must ensure that all relevant acoustic measurements of sound exposure be reported more systematically than in many previous studies. Specifically, behavioral responses need to be directly correlated with the physical parameters (e.g., SPL, SEL) of the stimuli most likely to evoke the responses. Such research clearly requires greater knowledge of exposure parameters (including SPL over some duration) than currently exists for most studies.

The relationship between exposure SPL and/or SEL and behavioral reaction should be determined for representative species within each functional hearing group. Whether the relationship follows a dose-response-like function for various sound types, and under what conditions, is a significant and pressing open question.

We need more data on the magnitude and time course of behavioral responses to known noise exposures to test the validity of concepts outlined here, and to make progress toward identifying specific behavioral criteria. Duty cycle (the proportion of time when the noise is present) is also likely to be important. Magnitude and duration of response are the most readily quantified parameters that may be useful in determining whether a behavioral response is likely to have a biologically meaningful outcome. Noise exposure criteria should attempt to distinguish between minor, temporary behavioral changes and those with greater significance. This is necessary in order to focus on biologically significant behavioral responses (see NRC, 2005) and the exposure conditions that elicit them.

Considering the many contextual cues that free-ranging animals use to perceive and characterize sound sources and to determine a response, it is not surprising that our analysis revealed a high degree of variability in behavioral responses as a function of RL. Consequently, the logic of relying solely on exposure RL as the metric for behavioral responses is substantially diminished. A host of variables additional to RL may be important to marine mammals in assessing a sound and determining how to react. This argues for careful design and execution of controlled exposure experiments to replicate the signal of interest in as many dimensions as possible. Serious consideration should be given to developing a broad multi-variable approach to behavioral research that takes into account not only source type and exposure level but also distance, motion, and relative signal-to-noise ratio. Some studies are already developing data of the scale and quality needed for such an approach. This includes studies providing broad, long-term measurements of ambient sounds in areas cohabited by anthropogenic sources and marine wildlife. Where these studies include remotely deployed passive acoustic sensors and tagged animals, they approach what may become the new standard. As additional information becomes available, future noise exposure criteria may assess behavioral reactions not only according to RL measured with multiple acoustic parameters, range (near and far), relative motion (towards, parallel, etc.), and rate of change, but also in relation to the animal's activity or perceptual situation (e.g., neutral; threatened, as by a predator; or positive, related to food, mating, etc.).

The role of habituation and sensitization in behavioral reactions to noise exposure is a critical subject for future research. These processes can only be studied under controlled or well-defined conditions (as in Deecke et al., 2002). A key question is how habituation and sensitization develop with repeated exposure in specific ecologically relevant circumstances. For example, the pattern of habituation to a neutral stimulus

is likely to follow quite a different pattern from selective habituation to a harmless stimulus that is initially perceived as a threat (Deecke et al., 2002). Furthermore, it would be desirable to know if there are common acoustic features in sounds to which marine mammals become sensitized. For example, to which acoustic features of a threat, such as a vessel used to hunt animals, does an animal become sensitized?

Analyses of the behavior of various animal species in the presence of predators suggest that they have evolved anti-predator responses that mirror their responses to human disturbance. According to predation risk theory, various ecological considerations beyond simply disturbance magnitude are very likely involved in determining and predicting behavioral response (Frid & Dill, 2002).

The biological relevance of behavioral changes can only be determined in natural populations in which vital life history parameters (e.g., reproduction, growth, and survival rates) can be measured before and after noise exposure and in conditions where other potential stressors have been controlled (NRC, 2005). One important question is whether these life history parameters are the same in populations that have apparently habituated to exposure and remain in relatively noisy environments as they are in populations living in quieter conditions. Because of the apparently major influence of experience and the strong context-specificity of behavioral responses to noise, field measurements must be made for long periods following repeated or continual exposure. Longitudinal studies should be conducted to assess the time course of exposure to various existing sound sources known or suspected to cause relatively long-term (seasonal) habitat abandonment. Where possible, parallel studies should be done in neighboring areas with different levels of noise exposure. Such studies should allow for other non-acoustic factors likely to affect distribution such as predators, prey, and other important environmental covariates. These studies will often need to extend over long periods (many years) in order to be effective, and they should be planned and funded recognizing that. Ideally, such a study should start collecting data well in advance of the introduction of anthropogenic noise, and continue throughout the period of anticipated impact and for long enough thereafter to observe return to baseline.

Effects of Noise Exposure on Marine Mammal Hearing and Other Systems

Auditory Masking

Auditory masking is likely the most widespread effect of anthropogenic noise on populations of marine mammals. The principles of masking are reasonably well-known from laboratory studies in mammals, including marine mammals. To enable masking to be included in subsequent noise exposure criteria, however, data are needed on masking and its effects in real-world conditions for all functional hearing groups. Data are needed on the masking effects of natural and anthropogenic noise sources; on detection of simple, artificial stimuli; and, increasingly, on more complex, biologically meaningful signals. Directionality in the masking sound and/or the signal of interest is very likely to affect the severity of masking and needs to be considered. Baseline measurements are needed on functional communication ranges for different acoustic signals and on the reduction of those ranges caused by either natural or anthropogenic maskers. Also needed are additional field measurements of the behavioral adjustments that marine mammals make to offset masking effects (e.g., Lesage et al., 1999; Serrano & Terhune, 2002; Foote et al., 2004; Scheifele et al., 2005).

Temporary Threshold Shift (TTS)

TTS studies in marine mammals remain limited to a very few species and individuals, limiting the certainty with which they may be extrapolated within and among groups. A number of specific TTS studies are needed to improve future criteria. For instance, it is critical to future iterations of these noise exposure criteria that research on TTS-onset, TTS growth with noise exposure, and recovery rates expands to larger numbers of individuals and species, and to species in the low- and high-frequency cetacean groups. Presently, extrapolation procedures must be used because TTS data are unavailable for certain functional hearing groups. Additionally, certain highly precautionary procedures are used here in the estimation of PTS-onset because the growth rate of TTS with increasing exposure level is generally poorly understood, even for the few marine mammal species in which TTS has been studied. The relationship between auditory sensitivity and susceptibility to TTS/PTS should be determined by group.

To the extent possible, electrophysiological techniques should be used to obtain these TTS data to increase sample size and knowledge of recovery functions.

More data for pinnipeds also are needed, particularly for pulse exposures where extrapolations of cetacean data currently must be used. Particular emphasis should be placed on determining whether harbor seals have increased sensitivity to noise exposure relative to other pinniped species, as current information suggests, and if so, whether species closely related to the harbor seal also are more sensitive than are other pinnipeds.

To minimize the need for such extrapolation and reduce the assumptions required to predict PTS-onset, empirical data are needed on TTS growth rates up to greater shift magnitudes (10 to 30 dB). These data are needed for both pulse and nonpulse sound types, at a variety of exposure frequencies, in both single and multiple exposures. These results should further elucidate whether, and in what conditions, the "equal energy hypothesis" may be appropriate for comparing the effects of variable noise exposures in marine mammals. For pulse exposures, particular attention should be paid to whether TTS growth is directly related to overall noise energy, and whether the kurtosis of exposure is also a factor (see Erdreich, 1986; Thiery & Meyer-Bisch, 1988; Dunn et al., 1991; Hamernik et al., 1993, 2003).

A further topic for future research is determining whether using 40 dB of TTS as a proxy for PTS-onset is a precautionary approach, and whether TTSs on the order of 25 to 35 dB are fully recoverable in marine mammals as expected from terrestrial mammal data. To avoid any possibility of injury, such studies should continue to take a precautionary approach, using gradual increases in exposure level and duration.

A related question is how TTS recovers following noise cessation in variable conditions. Data on recovery functions and TTS magnitude are needed for representative species from each functional hearing group. Electrophysiological techniques may be particularly useful in this regard. These data may be useful in comparing basic auditory system responses to noise exposure and determining how summation procedures for multiple exposures should be modified to more precisely consider exposure intermittence. Levels of relatively long duration noise exposure causing asymptotic TTS, in which TTS values do not continue to increase in magnitude with exposure but may have longer-lasting effects, should be determined. Recovery functions from asymptotic TTS of various levels should be compared with recovery functions from non-asymptotic TTS.

Finally, the existence of a stapedial reflex in marine mammals and its possible role in mitigating TTS and other effects of intense noise exposure are areas of needed research. For certain noise exposures, particularly those with relatively low frequencies and long duration, the middle ear muscles (tensor tympani and stapedial) of terrestrial mammals may contract and reduce the amplifying function of the ossicular chain (Yost, 2000). This muscular contraction reduces the amount of acoustic energy transmitted into the cochlea via the stapes. This stapedial reflex has been demonstrated in humans exposed to intense sound (Davis et al., 1955) as well as echolocating bats exposed to their own intense outgoing clicks (Henson, 1965). The middle ears of marine mammals have some specialized adaptations relative to terrestrial mammals (see Wartzok & Ketten, 1999). In water, if bone conduction (rather than ossicular conduction) is the predominant transmission path, it is possible that a stapedial reflex, if present, may have limited or no protective function for intense acoustic exposures. Research is also needed on the role of meatal closure in pinnipeds during noise exposure. Such closures could be an alternative or additional way of reducing auditory sensitivity. Either mechanism could also affect the interpretation of threshold if performed during audiometric measurements.

Permanent Threshold Shift (PTS)

Sound exposures causing PTS-onset, used here to define injury from acoustic exposure, have not been measured in marine mammals. Instead, exposures that would cause PTS-onset are estimated from measured TTS-onset using assumptions about the growth of TTS with noise exposure level. Direct measurements of PTS in marine mammals are highly desirable for establishing future injury criteria, but they are unlikely to be obtained due to ethical, legal, and/or practical considerations. Data from modeling and exposure of cadavers to very intense acoustic stimuli give some indication of conditions causing PTS but do not reveal the exposure conditions that produce PTS *in vivo*, nor active processes that affect basilar membrane displacement. Consequently, our research recommendations for improving PTS-onset predictions for marine mammals involve more indirect measures.

One recommended type of indirect measure is to compare age-related hearing changes in captive individuals that have been involved in TTS experiments with those that have not. This comparison may provide some insight into the complex relationship between repeated TTS and PTS, which remains poorly understood for terrestrial mammals, including humans. One main impediment, however, is that confounding variables likely exist other than controlled noise exposure. For captive individuals used in TTS studies, absolute hearing should be tested both during and following sequences of noise exposure experiments. For captive individuals not used in TTS experiments, absolute hearing should be measured at regular intervals over extended periods. The latter group may more readily display natural age-related hearing loss (*presbycusis*) than the former, as well as potential sex differences. For both groups, efforts should be made to characterize long-term ambient noise conditions experienced by test animals.

Non-Auditory Effects of Noise Exposure

Lack of specific data on acoustic exposures causing non-auditory effects in marine mammals currently prevents deriving explicit exposure criteria for such effects. Research is underway, however, that may make this possible in future versions of the criteria. Non-auditory effects of noise are potentially significant but remain generally poorly understood.

A current hypothesis regarding non-auditory effects is that acoustic exposure may produce nitrogen bubbles in blood or other tissues. Hemorrhages, gas and fat emboli, and other lesions have been reported in some marine mammals exposed to mid-frequency military sonar (Jepson et al., 2003; Fernández et al., 2004, 2005). Substantial empirical questions remain, however. First among these is whether nitrogen bubbles are in fact responsible for the hemorrhages, emboli, and other lesions reported. Conversely, are enough nitrogen bubbles produced to pose a risk of related tissue injuries, under any set of circumstances, arising from high nitrogen supersaturation levels, acoustic exposure, and/or drastic changes in behavior? Do high levels of nitrogen supersaturation or gas or fat emboli occur in diving mammals that have not been exposed to intense anthropogenic sound? Do these or related phenomena occur in species other than beaked whales? If bubble formation is acoustically mediated, does it occur as a direct result of acoustic exposure of bubble precursors (nuclei) in tissue, or indirectly through changes in diving behavior? If the pathway is direct, how does bubble formation and/or growth occur? A more thorough understanding is needed of lipid biochemistry in tissues that may be particularly sensitive to acoustically mediated bubble formation (e.g., acoustic fats). Modeling studies are needed on tissue properties and their relevance to nitrogen bubble formation at specific frequencies of interest. These studies should consider the growth of discrete bubbles from precursors in various tissues, and the interaction among coalesced aggregations of acoustically activated bubbles.

If the pathway is indirect and mediated by behavior, is rapid surfacing more risky than remaining submerged too long and exceeding physiological limits? How does the dive profile affect the limits of nitrogen supersaturation in normal diving? Do high levels of nitrogen supersaturation and gas emboli occur in marine mammals that have voluntary control over depth, diving profile, and inter-dive interval? Resolution of these questions is likely to require interplay between modeling and empirical measurements (Zimmer & Tyack, 2007).

In conjunction with the above physiological modeling and measurements, controlled exposure experiments should be conducted with deep-diving marine mammals to determine behavioral responses to sound sources, including sonar. These experiments should use realistic source and received levels. If responses are identified, this may identify situations where it would be useful to conduct observational studies of responses during uncontrolled use of anthropogenic sound sources. Research should characterize the changes in diving behavior and should determine what they mean in terms of bubble formation or growth with continued exposure.

Other possible non-auditory effects of acoustic exposure should be investigated as well. Stress hormone levels associated with noise exposure should be more fully investigated. As of now, they have been investigated following exposure of captive odontocetes to high-level sound (Thomas et al., 1990c; Romano et al., 2004). The ability of animals to recruit effective stress responses should also be studied during chronic exposures—for example, in captive animals that live permanently in noisy vs quiet environments. Effects of noise exposure on marine mammal vestibular and cardiovascular systems should also be studied.

Particularly Sensitive Species

In rare circumstances, marine mammals (primarily beaked whales) have been known to strand and ultimately die following exposure to tactical, mid-frequency active sonar (see Cox et al., 2006; Nowacek et al., 2007). Our knowledge of these kinds of extreme reactions to acute exposures remains poor. However, the available information suggests that at least some species of beaked whales are particularly sensitive to this one specific category of sound sources.

