

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 <i>in situ</i> 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 <i>in situ</i> 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 <i>in situ</i> 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.

Response score	Received RMS sound pressure level (dB re: 1 $\mu$ Pa)											
	80 to < 90	90 to < 100	100 to < 110	110 to < 120	120 to < 130	130 to < 140	140 to < 150	150 to < 160	160 to < 170	170 to < 180	180 to < 190	190 to < 200
9												
8		<b>1.0</b> (3)	<b>7.0</b> (3)		<b>5.0</b> (2)		<b>1.0</b> (7)				<b>5.0</b> (13)	<b>1.5</b> (13)
7												
6				<b>3.0</b> (2, 10)	<b>1.0</b> (2)				<b>1.0</b> (9)	<b>6.0</b> (12)		
5					<b>1.0</b> (11)							
4				<b>1.0</b> (4)	<b>2.0</b> (4)							
3	<b>5.0</b> (1)		<b>4.0</b> (3, 5)	<b>134</b> (4, 6)	<b>1.0</b> (4)							
2				<b>15.0</b> (2, 3, 8)								
1			<b>1.0</b> (4)	<b>1.0</b> (2, 5)	<b>1.0</b> (2, 4)							
0			<b>8.0</b> (3, 4)	<b>2.0</b> (2, 4)	<b>1.0</b> (2, 4)		<b>1.0</b> (2)				<b>3.0</b> (13)	<b>1.5</b> (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 scores: 0 & 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 <i>in situ</i> 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 <i>in situ</i> 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 non-pulse sounds (laboratory)	Calibrated RL measurements made <i>in situ</i> 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 <i>in situ</i> 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 <i>in situ</i> 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.

Response score	Received RMS sound pressure level (dB re: 1 μPa)											
	80 to < 90	90 to < 100	100 to < 110	110 to < 120	120 to < 130	130 to < 140	140 to < 150	150 to < 160	160 to < 170	170 to < 180	180 to < 190	190 to < 200
9												
8												
7												
6	<b>0.3</b> (4)	<b>0.3</b> (4)	<b>0.9</b> (1, 2, 4, 5, 6, 7)	<b>3.3</b> (1, 2, 4, 5, 6, 7)	<b>1.0</b> (3, 7)		<b>52.1</b> (2)	<b>9.3</b> (2)	<b>4.6</b> (2)			
5												
4			<b>0.1</b> (4)	<b>0.1</b> (4)								
3												
2												
1												
0	<b>12.8</b> (1, 5)	<b>23.1</b> (1, 2, 5, 6)	<b>0.4</b> (4, 7)	<b>0.1</b> (7)	<b>0.3</b> (3)							



Courtesy: A. Friedlander

**Table 20.** Summary of behavioral responses by different species 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  $\mu$ Pa).

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 <i>in situ</i> 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 <i>in situ</i> 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 <i>in situ</i> 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 <i>in situ</i> 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.

Response score	Received RMS sound pressure level (dB re: 1 μPa)											
	80 to < 90	90 to < 100	100 to < 110	110 to < 120	120 to < 130	130 to < 140	140 to < 150	150 to < 160	160 to < 170	170 to < 180	180 to < 190	190 to < 200
9												
8												
7												
6			<b>1.0</b> (3)									
5												
4					<b>1.0</b> (2)	<b>5.0</b> (2)						
3					<b>1.0</b> (2)	<b>2.0</b> (2)						
2												
1												
0	<b>1.0</b> (3)	<b>1.0</b> (3)		<b>1.0</b> (2)	<b>5.0</b> (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 RLSs are given in dB SPL, which are RMS sound pressure levels (dB re: 20  $\mu$ Pa).

