National Marine Fisheries Service
Endangered Species Act Section 7 Biological Opinion

Action Agencies:	The United States Navy and NOAA's National Marine Fisheries Service
Activities Considered:	(1) U.S. Navy's Surveillance Towed Array Sensor System Low Frequency Active Sonar Training, Testing and Operations (2) NOAA's National Marine Fisheries Service, Office of Protected Resources' issuance of four letters of authorization for the U.S. Navy to "take" marine mammals incidental to Surveillance Towed Array Sensor System Low Frequency Active Sonar Training, Testing and Operations in areas of the Pacific Ocean for the period of August 15, 2014 through August 14, 2015 pursuant to the five-year MMPA regulation
Consultation Conducted By:	NOAA's National Marine Fisheries Service, Office of Protected Resources, Endangered Species Act Interagency Cooperation Division
Approved:	Donna S. Wieting Director, Office of Protected Resources
Date:	AUG 1 3 2014
PCTS Tracking Number:	FPR-2014-9082

This page is intentionally blank.

BIOLOGICAL OPINION

TABLE OF CONTENTS

Page

1 Introduction 1
1.1 Background 2
1.2 Consultation History 3
2 Description of the Proposed Action 4
2.1 U.S. Navy Proposed Activities 5
2.1.1 Deployment 6
2.1.2 Active Sonar Components 6
2.1.3 Passive Sonar Components 7
2.1.4 Mitigation and Monitoring 8
2.1.5 Mitigation 10
2.2 NMFS Permits and Conservation Division Proposed Activities (2014-2015) 14
2.2.1 Authorization of Incidental Take of Marine Mammals 14
2.2.2 Mitigation Requirements 17
2.2.3 Monitoring Requirements 23
2.2.4 Reporting Requirements 24
2.3 Action Area 26
3 Approach to the Assessment 29
3.1 Overview of NMFS' Assessment Framework 29
3.2 Risk Analysis for Endangered and Threatened Species 30
3.3 Risk Analysis for Designated Critical Habitat 31
3.4 Defining "Significance" 33
3.5 Treatment of "Cumulative Impacts" (in the sense of NEPA) 34
3.6 Evidence Available for the Consultation 36
3.7 A Brief Background on Sound 37
3.8 Overview of the Navy's Assessment Framework for SURTASS LFA 37
3.8.1 Effects on Sea Turtles Associated with Transmission of LFA Sonar 38
3.8.2 Non-Auditory Effects for Marine Mammals 40
3.8.3 Auditory Effects on Marine Mammals 43
3.8.4 Behavioral Change in Marine Mammals 46
3.8.5 Masking in Marine Mammals 48
3.8.6 Stranding 50
3.9 The Navy's Approach to Assessing Behavioral Response of Marine Mammals 51
3.9.1 Parametric Analysis 52

3.9.2 Use of The Acoustic Integration Model (AIM) to Estimate Marine Mammal Exposures to Low-frequency Active Sonar 53

4 Status of Listed Resources 56

4.1 Listed Resources Not Considered Further 57

4.1.1 Scalloped Hammerhead Shark - Indo-West Pacific DPS 58

4.1.2 North Pacific Right Whale Critical Habitat 69

4.1.3 Hawaiian Monk Seal Critical Habitat 69

4.2 Climate Change 72

4.3 Species Considered Further in this Opinion 78

4.3.1 Blue Whale 79

4.3.2 Fin Whale 86

4.3.3 Western North Pacific Gray Whale 92

4.3.4 Humpback Whale 97

4.3.5 North Pacific Right Whale 103

4.3.6 Sei Whale 106

4.3.7 Sperm Whale 111

4.3.8 Main Hawaiian Islands Insular Distinct Population Segment (DPS) of False Killer Whale 116

4.3.9 Hawaiian Monk Seal 121

4.3.10 Green Turtle 125

4.3.11 Hawksbill Turtle 133

4.3.12 Loggerhead Turtle 138

4.3.13 Olive Ridley Turtle 148

4.3.14 Leatherback Turtle 154

5 Environmental Baseline 161

5.1 Natural Phenomena 161

5.1.1 Oceanic Temperature Regimes 161

5.1.2 Ambient Noise 162

5.1.3 Natural Mortality 164

5.2 Human Activities 164

5.2.1 Whaling 165

5.2.2 Shipping 166

5.2.3 Fisheries 167

5.2.4 Contaminants 171

5.2.5 Pollution and Ocean Debris 172

5.2.6 Pulsed Sound Generated by Seismic Surveys 174

5.2.7 Scientific Research 175

5.2.8 Vessel Approaches – Commercial and Private Marine Mammal Watching 177

5.2.9 Ongoing SURTASS LFA Sonar Training, Operations, and Testing 178

5.2.10 Ongoing Navy Pacific Fleet Training and Testing Activities 184

5.3 The Impact of the Baseline on Listed Resources 185

6 Effects of the Action 188

6.1 Stressors Associated with the Proposed Action 188
6.1.1 Vessel Operation - Noise 188
6.1.2 Vessel Operation - Discharges 194
6.1.3 Vessel Operation – Ship Strike 195
6.1.4 Passive System Component - SURTASS 195
6.1.5 Active System Component - LFA sonar 195
6.1.6 High Frequency (HF/M3) Sonar 206
6.2 Mitigation to Minimize the Likelihood of Exposure 207
6.3 Exposure 208
6.3.1 Intensity 209
6.3.2 Exposure Duration 210
6.3.3 Exposure Frequency 210
6.3.4 Mitigation Measures to Minimize the Likelihood of Exposure 211
6.3.5 Exposure Estimates 213
6.4 Response Analysis 225
6.4.1 Cetaceans 226
6.4.2 Sea Turtles 252
6.4.3 Responses of Sea Turtles 255
6.4.4 Risk to Sea Turtles 256
6.4.5 Interaction of SURTASS LFA Sonar and Other Sound Sources 257
6.5 Cumulative Effects 260
6.6 Potential for Long-Term or Additive Impacts 262
6.7 Integration and Synthesis 265
6.7.1 Blue Whale 265
6.7.2 Fin Whale 265
6.7.3 Western Pacific Gray Whale 266
6.7.4 Humpback Whale 267
6.7.5 North Pacific Right Whale 268
6.7.6 Sei Whale 269
6.7.7 Sperm Whale 270
6.7.8 Main Hawaiian Island Insular False Killer Whale 272
6.7.9 Hawaiian Monk Seal 272
6.7.10 Sea Turtles 273

7 Conclusion 274

8 Incidental Take Statement 275

8.1 Amount and Extent of Take 276
8.1.1 Activity Levels Indicator of Take for Sea Turtles 279

8.2 Effect of the Take 280
8.2.1 Reasonable and Prudent Measures 280
8.3 Terms and Conditions 281
9 Conservation Recommendations 283
10 Reinitiation of Consultation 284
11 Information Quality Act 285
12 References 286

LIST OF TABLES

Page

Table 1. Estimated Level B Harrassment Authorized for Mission Area 1, East of Japan 14

Table 2. Estimated Level B Harrassment Authorized for Mission Area 2, North Phillipine Sea 15

Table 3. Estimated Level B Harrassment Authorized for Mission Area 3, West Phillipine Sea 15

Table 4. Estimated Level B Harrassment Authorized for Mission Area 4, Offshore Guam 15

Table 5. Estimated Level B Harrassment Authorized for Mission Area 5, Sea of Japan 15

Table 6. Estimated Level B Harrassment Authorized for Mission Area 6, East China Sea 16

Table 7. Estimated Level B Harrassment Authorized for Mission Area 7, South China Sea 16

Table 8. Estimated Level B Harrassment Authorized for Mission Area 8, Offshore Japan 25-40° N 16

Table 9. Estimated Level B Harrassment Authorized for Mission Area 9, Offshore Japan 10-25° N 16

Table 10. Estimated Level B Harrassment Authorized for Mission Area 10, Hawaii North 17

Table 11. Estimated Level B Harrassment Authorized for Mission Area 11, Hawaii South 17

Table 12. Offshore Biologically Important Areas (OBIAs). 21

Table 13. Source Characteristics used for AIM modeling 54

Table 14. Listed Resouces under NMFS' jurisdiction that may occur in the Action Area for this Consultation 57

Table 15. Authorized scientific research takes of various listed marine species in the action area. 176

Table 16. Summary of SURTASS LFA sonar missions and mitigation measure detections from 2002 through 2nd Quarter 2014. 180

Table 17. Summary of SURTASS LFA Sonar Operations and Preventative Measures During the Annual LOAs and ITS Reporting Period from 15 August 2010 through 14 August 2013 182

Table 18. Summary of SURTASS LFA Sonar Operations and Preventative Measures During the First Two Quarters (August through February) of the Annual LOAs and ITS Reporting Period from 15 August 2013 through 14 August 2014.....183

Table 19. Number of proposed SURTASS LFA sonar missions by area proposed to occur 15 August 2014 through 14 August 2015. 215

Table 20. Critical Parameters Used in the Navy's SURTASS LFA Sonar Model 216

Table 21. Percentage estimated of each ESA-listed marine mammal stock, based on a single 7-day mission in each potential operating area, that may be affected by the operation of SURTASS LFA sonar (with mitigation measures applied). 217

Table 22. Estimated blue whale exposures to SURTASS LFA sonar between 120dB and 180dB 219

Table 23. Estimated fin whale exposures to SURTASS LFA sonar between 120dB and 180dB. 219

Table 24. Estimated Western Pacific gray whale exposures to SURTASS LFA sonar between 120dB and 180dB. 220

Table 25. Estimated humpback whale exposures to SURTASS LFA sonar between 120dB and 180dB. 221

Table 26. Estimated North Pacific right whale exposures to SURTASS LFA sonar between 120dB and 180dB. 221

Table 27. Estimted sei whale exposures to SURTASS LFA sonar between 120dB and 180dB 222

Table 28. Estimated sperm whale exposures to SURTASS LFA sonar between 120dB and 180dB 223

Table 29. Estimated Main Hawaiian Insular false killer whale exposures to SURTASS LFA sonar between 120dB and 180dB. 223

Table 30. Estimated Hawaiian monk seal exposures to SURTASS LFA sonar between 120dB and 180dB 224

Table 31. The number of individuals that are likely to be "taken" as a result of their exposure to U.S. Navy Surveillance Towed Array Sensor System Low Frequency Active Sonar by mission area number. 278

LIST OF FIGURES

Page

Figure 1. SURTASS LFA sonar systems showing the active (source array) and passive (receive array) components. 7

Figure 2. High Frequency / Marine Mammal Mitigation (HF/M3) Sonar 9

Figure 3. High frequency (HF/M3) sonar detection and low frequency mitigation zone. 12

Figure 4. Example of 12nm (22km) Stand-off for SURTASS LFA.....14

Figure 5: Mission areas in western North Pacific Ocean for SURTASS LFA sonar employment.....27

Figure 6: Mission areas in central North Pacific Ocean for SURTASS LFA sonar employment.....28

Figure 7. SURTASS LFA Sonar Risk Continuum Function 52

Figure 8. Two Hypothetical Threshold Shifts, Temporary and Permanent 198

ACRONYMS AND KEY TERMS

AIM. Acoustic Integration Model. A mathematical model, written and implemented in software by William T. Ellison, that integrates dive depth, animal distribution, and three dimensional sound field data to calculate a probability distribution for the number of animals and the proportion of time animals are exposed to different levels of sound.

Focal whale(s). An individual animal (or group of animals) selected for intensive observation during a given playback experiment.

HLA. Horizontal Line Array of hydrophones, towed behind a moving vessel.

HF-HLA. The High-Frequency Horizontal Line Array of hydrophones, towed behind the PBV in Phase Ill.

LFA. An abbreviation for SURTASS Low-Frequency Active sonar, a system designed for active detection and tracking of submarines at longer ranges than conventional (higher frequency) active sonar systems. The system uses a vertical line array of sound projectors to broadcast specially designed low-frequency (100-500 Hz) sonar pulses at high power levels, and a towed horizontal line array of hydrophones to receive echoes of the pulses from distant targets. The SURTASS LFA vessel also carries specialized signal processing and display equipment, and sophisticated systems for modeling undersea sound propagation.

LF-HLA. The Low-Frequency Horizontal Line Array of hydrophones, towed behind the PBV in Phase Ill.

OV. Observation vessel used to follow focal whales.

OV-HLA. The horizontal line array of hydrophones towed by the RHIB observation vessel in Phase III.

OV focal. An individual animal (or group of animals) followed for intensive observation using a systematic protocol during a given playback experiment.

PBV. Playback Vessel. The playback vessel in all of the experiments described here is the R/V Cory Chouest, operated by the US Navy as a development, testing, and training platform for the SURTASS LFAsystem.

Pop-Up. An autonomous acoustic data recorder that sits on the sea floor and can record acoustic data continuously for up to 23 days. The device responds to a special signal from a surface vessel

to release itself from its anchor, and "pop up" to the surface for retrieval. The stored acoustic data are then downloaded for analysis.

RHIB. The Rigid Hulled Inflatable Boat used in Phase III as the vessel for OV focal follows, measurement of received level near a whale, and recording of focal whale song. This was also called the Observation Vessel or OV.

RL. Received Level. An abbreviation for received sound intensity level, in dB re 1 p.Pa, at any specified point in space.

SCB. Southern California Bight.

SL. Source Level. An abbreviation for source sound intensity level, in dB re 1 pPa, at a distance of 1 meter from a sound source.

SoCal. Southern California.

so sus. SOund SUrveillance System. A network of bottom-mounted hydrophone arrays operated as part of the US Navy's Integrated Undersea Surveillance System. SOSUS arrays are capable of detecting and directionalizing sounds made by fin and blue whales at ranges of hundreds of miles. The proposed research will use two decommissioned SOSUS arrays to detect and track whales in the research area.

SRP. Scientific Research Permit.

SUR TASS. **SURveillance Towed Array Sensor System**. An undersea surveillance system operated by the US Navy, designed primarily for passive acoustic detection and tracking of submarines. A SUR TASS vessel is equipped with a towed horizontal line array of hydrophones, and specialized signal processing, recording, and display systems.

TDR. **Time-depth recorder**. This is a self-contained data recorder that measures and records water temperature, depth and time every 2 seconds (or any other programmable rate). The recorder is part of a tag that is attached to a whale for up to one day. The tag also includes a VHF radio transmitter that can be used to locate the whale from a vessel or aircraft.

TL. Transmission Loss. The reduction in sound intensity, in dB, between two locations (e.g., between a soundsource and a target}, equal to the difference between source level (SL) and target received level
(TRL).