Gas bubble formation is a hypothesized pathway of this effect (e.g., Fernández et al., 2005), but it remains poorly understood and the precise mechanism underlying these strandings remains unknown (e.g., Cox et al., 2006). The controlled exposure experiments outlined above are essential to revealing the conditions and responses underlying this effect. Until such research is conducted, deriving science-based exposure criteria specifically for beaked whales or other deep-diving cetaceans exposed to active sonar will prove difficult or impossible.

One current hypothesis is that behavioral reactions influence beaked whale diving patterns in a way that induces physically debilitating or disorienting injuries (Cox et al., 2006). Both the specifics of this potential mechanism and whether it is specific to beaked whales remains unknown, however. Mammals, including some marine mammals, show strong avoidance responses when evading predators. Sounds from tactical mid-frequency

sonars somewhat resemble, in frequency band and modulation, the social signals of one of the only predators of large marine mammals, the killer whale. If beaked whales inherit a broad template for acoustic detection of these predators, as waterfowl do for visual detection of aerial predators (Lorenz, 1939; Tinbergen, 1948), they might respond to sonar as if it were a predator. Learning is required for selective habituation to safe stimuli that resemble those from predators (Deecke et al., 2002). Many of the strandings that coincide with sonar exercises have occurred in sites where killer whales are rare. Possibly these stranded animals have not had enough experience with either sonar or killer whales to learn the difference. Propagation of sound in the ocean may also degrade acoustic features that help differentiate the two classes of stimuli at a distance. It is plausible that this type of reaction could occur at relatively long distances from the source if the sound is alarming based on properties other than high RL.

Whether beaked whales in certain conditions mistake tactical mid-frequency sonar signals for killer whales and consequently change their behavior in a way that injures them is an empirical question. This should be carefully investigated using controlled experiments that take into account the relevant contextual variables discussed above. Additional baseline data on beaked whale physiology, life history, and behavior are also needed to appropriately address questions regarding the apparent sensitivity of these animals to certain kinds of anthropogenic sound. Finally, in some specific conditions, such as sonar training ranges, where sophisticated listening arrays make it possible to detect marine mammals over large ranges before and during active sonar operations, active or passive detection of marine mammal behavioral patterns may become increasingly possible. While these observations have limitations, given that they may be able to detect more individuals without requiring tagging efforts, they may be an important complement to directed experiments.

Some other species of marine mammals are unusually responsive to certain anthropogenic sounds, either generally or under particular conditions, and this can result in strong and sometimes large-scale avoidance. Examples include harbor porpoises and, in some but not all situations, beluga and bowhead whales (Finley et al., 1990; Richardson et al., 1999; Olesiuk et al., 2002; Miller et al., 2005). There is a need for additional behavioral and acoustic information to better characterize these extreme responses, the situations in which they occur, and whether similar responses can occur in other related species or in response to other similar stimuli.

Necessary Progressions of Marine Mammal Noise Exposure Criteria

The currently proposed noise exposure criteria are for individual sound exposures and individual marine mammals. The research recommended above is needed to substantiate and improve future iterations of these types of criteria. Future iterations of behavioral disturbance criteria may derive dose-response functions based on an ordinal scoring paradigm similar to that provided. This may occur for subcategories of sound sources within the general categories here (e.g., seismic signals as a subset of multiple pulses, vessel noise as a subset of nonpulses). It may also occur for subgroups of species within the broad categories recognized here (e.g., phocid vs otariid pinnipeds) and for other types of marine mammals not addressed here (e.g., sirenians, sea otters, polar bears).

Future iterations of these noise exposure criteria should also perhaps distinguish several different categories of response that are expected, for both theoretical and empirical reasons, to vary with RL in different ways. For example, if an animal responds to a sound as if it were from a predator (Frid & Dill, 2002), one would expect the dose-response function to have a very different shape as compared to that if the animal responds based on interference with the animal's ability to communicate acoustically or echolocate. Predicting whether a sound might trigger an anti-predator response would require more detailed analyses of acoustic parameters of the anthropogenic sound compared to signals of predators. Further, in some non-marine taxa, different anti-predator responses may be triggered depending on levels and other characteristics of acoustic stimuli (Spangler, 1988; Hoy, 1989) and may be modulated by the cost of the response as well as the perceived risk (Frid & Dill, 2002). Behavioral ecologists hypothesize that anti-predator behavior should balance risk of predation against cost of response, including cost of foregone benefits from alternative activities (Frid & Dill, 2002). These non-acoustic parameters must be taken into account in order to understand disturbance responses. The acoustic parameters affecting anti-predator behavior may involve detection thresholds, ambient noise conditions, source distance and source movement, as well as the more direct measures of received sound. In future studies, most or all of these parameters should be measured.

Additionally, further exposure criteria are needed to fully consider the effects of anthropogenic noise on other types of marine life, including the effects of single and multiple exposures on individual invertebrates, fish, and sea turtles as well as sirenians, sea otters, and polar bears. There

are fewer data to support criteria for marine biota other than cetaceans and pinnipeds, and criteria are perhaps as urgently (or more urgently) needed for some other groups. Some fish and most sea turtle species are considered threatened or endangered. The effects of anthropogenic noise on fish are also of particular importance given their central role as both predators and prey in many marine ecosystems and because of human dependence on fisheries.

Additional criteria are also needed for the cumulative effects of repetitive or long-term noise exposure on marine mammals. Ideally, spatiotemporal data on marine ambient noise and long-term exposure histories of individuals should be integrated with vital rate data for marine mammal populations to address this question. Considerably more data are needed on how noise impacts in single animals can be extended to the population level. Such measurements will likely require extensive measurements on a few representative species and conservative extrapolations within and between functional hearing groups.

Noise exposure criteria that consider ecosystem-level effects are needed as well. It is possible that the effects of noise exposure on some elements of local food webs may have a cascade effect to other elements within the web. No data are available on the ecological effects of underwater noise, even at a local scale. However, given the upward trend in human activities in many nearshore areas, such ecological effects should be anticipated.

Progress in each of these research areas will involve iterative processes that depend on the availability of relevant scientific data. Like the process of improving and expanding future noise exposure criteria, our ability to understand and predict the effects of anthropogenic noise exposure on marine ecosystems will continue to evolve over a period of many decades.

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Appendix A. Acoustic Measures and Terminology

This appendix provides a more detailed description of many key acoustic measurements and terms used throughout the noise exposure criteria. It is not intended as an exhaustive or instructive text on these exceedingly complex issues (for more detailed treatments, see Kinsler et al., 1982; ANSI, 1986, 1994; Richardson et al., 1995; Harris, 1998; NRC, 2003). Rather, it is intended to provide fairly straightforward definitions and equations related to the marine mammal noise exposure criteria.

Pulses and Nonpulse Sounds

The distinction between these two general sound types is important because they have differing potential to cause physical effects, particularly with regard to hearing (e.g., Ward, 1997).

Pulses, as used in the context of this paper, are defined as brief, broadband, atonal, transients (ANSI, 1986; Harris, 1998, Chapter 12). Examples of pulses (at least at the source) are explosions, gunshots, sonic booms, seismic airgun pulses, and pile driving strikes. These sounds are all characterized by a relatively rapid rise from ambient pressure to a maximal pressure value followed by a decay period that may include a period of diminishing, oscillating maximal and minimal pressures. The rapid rise-time characteristic of these sounds ensures that they are also broadband in nature, with the higher-frequency components being related to the rapidity of the rise-time. Pulses, either as isolated events or repeated in some succession, generally have an increased capacity to induce physical injury as compared with sounds that lack these features.

Nonpulse (intermittent or continuous) sounds can be tonal, broadband, or both. Some of these nonpulse sounds can be transient signals of short duration but without the essential properties of pulses (e.g., rapid rise-time). Examples of sources producing nonpulse sounds include vessels; aircraft; machinery operations, such as drilling or wind turbines; and many active sonar systems. The duration of such sounds, as received at a distance, can be greatly extended in highly reverberant environments. It is critical to note that a sound that has characteristics of a pulse at the source may, as a result of propagation effects, lose those characteristics at some (variable) distance and could be characterized as a nonpulse for certain receivers.

Pulses and nonpulses are distinguished here by an empirical approach based on several temporal weightings. Various exponential time-weighting functions applied in measuring pulse and nonpulse sounds may yield different measured received levels (RLs) (see Harris, 1998). By way of illustration, most sound level meters (SLM) provide options for applying either a slow or fast time constant (1,000 or 125 ms, respectively) for measuring nonpulses, or an impulse time constant (35 ms) appropriate for measuring pulses. If applied to a sound pulse, the slow or fast SLM settings result in lower sound pressure level (SPL) measurements than those obtained using the impulse setting. Each of these time constants was selected based on the physical properties of the human auditory system. It is clear that further empirical measures of temporal resolution in marine mammals are needed, particularly for animal taxa whose hearing extends to significantly higher or lower frequencies than in humans (see Chapter 5, "Research Recommendations"). Future noise criteria are expected to include distinctions between pulse and nonpulse sounds that may be more specifically appropriate for marine mammals than is this current simple approach. We note also the need for an explicit distinction and measurement standard, such as exists for aerial signals (ANSI, 1986).

Peak sound pressure is the maximum absolute value of the instantaneous sound pressure during a specified time interval and is denoted as P_{max} in units of Pascals (Pa). It is not an averaged pressure. Peak pressure is a useful metric for either pulse or nonpulse sounds, but it is particularly important for characterizing pulses (ANSI, 1986; Harris, 1998, Chapter 12). Because of the rapid rise-time of such sounds, it is imperative to use an adequate sampling rate, especially when measuring peak pressure levels (Harris, 1998, Chapter 18).

Peak-to-peak sound pressure is the algebraic difference between the maximum positive and maximum negative instantaneous peak pressure.

The ***mean-squared pressure*** is the average of the squared pressure over some duration. For nonpulse sounds, the averaging time is any convenient period sufficiently long enough to permit averaging the variability inherent in the type of sound. Note that some of the variability of the received sound typically arises simply from the

relative movement of a free-ranging animal and a source, whether the latter is moving or stationary.

Sound pressure levels (SPLs) are given as the decibel (dB) measures of the pressure metrics defined above. The ***root-mean-square*** (RMS) SPL is given as dB re: 1 µPa for underwater sound and dB re: 20 µPa for aerial sound. Peak SPLs are given as dB re: 1 µPa (peak) in water and dB re: 20 µPa (peak) in air. Peak-to-peak SPLs are dB re: 1 µPa (peak-to-peak) in water and dB re: 20 µPa (peak-to-peak) in air. ***Source level*** (SL) is the received level measured or estimated 1 m from the source.

Duration is the length of a sound, generally in seconds. Duration is important because it affects various acoustic metrics, including mean-square and/or RMS sound pressure (Madsen, 2005). Because of background noise and reverberation, duration can be difficult to define precisely. Various definitions of duration exist in the literature such as the time between the points on the pressure-time waveform P(t) determined to be either 10 dB (0.316 times) or 20 dB (0.1 times) below the instantaneous peak pressure (Hamernik & Hsueh, 1991). Malme et al. (1983, 1984) used a similar approach. Harris (1998, Chapter 11) suggested alternative constructs, including exponential time weighting. This topic is discussed below with regard to updating measurement standards for impulse sounds. Greene (1997) described a practical definition of pulse duration based on the interval over which 90% of the sound energy arrived at the receiver. This interval could also be used as the averaging time for mean-square pressure (Madsen, 2005). This approach has been widely used in measuring exposure duration and SPL values for seismic airgun and pile driving signals (e.g., McCauley et al., 1998; Blackwell et al., 2004b). Defined as such, duration is the interval between the 5% and 95% bounds of the time-integral of the instantaneous sound-pressure squared (sound exposure [E(t)] as defined below) while accounting for background noise and low-level reverberation (assumed to be continuous). That is, the background noise is measured over a period of time before the pulse occurs and then is subtracted from the cumulative sum-of-square pressures to determine the sum-of-square pressures from the impulsive sound alone. This is done by manually identifying a period of time (t_1, t_2) preceding the event, deemed to be representative of ambient noise. The mean-square pressure (in Pa^2) of the ambient $(P_{amb})^2$ is determined with the following relationship:

$$\overline{P_{amb}^2} = \frac{1}{t_2 - t_1}\int_{t_1}^{t_2} P^2(t)dt \qquad \text{(1) eq.}$$

The ***temporal (or event) sound exposure*** E_{temp} (t) (in Pa^2-s) is then calculated as

$$E_{temp}(t) = \int_{t_2}^{t}\left(P^2(t) - \overline{P_{amb}^2}\right)dt \qquad \text{(2) eq.}$$

The 0% sound exposure point (t_a) signifies the "start" of the acoustic event and the 100% sound exposure point (t_b) signifies the "end" of the event. These two points are where the E(t) curve begins to rise and where it levels off, respectively. Their selection can be difficult due to variation in ambient noise preceding (and overlapping) the acoustic event, as well as reverberation plus ambient noise following the event. Consequently, many investigators identify these points subjectively.

The ***sound exposure E(t)*** (in Pa^2-s), where $t \leq t_b$, is then calculated as

$$E(t) = \int_{t_a}^{t}\left(P^2(t) - \overline{P_{amb}^2}\right)dt \qquad \text{(3) eq.}$$

where $E_{100} = E(t_b)$ is 100% of the sound exposure. For the 5% point, E(t) is determined as $E_5 = 0.05 \bullet E_{100} = E(t_5)$, while E(t) for the 95% energy point is determined as $E_{95} = 0.95 \bullet E_{100} = E(t_{95})$. Thus, $E_{90} = E_{95} - E_5$ and duration $(T_d) = t_{95} - t_5$ (s) where the received pressure level greatly exceeds the ambient level, eq. 3 can be reduced to

$$E(t) = \int_{t_a}^{t}\left(P^2(t)\right)dt \qquad \text{(4) eq.}$$

Sound exposure level (SEL) is the decibel level of the cumulative sum-of-square pressures over the duration of a sound (e.g., dB re: 1 μPa^2-s) for sustained nonpulse sounds where the exposure is of a constant nature (i.e., source and animal positions are held roughly constant). However, this measure is also extremely useful for pulses and transient nonpulse sounds because it enables sounds of differing duration to be characterized in terms of total energy for purposes of assessing exposure risk.