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 <i>in situ</i> in and around breeding rookeries	Visual observations of movement and behavior of individuals in breeding rookeries before, during, and after rocket launches	Yes	Exposure RLSs 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 <i>in situ</i> in and around breeding rookeries	Visual observations of movement and behavior of individuals in breeding rookeries before, during, and after rocket launches	Yes	Exposure RLSs 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 <i>in situ</i> in and around breeding rookeries	Visual observations of movement and behavior of individuals in breeding rookeries before, during, and after rocket launches	Yes	Exposure RLSs 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 <i>in situ</i> 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 <i>in situ</i> 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 <i>in situ</i> 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 non-pulses; 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.

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



Courtesy: Peter M. Scheifele

## 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
<i>Acoustic measurements of relevant sound sources</i>	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.
<i>Ambient noise measurements</i>	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.
<i>“Absolute” hearing measurements</i>	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.
<i>Auditory scene analysis</i>	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.
<i>Marine mammal behavioral responses to sound exposure</i>	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.
<i>Effects of sound exposure on marine mammal hearing: masking, TTS, and PTS</i>	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.
<i>Effects of sound exposure on marine mammal non-auditory systems</i>	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.
<i>Particularly sensitive species: beaked whales</i>	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.

### Acknowledgments

The authors acknowledge three former members of the marine mammal noise exposure criteria group: Whitlow Au, Sam Ridgway, and Ronald Schusterman. Partial financial support for meetings of the noise exposure criteria group was provided by the U.S. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Ocean Acoustics Program.

The authors also appreciate the expertise, review, and assistance of the following people: Laurie Allen, Robyn Angliss, Dan Costa, Tara Cox, Ned Cyr, Adam Frankel, Ron Kastelein, Steve Leathery, Rob McCauley, David Mellinger, Richard Merrick, Douglas Nowacek, Deborah Palka, Michael Payne, Arthur Popper, Andrew

Solow, Sofie Van Parijs, Willem Verboom, Donna Wieting, and five anonymous reviewers.

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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  $\mu\text{Pa}$  for underwater sound and dB re: 20  $\mu\text{Pa}$  for aerial sound. Peak SPLs are given as dB re: 1  $\mu\text{Pa}$  (peak) in water and dB re: 20  $\mu\text{Pa}$  (peak) in air. Peak-to-peak SPLs are dB re: 1  $\mu\text{Pa}$  (peak-to-peak) in water and dB re: 20  $\mu\text{Pa}$  (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  $\text{Pa}^2$ ) of the ambient ( $P_{amb}$ )<sup>2</sup> is determined with the following relationship:

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

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

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

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  $\text{Pa}^2\text{-s}$ ), where  $t \leq t_b$ , is then calculated as

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

where  $E_{100} = E(t_b)$  is 100% of the sound exposure. For the 5% point,  $E(t)$  is determined as  $E_5 = 0.05 \cdot E_{100} = E(t_5)$ , while  $E(t)$  for the 95% energy point is determined as  $E_{95} = 0.95 \cdot E_{100} = E(t_{95})$ . 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

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

**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  $\mu\text{Pa}^2\text{-s}$ ) 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  $\mu\text{Pa}$  for  $n$  exposures and the reference pressure ( $p_{ref}$ ) is  $1 \mu\text{Pa}$  under water and  $20 \mu\text{Pa}$  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:  $1 \mu\text{Pa}^2\text{-s}$ , and the appropriate units for aerial SEL are dB re:  $(20 \mu\text{Pa})^2\text{-s}$ .

**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

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

where  $O$  is the expectation operator,  $\mu$  is the mean, and  $\Sigma$  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 non-permanent 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



## 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  $\mu$ Pa (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  $\mu$ Pa (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<sup>3</sup> 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  $\mu$ Pa (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  $\mu$ Pa. 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  $\mu$ Pa), but that avoidance and other more profound behavioral changes were generally not observed unless the RL was  $\geq$  160 dB re: 1  $\mu$ Pa.