TRL. Target Received Level. The sound intensity level, in dB re 1 p.Pa, experienced by a whale at any specific location relative to the LFA sound source. The proposed research uses sophisticated models of underwater sound propagation to determine the TRL for a whale at a particular location given the source level, and local bathymetric and temperature conditions. TRLs will be verified by direct measurements with a calibrated hydrophone array deployed near focal whale groups. In each playback experiment, the LFA source level will be adjusted in order to achieve a particular TRL at a focal group of whales.

TTS. Temporary Threshold Shift in hearing sensitivity. Temporary threshold shifts are known to occur in humans and some other animals as a result of exposure to loud sounds.

VHF/TDRtag. A small recoverable tag containing a time-depth recorder (TDR) and a VHF (very high frequency) beacon transmitter, used in Phase I. The tag, which measures 6cm x 3.5cm x l.Scm and weighs 40g, is attached to a whale using a cross-bow from ranges of tens of meters. A dart tip anchors the tag, which is held externally by a tether. The beacon tag can be tracked from a vessel or aircraft using a directional receiving antenna.

VLA. Vertical Line Array of hydrophones suspended from a vessel or buoy.

1 INTRODUCTION

The Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. 1531 *et seq*.) establishes a national program for conserving threatened and endangered species of fish, wildlife, plants, and the habitat upon which they depend. Section 7(a)(2) of the ESA requires Federal agencies to consult with the United States Fish and Wildlife Service (USFWS), NMFS, or both, to ensure that their actions are not likely to jeopardize the continued existence of endangered or threatened species or adversely modify or destroy their designated critical habitat. Section 7(b)(3) requires that at the conclusion of consultation, USFWS or NMFS provide an opinion stating how the agencies' actions will affect listed species and their critical habitat. If an incidental take is expected, section 7(b)(4) requires the consulting agency to provide an incidental take statement (ITS) that specifies the impact of any incidental taking and includes reasonable and prudent measures to minimize such impacts.

When a Federal agency's action "may affect" a protected species, that agency is required to consult formally with NMFS or the USFWS, depending upon the endangered species, threatened species, or designated critical habitat that may be affected by the action (50 CFR §402.14(a)). Federal agencies are exempt from this general requirement if they have concluded that an action "may affect, but is not likely to adversely affect" endangered species, threatened species, or designated critical habitat and NMFS or the USFWS concurs with that conclusion (50 CFR §402.14(b)).

The U.S. Navy submitted a request for formal ESA consultation based on their determination that multiple ESA-listed species under NMFS' jurisdiction may be affected by the proposed operation of the Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) sonar onboard four U.S. Navy vessels. NMFS's Permits and Conservation Division submitted a request for formal consultation based on their determination that the issuance of four Letters of Authorization pursuant to Federal regulations under the Marine Mammal Protection Act of 1972, as amended (MMPA; 16 U.S.C. 1361 *et seq.*), related to the U.S. Navy's use of SURTASS LFA sonar may affect several ESA-listed species.

For the actions described in this document, the action agencies are: (1) the United States Navy (U.S. Navy), which proposes to operate the SURTASS LFA sonar onboard four Tactical-Auxiliary General Ocean Surveillance (T-AGOS) vessels; and (2) NMFS Permits and Conservation Division, which proposes to issue four Letters of Authorization that would allow the U.S. Navy to "take" marine mammals incidental to those SURTASS LFA sonar activities. The consulting agency for these proposals is NMFS Office of Protected Resources, Endangered Species Act Interagency Cooperation Division.

1.1 Background

We issued the first five-year biological opinion on U. S. Navy SURTASS LFA sonar activities in 2002 (2002) and NMFS Permits and Conservation Division's promulgation of five-year regulations valid from 2002 through 2007 under the MMPA following the completion of a Final Overseas Environmental Impact Statement and Environmental Impact Statement (FOEIS/EIS) (Navy, 2001a, 2001b, 2001c) pursuant to the National Environmental Policy Act of 1969 (NEPA; ; 42 U.S.C. 4321 et seq.. The MMPA regulations allowed for the issuance of annual Letters of Authorization to the Navy that authorized the take of marine mammals incidental to the conduct of SURTASS LFA sonar operations from the R/V *Cory Chouest and the USNS IMPECCABLE*. Annual biological opinions were subsequently written on the issuance of each annual Letter of Authorization pursuant to the MMPA regulations. Each of the biological opinions completed from 2002 through 2007 concluded that the proposed actions would not jeopardize any ESA-listed species and would not result in the destruction or adverse modification of any critical habitat under NMFS jurisdiction (NMFS, 2002, 2003, 2004c, 2005b, 2007b).

On August 13, 2007, we completed a biological opinion on a second five-year period of SURTASS LFA sonar activities and a corresponding MMPA rule that was valid from August 2007 through August 2012, following the necessary analysis under NEPA (Navy, 2007c). In the 2007 biological opinion, we evaluated the potential effects of the U.S. Navy's use of SURTASS LFA sonar on up to four vessels and NMFS Permits and Conservation Division's promulgation of regulations under the MMPA (72 FR 46846, August 21, 2007) for the "take" of marine mammals over a five year period. This rule was again followed by the issuance of annual Letters of Authorization and related biological opinions. In the biological opinions completed from 2007 through 2011, we concluded that the proposed actions would not jeopardize any ESA-listed species and would not result in the destruction or adverse modification of any critical habitat under NMFS jurisdiction (NMFS, 2007b, 2008, 2009c, 2010a, 2011).

Following the Navy's NEPA analysis (Navy, 2012a), we completed a biological opinion on the third five-year period of SURTASS LFA sonar and the related MMPA rule on 13 August 2012, which is valid from August 15, 2012 through August 14, 2017 (NMFS, 2012b). These regulations (50 CFR § 218 Subpart X) are pursuant to section 101(a)(5)(A) of the MMPA and again make it possible for NMFS Permits and Conservation Division to issue annual Letters of Authorization that allow the U.S. Navy to take marine mammals incidental to the U.S. Navy's operations of SURTASS LFA sonar.

In summary, the MMPA regulations require the Navy to (1) conduct the activities in a manner that minimizes, to the greatest extent practicable, any adverse impacts on marine mammals and their habitat, (2) only take marine mammals by the method of take indicated in the regulation and Letters of Authorization, (3) maintain a running calculation/estimation of takes of each species over the effective period of these regulations, (4) ensure that no more than 12 percent of any

marine mammal stock would be taken by Level B harassment annually over the course of this five-year regulation, regardless of the number of LFA sonar vessels operating, (5) limit Level A harassment as defined by the MMPA to no more than six mysticetes (total), of any of the species listed in 50 CFR § 218.230(b)(1) over the course of the five-year regulations, (6) limit Level A harassment as defined by the MMPA to no more than 25 odontocetes (total), of any of the species listed in 50 CFR § 218.230(b)(2) over the course of the five-year regulations, and (7) limit Level A harassment as defined by the MMPA to no more than 25 pinnipeds (total), of any of the species listed in 50 CFR § 218.230(b)(3) over the course of the five-year regulations.

This rule was again followed by the issuance of annual Letters of Authorization and related biological opinions. On August 15, 2012, NMFS issued its biological opinion on the first Letters of Authorization on the effects of the U.S. Navy's use of SURTASS LFA sonar on up to four vessels and NMFS Permits and Conservation Division's proposed issuance of Letters of Authorization that would be valid from August 15, 2012 through August 14, 2013. In that biological opinion we concluded that SURTASS LFA activities were not likely to jeopardize any ESA-listed species and were not likely to result in the destruction or adverse modification of any critical habitat under NMFS jurisdiction (NMFS, 2012a). On August 15 2013, NMFS issued a biological opinion on the effects of the U.S. Navy's use of SURTASS LFA sonar on up to four vessels and NMFS Permits and Conservation Division's issuance of LOAs that are valid from August 15, 2013 through August 14, 2014. In that biological opinion we also concluded that SURTASS LFA activities were not likely to jeopardize any ESA-listed species and were not likely to result in the destruction or adverse modification of any critical habitat under NMFS jurisdiction.

1.2 Consultation History

- On March 28, 2014, we received a request from the U.S. Navy to initiate formal ESA section 7 consultation on SURTASS LFA sonar activities that would occur from 15 August 2014 through 14 August 2015.

- On April 1, 2014, we received a request from NMFS Permits and Conservation Division for formal ESA section 7 consultation on their proposal to issue four annual Letters of Authorization (LOAs) pursuant to the MMPA on the Navy's use of SURTASS LFA sonar aboard four vessels. The reinitiation package included draft LOAs.

- On April 29, 2014, we initiated consultation and subsequently prepared this biological opinion and incidental take statement in accordance with section 7(b) of the ESA of 1973, as amended (16 U.S.C. 1531, *et seq*.), and implementing regulations at 50 CFR § 402.

This opinion is based on information provided in the various documents cited above and the unclassified annual monitoring reports, final comprehensive reports, and quarterly reports for the 2013 to 2014 LOAs for the periods of August 2013 to May 2014 (Navy, 2013 -a, 2013 -b, 2013 -

c, 2013 -d) that the U.S. Navy submitted to comply with the reporting requirements of the incidental take statements contained in previous and current biological opinions on MMPA Letters of Authorization for the employment of the SURTASS LFA sonar system (Navy, 2003, 2004, 2005a, 2005b, 2006a, 2007a, 2007b, 2007c, 2008a, 2008b, 2009a, 2009b, 2010a, 2010b, 2011a, 2012a), various documents and maps that NMFS Permits and Conservation Division prepared in support of the proposed Letters of Authorization, previous biological opinions on SURTASS LFA sonar activities (NMFS, 2002, 2007a, 2008, 2009c, 2012b), and other published and unpublished scientific and technical information.

- On July 3, 2014, NMFS issued the final determination to list the Central and Southwest (SW) Atlantic Distinct Population Segment (DPS) and the Indo-West Pacific DPS of scalloped hammerhead shark (*Sphyrna lewini*) as threatened species under the Endangered Species Act (ESA). NMFS also issued a final determination to list the Eastern Atlantic DPS and Eastern Pacific DPS of scalloped hammerhead sharks as endangered species under the ESA.

- On July 28, 2014, the U.S. Navy provided additional information on the potential effects of SURTASS LFA sonar operations and training on Indo-West Pacific DPS of the scalloped hammerhead shark with a determination of may affect, not likely to adversely affect. The Navy proposes to employ SURTASS LFA sonar in nine mission areas of the northwest Pacific Ocean during the period August 2014 through August 2015. The location of this DPS coincides with the locations of the SURTASS LFA northwest Pacific mission areas.

2 DESCRIPTION OF THE PROPOSED ACTION

"Action" means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by Federal agencies. Interrelated actions are those that are part of a larger action and depend on the larger action for their justification. Interdependent actions are those that have no independent utility apart from the action under consideration. This opinion addresses three interdependent actions: (1) the U.S. Navy SURTASS LFA sonar training, testing and operational activities; (2) the regulations promulgated by NMFS's Permits and Conservation Division pursuant to the MMPA governing the U.S. Navy's "take" of marine mammals incidental to SURTASS LFA sonar activities; and (3) NMFS's Permits and Conservation Division's LOA issued pursuant to the regulations that authorizes the U.S. Navy to "take" marine mammals incidental to SURTASS LFA sonar activities (authorized by 50 CFR § 218.110) from August 2014 through August 2015. This Opinion supercedes the 2013 Biological Opinion.

This biological opinion considers two related actions proposed by the U.S. Navy and NMFS Permits and Conservation Division. The Navy proposes SURTASS LFA sonar training, testing and operations on up to four vessels. NMFS Permits and Conservation proposes to issue four Letters of Authorization to the U.S. Navy governing the take and importation of marine mammals (50 CFR § 218 Subpart X). The four Letters of Authorization would authorize the U.S. Navy to take marine mammals incidental to their training, testing, and operations of SURTASS

LFA sonar on four vessels. These Letters of Authorization would be effective for one year from 15 August 2014 to 14 August 2015. The purpose of the Navy's proposed action is to meet the United States' need for an improved ability to detect quieter and harder-to-find foreign submarines at long range to provide U.S. forces with adequate time to respond to potential submarine threats (Navy, 2012a).

This approach is consistent with Congress' intent that we coordinate and integrate the decision-making process under MMPA and ESA to the maximum extent practicable, so this opinion analyzes the training, testing and operational activities during the time and in the geographic area covered by the MMPA regulations, which are limited to "periods of not more than five consecutive years." 16 U.S.C. 1371(a)(5)(A)(i). Further, NMFS has determined to structure this consultation in this way to ensure that the effects of reasonably anticipated training, testing and operational activities may be analyzed close in time to their occurrence.

NMFS recognizes that while Navy SURTASS LFA training, testing and operational requirements change over time in response to global or geopolitical events and other factors, the general types of activities addressed by this consultation are expected to continue into the reasonably foreseeable future, along with the associated impacts. Therefore, as part of our effects analysis, we assumed that the activities proposed for the period of August 15, 2014 through August 14, 2014 and the remaining period of the five-year (2012-2017) MMPA rule would continue into the reasonably foreseeable future at levels similar to that assessed in this opinion, and we considered the direct and indirect effects of those assumed future activities, together with the effects of all interrelated and interdependent actions. This approach addresses the recent court decision in Intertribal Sinkyone Wilderness Council v. National Marine Fisheries Service et al., No. 1:12-cv-00420-NJV (N.D. Cal. Sept. 25, 2013), although we may consider a different approach in future actions.

Notwithstanding this analysis, however, NMFS would fully take into account all of the best available science and any change in the status of the species when and if the Navy applies for a new MMPA incidental take authorization for SURTASS LFA upon expiration of the five-year regulations considered in this opinion. The Navy would also need to initiate a new ESA consultation at that time.

2.1 U.S. Navy Proposed Activities

The Navy proposed to use SURTASS LFA sonar systems (LFA sonar or compact LFA [CLFA] sonar) on a maximum of four United States Naval ships during routine training and testing as well as during military operations in areas of the western and central North Pacific Ocean from August 15, 2014 through August 14, 2015. The vessels are: USNS IMPECCABLE (T-AGOS 23), USNS ABLE (T-AGOS 20), USNS EFFECTIVE (T-AGOS 21), and USNS VICTORIOUS (T-AGOS 19). No more than four LFA sonar systems are expected to be in use during this period.

2.1.1 Deployment

The Navy proposes an estimated combined total of 20 active sonar missions for four SURTASS LFA vessels in the following operational areas:

- Up to 16 missions in the northwestern Pacific Ocean, which includes the following mission areas: east of Japan; the North Philippine Sea; the west Philippine Sea; offshore Guam; the Sea of Japan; the East China Sea; the South China Sea; and offshore Japan (25° to 40° N and 10° to 25° N).