The SEL metric also enables integrating sound energy across multiple exposures from sources such as seismic airguns, pile driving, and most sonar signals. Several methods exist for summing energy over multiple exposures. We use a relatively straightforward approach here, specifically

$$SEL = 10\log_{10}\left\{\frac{\sum_{n=1}^{N}\int_{0}^{T} p_n^2(t)\,dt}{(p_{ref})^2}\right\} \quad (5)\ \text{eq.}$$

where instantaneous sound-pressure (p) is measured in μPa for *n* exposures and the reference pressure (p_{ref}) is 1 μPa under water and 20 μPa in air. This summation procedure essentially generates a single exposure "equivalent" value that assumes no recovery of hearing between repeated exposures. The appropriate units for underwater SEL are dB re: 1 μPa^2-s, and the appropriate units for aerial SEL are dB re: $(20\ \mu Pa)^2$-s.

Kurtosis is a statistical measure of the probability distribution of sound pressure amplitudes (Hamernik & Hsueh, 1991; Lei et al., 1994; Hamernik et al., 2003) that describes the shape of the amplitude distribution. In some regards, it appears to be a highly relevant metric in that impulsive sound with high kurtosis and high instantaneous peak pressure may be particularly injurious to some mammals (Hamernik et al., 2003). Kurtosis is related to the fourth central-moment and is defined for random variable X as

$$kurt(X) = \frac{O\left[(X-\mu)^4\right]}{\sigma^4} \quad (6)\ \text{eq.}$$

where O is the expectation operator, μ is the mean, and Σ is the standard deviation. When kurtosis is high, amplitude distribution is generally more centrally peaked and may have broader tails. Normal distributions have a kurtosis value of 3 independent of the mean or standard deviation.

Frequency-selective weighting is often employed to measure (as a single number) sound pressure or energy in a specific frequency band, with emphasis or de-emphasis on particular frequencies as a function of the sensitivity to those frequencies. For aerial hearing in humans, A-weighting is derived from the inverse of the idealized 40-phon equal loudness hearing function across frequencies standardized to 0 dB at 1 kHz (Harris, 1998), providing level measures denoted as dB(A). C-weighting is determined from the inverse of the idealized 100-phon equal loudness hearing function (which differs in several regards from the 40-phon function) standardized to 0 dB at 1 kHz (Harris, 1998); level measures are denoted as dB(C).

Absent equal-loudness contours for marine mammals, special weighting functions based loosely on human weighting functions and general knowledge of functional hearing bandwidth, were developed here for the five functional marine mammal hearing groups (see the "Marine Mammal Functional Hearing Groups" section in Chapter 2). M-weighting has a mathematical structure similar to the C-weighting used in human hearing, which reflects the fact that sounds must be more intense at high and low frequencies for them to have equal auditory effect. C-weighting is most appropriate for determining the effects of intense sounds—that is, those with loudness equal to that of a tone 100 dB above threshold at 1,000 Hz. The M-weighting was designed to do much the same for the different marine mammal groups with the only difference being their varying low- and high-frequency cutoffs. The M-weighting for marine mammals, like the C-weighting used in humans, rolls off at a rate of 12 dB per octave.

The general expression for M-weighting (M[*f*]), using estimated frequency cutoffs for each functional marine mammal hearing group, is given as

$$M(f) = 20\log_{10}\frac{R(f)}{\max\{|R(f)|\}} \quad (7)\ \text{eq.}$$

where

$$R(f) = \frac{f_{high}^2 f^2}{(f^2 + f_{high}^2)(f^2 + f_{low}^2)} \quad (8)\ \text{eq.}$$

The estimated lower and upper "functional" hearing limits (f_{low} and f_{high}) for each of the five functional marine mammal hearing groups and the names of the frequency-weighting functions are given in Table 2. The weighting functions de-emphasize frequencies that are near the lower and upper frequency ends of the estimated hearing range as indicated by the negative relative values in Figure 1.

Audition (hearing) is a well-developed and primary sensory modality for most, if not all, marine vertebrates (Schusterman, 1981; Tyack, 1998; Fay & Popper, 2000). The vertebrate hearing system involves coding, processing, integrating, and responding to sound in a variety of ways, some not outwardly evident (Yost, 2000).

Hearing (auditory) threshold is most commonly measured by behavioral or electrophysiological responses and is defined as the SPL of the quietest sound audible in some percentage of experimental trials. In air, measurements are often conducted in specially constructed sound chambers. When that is not possible, tests must be conducted in low background noise conditions to yield meaningful threshold data.

Sensation level represents the difference (in dB) between the overall level of a sound and the receiver's auditory threshold at similar sound frequencies. It is particularly useful as a means of comparing the relative exposure level of a sound

for individuals that may have different hearing capabilities. Sensation level is sometimes abbreviated SL, but this is not done in this document to avoid confusion with the very different concept of source level.

Auditory masking is the partial or complete reduction in the audibility of signals due to the presence of interfering noise (see Buus, 1997). The degree of masking depends on the spectral and temporal relationships between signals and masking noise as well as their respective RLs (e.g., Fletcher, 1940).

Sound localization is the determination of source location based on features of received sounds. This critical, complex process of the auditory system can involve the detection of sounds produced directly by a source (passive listening) or the detection of echoes reflected off a target (as in the case of biosonar).

Auditory scene analysis is the process by which the auditory system sorts out related elements of a complex acoustic environment into those arising from discrete sound sources. This process is similar to psychological processes underlying visual perception whereby many different visual images are perceived as discrete elements within a visual scene.

Temporary Threshold Shift (TTS) is a reversible elevation in hearing threshold (i.e., a nonpermanent reduction in hearing sensitivity) most commonly resulting from noise exposure.

Permanent Threshold Shift (PTS) is a permanent elevation in hearing threshold (i.e., an unrecoverable reduction in hearing sensitivity). PTS can occur from a variety of causes, but it is most often the result of intense and/or repeated noise exposures. In that case it is also referred to as noise induced hearing loss (NIHL) or noise induced permanent threshold shift (NIPTS).

Courtesy: A. Friedlander

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Appendix B. Studies Involving Marine Mammal Behavioral Responses to Multiple Pulses

Low-Frequency Cetaceans/Multiple Pulses (Cell 2)

Numerous field observations have been made of low-frequency cetaceans reacting to multiple pulses, either opportunistically exposed to ongoing operations or by intentional exposures. A moderate number of species and experimental conditions have been considered, but the source was usually a seismic airgun or arrays of these intense sources. Some studies focused on migrating animals observed from fixed observation platforms or in/near migration corridors.

The general results of the severity scaling analysis for this condition suggest the onset of more significant behavioral disturbances from multiple pulses for migrating bowhead whales at RLs (RMS over pulse duration) around 120 dB re: 1 µPa (Richardson et al., 1999). For all other low-frequency cetaceans (including feeding bowhead whales), this onset was at RLs around 150 to 160 dB re: 1 µPa (Malme et al., 1983, 1984; Richardson et al., 1986; Ljungblad et al., 1988; Todd et al., 1996; McCauley et al., 1998). There is essentially no overlap in the RLs associated with the onset of behavioral responses by members of these two groups based on the information currently available.

Seismic airguns operated near bowhead whales generally initiate avoidance reactions as well as changes in locomotion and respiration (Reeves et al., 1984; Richardson et al., 1985, 1986, 1999; Ljungblad et al., 1988). During the autumn migration, avoidance behavior has been observed at relatively great (20+ km) ranges from source operations (Koski & Johnson, 1987; Richardson et al., 1999). Ljungblad et al. (1988) did not investigate behavioral reactions over such ranges. During the summer, feeding bowheads exhibited subtle behavioral responses but not active avoidance at distances beyond 6 km from airgun sources (Richardson et al., 1986; see also Miller et al., 2005).

Richardson et al. (1999) studied autumn-migrating bowhead whale and found avoidance by most individual whales at distances up to 20 km and some avoidance at 20 to 30 km. Seismic surveys using airgun arrays with 6 to 16 guns and total volumes of 560 to 1,500 in^3 were conducted in shallow (generally < 20 m) water of the Alaskan Beaufort Sea. Whales in their westward autumn migration over three seasons (1996 to 1998) were detected with aerial surveys on days with and without seismic survey activity. Using the observations of dozens of migrating whales during periods when airguns were not active, we were able to calculate the percentage of observed whales during seismic surveys that demonstrated avoidance behavior at various RLs (see Table 7). These results indicate that migrating bowhead whales in the Richardson et al. (1999) study often avoided areas where RLs exceeded 120 to 130 dB re: 1 µPa (RMS over pulse duration).

In contrast, Richardson et al. (1986) observed quite different movement patterns of bowhead whales exposed to seismic airgun sounds on their summer feeding grounds in the Canadian Beaufort Sea. Received levels from a single seismic airgun (0.66-L) were measured *in situ* near individual whales being observed 3 to 5 km from the sound source, and ranged from 118 to 133 dB re: 1 µPa. Visual orientation by groups of whales during airgun exposure was observed on two of five occasions; only minor changes in swimming and respiration patterns were observed. Richardson et al. (1986) also made opportunistic observations of groups of bowhead whales near a seismic vessel operating an airgun array. At the highest RLs, some measurements exceeded the dynamic range of the recording equipment and are considered exposure minima, although this was not the case for most measurements relevant to the behavioral observations. From these observations and the controlled exposure to sounds from a single airgun, Richardson et al. (1986) concluded that some whales responded subtly by changing diving and breathing patterns at relatively low RLs (*ca*. 120 to 140 dB re: 1 µPa), but that avoidance and other more profound behavioral changes were generally not observed unless the RL was ≥ 160 dB re: 1 µPa.

Ljungblad et al. (1988) conducted a series of acoustic experiments on behavioral reactions of bowhead whales exposed to sounds from ships with operating airgun(s). Experiment 1 was conducted on a group of eight whales. When a seismic vessel approached to within 3.5 km (max. RL near observed individuals was 142 dB re: 1 µPa), the bowhead whales coalesced and moved in a tight group away from the approaching vessel.

Experiment 2 involved a group of three bowhead whales that demonstrated startle responses at the onset of sounds from an airgun 7 km away (max. measured RL was 165 dB re: 1 μPa). Behavior returned to pre-exposure values shortly after the operation was terminated. Experiment 3 involved a group of seven bowhead whales that demonstrated avoidance behavior at ranges of ~3.5 km (max. measured RL of 178 dB re: 1 μPa). Experiment 4 involved a group of 50 bowhead whales. Behavioral reactions were first observed at ranges of about 8 km (max. measured RLs of 157 dB re: 1 μPa) and avoidance behavior was noted at ~3 km (RLs ~165 dB re: 1 μPa). Avoidance behavior in this instance similarly abated shortly following cessation of exposure (and was thus assigned a behavioral score of 6).

Recent work on summering bowhead whales by Miller et al. (2005) also found that avoidance responses were limited to distances of at most a few kilometers and RLs exceeding 160 dB re: 1 μPa. Miller et al. conducted a monitoring program over two summers for various marine mammals offshore of the Mackenzie Delta in the Southeast Beaufort Sea before and during seismic surveys. They presented observational data from both vessel-based and aerial observations of bowhead whales, belugas, and several pinniped species. The general methodology is briefly discussed here as well as data on behavioral responses by low-frequency cetaceans (bowhead whales) and the corresponding rank on the severity scale. The airgun operations involved 3-D seismic profiling from a 67-m vessel using two identical 2,250 in^3 sleevegun arrays, each with 24 airguns. Shots were at 8-s intervals and at a depth of 5 m below the surface of the water. Surveys were conducted in very shallow water (13 m average). Acoustic monitoring with calibrated hydrophones across the 10 Hz to 24 kHz bandwidth was conducted while seismic operations were underway. Physical properties of the operational environment, and hence sound propagation in the shallow water environments, were highly variable, but RLs as a function of range from active airgun arrays were measured. Vessel-based observers and aerial surveyors used line-transect methods to monitor marine mammals in and adjacent to seismic operational areas, both before and during shooting. Bowhead whales observed during the periods coincident with seismic operations were presumed to be feeding (i.e., not migrating). Many bowheads (355 individuals in 232 groups) were seen by marine mammal observers aboard the seismic vessel. Sighting rates were lower and mean sighting distances were somewhat larger during seismic operations than at times when the airguns were not operating, but the zone of avoidance around active airguns was very limited. The approximate difference in mean sighting distance was ~600 m. Similarly, the aerial surveyors did not detect any large-scale avoidance of the airgun operations by bowheads. These observations were generally consistent for both years in which measurements were made and are generally consistent with the observations of Richardson et al. (1986) in the same region and season (summer). Animals not exhibiting observable behavioral reactions (response score: 0) were consistently sighted in areas where RLs very likely ranged from 130 to 180 dB re: 1 μPa. The general lack of sightings within a small area around the seismic vessel suggests behavioral avoidance (response score: 6) at RLs exceeding 180 dB re: 1 μPa. Exposures were not estimated to exceed 190 dB re: 1 μPa. The entire study was treated as a single observation for the purposes of the behavioral analysis. Half of the "observation" was scored as avoidance behavior and half as no response, with exposure RL bins from 130 to 190 dB re: 1 μPa (Table 6).

The combined data for bowhead whale avoidance of airgun sounds (Richardson et al., 1986, 1999; Ljungblad et al., 1988; Miller et al., 2005) indicated that, when migrating, these animals can be particularly prone to behavioral disturbance, with the onset of significant responses occurring at approximately 120 dB re: 1 μPa (RMS over pulse duration) (Table 6). In contrast, when feeding, they may show subtle effects at low RLs but only tend to display active avoidance at RLs exceeding 160 dB re: 1 μPa.