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  $\mu$ Pa), 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  $\mu$ Pa). 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  $\sim$ 3.5 km (max. measured RL of 178 dB re: 1  $\mu$ Pa). 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  $\mu$ Pa) and avoidance behavior was noted at  $\sim$ 3 km (RLs  $\sim$ 165 dB re: 1  $\mu$ Pa). 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  $\mu$ Pa. 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<sup>2</sup> 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  $\sim$ 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  $\mu$ Pa. 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  $\mu$ Pa. Exposures were not estimated to exceed 190 dB re: 1  $\mu$ Pa. 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  $\mu$ Pa (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  $\mu$ Pa (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  $\mu$ Pa.

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  $\mu$ Pa (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  $\mu$ Pa, 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<sup>3</sup>) 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  $\mu$ Pa (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  $\mu$ Pa [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<sup>3</sup>) 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  $\mu$ Pa 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  $\mu$ Pa (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  $\mu$ Pa 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  $\mu$ Pa) 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  $\mu$ Pa 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  $\mu$ Pa. There was also one observation of a whale exposed to 179 dB re: 1  $\mu$ Pa; 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  $\mu$ Pa. 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  $\mu$ Pa. The third observation (2 d later) involved three individuals; the survey vessel was 94 km away and measured RL was 130 dB re: 1  $\mu$ Pa. 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  $\mu$ Pa (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  $\mu$ Pa 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  $\mu$ Pa. 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  $\mu$ Pa), but temporary avoidance behavior (response score: 6) was observed for successive exposures at 174 and 178 dB re: 1  $\mu$ Pa.

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  $\mu$ Pa (peak-to-peak) and in the two bottlenose dolphins at 196 and 209 dB re: 1  $\mu$ Pa (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  $\mu$ Pa (peak-to-peak); bottlenose dolphin at 229 dB re: 1  $\mu$ Pa [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  $\mu$ Pa (peak-to-peak) and significant differences in aldosterone and monocyte counts in the dolphin for exposures exceeding 225 dB re: 1  $\mu$ Pa (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 non-pulses. 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  $\mu$ Pa range (RMS over pulse duration) generally have limited

potential to induce avoidance behavior in pinnipeds, whereas RLs exceeding 190 dB re: 1  $\mu$ Pa are likely to elicit responses, at least in some ringed seals (Harris et al., 2001; Blackwell et al., 2004b; Miller et al., 2005).

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<sup>3</sup>, 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  $\mu$ Pa (RMS), 157 dB re: 1  $\mu$ Pa (peak), and 145 dB re: 1  $\mu$ Pa<sup>2</sup>-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  $\mu$ Pa (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  $\mu$ Pa.

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  $\mu$ Pa (RMS), 111 dB re: 20  $\mu$ Pa (peak), and 87 dB re: (20  $\mu$ Pa)<sup>2</sup>-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.

## 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  $\mu$ Pa and an increasing probability of avoidance and other behavioral effects in the 120 to 160 dB re: 1  $\mu$ Pa 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  $\mu$ Pa range and clear avoidance at 120 to 140 dB re: 1  $\mu$ Pa.

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  $\mu$ Pa 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  $\mu$ Pa. 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  $\mu$ Pa-m). In two cases for RLs 100 to 110 dB re: 1  $\mu$ Pa, there was no observed behavioral reaction. Avoidance behavior was observed in two cases where RLs were 110 to 120 dB re: 1  $\mu$ Pa.

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  $\mu$ Pa 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  $\mu$ Pa.