- Up to 4 missions in the north-central Pacific Ocean, which includes the northern and southern Hawaii mission areas located within the Navy's Hawaii Range Complex.

Each vessel is expected to spend approximately 54 days in transit between ports and operation areas as well as up to 240 days performing active sonar operations. Between missions, an estimated total of 71 days per year will be spent in port for upkeep and repair to maintain both the material condition of the vessel, its systems, and the morale of the crew. The maximum number of actual sonar transmission hours per vessel for the period will not exceed 432 hours. Therefore, the total number of active sonar hours will not exceed 1,728 hours over the one-year period.

2.1.2 Active Sonar Components

The SURTASS LFA sonar systems are long-range sensors that operate in the low frequency (LF) band (between 100 and 500 Hertz (Hz)). SONAR is an acronym for SOund NAvigation and Ranging, and its definition includes any system that uses underwater sound, or acoustics, for observations, monitoring, and communications. Sonar systems are used for many purposes, ranging from commercial off-the-shelf "fish finders" to military anti-submarine warfare (ASW) systems for detection and classification of submarines. More information on the LFA sonar and CLFA sonar systems can be found in the SURTASS LFA sonar 2012 Supplemental Environmental Impact Study / Supplemental Overseas EIS (SEIS/SOEIS) (Navy, 2012a).

The characteristics and operating features of the active acoustic component of LFA sonar are:

- The acoustic source is a vertical line array (VLA) of up to 18 source projectors suspended beneath the vessel (Figure 2). The LFA sonar's transmitted beam is omnidirectional (360 degrees) in the horizontal, with a narrow vertical beamwidth that can be steered above or below the horizontal.

- The source frequency is between 100 and 500 Hz. A variety of signal types can be used, including continuous wave (CW) and frequency-modulated (FM) signals.

- The source level (SL) of an individual source projector of the SURTASS LFA sonar array is approximately 215 dB re: 1 micro Pascal (µPa) at 1 meter (m) (rms) or less. As

measured by sound pressure level (SPL), the sound field of the array can never be higher than the SL of an individual source projector.

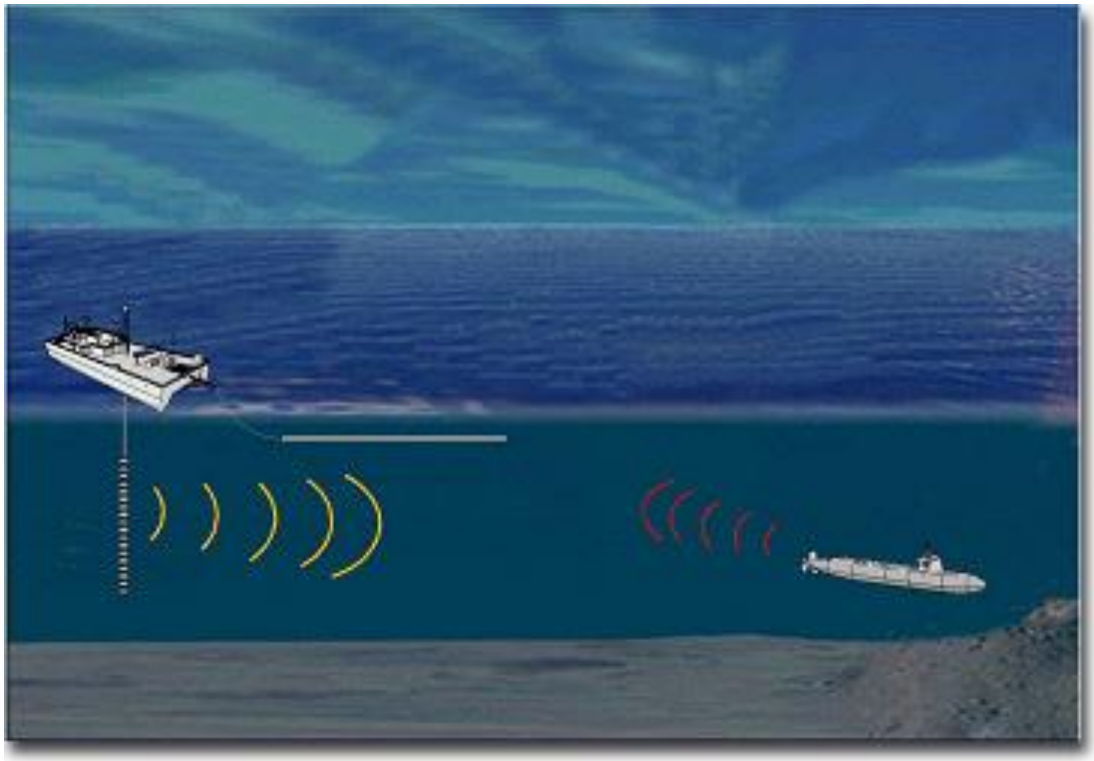

Figure 1. SURTASS LFA sonar systems showing the active (source array) and passive (receive array) components.

2.1.3 Passive Sonar Components

The SURTASS passive, or listening, part of the system detects returning echoes from submerged objects, such as threat submarines, through the use of hydrophones. These devices transform mechanical energy (received acoustic sound waves) to an electrical signal that can be analyzed by the processing system of the sonar. Advances in passive acoustic technology have led to the development of the SURTASS Twin-line (TL-29A) horizontal line array, a shallow water variant of the single line SURTASS system. TL-29A consists of a "Y" shaped array with two apertures. The array is approximately 1/5th the length of a standard SURTASS array, or approximately 305 m (1,000 ft) long. The TL-29A delivers enhanced capabilities, such as its ability to be towed in shallow water environments in the littoral zone, to provide significant directional noise rejection, and to resolve bearing ambiguities without having to change vessel course. The SURTASS TL-29A horizontal line array provides improved littoral capability. The passive capabilities of all SURTASS LFA sonar/CLFA sonar vessels have been upgraded with the installation of the TL-29A array.

The SURTASS LFA sonar vessel typically maintains a speed of at least 5.6 kilometers per hour (kph) (3 knots (kt)) through the water to tow the horizontal line array. The return signals, which are usually below background or ambient noise level, are then processed and evaluated to identify and classify potential underwater threats.

2.1.4 Mitigation and Monitoring

The following mitigation and mitigation monitoring measures are required when SURTASS LFA sonar is transmitting to reduce the potential for injury to marine animals. The 180-decibels (dB) isopleth mitigation zone covers the ocean volume ensonified to received levels of 180 dB re: 1 μPa_{rms} (sound pressure level) and greater by the SURTASS LFA sonar transmit array. Prior to commencing and during SURTASS LFA sonar transmissions, the Navy will determine the propagation of LFA sonar signals in the ocean and the distance from the SURTASS LFA sonar source to the 180-dB re: 1 μPa isopleths (the LFA sonar mitigation zone). As an added measure, NMFS requires a ‘‘buffer zone’’ that extends an additional 1 km (0.54 nmi) beyond the 180-dB LFA sonar mitigation zone.

2.1.4.1 Visual Monitoring

Visual monitoring for marine mammals and sea turtles from the vessel during daylight hours by personnel trained to detect and identify marine mammals and sea turtles includes daytime observations from the SURTASS LFA sonar vessels. Visual monitoring begins 30 minutes before sunrise or 30 minutes before the first SURTASS LFA sonar transmission. Monitoring continues for at least 15 minutes after completion of the SURTASS LFA sonar transmission exercise or until 30 minutes after sunset. The objective of these visual observations is to ensure that no marine mammals or sea turtles approach the active acoustic source close enough to enter the LFA sonar mitigation and buffer zones and to maintain a record of observed marine mammals and/or sea turtles.

Visual observations are made by designated ship personnel trained in detecting and identifying marine mammals and sea turtles. The trained observers maintain a topside watch and marine mammal/sea turtle observation log during any active SURTASS LFA sonar transmissions. The number and identification of marine mammals/sea turtles sighted, date/time of sighting, bearing and range from vessel, as well as any unusual behavior, is entered into the log.

If a potentially affected marine mammal or sea turtle is sighted outside of the LFA sonar mitigation and buffer zones, the observer notifies the military detachment (MILDET) Officer in Charge (OIC). The MILDET OIC then notifies the sonar operator for the high frequency /marine mammal mitigation (HF/M3) sonar (discussed in more detail below) to determine the range and projected track of the animal. If it is determined that the animal will travel into the LFA sonar mitigation and buffer zones, the MILDET OIC orders the delay or suspension of SURTASS LFA sonar transmissions when the animal is observed entering the LFA sonar mitigation and buffer zone. If the animal is visually observed anywhere within the LFA sonar mitigation and buffer zone, the MILDET OIC orders the immediate suspension of SURTASS LFA sonar

transmissions. The observer continues visual monitoring and recording until the animal is no longer seen. All sightings are recorded in the sighting log and provided as part of the Long-term Monitoring (LTM) Program.

2.1.4.2 Passive Acoustic Monitoring

Passive acoustic monitoring will be conducted using the passive (low frequency) SURTASS towed horizontal line array to listen for sounds generated by marine mammals as an indicator of their presence. If a received sound is estimated to be from a marine mammal within proximity of the SURTASS LFA sonar vessel, the monitoring technician notifies the MILDET OIC, who in turn alerts the HF/M3 sonar operator (discussed in more detail below) and visual observers (during daylight hours) to the presence of a marine mammal. If passive acoustic monitoring detects a marine mammal prior to or during sonar transmissions, the MILDET OIC orders a delay or suspension of SURTASS LFA sonar transmissions when the HF/M3 sonar and/or visual observers indicate that the animal enters the LFA sonar mitigation and buffer zones. All contacts are recorded in the log and provided as part of the LTM Program.

2.1.4.3 Active Acoustic Monitoring

Active acoustic monitoring uses the HF/M3 sonar, which is a Navy-developed, enhanced HF commercial sonar designed to detect, locate, and track marine mammals (and possibly sea turtles), that may pass close enough to the SURTASS LFA sonar's transmitter array to enter the LFA sonar mitigation and buffer zones.

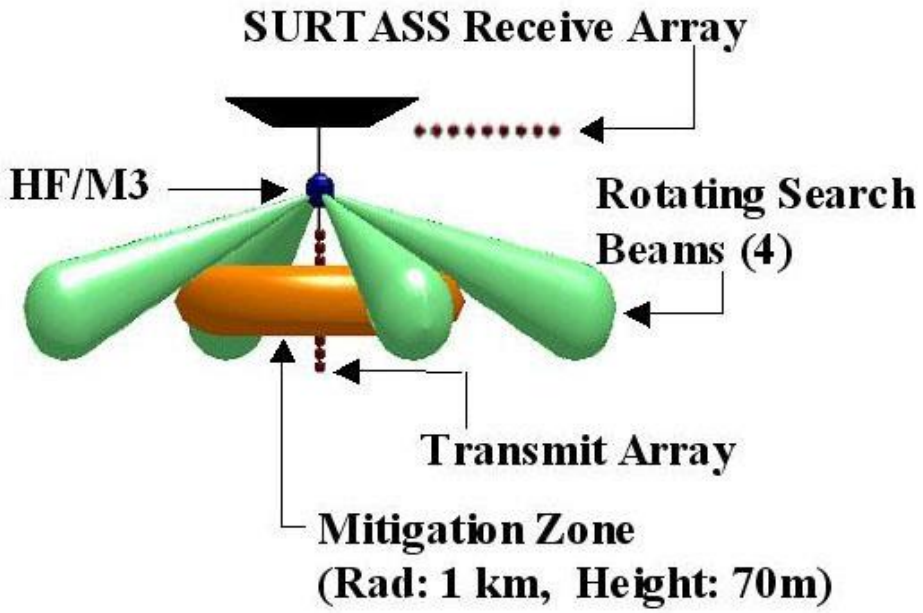

Figure 2. High Frequency / Marine Mammal Mitigation (HF/M3) Sonar

HF/M3 acoustic monitoring begins 30 minutes before the first SURTASS LFA sonar transmission of a given mission is scheduled to commence and continues until sonar transmissions are terminated. Prior to full-power sonar operations, the power level of the HF/M3 sonar is ramped up over a period of 5 minutes from the source level of 180 dB re: 1 µPa at 1 m (rms) in 10-dB increments until full power (if required) is attained to ensure that there are no inadvertent exposures of local animals to received levels ≥180 dB re: 1 µPa at 1 m (rms) from the HF/M3 sonar.

If the HF/M3 sonar detects a contact outside the LFA sonar mitigation and buffer zones, the HF/M3 sonar operator determines the range and projected track of the animal. If the operator determines that the animal will pass within the LFA sonar mitigation and buffer zones, the sonar operator notifies the MILDET OIC. The MILDET OIC then orders the delay or suspension of LFA sonar transmissions when the animal is predicted to enter the LFA sonar mitigation and buffer zones. Also, if a marine mammal or sea turtle is detected by the HF/M3 sonar within the LFA sonar mitigation and buffer zone, the operator notifies the MILDET OIC, who orders the immediate delay or suspension of transmissions. All contacts are recorded in the log and reported as part of the Navy's Long-Term Monitoring (LTM) Program. All acoustic contacts are recorded in a log and provided as part of the LTM Program.

2.1.4.4 Resumption of SURTASS LFA Sonar Transmissions

The Navy may resume/commence SURTASS LFA sonar transmissions 15 minutes after 1) all marine mammals have left the area of the LFA sonar mitigation zone and the 1-km buffer zone; and/or 2) there is no further detection of any marine mammal within the LFA sonar mitigation zone plus the 1-km buffer zone as determined by the passive or active acoustic or visual monitoring protocols.

2.1.5 Mitigation

Mitigation, as defined by the Council on Environmental Quality (CEQ), includes measures undertaken to minimize impacts by limiting the degree or magnitude of a proposed action and its implementation. The objectives of the mitigation and monitoring measures presented for use when SURTASS LFA sonar is transmitting are to effect the least practicable adverse impact on marine mammal species or stocks and to reduce the likelihood of adverse effects to ESA-listed marine species or adverse effects to their designated critical habitats. Under the terms and conditions of current and past biological opinions and incidental take statements, the Navy is required to carry out all mitigation and monitoring requirements contained in the Letters of Authorization issued under section 101(a)(5)(A) of the MMPA. Summaries of the results of these requirements are documented in quarterly, annual, and comprehensive reports.

2.1.5.1 HF/M3 Sonar

The Navy will conduct HF active acoustic monitoring (through the use of HF/M3 sonar) to detect, locate, and track marine mammals (and possibly sea turtles) that pass close enough to the SURTASS LFA sonar array to exceed the 180-dB mitigation criterion. This HF/M3 sonar

operates with a similar power level, signal type, and frequency as HF "fish finder" type sonars used worldwide by both commercial and recreational fishermen.