Low-frequency cetaceans, other than migrating bowhead whales, appear to be much more tolerant of exposure to multiple pulses, although data are limited to a few species and (primarily) airgun sources. Available data for species other than bowheads include reactions to opportunistic and intentional exposures of humpback whales (Malme et al., 1985; Todd et al., 1996; McCauley et al., 1998, 2000) and gray whales (Malme et al., 1983, 1984, 1986, 1988; also see review by Moore & Clarke, 2002). Todd et al. (1996), Malme et al. (1983, 1984), and McCauley et al. (1998) are included in the behavioral scoring analysis here because they contain sufficient information on exposures and individual responses of low-frequency cetaceans other than bowhead whales.

Todd et al. (1996) analyzed the impact of construction activity (explosions and drilling) on the entanglement of three foraging humpback whales off Newfoundland. They conducted observations of whale behavior during and following explosions and obtained acoustic measurements of underwater sound signatures. The data suggest few short-term changes in movement and behavior patterns in response to discrete exposures;

however, repeated exposures to high levels may have resulted in sensory impairment in whales and perhaps greater susceptibility to entanglement in fishing gear.

Malme et al. (1983, 1984) documented behavioral reactions of migrating gray whales to seismic pulses from both single airguns and an array. Only land-based observers were used, which meant that the observers could not have affected the whales' behavior. Both phases of the investigation yielded the general conclusion that RLs exceeding 160 dB re: 1 μPa (on an approximate RMS basis) were required to cause migrating gray whales to avoid airgun sounds, although statistically significant reactions that were less profound occurred at much larger ranges and lower levels. From their empirical phase II results, Malme et al. (1984) calculated 10, 50, and 90% probabilities of gray whale avoidance reactions in these conditions to be 164, 170, and 180 dB re: 1 μPa, respectively.

McCauley et al. (1998) made behavioral observations of migrating humpback whales off western Australia during seismic operations with a single airgun and several airgun array configurations. Seismic track lines were oriented perpendicular to the migration paths of humpback whales moving through the area. Aerial surveys were conducted to determine the presence of humpback whales moving through the survey area. Detailed observational data were presented for individuals and groups of whales; RLs were measured at variable ranges. The seismic survey did not appear to grossly affect the migration of humpback whales through the area; however, avoidance behavior was observed to begin at ranges from 5 to 8 km and to be almost universal at ranges of 1 to 4 km. Exposures to a single airgun (20 in^3) were extrapolated to equivalent ranges for exposure to full arrays based on empirical measurements. The data indicated an onset of behavioral avoidance at ~159 dB re: 1 μPa (peak-to-peak), roughly equivalent to the full array at 5 km. General behavioral avoidance (most individuals) occurred at a range of about 1 km for the single gun (~168 dB re: 1 μPa [peak-to-peak]), equivalent to the full array at about 3 km. Some individual whales did approach closer than the typical 3-km stand-off range; these may have been males investigating the presence of the low-frequency source.

In addition to presenting again the results given in the McCauley et al. (1998) paper, McCauley et al. (2000) provide additional behavioral observations of 16 humpback whale pods that approached as a single airgun (Bolt PAR 600b 20-in^3) was operated. These whales were also observed after termination of airgun operations. These trials were conducted in a large embayment (Exmouth Gulf) as the animals were engaged in a variety of resting and social behaviors. Five trials were excluded from consideration in our analysis, but behavioral observations were reported for the remaining 11. Of these, ten included cow pods of various sizes, and one was a lone male. Since the cow pods were not migrating and were not individually identified, a single behavioral observation is included in Table 7 for the ten observations. The results for the cow pods were very consistent, indicating clear avoidance (severity score = 6) of the airgun at exposures in the 140 to 150 dB re: 1 μPa range (RMS over pulse duration). The lone male essentially ignored the airgun until within *ca.* 100 m, when the received level approached 180 dB re: 1 μPa (RMS); this response may have had as much or more to do with the presence of the vessel than exposure to the airgun sound. Noting this contextual complexity here, a single observation for this individual is reported in the 170 to 180 dB re: 1 μPa exposure bin in Table 7 as general avoidance (severity score = 6).

Mid-Frequency Cetaceans/Multiple Pulses (Cell 5)

A limited number of behavioral observations have been made of mid-frequency cetaceans exposed to multiple pulses. Field observations have involved sperm whales and a few other odontocete species exposed to seismic airguns and small explosives (Madsen & Møhl, 2000; Madsen et al., 2002; Miller et al., 2005). Laboratory investigations have considered behavioral responses to various kinds of multiple pulse sources (Akamatsu et al., 1993). As in most criteria cells, a number of reported observations were not scored and reported here due to lack of relevant information and difficulties in accounting for various contextual variables. A summary of those studies used and others considered is given in Table 8; the severity scaling analysis for Cell 5 is shown in Table 9.

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

Field Observations (Cell 5)

Madsen & Møhl (2000) investigated sperm whale responses to small underwater detonators that included 1-g TNT charges, producing a 1-ms

broadband (300 Hz to 15 kHz) pulse; several charges were triggered per day. Echolocation click behavior was monitored, and one whale was localized acoustically. This individual demonstrated no modulation of vocal behavior when exposed to an RMS-equivalent RL of ~173 dB re: 1 µPa. There was also one observation of a whale exposed to 179 dB re: 1 µPa; it continued breathing normally with no visible response.

Madsen et al. (2002) studied responses of sperm whales in Norway to sounds associated with distant seismic survey operations. Calibrated RLs for individuals and correlated acoustic behavior are reported for three discrete sightings over a 5-d period. The first observation involved three sperm whales tracked by acoustic localization within a dispersed array of calibrated hydrophones, which also recorded airgun sounds from an array operating 40 km away. RL at the position of the whale was estimated to be 123 dB re: 1 µPa. The second observation (3 d later) involved a single sperm whale recorded before, during, and after airgun exposure at a range of 86 km; measured RL was 130 dB re: 1 µPa. The third observation (2 d later) involved three individuals; the survey vessel was 94 km away and measured RL was 130 dB re: 1 µPa. No change in sperm whale acoustic behavior was noted in any of these cases. The authors also played artificial codas and noticed that two whales directed their sonar beams at the speaker, but insufficient information is given to associate this response with a particular RL.

Miller et al. (2005) documented behavioral reactions of various marine mammal species, including belugas, to airgun operations. The general methodology is detailed above (see the "Cell 2" section). Owing to their normal seasonal patterns in the Beaufort Sea, belugas were most abundant in the Miller et al. (2005) study area prior to the start of seismic operations. There were relatively few vessel-based sightings, most of which were made when airguns were not active. Many belugas were observed during aerial surveys, however, and these data were used to compare beluga sightings within concentric 10-km bands around the active seismic source with sighting rates in non-airgun conditions. During airgun operations, Miller et al. detected significantly fewer animals 10 to 20 km from seismic operations and an unexpectedly high number of sightings in the 20- to 30-km zone. This was suggestive of behavioral avoidance of seismic operations at distances up to 20 km. These observations may in part explain why so few animals were observed by shipboard marine mammal observers. Miller et al. noted that the apparent avoidance of seismic operations was much greater than expected if the whales were responding to non-airgun sounds associated with vessel operation. For the purposes of our behavioral analyses, the combined beluga results were treated as a single observation that was subdivided equally into either avoidance behavior or no observable response. Belugas exposed to RLs of 100 to 120 dB re: 1 µPa (RMS over pulse duration) were determined to have had no observable reaction (response score: 0) to seismic exposures. RLs between 120 and 150 dB re: 1 µPa were determined to have induced temporary avoidance behavior (response score: 6) in belugas, based on the vessel-based and aerial observations. Based on both the vessel-based and aerial surveys, exposures apparently did not exceed 150 dB re: 1 µPa. Weighted behavioral response scores for each of these five exposure RL bins are given in Table 7.

Several studies involved behavioral reactions of free-ranging, mid-frequency cetaceans but lacked specific measures to be included directly in our analyses. André et al. (1997) exposed sperm whales to various stimuli, including two pulse sounds (recorded coda playbacks and a 10-kHz pulse). A significant number of exposed whales exhibited vocal modulations and modified diving behavior, but insufficient information is available on received exposures of individual whales. Stone (2003) compiled a large database of sighting data of several mid-frequency cetacean species observed from seismic survey vessels. Sighting rates of small odontocetes were significantly lower when airguns were firing, and they were sighted at greater distances from vessels, indicating avoidance behavior. The study sponsors (JNCC) kindly provided raw data for use in our quantitative avoidance analyses, but they are not included due to difficulties in estimating exposure RL for individual sightings. (See also Stone & Tasker, 2006, for a recently published account.)

Laboratory Observations (Cell 5)

Akamatsu et al. (1993) investigated avoidance behavior in two captive false killer whales exposed to 15 different kinds of sounds, including pulse sequences (manual strikes on a metal pipe once every 2 s) in the 24 to 115 kHz range. For this stimulus, no avoidance was seen following the first exposure (174 dB re: 1 µPa), but temporary avoidance behavior (response score: 6) was observed for successive exposures at 174 and 178 dB re: 1 µPa.

Finneran et al. (2000) observed behavioral responses of two captive bottlenose dolphins and a beluga whale during TTS experiments involving a series of impulsive exposures designed to replicate distant explosions. Each animal exhibited alterations of nominal trained behaviors (reluctance to return to experimental stations) during the experiment; the onset of behavioral disturbance

occurred in the beluga at 220 dB re: 1 μPa (peak-to-peak) and in the two bottlenose dolphins at 196 and 209 dB re: 1 μPa (peak-to-peak), respectively. In a related study, Finneran et al. (2002b) observed behavioral responses of a bottlenose dolphin and a beluga whale after exposure to impulsive sounds produced by a water gun. Both individuals showed a similar reluctance to return to experimental stations (beluga at 202 dB re: 1 μPa (peak-to-peak); bottlenose dolphin at 229 dB re: 1 μPa [peak-to-peak]). Romano et al. (2004) studied physiological responses to these exposures in these same animals. They observed clear neuro-immune responses in the beluga at exposures above 222 dB re: 1 μPa (peak-to-peak) and significant differences in aldosterone and monocyte counts in the dolphin for exposures exceeding 225 dB re: 1 μPa (peak-to-peak).

High-Frequency Cetaceans/Multiple Pulses (Cell 8)

Based on our source type distinction (see Chapter 2), virtually all sound sources used in behavioral studies of high-frequency cetaceans (e.g., acoustic harassment devices [AHDs] and acoustic deterrent devices [ADDs]) would be characterized as nonpulses. While individual elements produced by some of these sources would be characterized as pulses, and sequences of them as multiple pulses, they are generally emitted in such rapid fashion that mammalian auditory systems are likely to perceive them as nonpulses. Further, some AHDs, ADDs, and all other sources used in behavioral studies with high-frequency cetaceans lack the characteristics of pulses. Due to the lack of data, it is not possible to present any behavioral response data on multiple pulses for high-frequency cetaceans; available data for nonpulse sounds are considered elsewhere (see the "High-Frequency Cetaceans/Nonpulses [Cell 9]" sections of Chapter 4 and Appendix C). We note the need for behavioral research on these animals using sound sources unequivocally classified as pulses.

Pinnipeds in Water/Multiple Pulses (Cell 11)

Information on behavioral reactions of pinnipeds in water to multiple pulses is derived from studies using small explosives similar to those used in fisheries interactions, construction activity, 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. Our general finding is that exposures in the ~150 to 180 dB re: 1 μPa range (RMS over pulse duration) generally have limited potential to induce avoidance behavior in pinnipeds, whereas RLs exceeding 190 dB re: 1 μPa are likely to elicit responses, at least in some ringed seals (Harris et al., 2001; Blackwell et al., 2004b; Miller et al., 2005).

Harris et al. (2001) documented responses of pinnipeds (primarily ringed seals, but a few bearded and spotted seals) and obtained calibrated measures of RLs within defined spatial zones during operation of a single airgun, an 11-airgun array totaling 1,320 in^3, and during control periods. Visual observations from the seismic vessel were limited to the area within a few hundred meters, and 79% of the seals observed were within 250 m of the vessel. During daylight, seals were observed at nearly identical rates with no airguns, one airgun, or when a full airgun array was firing. Seals were significantly further away during full array operations compared to the other two conditions. Also, there was some avoidance within 150 m of the vessel in these conditions (0.37 seals seen per hour in control periods compared to 0.21 seals/h during full array operations). Seismic operations were not believed to cause many, if any, seals to desert the operational area.

Blackwell et al. (2004b) investigated behavioral reactions of ringed seals to impact sounds associated with the driving of steel pipes in the construction of an oil production facility. Multiple strikes were recorded under water at distances up to 3 km from the source. Unweighted peak pressure level, SPL, and SEL measurements were made at various distances. At the closest point (63 m), RLs were 151 dB re: 1 μPa (RMS), 157 dB re: 1 μPa (peak), and 145 dB re: 1 μPa^2-s (SEL). Pulses had measurable components extending to over 10 kHz, although more than 95% of the energy in the signals was below 225 Hz. A frequency-weighting metric somewhat similar to that proposed here was applied to the recorded signals in estimating audibility ranges. Individuals demonstrated no or low-level behavioral responses to pile-driving sounds, but were somewhat responsive to helicopter overflights. Blackwell et al. noted, however, that their data were collected after a prolonged period of intensive construction activity and may reflect the least responsive part of the original population of seals that may have already habituated to the noise source. Individual observations in which helicopters were not present are considered in our behavioral analysis, weighted by the total number of relevant observations (Table 11). Aerial measurements of multiple pulse exposures were also obtained in this study and are considered in the relevant condition below.

Miller et al. (2005) documented behavioral reactions of various marine mammal species, including pinnipeds in water, to seismic airgun

operations. The general methodology is detailed above (see the "Cell 2" section). The vast majority (> 90%) of the seals were ringed seals and the remainder were bearded seals. Vessel-based observers saw seals around the vessel, and often quite close to it, throughout the period of seismic operations. Seals were observed significantly further away during airgun operations in the first summer, whereas the reverse pattern was actually the case in the second season. Combined, the results suggest essentially no observable behavioral response in pinnipeds exposed to seismic signals in these specific conditions. Based on the acoustic measurements that were conducted and the areas in which these pinnipeds were observed, RLs were likely 170 to 200 dB re: 1 μPa (RMS over pulse duration). A single observation of no reaction (response score: 0) for pinnipeds in water is reported for this study and is weighted equally across these exposure RL bins (Table 8).