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  $\mu$ Pa-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  $\mu$ Pa 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  $\mu$ Pa-m). Measured RLs ranged from 120 to 150 dB re: 1  $\mu$ Pa. 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  $\mu$ Pa. 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  $\mu$ Pa @ 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  $\mu$ Pa, though not specified in sufficient detail to include in the analysis here. For 11 playbacks, exposures were between 120 and 130 dB re: 1  $\mu$ Pa 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  $\mu$ Pa (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  $\mu$ Pa-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  $\mu$ Pa, while others failed to exhibit such responses for exposure RLs from 120 to 150 dB re: 1  $\mu$ Pa. Contextual variables other than exposure RL, and probable species differences, are the likely reasons for this variability. 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  $\mu$ Pa 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  $\sim$ 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 to 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  $\sim$ 100 dB re: 1  $\mu$ Pa 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  $\sim$ 90 dB re: 1  $\mu$ Pa 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  $\mu$ Pa/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  $\sim$ 101 to 105 dB re: 1  $\mu$ Pa 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  $\sim$ 120 dB re: 1  $\mu$ Pa 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  $\sim$ 116 dB re: 1  $\mu$ Pa 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  $\sim$ 121 dB re: 1  $\mu$ Pa 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  $\mu$ Pa in the 20- to 1,000-Hz band). A single beluga vocal response was noted at RL = 116 dB re: 1  $\mu$ Pa 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  $\mu$ Pa).

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  $\mu$ Pa/Hz. Over a bandwidth of 100 to 6,000 Hz, equivalent broadband source levels are ~157 dB re: 1  $\mu$ Pa-m; RLs at a range of 450 m are ~104 dB re: 1  $\mu$ Pa.

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  $\mu$ Pa. Dolphin vocal responses were observed before vessel sounds were audible, and apparently occurred with RLs in the 110 to < 120 dB re: 1  $\mu$ Pa 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  $\mu$ Pa/Hz in the 1- to 22-kHz band (broadband RL ~128 dB re: 1  $\mu$ Pa) as opposed to ~65 dB re: 1  $\mu$ Pa/Hz in the same band (broadband RL ~108 dB re: 1  $\mu$ Pa) 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<sup>2</sup>. 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  $\mu$ Pa.

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  $\mu$ Pa-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  $\mu$ Pa (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  $\mu$ Pa bin for this single observation of the group.

Awbrey & Stewart (1983) played back semi-submersible drillship sounds (source level: 163 dB re: 1  $\mu$ Pa-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  $\mu$ Pa 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  $\mu$ Pa-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  $\mu$ Pa 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  $\mu$ Pa) 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  $\mu$ Pa. 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  $\mu$ Pa), at least for initial exposures. This observation is also evident in the severity scaling analysis for Cell 9 (Table 19). All recorded exposures exceeding 140 dB re: 1  $\mu$ Pa induced profound and sustained avoidance behavior in wild harbor porpoises. 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  $\mu$ Pa 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  $\mu$ Pa) 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 wide-band 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  $\mu$ Pa @ 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  $\mu$ Pa). 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  $\mu$ Pa-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  $\mu$ Pa. 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  $\mu$ Pa-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  $\mu$ Pa; mean closest approach distance was consistent with exposure RLs of ~125 dB re: 1  $\mu$ Pa.

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  $\mu$ Pa/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  $\mu$ Pa) 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  $\mu$ Pa generally do not appear to induce strong behavioral responses in pinnipeds exposed to nonpulse sounds in water; no data exist regarding exposures at higher levels. The severity 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  $\mu$ Pa-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  $\mu$ Pa. 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  $\mu$ Pa-m max. source level, ramped up from 165 dB re: 1  $\mu$ Pa-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  $\mu$ Pa (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  $\mu$ Pa 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  $\mu$ Pa-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  $\mu$ Pa-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  $\mu$ Pa/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 non-pulse 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  $\mu$ Pa-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  $\mu$ Pa. 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  $\mu$ Pa 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  $\mu$ Pa exposure bins, and a single observation indicating avoidance behavior (6) is shown in the 100 to 110 dB re: 1  $\mu$ Pa 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  $\mu$ Pa) 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  $\mu$ Pa 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  $\mu$ Pa. 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  $\mu$ Pa; 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  $\mu$ Pa)<sup>2</sup>-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  $\mu$ Pa 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  $\mu$ Pa (unweighted aerial SEL value was 130 dB re: [20  $\mu$ Pa]<sup>2</sup>-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 IAS 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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