The HF/M3 sonar is located near the top of the SURTASS LFA sonar vertical line array (Figure 3). The HF/M3 sonar computer terminal for data acquisition/processing/display is located in the SURTASS Operations Center (SOC) aboard the SURTASS LFA sonar vessel. The general characteristics of the HF/M3 sonar are:

- Frequency: 30 to 40 kHz;
- Bandwidth: variable (1.5 to 6 kHz nominal);
- Duty Cycle: 3 to 4 percent (nominal);
- Nominal Source Level: 220 dB re: 1 µPa at 1 m;
- Pulse Length: variable (10 to 40 msec nominal);
- Pulse Repetition Rate: set by maximum search range (3 to 4 sec nominal);
- Source Ramp-Up: five-minute period;
- Detection Volume: 4 equally spaced swept 8º (horizontal) x 10º (vertical) beams making up a 10º (vertical) sector sweep through full 360º (horizontal) around the source (i.e., omnidirectional in the horizontal, 10º vertical beamwidth); nominal time for full 360º sweep 45 to 60 seconds;
- Maximum Detection Range: nominally 2 km (1.08 nmi).
- Operational Depth Capability: compatible with maximum deployed depth of SURTASS LFA sonar source array;
- Vertical Steering: ±10º; and
- Receiver Gain: 23 dB (nominal versus omnidirectional noise).

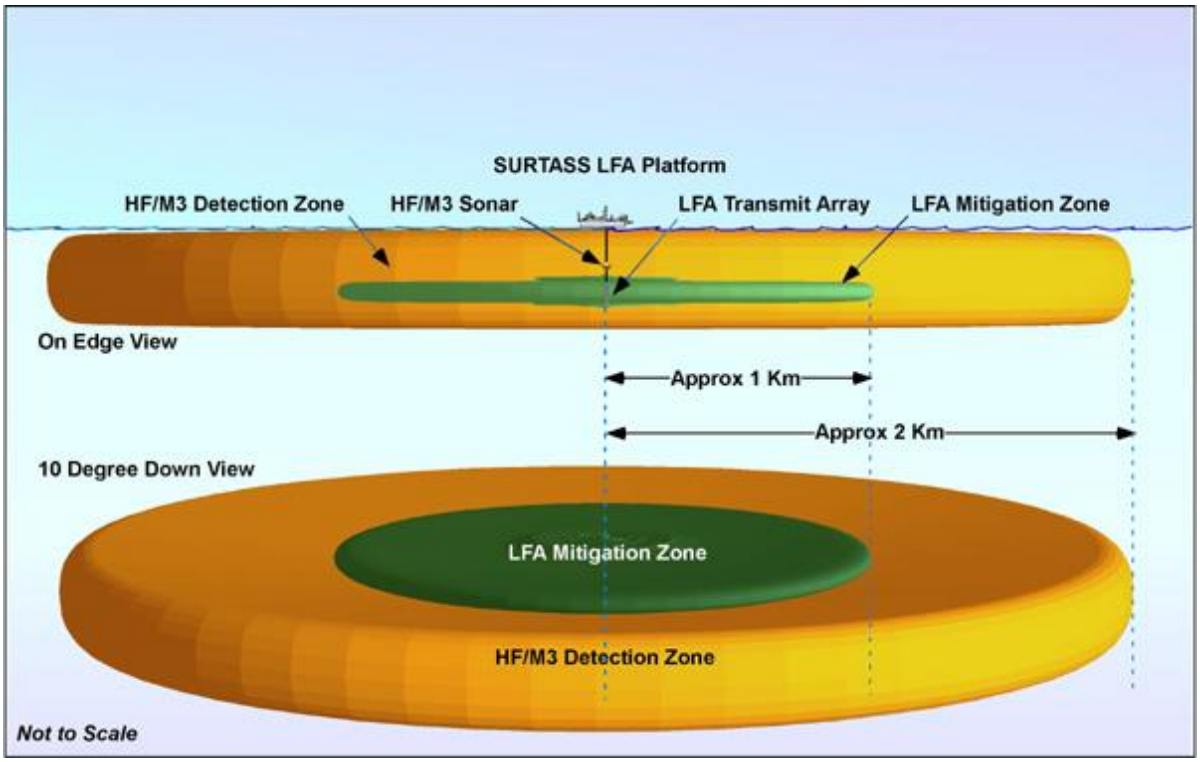

Figure 3. High frequency (HF/M3) sonar detection and low frequency mitigation zone.

The HF/M3 sonar will operate 30 minutes prior to commencement of LFA sonar transmissions and continuously while the SURTASS LFA sonar is active. Detection of a marine animal by the HF/M3 sonar automatically triggers an alert to the SOC Watch Supervisor, who has the HF/M3 tracking team immediately evaluate the detection.

Analysis and testing of the HF/M3 sonar operating capabilities indicates that this system substantially increases the probability of detecting marine mammals within the LFA sonar mitigation and buffer zones and provides an excellent monitoring capability (particularly for medium-large marine mammals) beyond the LFA sonar mitigation and buffer zones, out to approximately 2 km (1.08 nmi) (Ellison & Stein, 1999). In fact, the Navy's tests have shown that the HF/M3 system is nearly 100 percent effective using multiple pings in detecting marine mammals of any size.

The HF/M3 system also increases the likelihood of detecting marine turtles. Because the HF/M3 sonar is positioned at the top of the LFA vertical array, sea turtles would first have to swim through the 1-km HF/M3 sonar detection zone (*i.e.*, the very same 1-km buffer zone for marine mammals) where detection is highly likely, before entering the LFA sonar mitigation zone.

2.1.5.2 Coastal and Dive Site Restrictions

Based on the analyses presented in SURTASS LFA sonar NEPA documents (Navy, 2001b, 2001c, 2007c, 2012a), NMFS and the Navy have developed the following geographic restrictions for the deployment of SURTASS LFA sonar. During SURTASS LFA sonar operations, the sound field produced by the LFA sonar will not exceed 180 dB re: 1 µPa at 1 m (rms) sound pressure level within 22 km (12 nmi) of any coastline (including offshore islands). This geographic restriction is known as the "coastal standoff range" for use of SURTASS LFA sonar. Additionally, the Navy will operate SURTASS LFA sonar so that the produced sound field will not exceed received levels of 145 dB re: 1 μPa_{rms} within known recreational and commercial dive sites.

2.1.5.3 Offshore Biologically Important Areas (OBIAs) for Marine Mammals

For the use of SURTASS LFA sonar, OBIAs (Figure 4) for SURTASS LFA sonar operations are defined as those areas of the world's oceans outside of the geographic stand-off range of any coastline (i.e., 22 km/12 nmi) with: (1) high densities of marine mammals; (2) known/defined breeding/calving grounds, foraging grounds, migration routes; or (3) small, distinct populations of marine mammals with limited distributions.

The 2012 NMFS MMPA Rule for the operation of SURTASS LFA sonar requires the Navy to refrain from operating SURTASS LFA sonar within any OBIA for SURTASS LFA sonar during biologically important seasons and further requires that the SURTASS LFA sonar vessel ensures that the 180 dB re: 1 μPa_{rms} isopleth remains at least 1 km (0.54 nmi) seaward of the outer perimeter of the OBIA (50 CFR § 218 Subpart X).

Under the 2012 MMPA regulation, has designated 22 OBIAs. A detailed description of the OBIA derivation process, assessment conclusions, and specific OBIA boundary coordinates may be found in the Final SEIS/SOEIS for SURTASS LFA sonar (Navy, 2012a) and in the final rulemaking for SURTASS LFA sonar. However, all of the OBIAs will be outside of the U.S. Navy's proposed operational areas during the effective period of the proposed Letters of Authorization except the Hawaiian Islands Humpback Whale National Marine Sanctuary—Penguin Bank OBIA located in the North-central Pacific Ocean.

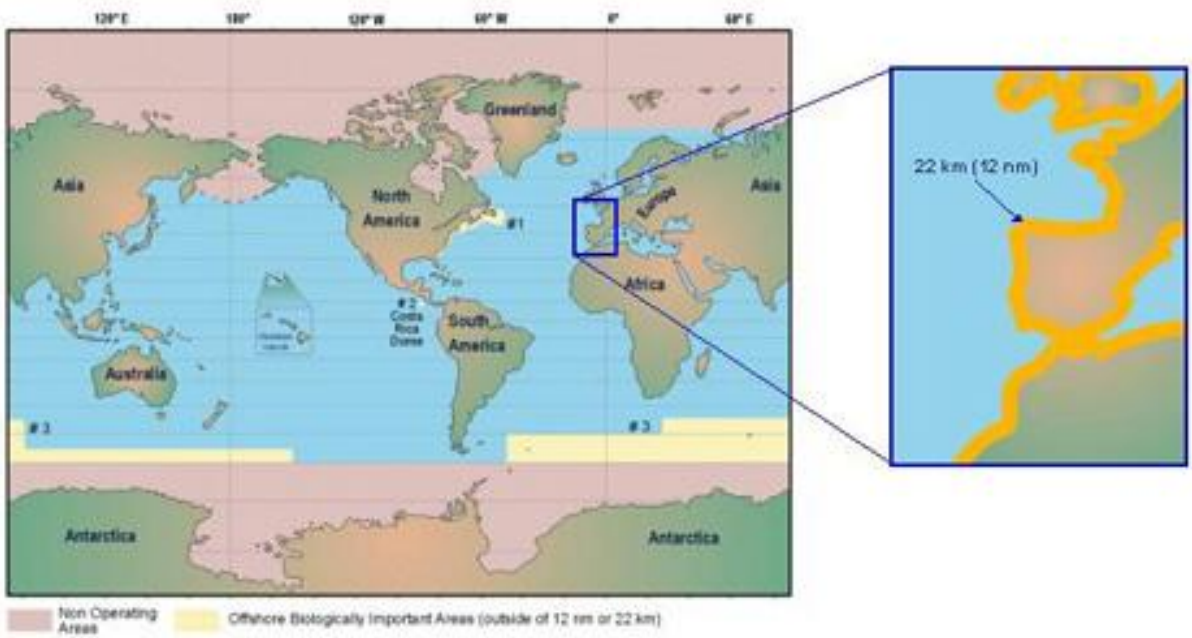

Figure 4. Example of 12nm (22km) Stand-off for SURTASS LFA

2.2 NMFS Permits and Conservation Division Proposed Activities (2014-2015)

NMFS Permits and Conservation Division proposes to issue four Letters of Authorization (one for each SURTASS LFA sonar vessel) pursuant to the MMPA implementing regulations and the regulations governing taking of marine mammals incidental to SURTASS LFA sonar activities that would authorize the incidental "take" of marine mammals by the U.S. Navy. The Letters of Authorization will include requirements for monitoring, mitigation, reporting, geographical restrictions, and long-term monitoring for SURTASS LFA sonar operations. The MMPA specifically limits the NMFS authority to allow for incidental, but not intentional takes, during periods of not more than five consecutive years (MMPA 101 (a)(5)(A)(i)). Each Letter of Authorization is valid from August 15, 2014 through August 14, 2015 pursuant to MMPA regulations at 50 CFR § 218 Subpart X. The Letters of Authorization will allow the U.S. Navy to incidentally, but not intentionally, take marine mammals by harassment within the action area.

2.2.1 Authorization of Incidental Take of Marine Mammals

The Holder of the Authorization must maintain a running calculation/estimation of takes of each species over the effective period of these regulations. The take, by Level B harassment, that occurs during the year covered by the Authorization may not exceed 12 percent of any marine mammal stock listed in the following tables. ESA-listed marine mammal species are in bold and highlighted.

Table 1. Estimated Level B Harrassment Authorized for Mission Area 1, East of Japan

East of Japan—Mission Area 1 1 Mission

Animal	Take Estimates Level B harassment
Fin whale	5
North Pacific right whale	—
Sei whale	7
Sperm whale	24

Table 2. Estimated Level B Harrassment Authorized for Mission Area 2, North Phillipine Sea

North Philippine Sea—Mission Area 2 3 Missions	
Animal	**Take Estimates Level B harassment**
Blue whale	2
Fin whale	18
Humpback whale	78
North Pacific right whale	2
Sperm whale	90

Table 3. Estimated Level B Harrassment Authorized for Mission Area 3, West Phillipine Sea

West Philippine Sea—Mission Area 3 3 Missions	
Animal	**Take Estimates Level B harassment**
Blue whale	2
Fin whale	8
Humpback whale	18
Sperm whale	108

Table 4. Estimated Level B Harrassment Authorized for Mission Area 4, Offshore Guam

Offshore Guam—Mission Area 4 3 Missions	
Animal	**Take Estimates Level B harassment**
Blue whale	2
Fin whale	2
Humpback whale	2
Sei whale	18
Sperm whale	98

Table 5. Estimated Level B Harrassment Authorized for Mission Area 5, Sea of Japan

Sea of Japan—Mission Area 5 2 Missions

Animal	Take Estimates Level B harassment
Fin whale	73
North Pacific right whale	1
Western North Pacific gray whale	2
Sperm whale	120

Table 6. Estimated Level B Harrassment Authorized for Mission Area 6, East China Sea

East China Sea—Mission Area 6 1 Mission	
Animal	**Take Estimates Level B harassment**
Fin whale	8
North Pacific right whale	-
Western North Pacific gray whale	-
Sperm whale	30

Table 7. Estimated Level B Harrassment Authorized for Mission Area 7, South China Sea

South China Sea—Mission Area 7 1 Mission	
Animal	**Take Estimates Level B harassment**
Fin whale	4
North Pacific right whale	1
Western North Pacific gray whale	1
Sperm whale	13

Table 8. Estimated Level B Harrassment Authorized for Mission Area 8, Offshore Japan 25-40° N

Offshore Japan 25-40° N—Mission Area 8 1 Mission	
Animal	**Take Estimates Level B harassment**
Fin whale	6
Sei whale	6
Sperm whale	49
Hawaiian monk seal	1

Table 9. Estimated Level B Harrassment Authorized for Mission Area 9, Offshore Japan 10-25° N

Offshore Japan 10-25° N—Mission Area 9 1 Mission

Animal	Take Estimates Level B harassment
Blue whale	1
Fin whale	1
Sei whale	5
Sperm whale	42

Table 10. Estimated Level B Harrassment Authorized for Mission Area 10, Hawaii North

Hawaii North—Mission Area 10 2 Missions	
Animal	**Take Estimates Level B harassment**
Blue whale	14
Fin whale	7
Humpback whale	10
Sei whale	1
False killer whale (Main Hawaiian Islands Insular)	2
Sperm whale	224
Hawaiian monk seal	10

Table 11. Estimated Level B Harrassment Authorized for Mission Area 11, Hawaii South

Hawaii South—Mission Area 11 2 Missions	
Animal	**Take Estimates Level B harassment**
Blue whale	9
Fin whale	5
Humpback whale	12
Sei whale	2
False killer whale (Main Hawaiian Islands Insular)	2
Sperm whale	74
Hawaiian monk seal	5

2.2.2 Mitigation Requirements

The Holder of the incidental take authorization, and any individuals operating under the holder's authority, must conduct the activity identified in 50 CFR § 218.230 and Condition 3 of the Authorization in a manner that minimizes, to the greatest extent practicable, adverse impacts on marine mammals, their habitats, and the availability of marine mammals for subsistence. When conducting operations identified in 50 CFR § 218.230, the following mitigation measures must be implemented:

(a) The Holder of this Authorization, and any individuals operating under his authority, must not broadcast the SURTASS LFA sonar signal at a frequency greater than 500 Hz.