Several other studies were deleted from our analysis due to a lack of certain information. Two studies investigated small firecracker-like explosives (called "seal bombs") and their effect on the underwater behavior of pinnipeds around fishing gear (Shaughnessy et al., 1981; Mate & Harvey, 1987). Initially, these explosives tend to induce the desired avoidance behavior, but this response fades quickly due to habituation (see Richardson et al., 1995). Mate & Harvey (1987) reported fairly extensive descriptions of startle and temporary avoidance data as well as some information on exposure conditions. Besides the challenging matter of interpreting the apparently rapid habituation to this sound source, however, data are lacking that relate discrete exposures with defined behavioral responses of specific individual pinnipeds. For these reasons, we excluded data on responses to seal bombs from our analysis. Moulton et al. (2003, 2005) conducted surveys of ringed seal distribution before and during the construction and operation of the same oil production facility described by Blackwell et al. (2004a, 2004b). Sound sources included nonpulse as well as multiple pulse sources (including impact pile-driving). Their observations across multiple seasons indicated little or no behavioral avoidance of the area in response to various industrial activities. Due to difficulties with control observations across seasons and the lack of information about discrete exposures and individual reactions, however, we excluded the Moulton et al. (2003, 2005) data from our analysis. A final study for which available data were insufficient for inclusion here is Thompson et al. (1998). That telemetry study seemed to show much higher responsiveness of gray and harbor seals to airgun sounds than has been demonstrated in other studies, which relied on visual observations. Thus, future studies may show some seals to be more responsive to multiple pulses than Table 11 would suggest.

Pinnipeds in Air/Multiple Pulses (Cell 11)

The effects of multiple aerial pulses on pinnipeds are among the least well-documented of the conditions we considered. Most of the available data on responses to pulses are 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 launches are not repeated so frequently that the exposure can be considered as involving multiple pulses, and many of the exposures include nonpulse components. However, they are discussed in some detail in this appendix (as well as in Appendix C for nonpulses within these studies) along with several other studies potentially relevant to Cell 14 but ultimately not used in the analysis here. Consequently, the quantitative information analyzed for reactions of pinnipeds in air exposed to multiple pulses (see Table 12 for summary and Table 13 for severity scaling analysis) focuses on the aerial data of Blackwell et al. (2004b). These extremely limited data suggest very minor, if any, observable behavioral responses for exposures ranging from 60 to 80 dB re: 20 μPa.

Blackwell et al. (2004b) reported behavioral reactions of ringed seals to aerial impact sounds from pile-driving (described above). Multiple strikes were recorded in air at distances up to 500 m from the source. Unweighted SPL, peak sound pressure levels, and SEL measurements were made at various distances. At the closest point (63 m) average RLs were 93 dB re: 20 μPa (RMS), 111 dB re: 20 μPa (peak), and 87 dB re: (20 μPa)^2-s (SEL). Mean pulse durations were between 0.17 and 0.63 s, with measurable energy to over 10 kHz, but with 95% of the energy occurring between 89 and 3,534 Hz. A frequency-weighting metric somewhat similar to that proposed here was applied to the recorded signals in estimating audibility ranges. Individuals demonstrated very limited behavioral responses to pile-driving sounds in some conditions (most appeared either "indifferent or curious") but were more responsive to helicopter overflights. Data were collected after prolonged construction activities, and some habituation probably had taken place already. Individual observations for which helicopters were not present are considered in the behavioral analysis here and weighted by the total number of relevant observations (Table 13) to equal a single observation for the study.

Perry et al. (2002) measured the effects of repeated (0 to 5/d) sonic booms from Concorde aircraft on harbor and gray seals on Sable Island, Nova Scotia. They measured the number of animals on shore before and after booms as well as the frequency of various behaviors. Additionally, they compared heart rates in exposure and control conditions using recording devices deployed on the animals. They reported received sound overpressure of booms on the breeding beaches of both pinniped species. Observed effects on animal presence, behavior, and heart rate were generally minor and not statistically significant; animals were largely tolerant of the sounds but became somewhat more alert following them. However, Perry et al. (2002) note that there is a long history of sonic booms from aircraft in the area and the animals are likely habituated to their presence. Due to this complication and the lack of explicit received SPL measures at exposed individuals, we did not score the results of Perry et al. (2002) here.

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Appendix C. Studies Involving Marine Mammal Behavioral Responses to 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, 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; Richardson et al., 1990b; McCauley et al., 1996; Frankel & Clark, 1998; Borggaard et al., 1999; Biassoni et al., 2000; Croll et al., 2001; Palka & Hammond, 2001; Nowacek et al., 2004). Observations from several related studies (Dahlheim, 1987; Frankel & Clark, 2000, 2002; Schick & Urban, 2000; Moore & Clarke, 2002; Jahoda et al., 2003; Mobley, 2005) were reviewed briefly but not analyzed here because key information was lacking.

These papers generally indicate no (or very limited) responses at RLs 90 to 120 dB re: 1 μPa and an increasing probability of avoidance and other behavioral effects in the 120 to 160 dB re: 1 μPa range (Table 14). However, the data also indicate 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.

Baker et al. (1982) investigated behavioral responses of individual humpback whales to vessel traffic in southeast Alaska. Individual RLs were not reported, but sufficient information regarding individual ranges was obtained to approximate exposures given that the acoustic characteristics of identical classes of vessel classes involved were measured in similar conditions by Miles & Malme (1983). Results indicate some behavioral avoidance when RL was in the 110 to 120 dB re: 1 μPa range and clear avoidance at 120 to 140 dB re: 1 μPa.

Malme et al. (1983, 1984) used playback methods to document behavioral reactions of migrating gray whales to intermittent sounds of helicopter overflights and continuous sounds from drilling rigs and platforms. Both phases of the investigation yielded the general conclusion that RLs exceeding 120 dB re: 1 μPa induced demonstrable behavioral reactions (avoidance). Malme et al. (1984) calculated 10%, 50%, and 90% probabilities of gray whale avoidance reactions in these conditions as 110, 120, and 130 dB re: 1 μPa. Malme et al. (1986) observed the behavior of feeding gray whales during four experimental playbacks of drilling sounds (50 to 315 Hz; 21-min overall duration and 10% duty cycle; source levels 156 to 162 dB re: 1 μPa-m). In two cases for RLs 100 to 110 dB re: 1 μPa, there was no observed behavioral reaction. Avoidance behavior was observed in two cases where RLs were 110 to 120 dB re: 1 μPa.

Richardson et al. (1990b) performed 12 playback experiments in which bowhead whales in the Alaskan Arctic were exposed to drilling sounds. Low-frequency source characteristics and transmission loss were well-characterized, enabling RL estimates to be made for individual cases. Whales generally did not respond to exposures in the 100 to 130 dB re: 1 μPa range, although there was some indication of minor behavioral changes in several instances.

Using different detection and sampling techniques, McCauley et al. (1996) reported several cases of humpback whales responding to vessels in Hervey Bay, Australia, along with measurements of noise RL. Not all cases reported provided sufficient information to associate a response or lack of response with exposure, but in three cases, individual responses and noise RL were reported. Results indicated clear avoidance at RLs between 118 to 124 dB re: 1 μPa.

Palka & Hammond (2001) analyzed line transect census data in which the orientation and distance off transect line were reported for large numbers of minke whales. General additive models were used to estimate the range at which cetaceans respond to the noise of the research vessel by approach or avoidance. The typical avoidance distance for 272 minke whales in the Gulf of Maine was 717 m; for 352 minke whales in the North Sea, it was 563 m; and for 493 minke whales in the Northeastern Atlantic, it was 695 m. Received levels were estimated based on a nominal source level for that class of research vessel (*ca*. 170 to 175 dB re: 1 μPa-m) and an assumption of spherical (20 log R) spreading loss (54 dB loss @ 500 m; likely

reasonable for these conditions). These data are represented in Table 14 by the 110 to 120 dB re: 1 µPa exposures and a relatively low (less severe) behavioral response score of three (i.e., minor changes in locomotion speed, direction, and/or diving profile).

Several additional studies have used playback experiments with active sound sources to investigate the behavioral reactions of low-frequency cetaceans to nonpulse sources. Biassoni et al. (2000) and Miller et al. (2000) report behavioral observations for humpback whales exposed to a low-frequency sonar stimulus (160- to 330-Hz frequency band; 42-s tonal signal repeated every 6 min; source levels 170 to 200 dB re: 1 µPa-m). Measured RLs ranged from 120 to 150 dB re: 1 µPa. In nine cases, individual whales continued singing throughout exposures, while in four instances, individuals ceased calling when they joined another whale. The cessation of song and joining another individual is typical of normal mysticete social interactions (Tyack, 1981). Consequently, these events were not scored as a vocal response to the playback but as a moderate orienting behavior (severity score = 2). For the remaining five playbacks, individual whales stopped singing during exposure without joining other whales (severity scale = 4). Although singers also stop spontaneously under control conditions, the latter five experimental trials were considered vocal cessation resulting from sound exposure (Biassoni et al., 2000). However, there are insufficient data to compare control and experimental cases for spontaneous rates of cessation. Analysis of all singers indicated an increase in song duration during exposure due to increased repetition of elements of the song. Since it was possible that some individual whales were represented multiple times within the playbacks, the Biassoni et al. (2000) and Miller et al. (2000) data were scored as a single behavioral observation. The 18 individual observations were weighed inversely by the total number (1/18) in Table 15.

Croll et al. (2001) investigated responses of foraging fin and blue whales to the same LFA sonar stimulus off southern California. Unlike the previous two studies, where individual experimental subjects were tracked on a behavioral scale, this study used sighting data on an ecological scale. Playbacks and control intervals with no transmission were used to investigate behavior and distribution on time scales of several weeks and spatial scales of tens of kilometers. Sightings and whale diving behavior were not random but were related to environmental features such as the continental shelf break and its effects on prey abundance rather than operation and location of the nonpulse sonar source. The general conclusion was that whales remained feeding within a region for which 12 to 30% of exposures exceeded 140 dB re: 1 µPa. A single observation was scored for this study because individual responses were not reported.

Frankel & Clark (1998) conducted playback experiments with wintering humpback whales around the Big Island of Hawai'i. The sound source was a single speaker producing a low-frequency "M-sequence" (sine wave with multiple-phase reversals) signal in the 60 to 90 Hz band. This was similar in bandwidth to the ATOC source, but had a much lower output level (172 dB re: 1 µPa @ 1 m). A vertical line array of calibrated hydrophones was deployed from a spar buoy to measure received signals *in situ*. Detailed observations of many behavioral patterns (including respiration, diving, and general movements) were recorded before, during, and after playback (n = 50) and control (n = 34) sequences. A single trial also involved playback of humpback foraging sounds. Most of the playback sequences involved very low-level RLs, *ca*. 90 to 120 dB re: 1 µPa, though not specified in sufficient detail to include in the analysis here. For 11 playbacks, exposures were between 120 and 130 dB re: 1 µPa and included sufficient information regarding individual responses. During eight of the trials, there were no measurable differences in tracks or bearings relative to control conditions, whereas on three occasions, whales either moved slightly away from (n = 1) or towards (n = 2) the playback speaker during exposure. Because it was not possible to determine whether the same individual whales were represented more than once in the playback sequences, a single observation was recorded for Frankel & Clark (1998), with 0.73 of this observation (8/11) scored as a 0 (no response) and 0.27 (3/11) scored as a 3 (minor changes in locomotion speed, direction, and/or diving). A final important observation from the detailed statistical analysis by Frankel & Clark was that the presence of the source vessel itself had a greater effect than did the M-sequence playback.

Finally, Nowacek et al. (2004) used controlled exposures to demonstrate behavioral reactions of northern right whales to various nonpulse sounds. Playback stimuli included ship noise; social sounds of conspecifics; and a complex, 18-min "alert" sound consisting of repetitions of three different artificial signals (alternating 1-s pure tones [500 and 850 Hz]; a 2-s, tonal, frequency downsweep [4,500 to 500 Hz]; and a pair of 1-s pure tones [1,500 Hz and 2,000 Hz] amplitude modulated at 120 Hz). A total of ten whales were tagged with calibrated instruments that measured received sound characteristics and concurrent animal movements in three dimensions. Five out of six exposed whales reacted strongly to alert

signals at measured RLs between 130 and 150 dB re: 1 μPa (i.e., ceased foraging and swam rapidly to the surface; severity scale = 7). Two of these individuals were not exposed to ship noise and are given as a discrete observation in Table 15, whereas the other four were exposed to both stimuli and thus weighted as 0.5 (1/2) observations for the respective RL and severity score. These whales reacted mildly to conspecific signals (not scored here because of biological signals). Seven whales, including the four exposed to the alert stimulus, had no measurable response to either ship sounds or actual vessel noise. This study by Nowacek et al. included the careful experimental design, controls, and detailed information on exposure and individual behavioral response that were required for behavioral analysis. More studies of this type and rigor are urgently needed (see Chapter 5).