(b) Through mitigation described under 50 CFR § 218.234 and Condition 9 (Mitigation Monitoring) of this Authorization, the Holder of this Authorization and any individuals operating under his authority must ensure, to the greatest extent practicable, that no marine mammal is subjected to a sound pressure level of 180 dB re: 1 µPa (rms) or greater.

(c) **LFA Sonar Mitigation Zone**: Prior to commencing and during SURTASS LFA sonar transmissions, the Holder of this Authorization will use near real-time environmental data and underwater acoustic prediction models to determine the propagation of the SURTASS LFA sonar signals in the mission area. The Holder must determine the distance from the SURTASS LFA sonar source to the 180-dB re: 1 µPa isopleth (rms) (*i.e.*, the LFA sonar mitigation zone) to comply with Condition 8(b).

    (i) The Holder will update these sound field estimates every 12 hours or more frequently when meteorological or oceanographic conditions change.

(d) **Additional 1-Kilometer (km) Buffer Zone**: The Holder of this Authorization will establish a 1-km buffer zone around the LFA sonar mitigation zone.

(e) **Ramp-Up Procedures for the HF/M3 System**: The Holder of this Letter of Authorization and any individuals operating under his authority, will ramp up the High Frequency / Marine Mammal Monitoring (HF/M3) active sonar referenced in 50 CFR § 218.234 from a power level beginning at a maximum source sound pressure level of 180 dB re: 1 µPa (rms) in 10-dB increments to operating levels over a period of no less than five minutes:

    (i) At least 30 minutes prior to any SURTASS LFA sonar transmission;

    (ii) Prior to any SURTASS LFA sonar calibrations or testing that are not part of regular SURTASS LFA sonar transmissions described in 50 CFR § 218.230; and

    (iii) Anytime after individuals have powered down the HF/M3 active sonar source for more than two minutes.

    (iv) Once individuals detect a marine mammal, they will not increase the HF/M3 active sonar system's sound pressure. Resumption of the ramp-up of HF/M3 sonar system would not occur until marine mammals are no longer detected by the HF/M3 active sonar system, passive acoustic monitoring or visual monitoring described in Condition 9.

(f) **Suspension/Delay for SURTASS LFA Sonar Transmissions**: If the Holder of this Authorization and any individuals operating under his authority, detects a marine mammal through monitoring required under 50 CFR § 218.235 and Condition 9 within either the LFA sonar mitigation zone or the 1-km buffer zone, the Holder will immediately suspend or delay SURTASS LFA sonar transmissions.

(g) **Resumption of SURTASS LFA Sonar Transmissions**: The Holder of this Authorization and any individuals operating under his authority may resume/commence SURTASS LFA sonar transmissions 15 minutes after:

(i) All marine mammals have left the area of the LFA sonar mitigation zone and the 1-km buffer zone; and/or

(ii) There is no further detection of any marine mammal within the LFA sonar mitigation zone plus the 1-km buffer zone as determined by the passive or active acoustic or visual monitoring protocols described in 50 CFR § 218.235 and Condition 9.

(h) **Geographic Restrictions**: The Holder of this Authorization and any individuals operating under his authority will not operate SURTASS LFA sonar such that the SURTASS LFA sonar sound field exceeds 180 dB re: 1 μ Pa (rms):

(i) At a distance of less than or equal to 22 km (14 miles (mi); 12 nautical miles (nmi)) from any coastline, including offshore islands.

(ii) At a distance of less than or equal to 1 km (0.62 mi; 0.54 nm) seaward of the outer perimeter of any Offshore Biologically Important Area (OBIA) for marine mammals designated in 50 CFR § 218.234(f)(2) and described in Condition 8(h)(iii) during the period specified.

(iii) The OBIAs for marine mammals (with specified periods) for SURTASS LFA sonar routine training, testing, and military operations are listed in Table 12 below:

(i) **Operational Exception for SURTASS LFA Sound Field in OBIAs**: During military operations, SURTASS LFA sonar transmissions may exceed 180 dB re: 1 μPa (rms) within the boundaries of an OBIA, including operating within an OBIA, when the Holder of this Authorization determines that it is: 1) operationally necessary to continue tracking an existing underwater contact; or 2) operationally necessary to detect a new underwater contact within the OBIA. This exception does not apply to routine training and testing with the SURTASS LFA sonar systems.

(j) **Mission Planning**: The Holder of this Authorization must maintain a running calculation/estimation of takes of each species and stocks over the effective period of these regulations. The Holder of this Authorization will plan all SURTASS LFA sonar missions to ensure that no more than 12 percent of any marine mammal stock listed in 50 CFR § 218.230(b)(1) through (3) would be taken by Level B harassment annually. This annual per-stock cap of 12 percent applies regardless of the number of LFA sonar vessels operating. The Holder of this Authorization must coordinate with the Holder of the Letters of Authorization issued to the USNS VICTORIOUS, USNS EFFECTIVE, and the USNS IMPECCABLE, to ensure that this condition is met for all vessels combined.

2.2.2.1 Mitigation Monitoring

The Holder of the incidental take authorization, and any individuals operating under the holder's authority, must:

(a) Perform the following for visual mitigation monitoring:

(i) Marine mammal biologists qualified in conducting at-sea marine mammal visual monitoring from surface vessels will train and qualify designated ship personnel as lookouts to conduct at-sea visual monitoring.

(ii) Marine mammal biologists will train the lookouts in the most effective means to ensure quick and effective communication within the command structure to facilitate implementation of protective measures if they observe marine mammals.

(iii) Conduct visual monitoring from the ship's bridge during daylight hours (30 minutes before sunrise until 30 minutes after sunset) during operations that employ SURTASS LFA sonar in the active mode. Maintain a topside watch with standard binoculars (7x) and with the naked eye.

(b) Perform the following for passive acoustic monitoring:

(i) Use the low frequency, passive SURTASS sonar system to listen for vocalizing marine mammals.

(c) Perform the following for active acoustic monitoring:

(i) Use the HF/M3 active sonar to locate and track marine mammals in relation to the SURTASS LFA sonar vessel and the sound field produced by the SURTASS LFA sonar source array, subject to the ramp-up requirements in § 218.234(e) and Condition 8(e).

10. Mitigation monitoring under Conditions 9(a), (b), and (c) must:

(a) Commence at least 30 minutes before the first SURTASS LFA sonar transmission (30 minutes before sunrise for visual monitoring);

(b) Continue between sonar transmissions (pings); and

(c) Continue either for at least 15 minutes after completion of the SURTASS LFA sonar transmission exercise (30 minutes after sunset for visual monitoring) or if marine mammals are showing abnormal behavioral patterns, for a period of time until behavior patterns return to normal or conditions prevent continued observations.

Table 12. Offshore Biologically Important Areas (OBIAs).

OBIA No.	Area Name	Water Body	Significant Marine Mammal Species	Seasonal Restrictions
1	Georges Bank	Northwest Atlantic Ocean	North Atlantic right whale	Year-round
2	Roseway Basin Right Whale Conservation Area	Northwest Atlantic Ocean	North Atlantic right whale	Canadian Restriction: June through December
3	Great South Channel, U.S. Gulf of Maine, and Stellwagen Bank National Marine Sanctuary [1]	Northwest Atlantic Ocean/ Gulf of Maine	North Atlantic right whale	January 1 to November 14
4	Southeastern U.S. Right Whale Seasonal Habitat	Northwest Atlantic Ocean	North Atlantic right whale	15 November to 15 April
5	North Pacific Right Whale Critical Habitat	Northeastern Pacific Ocean/Gulf of Alaska and Bering Sea	North Pacific right whale	March through August
6	Silver Bank and Navidad Bank	Northwestern Atlantic Ocean/Caribbean Sea	Humpback whale	December through April
7	Coastal Waters of Gabon, Congo and Equatorial Guinea	Southeastern Atlantic Ocean	Humpback and blue whales	June through October
8	Patagonian Shelf Break	Southwestern Atlantic Ocean	Southern elephant seal	Year-round
9	Southern Right Whale Seasonal Habitat	Southwestern Atlantic Ocean	Southern right whale	May through December
10	Central California National Marine Sanctuaries	Northeastern Pacific Ocean	Blue and humpback whales	June thru November
11	Antarctic Convergence Zone	Southern Ocean	Blue, fin, sei, minke, humpback, and Southern right whales	October through March
12	Piltun and Chayvo Offshore Feeding Grounds—Sea of Okhotsk	Northwestern Pacific Ocean/Sea of Okhotsk	Western Pacific gray whale	June through November

OBIA No.	Area Name	Water Body	Significant Marine Mammal Species	Seasonal Restrictions
13	Coastal Waters off Madagascar	Western Indian Ocean	Humpback and blue whales	July through September for humpback whale breeding / November through December for migrating blue whales
14	Madagascar Plateau, Madagascar Ridge, and Walters Shoal	Western Indian Ocean	Pygmy blue, humpback, and Bryde's whales	November through December
15	Ligurian-Corsican-Provençal Basin and Western Pelagos Sanctuary	North-central Mediterranean Sea	Fin whale	July to August
16	Hawaiian Islands Humpback Whale National Marine Sanctuary —Penguin Bank	North-Central Pacific Ocean	Humpback whale	November through April
17	Costa Rica Dome	Eastern Tropical Pacific Ocean	Blue and humpback whales	Year-round
18	Great Barrier Reef Between 16°S and 21°S	Coral Sea/Southwestern Pacific Ocean	Humpback and dwarf minke whales	May through September
19	Bonney Upwelling	Eastern Indian Ocean	Blue, pygmy blue, and Southern right whales	December through May
20	Northern Bay of Bengal and Head of Swatch-of-No-Ground (SoNG)	Bay of Bengal/Northern Indian Ocean	Bryde's whale (small form)	Year-round
21	Olympic Coast National Marine Sanctuary and the Prairie, Barkley Canyon, and Nitnat Canyon	Northeastern Pacific Ocean	Humpback whale	Olympic Coast National Marine Sanctuary OBIA: December, January, March, April, and May / The Prairie, Barkley Canyon, and Nitnat Canyon: June to September
22	Abrolhos Bank	Southwest Atlantic Ocean	Humpback whale	August through November

[1] *The boundary of OBIA #3 encompasses the northern critical habitats of the North Atlantic right whale, Stellwagen Bank National Marine Sanctuary, and areas within the Gulf of Maine.*

2.2.3 Monitoring Requirements

Section 101(a)(5)(A) of the MMPA states that in order to issue a Letter of Authorization for an activity, NMFS must set forth "requirements pertaining to the monitoring and reporting of such taking". The MMPA implementing regulations at 50 CFR § 216.104 (a)(13) indicate that requests for Letters of Authorization must include the suggested means of accomplishing the necessary monitoring and reporting that will result in increased knowledge of the species, the level of taking, or impacts on populations of marine mammals that are expected to be present.

The Holder of the Letter of Authorization and any individuals operating under the holder's authority, for activities described in 50 CFR § 218.230 must:

(a) Cooperate with NMFS and any other federal agency for monitoring the impacts of the activity on marine mammals; and

(b) Designate qualified on-site individuals to conduct the mitigation, monitoring, and reporting activities specified in this Letter of Authorization.

The Holder of this Authorization and any individuals operating under his authority will conduct all monitoring required under the Letter of Authorization to increase knowledge of the affected marine mammal species. The Holder of this Authorization must:

(a) Complete consideration of the Scientific Advisory Group's final report on the different types of monitoring/research that could increase the understanding of the potential effects of SURTASS low-frequency active sonar transmissions on beaked whales and/or harbor porpoises.

(b) Continue to assess data from the Marine Mammal Monitoring Program and work toward making some portion of that data, after appropriate security reviews, available to scientists with appropriate clearances. Any portions of the analyses conducted by these scientists based on these data that are determined to be unclassified after appropriate security reviews should be made publicly available.

(c) Continue to explore the feasibility of coordinating with other Navy fleet assets and/or range monitoring programs to include the use of SURTASS passive sonar (towed horizontal line arrays) to augment the collection of marine mammal vocalizations before, during, and after designated exercises.

(d) Continue to collect ambient noise data and explore the feasibility of declassifying and archiving the ambient noise data for incorporation into appropriate ocean noise budget efforts.

2.2.4 Reporting Requirements

The Holder of the incidental take authorization and any individuals operating under the holder's authority must:

(a) Provide a status update to NMFS when the Holder submits the next annual application on efforts to assess the data collected by its undersea arrays and progress toward making some portion of that data, after appropriate security reviews, available to scientists with appropriate clearances.

(b) Draft a plan of action outlining a strategy for implementing the SAG's recommendations for going forward with beaked whale and/or harbor porpoise research; or describe in writing why such research is not feasible/or is unlikely to increase the understanding of the potential effects of low-frequency active sonar transmissions on beaked whales and/or harbor porpoises, to be followed by a meeting with NMFS to discuss any other potential options.

(c) Systematically observe SURTASS LFA sonar operations for injured or disabled marine mammals and monitor the principal marine mammal stranding networks and other media to correlate analysis of any whale strandings that could potentially be associated with SURTASS LFA sonar operations. The Holder and any individuals operating under his authority shall:

(i) Ensure that NMFS is notified immediately or as soon as clearance procedures allow if an injured, stranded, or dead marine mammal is found during or shortly after, and in the vicinity of, any SURTASS LFA operations. The Holder will report the incident to the Incidental Take Program Supervisor, Permits and Conservation Division, Office of Protected Resources, NMFS.

(ii) Provide NMFS with species or description of the animal(s), the condition of the animal(s) (including carcass condition if the animal is dead), location, time of first discovery, observed behaviors (if alive), and photo or video (if available).

(iii) In the event that an injured, stranded, or dead marine mammal is found by the Holder and any individuals operating under his authority, that is not in the vicinity of, or found during or shortly after SURTASS LFA sonar operations, the Holder and any individuals operating under his authority, will report the same information to NMFS as listed above as soon as operationally feasible and clearance procedures allow.

(d) In the event of a ship strike by the SURTASS LFA sonar vessel, at any time or place, the Holder and any individuals operating under his authority, must:

(i) Immediately, or as soon as clearance procedures allow, report to the NMFS the species identification (if known), location (lat/long) of the animal (or the strike if the animal has disappeared), and whether the animal is alive or dead (or unknown).

(ii) Report the incident to the Incidental Take Program Supervisor, Permits and Conservation Division, Office of Protected Resources, NMFS.

(iii) Report to the NMFS as soon as operationally feasible the size and length of the animal, an estimate of the injury status (e.g., dead, injured but alive, injured and moving, unknown, etc.).

(iv) Report to the NMFS the vessel class/type and operational status, vessel length, speed, and vessel heading as soon as feasible.

(v) Provide the NMFS a photo or video, if equipment is available.

(e) Submit classified and unclassified quarterly mission reports to the Director, Office of Protected Resources, NMFS no later than 30 days after the end of each quarter beginning on the date of effectiveness of a Letter of Authorization or as specified in the appropriate Letter of Authorization. Each quarterly mission report will include all active-mode missions completed during that quarter. At a minimum, each classified mission report must contain the following information:

(i) Dates, times, and location of each vessel during each mission;

(ii) Information on sonar transmissions during each mission and records of any delays or suspensions;

(iii) Location of the SURTASS LFA sonar mitigation and buffer zones in relation to the LFA sonar array;

(iv) Marine mammal observations including animal type and/or species, number of animals sighted, date and time of observations, type of detection (visual, passive acoustic, HF/M3 sonar), bearing and range from vessel, abnormal behavior (if any), and remarks/narrative (as necessary).

(v) The report will include the Navy's estimates of the percentages of marine mammal stocks affected (both for the quarter and cumulatively for the year covered by the Authorization) by SURTASS LFA sonar operations (both within and outside the LFA sonar mitigation zone), using predictive modeling based on operating locations, dates/times of operations, system characteristics, oceanographic environmental conditions, and animal demographics.

(vi) In the event that no SURTASS LFA sonar missions are completed during a quarter, a report of negative activity will be provided.

(f) Submit an annual, unclassified report to the Director, Office of Protected Resources, NMFS, no later than 45 days after expiration of this Authorization. At a minimum, the annual report will contain the following:

(i) An unclassified summary of the year's quarterly reports;

(ii) The Navy's estimates of the percentages of marine mammal stocks affected by SURTASS LFA sonar operations (both within and outside the LFA sonar mitigation zone), using predictive modeling based on operating locations, dates/times of operations, system characteristics, oceanographic environmental conditions, and animal demographics.

(iii) An analysis of the effectiveness of the mitigation measures with recommendations for improvements, where applicable;

(iv) An assessment of any long-term effects from SURTASS LFA sonar operations; and

(v) Any discernible or estimated cumulative impacts from SURTASS LFA sonar operations.

2.3 Action Area

"Action area" refers to all areas to be affected directly or indirectly by the proposed Federal action and not merely the immediate area involved in the action (50 CFR § 402.02). The action area for this biological opinion includes areas within the Pacific Ocean:

- East of Japan; the North Philippine Sea; the west Philippine Sea; offshore Guam; the Sea of Japan; the East China Sea; the South China Sea; and offshore Japan (25° to 40° N and 10° to 25° N) See Figure 4.

- The central North Pacific Ocean, which includes the North and South Hawaii mission areas within the Hawaii Range Complex. See Figure 5.

- The proposed action may occur in any marine waters within these areas that are (1) more than 22 km (12 nmi) from any coastline, and (2) are not within the Hawaiian Islands Humpback Whale National Marine Sanctuary—Penguin Bank OBIA (OBIA #16) located in the north-central Pacific Ocean (Table 12) from November to April, and areas subject to the 180 dB restrictions delineated in previous sections, unless the 145 dB operational exemption applies.

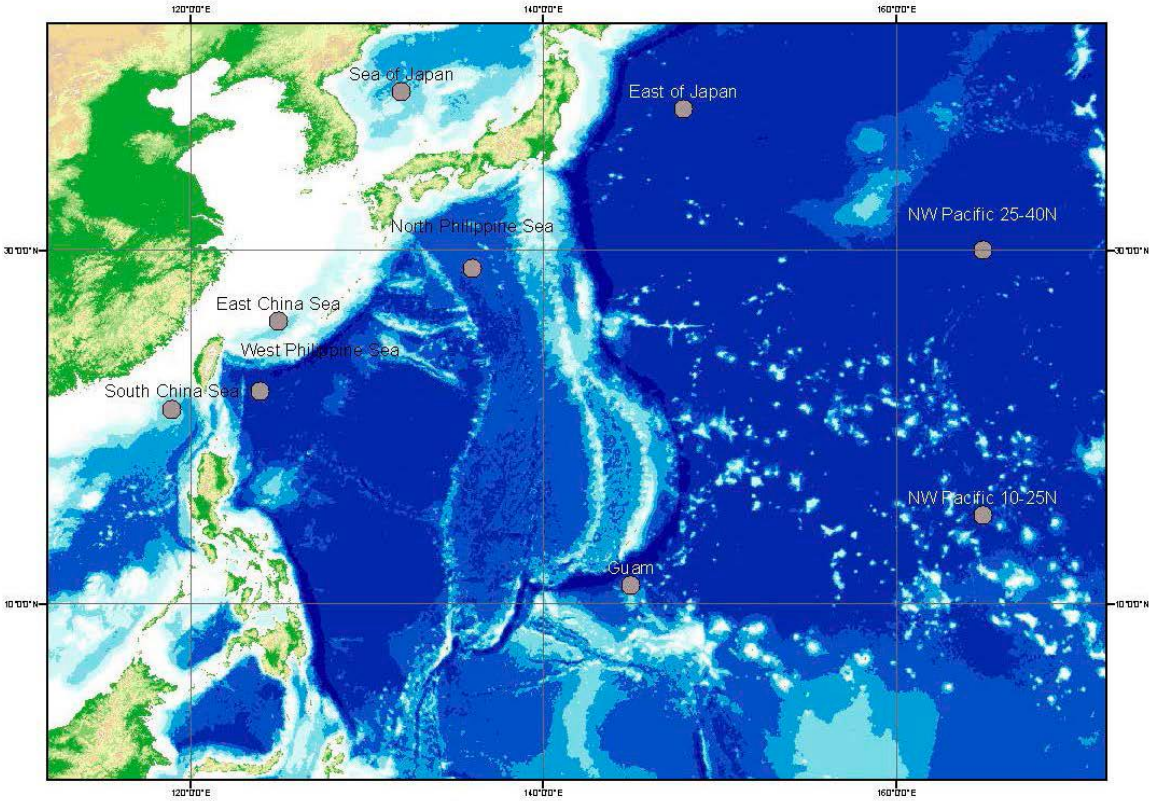

Figure 5: Mission areas in western North Pacific Ocean for SURTASS LFA sonar employment.

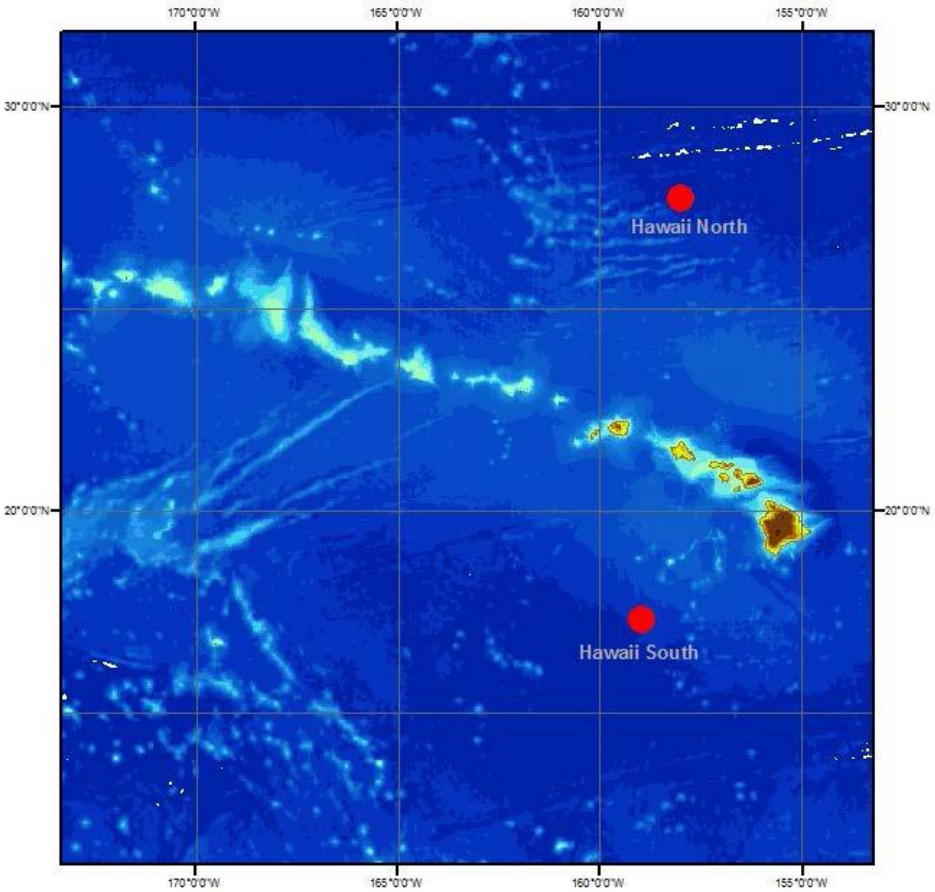

Figure 6: Mission areas in central North Pacific Ocean for SURTASS LFA sonar employment.

3 APPROACH TO THE ASSESSMENT

Section 7(a)(2) of the ESA requires Federal agencies, in consultation with NMFS, to insure that their actions are not likely to jeopardize the continued existence of endangered or threatened species, or adversely modify or destroy their designated critical habitat. The jeopardy analysis considers both survival and recovery of the species. The adverse modification analysis considers the impacts on the conservation value of designated critical habitat.

"To jeopardize the continued existence of a listed species" means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 CFR §402.02).

This biological opinion does not rely on the regulatory definition of "destruction or adverse modification" of critical habitat at 50 C.F.R. 402.02. Instead, we have relied upon the statutory provisions of the ESA to complete the following analysis with respect to critical habitat.[1]

3.1 Overview of NMFS' Assessment Framework

We will use the following approach to determine whether the action is likely to jeopardize listed species or destroy or adversely modify critical habitat:

- Identify the rangewide status of the species and critical habitat likely to be adversely affected by the action.
- Describe the environmental baseline in the action area. The environmental baseline includes the past and present impacts of Federal, state, or private actions and other human activities *in the action area*. It includes the anticipated impacts of Federal projects that have already undergone formal or early section 7 consultation and the impacts of state or private actions that are contemporaneous with the consultation in process.
- Analyze the effects of the action on both species and their habitat. In this step, we consider how the action would affect the species' reproduction, numbers, and distribution or, in the case of salmon and steelhead, their viable salmonid population (VSP) parameters. We also evaluate the action's effects on critical habitat features.
- Describe any cumulative effects in the action area. Cumulative effects, as defined in our implementing regulations (50 CFR §402.02), are the effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area. Future Federal actions that are unrelated to the action are not considered because they require separate section 7 consultation.

We integrate and synthesize the above factors to assess the risk that the action poses to species and critical habitat. In this step (*Integration and Synthesis*), we add the effects of the action

[1] Memorandum from William T. Hogarth to Regional Administrators, Office of Protected Resources, NMFS (Application of the "Destruction or Adverse Modification" Standard Under Section 7(a)(2) of the Endangered Species Act) (November 7, 2005).

(Section 6) to the *Environmental Baseline* (Section 5) and the *Cumulative Effects* (Section 6.10) to assess whether the action could reasonably be expected to: (1) reduce appreciably the likelihood of both survival and recovery of the species in the wild by reducing its numbers, reproduction, or distribution; or (2) reduce the conservation value of designated or proposed critical habitat. These assessments are made in full consideration of the *Status of the Species* and critical habitat (Section 4).

Reach jeopardy and adverse modification C*onclusion*. In this step (Section 8) we state our conclusions regarding jeopardy and the destruction or adverse modification of critical habitat are presented in Section 8. These conclusions flow from the logic and rationale presented in Section 7 (*Integration and Synthesis*).

If necessary, **define a reasonable and prudent alternative to the action**. If, in completing the last step in the analysis, we determine that the action under consultation is likely to jeopardize the continued existence of listed species or destroy or adversely modify designated critical habitat, we must identify a reasonable and prudent alternative (RPA) to the action. The action as conducted in accordance with the RPA must not be likely to jeopardize the continued existence of listed species nor adversely modify their designated critical habitat and it must meet other regulatory requirements.

3.2 Risk Analysis for Endangered and Threatened Species

Our jeopardy determinations must be based on an action's effects on the continued existence of threatened or endangered species as those "species" have been listed, which can include true biological species, subspecies, or distinct population segments of vertebrate species. Because the continued existence of listed species depends on the fate of the populations that comprise them, the viability (that is, the probability of extinction or probability of persistence) of listed species depends on the viability of the populations that comprise the species. Similarly, the continued existence of populations are determined by the fate of the individuals that comprise them; populations grow or decline as the individuals that comprise the population live, die, grow, mature, migrate, and reproduce (or fail to do so).

Our risk analyses reflect these relationships between listed species and the populations that comprise them, and the individuals that comprise those populations. Our risk analyses begin by identifying the probable risks actions pose to listed individuals that are likely to be exposed to an action's effects. Our analyses then integrate those individuals risks to identify consequences to the populations those individuals represent. Our analyses conclude by determining the consequences of those population-level risks to the species those populations comprise.

We measure risks to listed individuals using the individual's "fitness," which are changes in an individual's growth, survival, annual reproductive success, or lifetime reproductive success. In particular, we examine the scientific and commercial data available to determine if an

individual's probable response to an Action's effects on the environment (which we identify in our *response analyses*) are likely to have consequences for the individual's fitness.