We reviewed additional studies concerning low-frequency cetaceans and nonpulse sounds but did not include them in the analysis here, generally due to the absence of key information. Dahlheim (1987) exposed gray whales to playbacks of outboard noise, gray whale calls, and tonal sounds. Whales significantly increased calling rate and modified call structure for sources other than the test tone (the latter caused all vocalization to cease). During and following longer duration playbacks of oil drilling and killer whale sounds with more precise tracking of gray whale locations, individuals spent more time milling, and whales remained farther offshore during killer whale playbacks. Unfortunately, insufficient information is presented to associate changes with specific RLs. Borggaard et al. (1999) measured the effects of industrial activity on several mysticete species in Newfoundland, but insufficient information is reported on individually discernible responses. Schick & Urban (2000) applied statistical methods to assess spatial avoidance of active drilling rigs by bowhead whales, but no acoustic data are reported. Moore & Clarke (2002) synthesized previously published data (all considered separately above) on numerous nonpulse sources, in order to assess the avoidance probability of gray whales for various exposure RLs. Jahoda et al. (2003) studied individual responses of fin whales ($n = 25$) to close rapid approaches of small vessels; 18 observations included control and experimental data. Clear behavioral responses were observed, but neither RL nor range from source to individuals were given. Results are further complicated by whale tagging attempts from the vessel. Frankel & Clark (2000) and Mobley (2005) investigated the distribution of humpback whales in Hawai'i in relation to the operation of a low-frequency tomographic source (*ca.* 75 Hz; 37.5-Hz nominal bandwidth; 20-min duration every 2 h during daylight hours; source level: 195 dB re: 1 μPa-m). Frankel & Clark (2000) observed whales from a land station and determined that the average distance between the sound source and the whale groups sighted was significantly greater during source operation. These and other data were also considered in the context of other factors affecting humpback whale distribution off the island of Kaua'i. Mobley (2005) conducted aerial surveys in each of three years (2001, source off; 2002 & 2003, source on) during the peak season of humpback residency. Abundance and distribution of whales were very similar in the area surrounding the source over all three years; small differences in sighting rates, sighting location depth, and distances from the source and shore were not statistically significant. Frankel & Clark (2002) and Mobley (2005) lack explicit data on RLs associated with individual behavioral observations, which precludes their inclusion here.

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 ADD/AHDs (Watkins & Schevill, 1975; Morton & Symonds, 2002; Monteiro-Neto et al., 2004), industrial activities (Awbrey & Stewart, 1983; Richardson et al., 1990b), mid-frequency active military 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. 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. The relevance of behavioral reactions of trained, food-reinforced captive animals exposed to noise in assessing reactions of free-ranging marine mammals is not well-known, however (discussed below).

The combined field and laboratory data for mid-frequency cetaceans exposed to nonpulse sounds do not lead us to a clear conclusion about

RLs coincident with various behavioral responses (see severity scaling, Table 17). In some settings, individuals in the field showed profound (and what we regard here as significant) behavioral responses to exposures from 90 to 120 dB re: 1 µPa, while others failed to exhibit such responses for exposure RLs from 120 to 150 dB re: 1 µPa. Contextual variables other than exposure RL, and probable species differences, are the likely reasons for this variability. Context, including the fact that captive subjects were often directly reinforced with food for tolerating noise exposure, 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 µPa before inducing behavioral responses.

Field Observations (Cell 6)

The most extensive series of observations regarding vessels and watercraft is from LGL and Greeneridge (1986) and Finley et al. (1990), who documented belugas and narwhals (*Monodon monoceros*) congregated near ice edges reacting to the approach and passage of ice-breaking ships. Over a 3-y period (1982 to 1984), they used both ice-based local observations of whales and aerial surveys, and also made detailed acoustic measurements. The survey method made it difficult to assess independent groups of animals. Some large-scale groupings could be identified for several different "disturbance" periods, however. Pre-disturbance group size was ~3; we divided reported numbers of disturbed "herds" by three to estimate the number of independent groups. Aerial surveys in 1984 lumped sightings by minute, which corresponded to about 3.4 km in distance. We considered this distance sufficient to treat each minute as an independent unit for avoidance analysis. The responses of both species over a 3-y period were generally similar to responses they make to predators as described by Inuit hunters.

Beluga whales responded to oncoming vessels by (1) fleeing at speeds of up 20 km/h from distances of 20 to 80 km, (2) abandoning normal pod structure, and (3) modifying vocal behavior and/or emitting alarm calls. Narwhals, in contrast, generally demonstrated a "freeze" response, lying motionless or swimming slowly away (as far as 37 km down the ice edge), huddling in groups, and ceasing sound production. There was some evidence of habituation and reduced avoidance 2 to 3 d after onset. Due to the detailed and extensive nature of these observations, data from each season, and how they are interpreted here, are given in detail.

The 1982 season observations by LGL & Greeneridge (1986) involved a single passage of an icebreaker with both ice-based and aerial measurements on 28 June 1982. Four groups of narwhals (n = 9 to 10, 7, 7, and 6) responded when the ship was 6.4 km away (exposure RLs of ~100 dB re: 1 µPa in the 150- to 1,150-Hz band). At a later point, observers sighted belugas moving away from the source at > 20 km (exposure RLs of ~90 dB re: 1 µPa in the 150- to 1,150-Hz band). The total number of animals observed fleeing was about 300, suggesting approximately 100 independent groups (of three individuals each), which is the sample size used here. No whales were sighted the following day, but some were sighted on 30 June, with ship noise audible at spectrum levels of approximately 55 dB re 1 µPa/Hz (up to 4 kHz).

Observations during 1983 (LGL & Greeneridge, 1986) involved two ice-breaking ships with aerial survey and ice-based observations during seven sampling periods. As the first vessel approached at a distance of about 65 km, ice-based observers noted reactions from both narwhals (seven groups) and belugas (eight groups) (exposure RLs of ~101 to 105 dB re: 1 µPa in the 20- to 1,000-Hz band). After 22 h without operation, the vessel commenced ice-breaking, and a second icebreaker approached (exposure RLs of ~120 dB re: 1 µPa in the 20- to 1,000-Hz band). This resulted in the rapid movement of > 225 belugas (estimated as a sample size of 75 for this analysis); belugas were neither seen nor heard for the remainder of the second observation period. Behavioral responses were also observed for 10 groups of narwhals. A total of 73 narwhals were seen and/or heard, but their reactions are not clearly reported and are thus excluded from analysis here. At the onset of the third sampling period, following a 4.5-h silent interval, four narwhal groups were observed in nominal social behavior (diving and vocalizing). An ice-breaking vessel operated intermittently, but no change was observed in narwhal behavior. Belugas in the area did modify vocalization parameters during operations (exposure RLs of ~116 dB re: 1 µPa in the 20- to 1,000-Hz band). A 6-h quiet interval was followed by 10.5 h of ice-breaking operation, but bad weather precluded animal observations. After an additional 9-h hiatus, ice-breaking commenced again by both vessels (exposure RLs of ~121 dB re: 1 µPa in the 20- to 1,000-Hz band). Ice-based observers documented 14 narwhals and 11 belugas leaving the area, and aerial surveys indicated 80% of 673 belugas moving away from sound sources (estimated number of groups calculated as [.8]*[673/3] = 179.5). As noise levels from ice-breaking operations diminished, a total of 45 narwhals returned to the area and engaged in diving and foraging behavior. The sixth observation period followed 6.5 h without a vessel in the area,

during which 30 belugas (estimated as 10 groups) and 15 narwhals (estimated as five groups) were observed diving in the area (exposure RLs of ~105 dB re: 1 µPa in the 20- to 1,000-Hz band). A single beluga vocal response was noted at RL = 116 dB re: 1 µPa in the 20- to 1,000-Hz band. Aerial surveys indicated dense concentrations of narwhals (n = 50) and belugas (n = 400) apparently foraging well away from the disturbance site. During the final sampling period, following an 8-h quiet interval, no reactions were seen from 28 narwhals and 17 belugas (exposure RLs ranging up to 115 dB re: 1 µPa).

The final season (1984) reported in LGL & Greeneridge (1986) involved aerial surveys before, during, and after the passage of two ice-breaking ships. The lack of ice camps precluded acoustic measurements as well as behavioral observations. A preliminary survey was conducted the day before operations, and an additional aerial survey was conducted as both ships commenced operating. During operations, no belugas and few narwhals were observed in an area approximately 27 km ahead of the vessels, and all whales sighted over 20 to 80 km from the ships were swimming strongly away. Additional observations confirm the remarkable spatial extent of avoidance reactions to this sound source in this context. In the absence of acoustic measurements, however, it was necessary to estimate RLs from the detailed data from the same ice-breaking vessel during the previous season.

Behavioral responses at fairly low exposure RLs are suggested by studies of some other mid-frequency cetaceans as well. Gordon et al. (1992) conducted opportunistic visual and acoustic monitoring of sperm whales in New Zealand exposed to nearby whale-watching boats (within 450 m). Individuals could not be used as the units of analysis because it was difficult to re-sight specific individuals during both exposure and control conditions. Sperm whales respired significantly less frequently, had shorter surface intervals, and took longer to start clicking at the start of a dive descent when boats were nearby than when they were absent. Noise spectrum levels of whale-watching boats ranged from 109 to 129 dB re: 1 µPa/Hz. Over a bandwidth of 100 to 6,000 Hz, equivalent broadband source levels are ~157 dB re: 1 µPa-m; RLs at a range of 450 m are ~104 dB re: 1 µPa.

Palka & Hammond (2001) applied a General Additive Model to line transect data to estimate the range at which mid-frequency cetaceans typically responded to the noise of research vessels. The subjects were Atlantic white-sided dolphins in the Gulf of Maine and white-beaked dolphins (*Lagenorhynchus albirostris*) in the North Sea. The white-sided dolphins exhibited simple avoidance behavior (as indicated by their orientations) out to an estimated range of 592 m based on 85 group sightings (n > 1). White-beaked dolphins actually approached vessels between 150 and 300 m away, but demonstrated avoidance at distances of 300 to 700 m. Typical avoidance distance was estimated as 716 m based on 48 groups sighted.

Buckstaff (2004), using repeated samples of the behavior of 14 individual bottlenose dolphins, observed 1,233 vessel approaches (within 400 m) near Sarasota, Florida. Dolphin whistle rates became elevated before vessel noise was detectable to the researcher listening via towed hydrophones. Vessel RLs measured near dolphin subjects ranged from 113 to 138 dB re: 1 µPa. Dolphin vocal responses were observed before vessel sounds were audible, and apparently occurred with RLs in the 110 to < 120 dB re: 1 µPa category.

Morisaka et al. (2005) compared whistles from three populations of Indo-Pacific bottlenose dolphins (*Tursiops aduncus*). One population was exposed to vessel noise with spectrum levels of ~85 dB re: 1 µPa/Hz in the 1- to 22-kHz band (broadband RL ~128 dB re: 1 µPa) as opposed to ~65 dB re: 1 µPa/Hz in the same band (broadband RL ~108 dB re: 1 µPa) for the other two sites. Dolphin whistles in the noisier environment had lower fundamental frequencies and less frequency modulation, suggesting a shift in sound parameters as a result of increased ambient noise.

Morton & Symonds (2002) used census data on killer whales in British Columbia to evaluate avoidance of nonpulse AHD sources. They considered unusually long time scales, comparing pre-exposure data from 1985 to 1992, exposure from 1993 to 1998, and post-exposure from 1999 to 2000. The response data were simply presence or absence, making it difficult to assess RLs. Using some monitoring and reasonable assumptions, however, they estimated audibility ranges throughout the complex study area. Avoidance ranges were *ca.* 4 km. Also, there was a dramatic reduction in the number of days "resident" killer whales were sighted during AHD-active periods compared to pre- and post-exposure periods and a nearby control site. Morton & Symonds did not indicate how many pods were involved in their analysis. Consequently, we assume a single independent group in our analysis.

Monteiro-Neto et al. (2004) studied avoidance responses of tucuxi (*Sotalia fluviatilis*) to Dukane® Netmark ADDs. Source characteristics are not given, but identical devices were used by Culik et al. (2001), and acoustic parameters are reported in detail there (and in the "Cell 9" section). In a total of 30 exposure trials, ~5 groups each demonstrated

significant avoidance compared to 20 pinger off and 55 no-pinger control trials over two quadrats of about 0.5 km^2. Neither avoidance range nor RLs are given, but based upon a central distance from the quadrat of 10 m, and assuming 15 log R transmission loss in this shallow environment (water depth 1 to 5 m), estimated exposure RLs were ~115 dB re: 1 µPa.

The only specific situation involving exposure of wild marine mammals to active mid-frequency military sonar for which exposure conditions are known with any degree of specificity involved incidental exposure of killer whales to sounds from the naval vessel USS *Shoup* (NRL, 2004a, 2004b; NMFS, 2005). A group (J-pod) of southern resident killer whales in the eastern Strait of Juan de Fuca and Haro Strait, Washington, was observed by researchers before, during, and after the approaching USS *Shoup* transmitted sonar signals from its 53C sonar at a source level of *ca*. 235 dB re: 1 µPa-m once every *ca*. 28 s for several hours. At its point of closest approach, the mean direct-path RL within a specified area around the animals was *ca*. 169 dB re: 1 µPa (NRL, 2004a, 2004b). As indicated by NMFS (2005), there is some discrepancy in interpretation of the behavioral responses among researchers who were either on the water or who observed video recordings of behavioral responses. The lead researcher following and observing the animals during the event indicated that individuals in the group demonstrated abnormal avoidance behavior, most dramatically at the point of closest approach. However, the behavior of the whales apparently returned to normal within a short period following cessation of sonar transmissions. A severity score of 6 (mild/moderate avoidance) is subsequently reported in the 160 to 170 dB re: 1 µPa bin for this single observation of the group.

Awbrey & Stewart (1983) played back semi-submersible drillship sounds (source level: 163 dB re: 1 µPa-m) to belugas in Alaska. They reported avoidance reactions at 300 and 1,500 m and approach by groups at a distance of 3,500 m (RLs ~110 to 145 dB re: 1 µPa over these ranges assuming a 15 log R transmission loss). Similarly, Richardson et al. (1990b) played back drilling platform sounds (source level: 163 dB re: 1 µPa-m) to belugas in Alaska. They conducted aerial observations of eight individuals among ~100 spread over an area several hundred meters to several kilometers from the sound source and found no obvious reactions. Moderate changes in movement were noted for three groups swimming within 200 m of the sound projector.

A number of additional studies (Rendell & Gordon, 1999; Chilvers & Corkeron, 2001; Bordino et al., 2002; Williams et al., 2002; Cox et al., 2003; Hastie et al., 2003; Lusseau, 2003; Foote et al., 2004; Scheifele et al., 2005) were reviewed in detail. The results were excluded from Table 17 due to limited or no information on animal numbers and/or location relative to the source, acoustic properties of sources, propagation variables, or received exposure conditions. The general observations of each study are given here briefly. Hastie et al. (2003) documented increased swimming and diving synchrony of bottlenose dolphins off northern Scotland in the presence of vessel traffic. Lusseau (2003) observed effects on behavior of New Zealand bottlenose dolphins within 400 m of boats. Chilvers & Corkeron (2001) considered differences in behavior of bottlenose dolphins that do and do not forage around trawlers. Williams et al. (2002) observed that some killer whales adopt erratic movement patterns, suggestive of avoidance, when whale-watching vessels accelerate to intersect the whale's course. RLs of vessel sound increased approximately 14 dB with increased speed associated with leapfrogging. Bordino et al. (2002) determined that ADDs were initially effective at reducing by-catch of Franciscana dolphins in Argentine subsistence gillnet fisheries. Cox et al. (2003) investigated reactions of bottlenose dolphins to Dukane® NetMark 1000 ADDs attached to commercial gillnets and found very limited to no behavioral avoidance. A group of long-finned pilot whales (*Globicephala melas*) demonstrated significant elevations of whistle rates following each exposure to mid-frequency military sonar reported to be at a "high" level (Rendell & Gordon, 1999).

Finally, two recent papers deal with important issues relating to changes in marine mammal vocal behavior as a function of variable background noise levels. Foote et al. (2004) found increases in the duration of killer whale calls over the period 1977 to 2003, during which time vessel traffic in Puget Sound, and particularly whale-watching boats around the animals, increased dramatically. Scheifele et al. (2005) demonstrated that belugas in the St. Lawrence River increased the levels of their vocalizations as a function of the background noise level (the "Lombard Effect"). (See also Parks et al., 2007, for a related new paper on mysticetes.) These papers demonstrate some degree of plasticity in the vocal signal parameters of marine mammals in response to the ambient condition (likely affected by the presence of human sound sources). These studies were not particularly amenable to the kind of analysis conducted in the severity scaling. We note the particular importance of direct measurements of noise impacts on marine mammal vocalization and communication systems.

Laboratory Observations (Cell 6)
Several researchers conducting laboratory experiments on hearing and the effects of nonpulse sounds on hearing in mid-frequency cetaceans have reported concurrent behavioral responses. Nachtigall et al. (2003) reported that noise exposures up to 179 dB re: 1 µPa and 55-min duration affected the trained behaviors of a bottlenose dolphin participating in a TTS experiment. Finneran & Schlundt (2004) provided a detailed, comprehensive analysis of the behavioral responses of belugas and bottlenose dolphins to 1-s tones (RLs 160 to 202 dB re: 1 µPa) in the context of TTS experiments. Romano et al. (2004) investigated the physiological responses of a bottlenose dolphin and a beluga exposed to these tonal exposures and demonstrated a decrease in blood cortisol levels during a series of exposures between 130 and 201 dB re: 1 µPa. Collectively, the laboratory observations suggested the onset of behavioral response at higher RLs than did field studies (see Table 16). The differences were likely related to the very different conditions and contextual variables between untrained, free-ranging individuals vs laboratory subjects that were rewarded with food for tolerating noise exposure.

High-Frequency Cetaceans/Nonpulses (Cell 9)

Numerous controlled studies have been conducted recently on the behavioral reactions of high-frequency cetaceans to various nonpulse sound sources both in the field (Culik et al., 2001; Johnston, 2002; Olesiuk et al., 2002) and in laboratory settings (Kastelein et al., 1997, 2000, 2005, 2006a). However, only one high-frequency species (harbor porpoise) has been extensively studied. For that species, sufficient data are available to estimate behavioral response magnitude vs received exposure conditions. The original studies were attempts to reduce harbor porpoise by-catch by attaching warning pingers to fishing gear. More recent studies consider whether ADDs and AHDs 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) 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 µPa), 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 µPa induced profound and sustained avoidance behavior in wild harbor porpoises. Harbor porpoises also tend to avoid boats, although Dall's porpoises do not (Richardson et al., 1995). Whether this apparently high degree of behavioral sensitivity by harbor porpoises to anthropogenic sounds extends to other high-frequency cetacean species (or to nonpulse sources other than ADDs, AHDs, and boats) is unknown. However, given the lack of information to the contrary, such a relationship should be assumed as a precautionary measure.

Habituation to sound exposure was noted in some but not all studies. In certain field conditions, strong initial reactions of high-frequency cetaceans at relatively low RLs appeared to wane rather rapidly with repeated exposure (Cox et al., 2001). In contrast, several laboratory observations showed little or no indication of reduced behavioral sensitivity as a function of exposure experience (Kastelein et al., 1997, 2005).

Field Observations (Cell 9)
Kraus et al. (1997) found (and Barlow & Cameron, 2003, later confirmed) that ADDs can affect by-catch rates of harbor porpoises in commercial fishing applications. Kraus et al. (1997) found that nets with Dukane® pingers (10-kHz fundamental frequency, 300-ms duration, 132 dB re: 1 µPa source level) were sufficiently avoided that significantly fewer porpoises were entangled than in nets lacking pingers. Their observations suggest an ADD avoidance range of at least 10 m (exposure RL ~110 dB re: 1 µPa) but are not explicit enough in documenting exposure conditions or individual responses to include in the behavioral scoring analysis here.

Culik et al. (2001) conducted behavioral observations of groups of harbor porpoises near Vancouver Island before, during, and after the removal of a PICE pinger (eight different wideband swept frequency signals between 20 and 160 kHz; 300-ms duration at random intervals [5 to 30 s]; max. broadband SL = 145 dB re: 1 µPa @ 1 m). Source characteristics of the alarm were known, but propagation measurements were not made *in situ*. Exposure RLs are estimated here based on source characteristics and assumptions regarding propagation, allowing for measures of similar sources in similar conditions. A large exclusion zone of approximately 530-m radius surrounding active acoustic alarms was observed (corresponding to exposure RLs of ~90 to 100 dB re: 1 µPa). Individual sighting and avoidance data during CEE active and control conditions were scored for individuals within and outside this exclusion zone (see Table 18).

Johnston & Woodley (1998) conducted an extensive survey of AHDs used in the Bay of Fundy to exclude pinnipeds from salmon aquaculture sites. Based on the behavioral observations of Olesiuk

et al. (1996), Johnston & Woodley (1998) determined that harbor porpoises were likely being excluded from extensive areas of important habitat as a result of overlapping AHD deployments. This study lacked the discrete observational data necessary for analysis here, but two subsequent studies contained such measurements for harbor porpoises exposed to AHDs.

Olesiuk et al. (2002) conducted a controlled exposure in which they deactivated an array of four Airmar® AHDs for 3 wk and then activated the array for three 3-wk intervals over an 18-wk period. Source characteristics are known (10-kHz fundamental frequency; 194 dB re: 1 µPa-m (peak-to-peak) source level; series of 1.8-ms pulses, repeated at 40-ms intervals grouped into 2.3-s trains separated by 2.1-s quiet periods). However, no empirical acoustic measurements were obtained. Exposure RLs were estimated here based on source characteristics and simple assumptions about the propagation of high-frequency sounds in shallow-water environments. Active AHDs resulted in clear avoidance behavior by individuals and groups of harbor porpoises. The sighting rate during active periods as a percent of that during control periods was only 1.4% at ranges from 400 to 599 m, 2.5 to 3.3% at 600 to 2,499 m, and 8.1% at 2.5 to 3.5 km. RLs at 3.5 km were estimated to be ~123 dB re: 1 µPa. Avoidance data during active and control periods were scored here for individuals within the various exposure zones (Table 18).

Johnston (2002) observed similar harbor porpoise behavioral avoidance of active AHDs. They used an Airmar® dB II Plus AHD System (10-kHz fundamental frequency; 180 dB re: 1 µPa-m source level, producing a short train of 2.5-ms signals repeated every 17 s) deployed from a small boat. They sighted fewer animals when the AHD was active, and these animals were significantly further away than during control phases. Approximate exposure RLs at the point of closest approach were estimated here as ~128 dB re: 1 µPa; mean closest approach distance was consistent with exposure RLs of ~125 dB re: 1 µPa.

Additional field observations of harbor porpoises suggest that their apparently high degree of behavioral sensitivity extends to sources other than ADDs and AHDs. Koschinski et al. (2003) observed behavioral responses of harbor porpoises to simulated wind turbine noise (max. energy between 30 and 800 Hz; spectral density source levels of 128 dB re: 1 µPa/Hz at 80 and 160 Hz). They sighted harbor porpoises at greater ranges during playbacks of simulated wind turbine noise and observed that animals more frequently used echolocation signals during industrial activity. These data are not scored here, however, due to limited available information about noise exposure conditions and individual behavioral responses.

Finally, while their study was not considered in the severity scaling here, we note the importance of the Cox et al. (2001) observations regarding harbor porpoise habituation. They found that wild porpoises were initially displaced by approximately 208 m from active ADDs, but this displacement decreased by 50% in 4 d, and reached control levels in 10 to 11 d. Because of the potential for habituation, it should be noted that many or most of the field observations reported here, other than those that involve long-duration deployments, are likely most relevant for naïve individuals.

Laboratory Observations (Cell 9)

Relatively extensive laboratory data are available on captive, individual high-frequency cetaceans exposed to some of the same acoustic alarms (ADDs and AHDs) and scaring devices deployed in field applications. We applied our behavioral scoring paradigm to data from each of the captive studies conducted by Kastelein and colleagues, which included relatively detailed information on individual responses and directly measured exposure RLs.

Kastelein et al. (1997) recorded behavioral responses (location, swimming speed, and respiration patterns) of a naïve, captive harbor porpoise exposed to a variety of clicks, tones, and frequency sweeps. All of the relatively low exposure RLs (~90 to 115 dB re: 1 µPa) resulted in strong behavioral avoidance (subjects basically swam rapidly as far from the devices as possible within the enclosure) as well as changes in swimming speed and breathing patterns. Although this response quickly abated following noise cessation, no habituation was observed across multiple exposure events. Data from individual exposure trials were presented by Kastelein et al. and are analyzed here. To avoid pseudoreplication, these data are inversely weighted by the total number of trials to approximate a single exposure for the individual. Based on harbor porpoise hearing measurements (Andersen, 1970) and the Kastelein et al. (1997) data on behavioral reactions, Taylor et al. (1997) estimated zones of noise influence (audibility, behavioral disturbance, and hearing damage) for free-ranging harbor porpoises.

Subsequently, Kastelein et al. (2000) exposed two naïve subjects to three different nonpulse sources and observed generally similar behavioral avoidance in all conditions. Pooled data for each subject were scored and reported here; pooled data for each alarm in the dose-response analysis were weighted to equate with a single exposure event for each individual. Kastelein et al. (2001) later measured similar behavioral responses of

the same two individual harbor porpoises to three different acoustic alarms, but these data were not included in this analysis because subjects were no longer naïve to controlled noise exposures.

Kastelein et al. (2005) exposed two additional naïve harbor porpoises to various sounds associated with underwater data transmission systems (clicks, tones, sweeps, and impulsive distance sensors with a range of source characteristics). They directly measured source levels of each sound type and RLs at numerous positions within the experimental pool. Observed behavioral responses (avoidance and changes in swimming and respiration patterns) were very similar to those during the previous Kastelein et al. (1997, 2000, 2001) studies. Pooled data for each individual response and source type were analyzed here in the same manner as we applied to the Kastelein et al. (2000) measurements. Kastelein et al. (2006a) exposed yet another naïve individual harbor porpoise and reported very similar findings, which we incorporated as a single pooled result, with all exposures equally weighted.

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 our analysis include field exposures of harbor seals to AHDs (Jacobs & Terhune, 2002) and of translocated 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 data (see Table 20) suggest that exposures between ~90 and 140 dB re: 1 µPa 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 scaling for Cell 12 is 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. In contrast to the mid-frequency odontocetes, captive pinnipeds 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, whereas free-ranging subjects may have been more tolerant of exposures because of motivation to return to a safe location (Costa et al., 2003) or to approach enclosures holding prey items (Jacobs & Terhune, 2002).

Field Observations (Cell 12)

Jacobs & Terhune (2002) observed harbor seal reactions to Airmar® dB plus II AHDs (general source characteristics given in the "Cell 9" section above; source level in this study was 172 dB re: 1 µPa-m) deployed around aquaculture sites. From 1 to 10 AHDs were deployed around nine different sites. Jacobs & Terhune measured received SPLs around the AHDs and measured the behavior of seals in the surrounding area. Seals in this study were generally unresponsive to sounds from the AHDs. During two specific events, individuals came within 43 and 44 m of active AHDs and failed to demonstrate any measurable behavioral response; estimated exposure RLs based on the measures given were ~120 to 130 dB re: 1 µPa. These individual observations are weighted to represent a single observation for this study, scored (as 0), and reported in Table 21.

Costa et al. (2003) measured received noise levels from an ATOC sound source off northern California using acoustic data loggers placed on translocated elephant seals. Subjects were captured on land, transported to sea, instrumented with archival acoustic tags, and released such that their transit would lead them near an active ATOC source (at 939-m depth; 75-Hz signal with 37.5-Hz bandwidth; 195 dB re: 1 µPa-m max. source level, ramped up from 165 dB re: 1 µPa-m over 20 min) on their return to a haulout site. Costa et al. provided a wide range of detailed quantitative measures of individual diving behavior, responses, and exposure RLs in well-characterized contexts; this kind of information was ideal for the present purposes. Dive depth and duration, descent/ascent velocity, surface interval, and exposure RL were recorded from a total of 14 seals. An additional three seals were exposed to the ATOC source during translocations and behavioral observations were made, but exposure RLs were unavailable. Seven control seals were instrumented similarly and released when the ATOC source was not active. Received exposure levels of the ATOC source for experimental subjects averaged 128 dB re: 1 µPa (range 118 to 137) in the 60- to 90-Hz band. None of the instrumented animals terminated dives or radically altered behavior upon exposure, but some statistically significant changes in diving parameters were documented in nine individuals. The behavioral scores assigned here for statistically significant responses were either three or four depending on the change in diving behavior during exposure relative to mean values for the same individuals before and after exposure (< 50% change scored 3; > 50% change scored 4). Translocated northern elephant seals exposed to this particular nonpulse source (ATOC) began to demonstrate

subtle behavioral changes at ~120 to 140 dB re: 1 μPa exposure RLs (Table 21).