When individual, listed plants or animals are expected to experience reductions in fitness, we would expect those reductions to also reduce the abundance, reproduction rates, or growth rates (or increase variance in one or more of these rates) of the populations those individuals represent (S. C. Stearns, 1992). Reductions in one or more of these variables (or one of the variables we derive from them) is a *necessary* condition for reductions in a population's viability, which is itself a *necessary* condition for reductions in a species' viability. Therefore, when listed plants or animals exposed to an Action's effects are *not* expected to experience reductions in fitness, we would not expect that Action to have adverse consequences on the viability of the populations those individuals represent or the species those populations comprise (Anderson, 2000; Mills & Beatty, 1979; S. C. Stearns, 1992). As a result, if we conclude that listed plants or animals are *not* likely to experience reductions in their fitness, we would conclude our assessment because an Action that is not likely to affect the fitness of individuals is not likely to jeopardize the continued existence of listed species.

If, however, we conclude that individual listed plants or animals are likely to experience reductions in their fitness, our assessment tries to determine if those fitness reductions are likely to be sufficient to reduce the viability of the populations those individuals represent (measured using changes in the populations' abundance, reproduction, spatial structure and connectivity, growth rates, or variance in these measures to make inferences about the population's extinction risks). In this step of our analyses, we use the population's base condition (established in the *Environmental Baseline* and *Status of Listed Resources* sections of this Opinion) as our point of reference. Finally, our assessment tries to determine if changes in population viability are likely to be sufficient to reduce the viability of the species those populations comprise. In this step of our analyses, we use the species' status (established in the *Status of the Species* section of this Opinion) as our point of reference and we use our understanding of the general patterns and processes by which species become extinct to help inform our decision about whether changes in the performance of one or more populations are likely to affect the viability of the species those populations comprise.

3.3 Risk Analysis for Designated Critical Habitat

Our "destruction or adverse modification" determinations must be based on an action's effects on the conservation value of habitat that has been designated as critical to threatened or endangered species[2]. If an area encompassed in a critical habitat designation is likely to be exposed to the *direct or indirect consequences of the action on the natural environment*, we ask

[2] We are aware that several courts have ruled that the definition of destruction or adverse modification that appears in the section 7 regulations at 50 CFR §402.02 is invalid and do not rely on that definition for the determinations we make in this Opinion. Instead, as we explain in the text, we use the "conservation value" of critical habitat for our determinations which focuses on the designated area's ability to contribute to the conservation or the species for which the area was designated.

if primary or secondary constituent elements included in the designation (if there are any) or physical or biotic phenomena that give the designated area value for the conservation are likely to respond to that exposure.

In this step of our assessment, we identify (a) the spatial distribution of stressors and subsidies produced by an action; (b) the temporal distribution of stressors and subsidies produced by an action; (c) changes in the spatial distribution of the stressors with time; (d) the intensity of stressors in space and time; (e) the spatial distribution of physical and biological features of designated critical habitat; and (f) the temporal distribution of constituent elements of designated critical habitat.

If primary constituent elements of designated critical habitat (or physical, chemical, or biotic phenomena that give the designated area value for the conservation of listed species) are likely to respond given exposure to the *direct or indirect consequences of the proposed action on the natural environment*, we ask if those responses are likely to be sufficient to reduce the quantity, quality, or availability of those constituent elements or physical, chemical, or biotic phenomena.

In this step of our assessment, we must identify or make assumptions about (a) the habitat's probable condition before any exposure as our point of reference (that is part of the impact of the *Environmental Baseline* on the conservation value of the designated critical habitat); (b) the ecology of the habitat at the time of exposure; (c) where the exposure is likely to occur; and (d) when the exposure is likely to occur; (e) the intensity of exposure; (f) the duration of exposure; and (g) the frequency of exposure.

In this step of our assessment, we recognize that the conservation value of critical habitat, like the base condition of individuals and populations, is a dynamic property that changes over time in response to changes in land use patterns, climate (at several spatial scales), ecological processes, changes in the dynamics of biotic components of the habitat, etc. For these reasons, some areas of critical habitat might respond to an exposure when others do not. We also consider how designated critical habitat is likely to respond to any interactions and synergisms between or cumulative effects of pre-existing stressors and proposed stressors.

If the quantity, quality, or availability of the primary constituent elements of the area of designated critical habitat (or physical, chemical, or biotic phenomena) are reduced, we ask if those reductions are likely to be sufficient to reduce the conservation value of the designated critical habitat for listed species in the action area. In this step of our assessment, we combine information about the contribution of constituent elements of critical habitat (or of the physical, chemical, or biotic phenomena that give the designated area value for the conservation of listed species, particularly for older critical habitat designations that have no constituent elements) to the conservation value of those areas of critical habitat that occur in the action area, given the physical, chemical, biotic, and ecological processes that produce and maintain those constituent elements in the action area. We use the conservation value of those areas of designated critical

habitat that occur in the action area as our point of reference for this comparison. For example, if the critical habitat in the action area has limited current value or potential value for the conservation of listed species, that limited value is our point of reference for our assessment.

If the conservation value of designated critical habitat in an action area is reduced, the final step of our analyses asks if those reductions are likely to be sufficient to reduce the conservation value of the entire critical habitat designation. In this step of our assessment, we combine information about the constituent elements of critical habitat (or of the physical, chemical, or biotic phenomena that give the designated area value for the conservation of listed species, particularly for older critical habitat designations that have no constituent elements) that are likely to experience changes in quantity, quality, and availability given exposure to an action with information on the physical, chemical, biotic, and ecological processes that produce and maintain those constituent elements in the action area. We use the conservation value of the entire designated critical habitat as our point of reference for this comparison. For example, if the entire designated critical habitat has limited current value or potential value for the conservation of listed species that limited value is our point of reference for our assessment.

3.4 Defining "Significance"

In biological opinions, we focus on potential physical, chemical, or biotic stressors that are "significant" in the sense of being distinct from ambient or background. We then ask if

a. exposing individuals to those potential stressors is likely to represent a "significant" negative experience in the life history of individuals that have been exposed; and if
b. exposing individuals to those potential stressors is likely to cause the individuals to experience "significant" physical, chemical, or biotic responses; and if
c. any "significant" physical, chemical, or biotic response are likely to have "significant" consequence for the fitness of the individual animal; and if
d. exposing the physical, chemical, or biotic phenomena that we identified as constituent elements in a critical habitat designation or, in the case of critical habitat designations that do not identify constituent elements, those physical, chemical or biotic phenomena that give designated critical habitat value for the conservation of endangered or threatened species is likely to represent a "significant" change in the quantity, quality, or availability of the physical, chemical, or biotic resource; and if
e. any "significant" change in the quantity, quality, or availability of a physical, chemical, or biotic resource is likely to "significantly" reduce the conservation value of the designated critical habitat.

In all of these cases, the term "significant" means "clinically or biotically significant" rather than statistically significant because the presence or absence of statistical significance do not imply the presence or absence of clinical significance (Achinstein, 2001; Royall, 2004) (Johnson 1999).

For populations (or sub-populations, demes, etc.), we are concerned about whether the number of individuals that are likely to experience "significant" reductions in fitness and the nature of any fitness reductions are likely to have a "significant" consequence for the viability (= probability of demographic, ecological, or genetic extinction) of the population(s) those individuals represent. Here "significant" also means "clinically or biotically significant" rather than statistically significant.

For "species" (the entity that has been listed as endangered or threatened, not the biological species concept), we are concerned about whether the number of populations that are likely to experience "significant" reductions in viability (= increases in their extinction probabilities) and the nature of any reductions in viability are likely to have "significant" consequence for the viability (= probability of demographic, ecological, or genetic extinction) of the "species" those populations comprise. Here, again, "significant" also means "clinically or biotically significant" rather than statistically significant.

For designated critical habitat, we are concerned about whether the area that has been designated is likely to experience "significant" reductions in the quantity, quality, or availability of physical, chemical, or biotic resources that are likely to result in "significant" reductions in the conservation value (usually measured using the concept of "carrying capacity[3]") of the entire are contained in the designation.

3.5 Treatment of "Cumulative Impacts" (in the sense of NEPA)

The U.S. Council on Environmental Quality defined "cumulative effects" (which we refer to as "cumulative impacts" to distinguish between NEPA and ESA uses of the same term) as "the impact on the environment which results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions regardless of what agency (Federal or non-federal) or person undertakes such other actions" (40 CFR §1508.7). The effects analyses of biological opinions considered the "impacts" on listed species and designated critical habitat that result from the incremental impact of an action by identifying natural and anthropogenic stressors that affect endangered and threatened species throughout their range (the *Status of the Species*) and within an Action Area (the *Environmental Baseline*, which articulate the pre-existing *impacts* of activities that occur in an Action Area, including the past, contemporaneous, and future *impacts* of those activities). We assess the effects of a proposed action by adding their direct and indirect effects to the *impacts* of the activities we identify in an *Environmental Baseline* (50 CFR §402.02), in light of the impacts of the status of the listed species and designated critical habitat throughout their range; as a result, the results of our effects analyses are equivalent to those contained in the "cumulative impact" sections of NEPA documents.

[3] Carrying capacity refres to the largest number of individuals of a particular species that can survive over long periods of time in a given environment. This level depends on the effect of the limiting factors.

We considered potential cumulative impacts as part of our consultation. Specifically, we considered (1) impacts or effects that accumulate in the environment in the form of stressors or reservoirs of stressors and (2) impacts or effects that represent either the response of individuals, populations, or species to that accumulation of stressors in the environment or the accumulated responses of individuals, populations, and species to sequences of exposure to stressors. Further, we considered the potential impacts of these accumulative phenomema on an annual basis, over the duration of the five-year MMPA regulations, and under the assumption that these activities would continue into the reasonably foreseeable future. Given the ongoing nature of the activities, we assume that the type, amount, and extent of training, testing, and operations does not exceed maximum levels assessed in the action.

In the sense of Item 1, which captures the normal usage of "cumulative impacts," we concluded that phenomena like sound do not accumulate (sound energy rapidly transforms into other forms of energy), although phenomena like the acreage of habitat destroyed and concentrations of toxic chemicals, sediment, and other pollutants accumulate. If there is sufficient time between exposures of individuals to sound stressors below levels for permanent injury, individuals would have ability to recover. We conclude that the probability of a vessel strike accumulated, in the sense that the probabilities of collisions associated with multiple transits are higher than the probabilities associated with a single transit. We factored those considerations into our estimation of the probability of a collision associated with multiple transits.

In the sense of Item 2, we considered phenomena that accumulate in individuals and individually contribute or collectively determine the probable fitness of the individuals that comprise a population. These include, the passage of time and its corollary, the loss of time (specifically, the loss of time to reproduce, to forage, and to migrate, etc.); reproductive success; longevity; energy debt, including allostatic loading; body burdens of toxic chemicals; the fitness costs of behavioral decisions (canonical costs); injuries and tissue damage; and overstimulation of sensory organs (which would include noise-induced losses of hearing sensitivity).

At the level of populations, phenomena that "accumulate" include population abundance; the number or percent of individuals in a population with lifetime reproductive success greater than 2.0; the number or percent of individuals in a population with lifetime reproductive success equal to 2.0; the number or percent of individuals in a population with lifetime reproductive success less than 2.0; the number or percent of individuals that emigrate from a population per unit time; the number or percent of individuals that immigrate into a population per unit time; mortality within a particular age or stage over generation time; and the reservoir of juveniles in a population that have a high probability of surviving to the age of reproduction (population momentum or its absence).

At the species level, we accumulate those phenomena that allow us to estimate the extinction risks facing a species. These include increases or decreases in the number of occurrences or

populations; the extinction probability of particular occurrences; variance in the rates of population growth or decline; and demographic stochasticity.

Cummulative effects also include effects of future State, tribal, local, or private actions that are reasonably certain to occur in the action area considered in this biological opinion. Future Federal actions that are unrelated to the action are not considered in this section because they require separate consultation pursuant to Section 7 of the ESA.

3.6 Evidence Available for the Consultation

In 2002, NMFS' Endangered Species Division completed its first biological opinion on the U.S. Navy's proposed employment of the SURTASS LFA sonar system and NMFS' Permits and Conservation Division's proposal to authorize the "take" of marine mammals pursuant to the Navy's employment of that sonar system. From 2002 to 2012, inclusive, NMFS completed biological opinions on each annual letter of authorization the Permits and Conservation Division issued to the U.S. Navy for annual SURTASS LFA sonar missions. Additionally, programmatic biological opinions were completed in 2002, 2007, and in 2012. Thus, this opinion builds upon the earlier biological opinions NMFS has prepared on the employment of the SURTASS LFA sonar system and uses the evidence we collected, analyzed, and synthesized for those earlier opinions as its foundation. For this current opinion, 2014, we first identified new lines of evidence that became available since we completed the earlier opinions on the potential effects of the SURTASS LFA sonar system on endangered species, threatened species, and critical habitat that has been designated for them.

Since we completed our biological opinion on the 2012 to 2017 set of regulations and the initial Letters of Authorization that NMFS issued to authorize the "take" of marine mammals for the use of SURTASS LFA sonar system during 2012 to 2013 and 2013 through 2014 (NMFS, 2007a), new evidence has become available on several issues that are important for this assessment. We continue to expand upon the multiple years of data derived from the actual operations of the SURTASS LFA sonar system, including data on the effectiveness of the mitigation protocols associated with the sonar system (Navy, 2003, 2004, 2005a, 2006a, 2007a, 2007b, 2008b, 2009b, 2010b, 2011a). Additional research conclusions on the potential effects of the SURTASS LFA sonar system on fish species have been published (Doksaeter, Handegard, Godo, Kvadsheim, & Nordlund, 2012; Halvorsen, Wysocki, & Popper, 2006a; Kane et al., 2010; A. N. Popper et al., 2007; Sivle et al., 2012); although no ESA-listed fish species occur in the action area for this consultation. Also, a new study on gas bubble formation in marine mammals has been published (Sascha K Hooker et al., 2012). Other recent studies either were not applicable to the species analyzed in this opinion (R. Kastelein & Jennings, 2012); (Klinck et al., 2012); (J. E. Moore & Barlow, 2013); (Pirotta et al., 2012); (P. J. Miller et al., 2012; Rolland et al., 2012; Brandon L Southall et al., 2012), or otherwise had no bearing on this consultation (Dähne et al., 2013); (Kaschner, Quick, Jewell, Williams, & Harris, 2012) (P. J. Miller et al., 2012) (Van Der Hoop et al., 2013).

3.7 A Brief Background on Sound

Sound is a wave of pressure variations propagating through a medium (for the sonar considered in this opinion, the medium is marine water). Pressure variations are created by compressing and relaxing the medium. Sound measurements can be expressed in two forms: *intensity* and *pressure*. Acoustic intensity is the average rate of energy transmitted through a unit area in a specified direction and is expressed in watts per square meter (W/m^2). Acoustic intensity is rarely measured directly and is derived from ratios of *pressures*; the standard reference pressure for underwater sound is 1 micro Pascal (μPa); for airborne sound, the standard reference pressure is 20 μPa (W. John Richardson, Jr., Malme, & Thomson, 1995).