Several other field studies (discussed briefly below) were considered but not included in the behavioral analyses due to limited information about source and/or propagation characteristics, individual responses during and/or in the absence of exposure, or both. While studying cetaceans, Richardson et al. (1990b, 1991) made some observation of ringed and bearded seal responses to playbacks of underwater drilling sounds. Their findings generally suggested a fairly high degree of tolerance by exposed pinnipeds to these sounds. This contrasts to some extent with the results of Frost & Lowry (1988) who found some reduction in ringed seal densities around islands on which drilling was occurring. Norberg & Bain (1994) made detailed acoustic measurements of several arrays of Cascade Applied Sciences® AHDs (11.9- to 14.7-kHz frequency sweeps; 195 dB re: 1 μPa-m source level; 1-ms pulse produced in 57 to 58 discrete pulse chirps of 2.3-s total duration). These devices were placed on the Chittenden Locks in Puget Sound, Washington, in an effort to dissuade predation of wild steelhead trout by California sea lions. Behavioral responses of individual animals, however, were not reported. Norberg (2000) evaluated the behavioral responses of California sea lions to Airmar® AHDs (10-kHz fundamental frequency; 195 dB re: 1 μPa-m source level; short train of 2.5-ms signals repeated every 17 s) intended to reduce predation on salmonids in aquaculture facilities. Behavioral observations suggested limited behavioral deterrence by the devices (predation rates were similar in experimental and control conditions), but measures of RLs and individual response behavior are absent. Yurk (2000) also observed pinnipeds exposed to AHDs in the context of fisheries interactions. He determined that active AHDs were more effective than a mechanical barrier or altered lighting conditions in dissuading harbor seals from preying on fish under bridges. Again, however, insufficient information regarding received sounds and individual responses is available to consider these observations explicitly here. Koschinski et al. (2003) observed harbor seals during underwater playbacks of simulated wind turbine noise (maximum energy between 30 and 800 Hz; spectral density source levels of 128 dB re: 1 μPa/Hz at 80 and 160 Hz). Harbor seals were sighted at greater distances during playbacks than during control conditions. However, limited information on received exposures and individual behavioral responses precluded inclusion in our analysis. Moulton et al. (2003, 2005) studied ringed seals before and during the construction and operation of an oil production facility. They found little or no avoidance of the area around the various industrial sources, most of which emitted nonpulses. Because of the continuous exposure to multiple sound sources at varying distances, this study did not produce data on discrete exposures and responses.

Laboratory Observations (Cell 12)

Kastelein et al. (2006b) exposed nine captive harbor seals in a ~25 × 30 m enclosure to nonpulse sounds used in underwater data communication systems (similar to acoustic modems). Test signals were identical to those used by Kastelein et al. (2005) in harbor porpoise exposure studies (frequency modulated tones, sweeps, and bands of noise with fundamental frequencies between 8 and 16 kHz; 128 to 130 [± 3] dB re: 1 μPa-m source levels; 1- to 2-s duration [60-80% duty cycle]; or 100% duty cycle). They recorded seal positions and the mean number of individual surfacing behaviors during control periods (no exposure), before exposure, and in 15-min experimental sessions (n = 7 exposures for each sound type). Background noise and exposure RLs (in terms of L_{eq}; 32-s total time) were measured at numerous positions around the enclosure for each acoustic source. Acoustic discomfort was recognized based on movement out of areas that animals used during control periods. An acoustic discomfort threshold was calculated for the group of seals for each source type, and for each sound source this was *ca.* 107 dB re: 1 μPa. Seals generally swam away from each source, avoiding it by ~5 m, although they did not haul out of the water or change surfacing behavior. Seal reactions did not appear to wane over repeated exposure (i.e., there was no obvious habituation), and the colony of seals generally returned to baseline conditions following exposure.

For the behavioral analysis conducted here, the Kastelein et al. (2006b) results were interpreted as follows. Because the behavior of individuals within the same pool at the same time cannot be considered independent, the group of nine harbor seals was considered a single observation. Because of similarity of sources and exposure conditions and the close temporal timing of exposures, we combined observations across the four sound types and include a single observation within each appropriate 10-dB bin. Exposures between ~80 and 107 dB re: 1 μPa seemed insufficient to induce behavioral avoidance in the colony of seals, but higher exposures were considered sufficient. Consequently, single observations indicating no response (0) appear in the 80 to 90 and in the 90 to 100 dB re: 1 μPa exposure bins, and a single observation indicating avoidance behavior (6) is shown in the 100 to 110 dB re: 1 μPa condition (Table 21).

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, power-boat approaches, and construction noise. Unfortunately, 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 are not appropriate for quantitative analyses relating exposure level and scored behavioral response. 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 complicated, however, 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 for which there was clearly just nonpulse exposure were considered in the severity scaling analysis (Thorson et al., 1999, 2000b; Berg et al., 2002).

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 μPa) nonpulse sounds often leave haulout areas and seek refuge temporarily (minutes to a few hours) in the water. In contrast, pinnipeds exposed to distant launches at RLs ~60 to 70 dB re: 20 μPa tend 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 colonies studied to such noise events. Also, there are strong species differences, with harbor seals being much more responsive than northern elephant seals (e.g., Holst et al., 2005a, 2005b). Due to the limitations of the available data, it is not currently possible to make any further general characterizations regarding this condition.

A series of highly detailed, quantitative analyses on the behavior of pinnipeds exposed to the sounds of various large missile launches were reviewed. These sources generally produce sustained, generally low-frequency (little energy above 1,000 Hz) "rumbling" sounds lasting tens of seconds (nonpulse) associated with launch boosters, as well as a sonic boom (pulse) in flight as the rocket goes supersonic. Extensive research has been conducted on the effects of both sound types on pinnipeds. Nonpulse exposures are considered in this section, whereas behavioral responses to the pulse component of some of the same launches are considered in Appendix B. Because many measurements were made on the same few colonies of pinnipeds that were exposed to multiple launches, it is likely that some of the same individuals were resampled. Therefore, we weighted the combined results across studies for each species and breeding location into a single observation for the behavioral analysis here. That is, we considered each species in an individual breeding colony a single unit of observation across studies. The results were pooled accordingly in Table 22, but the studies are discussed longitudinally below. The studies discussed below reported exposure conditions on or near pinniped breeding rookeries during launches of different types of rockets using a variety of metrics, including A-weighted values and a frequency-weighting function derived from the harbor seal audiogram; we used unweighted SPL values for the analysis here.

Thorson et al. (1998) measured harbor seal responses and conducted AEP measurements on seals exposed to a Titan IV A-18 launch from Vandenberg Air Force Base (VAFB), California. They studied colonies both on the mainland at VAFB and on nearby Santa Cruz Island. Unfortunately, the launch occurred at night and during a period of relatively high tide, limiting both the number of seals present on the rookeries and the observation of individuals. However, behavioral monitoring over several days after the launch did not indicate any abandonment of the breeding rookeries at either site. Hearing measures (AEP) on individuals tested before and several hours after the launch did not indicate any loss of sensitivity.

Thorson et al. (1999) conducted similar observations of harbor seals at VAFB and also observed northern elephant seals, California sea lions, and northern fur seals at nearby San Miguel Island. Following the launch (of an Athena 2 IKONOS-1 missile), 33 harbor seals (including six pups) at the VAFB rookery entered the water. They began to return to the beach beginning 16 min after the launch, and no pups were observed to have died as a result of the event. This behavior was considered to represent both minor avoidance and a brief/minor potential or actual separation of females and dependent offspring (scored 6 here). The maximum unweighted SPL value was 119 dB re: 20 μPa. Individuals of the three pinniped species monitored on San Miguel Island reacted similarly. However, their responses were to the sonic boom generated by the rocket once airborne rather than to the nonpulse

sounds associated with the launch per se, and thus are not scored here.

Thorson et al. (2000a) conducted observations of harbor seal abundance, distribution, and haulout patterns at VAFB for several days before and after the launch of a Titan II G-13 missile from VAFB. This launch occurred during the middle of the night, precluding direct observation of seal reactions (and behavioral scoring here), although observations on subsequent days indicated generally nominal harbor seal presence and distribution in the area.

Thorson et al. (2000b) measured behavioral and auditory responses of harbor seals at VAFB and behavioral responses of northern elephant seals and California sea lions on San Miguel Island to the launch of a Titan IV B-28 missile from VAFB. They observed all 54 harbor seals at the VAFB site moving from the breeding rookery into the water within 2 min of the onset of the launch (47 entered the water immediately). The maximum unweighted SPL value near the rookery was 116 dB re: 20 μPa; this exposure was considered here to be consistent with a behavioral score of 6 for this group of seals. The sound persisted for several minutes, and the unweighted SEL value was 127 dB re: (20 μPa)^2-s. There was no difference in the hearing capabilities of three young seals tested using AEP methods before and after the missile launch. Neither the California sea lions nor elephant seals on San Miguel Island were observed to respond at all to the "faint" noise associated with the launch, corresponding to a severity scaling score of 0 (Table 23). These sounds were from the launch boosters (nonpulses) rather than sonic booms and were estimated here as ~60 to 70 dB re: 20 μPa based on the measurements and descriptions given.

Berg et al. (2001) obtained similar measurements of behavioral responses of harbor seals at VAFB and California sea lions and northern elephant seals at San Miguel Island to a Delta II EO-1 missile launch from VAFB. Observations were also made of southern sea otter (*Enhydra lutris nereis*) and California brown pelican (*Pelecanus occidentalis californicus*) responses. No harbor seals were hauled out on the VAFB rookery during this launch. Berg et al. note that subsequent harbor seal abundance and distribution in the days after the launch were within normal variability, and there appeared to be no lasting behavioral reactions. Elephant seals and California sea lions at San Miguel Island did not noticeably respond to sounds associated with the launch, which in this case were predominantly the sonic boom (pulse) component.

Berg et al. (2002) measured behavioral responses of harbor seals on VAFB rookeries to the launch of a Titan IV B-34 missile from a launch pad at VAFB ~8.6 km away. At the time of the launch, 38 seals were present at two haulout sites, all of which entered the water immediately following the onset of launch noise. More seals (n = 56) were present at the locations 90 min after the launch event, indicating the temporary and minor nature of the disturbance, and no injured animals were located. The avoidance behavior was coincident with a maximum unweighted SPL value near the rookeries of 119 dB re: 20 μPa (unweighted aerial SEL value was 130 dB re: [20 μPa]^2-s).

Finally, Berg et al. (2004) observed behavioral responses of California sea lions, northern elephant seals, and northern fur seals on San Miguel Island to the launch of an Atlas IIAS MLV-14 missile from VAFB. Received signals were sonic booms which had little to no effect on the behavior of the pinnipeds, other than minor orienting behaviors and movements in some of the California sea lions. These results are not scored here, in part because the sounds included pulses.

Other researchers have investigated the effects of other kinds of human activities (e.g., aircraft, motorboats, general human presence) as well as rocket launches on the haulout behavior, including avoidance, of pinnipeds (Allen et al., 1984; Suryan & Harvey, 1998; Born et al., 1999; Moulton et al., 2002). The combined results indicated that hauled-out pinnipeds in certain conditions can be disturbed, significantly in some cases, by the presence of various human activities. However, these studies lack either specific estimates of received noise exposure conditions or individual-specific behavioral responses or both. Additionally, multiple stimuli were generally simultaneously present, including the visual presence of sources, which preclude their inclusion here. Gentry et al. (1990) determined that northern fur seals were generally tolerant of underground explosions and other quarrying operations in relatively close proximity; only a few orienting behaviors were observed in response to the largest blasts. Some acoustic measurements were made, but individual behaviors or group responses and received exposure levels were not reported and were thus not scored here.

Holst et al. (2005a, 2005b) observed behavioral responses in three species of pinnipeds—harbor seal, California sea lion, and northern elephant seal—on San Nicolas Island to 47 small- and mid-sized missile launches over a 4-y period. They observed animal presence and distribution before launches and behavior during and following launches. Some of the missiles generated sonic booms, but the majority of the exposures were relatively low-frequency, long-duration rumbling sounds that would be categorized as nonpulses. During many launches, acoustic measurements were made near the animals whose behavior was videotaped. Peak, SPL, and SEL exposures were reported. This dataset has not been incorporated into the present analysis. However,

results indicated that California sea lions had mixed reactions to rocket launches, with some individuals exhibiting startle responses and increased vigilance and others showing virtually no reaction. Northern elephant seal reactions were minimal, consisting only of minor movements and orienting responses that rapidly subsided. Conversely, harbor seals were by far the most responsive of the pinnipeds observed, with many individuals entering the water from haulout sites following rocket launches and failing to return for periods of hours. No cases of long-term pup separation or of injury were documented. If those phenomena had occurred, they would be considered relatively severe in terms of the behavioral scoring paradigm given here and should also be considered as they relate to injury criteria. In California sea lions and northern elephant seals, there were significant correlations between behavioral responses and both the missile's closest distance and the RL of the launch sound near the pinnipeds (SEL). Corresponding relationships for harbor seals were weaker. Holst et al. (2005b) concluded that the temporary behavioral responses, even the relatively severe ones observed in harbor seals, do not appear to have substantial adverse effects on pinniped populations. This conclusion is based on the decades-long occurrence of missile launches and the presence of increasing numbers of pinnipeds of all three species in the area.

(Left to Right): David Kastak, Roger Gentry, Peter Tyack, Brandon Southall, Darlene Ketten, W. John Richardson, Jeanette Thomas, James Miller, Ann Bowles, James Finneran, Charles Greene, Jr., Paul Nachtigall, and William Ellison

This manuscript is respectfully dedicated to our co-author, David Kastak.

Dave was a brilliant scientist, but even more importantly, he was a man with great sincerity, integrity, and a sharp wit. He was an inspiration and mentor to many, and his significant, incisive research on marine mammal cognition and sensory systems over the past two decades provided advances that shaped and will continue to guide the future of these fields. Dave was a valued colleague and treasured friend to all of us. He will be missed but never forgotten.

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