Acousticians have adopted a logarithmic scale for sound intensities, which is denoted in decibels (dB). Decibel measurements represent the ratio between a measured pressure value and a reference pressure value (i.e., 1 μPa for underwater sound or 20 μPa for airborne sound). The logarithmic nature of the scale means that each 10 dB increase is a ten-fold increase in power (e.g., 20 dB is a 100-fold increase, 30 dB is a 1,000-fold increase). Humans perceive a 10 dB increase in noise as a doubling of sound level or a 10 dB decrease in noise as a halving of sound level. The term "sound pressure level" implies a decibel measure and a reference pressure that is used as the denominator of the ratio (dB re: 1μPa). Throughout this opinion, we use 1 micro Pascal (denoted re: 1μPa) as a standard reference pressure unless noted otherwise.

It is important to note that decibels underwater and decibels in air are not the same and cannot be directly compared. Due to the different densities of air and water and the different reference pressure standards of sound in water and air, a sound with the same intensity (i.e., power) in air and in water would be approximately 63 dB quieter in air. Thus, a sound that is 160 dB loud underwater would have the same effective intensity as a sound that is 97 dB loud in air.

Sound frequency is measured in cycles per second, or Hertz (abbreviated Hz), and is analogous to musical pitch; high-frequency sounds are perceived as high-pitched while low-frequency sounds are perceived as low-pitched sounds. Natural sounds in the ocean span a wide range of frequencies: from earthquake noise at 5 Hz to harbor porpoise clicks at 150,000 Hz. These sounds are so low or so high in pitch that humans cannot even hear them; acousticians call these infrasonic and ultrasonic sounds, respectively. A single sound may be made up of many different frequencies. Sounds made up of only a small range of frequencies are called "narrowband" and sounds with a broad range of frequencies are called "broadband"; airguns are an example of a broadband sound source and sonars are an example of a narrowband sound source.

3.8 Overview of the Navy's Assessment Framework for SURTASS LFA

We provide this overview of the Navy's assessment framework which was we reviewed during consulations and base our exposure and response analsysis in this Opinion. Potential effects on marine mammals or sea turtles from operation of LFA sonar include:

• **Non-auditory impacts**: Non-auditory impacts for marine mammals include direct acoustic impact on tissue, indirect acoustic impact on tissue surrounding a structure, and acoustically mediated bubble growth within tissues from supersaturated dissolved nitrogen gas. These types of impacts have the potential for resonance of the lungs/organs, tissue damage, and mortality. There are no data on the potential for anthropogenic sound to cause non-auditory injury in sea turtles.

• **Auditory impacts**: Auditory impacts include permanent threshold shift (PTS), which is a severe situation that occurs when sound intensity is very high or of such long duration that the result is a permanent hearing loss on the part of the listener. PTS constitutes an injury while temporary threshold shift (TTS) is a lesser hearing impact caused by underwater sounds of sufficient loudness that cause a transient condition in which an animal's hearing is impaired for a period of time. Since hearing is not permanently or irrevocably damaged, TTS is not considered an injury (W. John Richardson & Wursig, 1995) (B. L. Southall et al., 2007), although during a period of TTS, animals may be at some disadvantage in terms of detecting predators or prey.

• **Behavioral change**: Behavioral responses to intense sounds in a marine animal's environment vary from subtle changes in surfacing and breathing patterns to cessation of vocalization or even active avoidance or escape from regions of high sound levels (Wartzok, et al., 2004) (D'Spain & Wartzok, 2004).

• **Masking**: The presence of intense sounds in the environment can potentially interfere with an animal's ability to hear relevant sounds. This effect, known as "auditory masking", could interfere with the animal's ability to detect biologically-relevant sounds, such as those produced by predators, prey, or reproductively active mates. During auditory masking, an animal may, thus, not be able to escape predacious attack, locate food, or find a reproductive partner.

• **Stranding:** Stranding occurs when marine mammals passively (unintentionally) or purposefully come ashore either alive, but debilitated or disoriented, or dead. Although some species of marine mammals (pinnipeds) and sea turtles routinely come ashore during all or part of their life history, stranded marine mammals are differentiated by their diseased or ill state, helplessness ashore, and inability to cope with or survive their stranded situation (i.e., they are outside their natural habitat and survival envelope) (J. R. Geraci & V. J. Lounsbury, 2005).

3.8.1 Effects on Sea Turtles Associated with Transmission of LFA Sonar

No data are available on the potential for anthropogenic sound to cause non-auditory injury in sea turtles. Although it is known that sea turtles can hear LF sound, there is limited information on their behavioral and physiological responses to LF sound underwater (DoN, 2001, 2007, and 2012). Although not directly related to SURTASS LFA sonar effects, a review of effects of explosives on sea turtles was done by (Viada et al., 2008a). For explosive structure removals in the Gulf of Mexico, NMFS specified that the area within 915 meters (m) (3,000 feet [ft]) of the structure being removed must be clear of sea turtles. Therefore, using a value of 180-dB sound

pressure level (SPL) injury threshold for sea turtles (within approximately 1,000 m [3,281 ft] of the LFA array or the mitigation zone[4] for SURTASS LFA sonar) is conservative.

Very few studies exist on the potential effects of underwater sound on sea turtles and most of the available research examined the effects of sounds of much longer duration or of different types (e.g., seismic airgun) than LFA sonar signals. Additionally, very little is known about sea turtle hearing and what, if anything, may cause a sea turtle to incur permanent or even temporary loss of hearing. However, the few data available support the premise that using a value of 180-dB as the injury threshold for sea turtles is conservative. A sea turtle would have to be within the LFA mitigation zone (≥180 dB re 1 μPa [rms] received level [RL]) when the sonar was transmitting to be at risk of acoustic injury, including permanent or temporary loss of hearing. The probability of a sea turtle being within the 180-dB mitigation zone is considered highly unlikely because of the active acoustic and visual monitoring mitigation protocols and the operating parameters of the sonar (slow moving ship, low system duty cycle, and narrow bandwidth of LFA sonar signal).

A change in behavior that causes prolonged displacement of sea turtles from the site of their normal activities could be considered a deleterious effect. Spatial displacement is an example of how sound might negatively affect a turtle's behavior, causing it to move either vertically or horizontally in the water column. For example, a sea turtle could move vertically from depth where it may have been foraging to the surface, where anthropogenic LF sound would be weaker but exposure to predation may be increased. Horizontal displacement could occur when sea turtles, such as the non-pelagic green, olive ridley, or hawksbill turtles, respond to a stimulus by moving away from preferred benthic foraging habitat. Behavioral responses to anthropogenic activity, however, have not been extensively investigated. The majority of available research is on the response of sea turtles to underwater seismic noise. Studies of captive turtles exposed to sound from individual seismic airguns suggest that they may show startle or avoidance responses to airguns (O'Hara and Wilcox, 1990; McCauley et al., 2000; Bartol and Musick, 2003). The work by O'Hara and Wilcox (1990), McCauley et al. (2000), and (S. L. DeRuiter & Larbi Doukara, 2012) reported behavioral changes of sea turtles in response to exposure to seismic airgun transmissions. O'Hara and Wilcox (1990) reported avoidance behaviors by loggerheads in response to airguns with sound levels (RL) of 175 to 176 dB re 1 μPa (peak-to-peak). McCauley et al. (2000) reported noticeable increases in swimming behavior for both green and loggerhead turtles at RLs of 166 dB re 1 μPa (peak-to-peak). At 175 dB re 1 μPa (peak-to-peak) RL, both green and loggerhead sea turtles displayed increasingly erratic behavior (McCauley et al., 2000). DeRuiter and Doukara (2012) reported that basking loggerhead turtles interrupted basking behavior and dove in response to the sound seismic airguns; 49 (or 57%) of 86 observed turtles

[4] The LFA mitigation zone covers a volume ensonified to a received level ≥180 dB re 1 μPa (rms) around the SURTASS LFA sonar array, which is centered at a nominal depth of 122 m (400 ft) below the water surface. Based on spherical spreading, the LFA mitigation zone will vary between the approximate ranges of 750 to 1,000 m (2,461 to 3,481 ft) from the source array, over a depth of approximately 87 to 157 m (285 to 515 ft).

dove at or before their closest range to the airguns and at least 6 loggerheads dove immediately following an airgun shot, often showing a startle response. However, seismic airguns transmit impulsive signals characterized by a large frequency bandwidth, high energy, and short duration signals. Therefore, airgun signals cannot be directly compared with SURTASS LFA sonar, since the signal characteristics are very different, and the likelihood of effects on living tissue dissimilar as well. Based on the hearing data on sea turtles, it is possible that if a sea turtle where in proximity during SURTASS LFA sonar transmissions that the turtle would hear the LF transmissions. Given that the majority of sea turtles encountered in the oceanic areas in which SURTASS LFA sonar is proposed to operate would in high likelihood be transiting and not lingering in the open ocean, the possibility of significant behavior changes, especially from displacement, would be unlikely. Given the SURTASS LFA sonar operational parameters mentioned above, the potential for SURTASS LFA sonar to cause behavioral changes in sea turtles must be considered negligible.

No studies of the masking on sea turtles have been conducted. Masking effects may occur for sea turtle species that have critical hearing bandwidths at the same frequencies as the SURTASS LFA sonar, but masking would likely be temporary. The geographical restrictions imposed on all SURTASS LFA sonar operations would greatly limit the potential for exposure and masking to occur in sea turtles in areas such as nesting sites where they would be aggregated, especially in large numbers. Masking effects are not expected to be significant because of the nominal 7.5% duty cycle[5], the maximum 100-second (sec) sonar signal duration, constant movement of the T-AGOS vessels, limited 30 Hz sonar bandwidth, and the LFA sonar signals not remaining at a single frequency for more than ten seconds.

3.8.2 Non-Auditory Effects for Marine Mammals

Non-auditory effects from active sonar transmissions include direct acoustic impact on tissue, indirect acoustic impact on tissue surrounding a structure, and acoustically mediated bubble growth within tissues from supersaturated dissolved nitrogen gas (Nowacek, Thorne, Johnston, & Tyack, 2007); (B. L. Southall et al., 2007).

3.8.2.1 Direct Acoustic Effects

Physical effects, such as direct acoustic trauma or acoustically enhanced bubble growth, require relatively intense received energy that would only occur at short distances from high-powered sonar sources (Nowacek et al., 2007; Zimmer and Tyack, 2007). Resonance can occur in marine mammals but does not necessarily cause injury; any injury is not expected to occur below a received sound pressure level (RL) of 180 dB re 1 μPa (rms). Damage to the lungs and large sinus cavities of cetaceans from air space resonance is not regarded as a likely significant non-auditory injury because resonance frequencies of marine mammal lungs are below that of the LFA signal (Finneran, 2003). SURTASS LFA sonar transmissions are not expected to cause gas

[5] Average duty cycle (ratio of sound "on" time to total time) of the SURTASS LFA sonar active transmission mode is less than 20%. The typical duty cycle, based on historical LFA operational parameters since 2003 is nominally 7.5 to 10%. During the remaining 80 to 92.5% of the time, LFA transmitters would be off, thus adding no sound to the water.

bubble formation or beaked whale strandings. Further, transmission or employment of SURTASS LFA sonar has never been associated with any marine mammal or sea turtle stranding.

3.8.2.2 Gas Bubble Formation

Presently, there are discussions among researchers regarding the potential for marine mammals to suffer from a form of decompression sickness caused by in vivo nitrogen gas-bubble growth. Jepson et al. (2003, 2005) {Jepson, 2003 #71846} {Jepson, 2005 #36697} and Fernandez {Fernandez, 2005 #46703} reported results of necropsies of stranded beaked whales, some of which coincided with naval sonar exercises, which they interpreted as consistent with a decompression-like syndrome {Nowacek, 2007 #65246}.

Scientists have documented bone lesions (osteonecrosis), which may be a chronic result of nitrogen bubbles, in the rib and chevron bone articulations, nasal bones, and deltoid crests of sperm whale specimens from the Atlantic and Pacific Oceans dating from the late 1800s to 2003, (Moore and Early, 2004). This suggests that nonlethal pathologies related to gas bubbles may occur during the normal life span of, at least, the deep-diving sperm whale. Houser (2007) assessed the potential for nitrogen bubble formation in a trained dolphin. Based on repetitive dives to depths of 10, 30, 50, 70, and 100 m (32.8, 98.4, 164, 230, and 328 ft), ultrasound inspections were completed on the portal and innominate veins (i.e., the left and right brachiocephalic veins). Blood samples were also taken over a 20-minute (min) period at the end of each of the 50, 70, and 100 m (164, 230, and 328 ft) dives for the assessment of nitrogen partial pressure. There were no vascular bubbles found in any post-dive ultrasound. Nitrogen partial pressures from blood samples were not significantly elevated from those of the dolphin at rest (20 min post dive). Results suggest that repetitive, prolonged dives up to 100 m (328 ft) accumulate insufficient nitrogen to generate asymptomatic intravascular bubbles in bottlenose dolphins.

Zimmer and Tyack (2007) modeled nitrogen tension and bubble growth in beaked whales during normal diving behavior and for several hypothetical dive profiles to assess the risk of nitrogen bubble formation. These authors concluded that macroscopic bubbles are unlikely to pose a risk of decompression-like syndrome from a simple interruption of a normal deep foraging dive, even when accompanied by an unrealistic ascent rate. Zimmer and Tyack (2007) concluded, contrary to the findings of Jepson et al. (2003), that the interruption and rapid ascent from a regular deep foraging dive is unlikely to pose a risk of decompression-like syndromes; they suggested that gas bubble lesions in stranded beaked whales reported by Jepson et al. (2003, 2005) and Fernandez et al. (2005) might be caused by repetitive dives of short to medium surfacing duration without exceeding the depth of alveolar collapse. Also, Zimmer and Tyack (2007) found that the longer the dive time compared to surfacing time, the greater the risk; the authors suggested the hypothesis that beaked whales have an avoidance response to killer whales and great white sharks, which are their primary near-surface predators, resulting in their swimming at depths of

approximately 25 m (82 ft) without exceeding alveolar collapse. This hypothesis requires more behavioral and physiological research.

Baird et al. (2008) investigated the variation in diving behavior from time-depth recorders on six Blainville's and two Cuvier's beaked whales. Both species demonstrated ascent rates from dives deeper than 800 m (2,625 ft) that were significantly slower than decent rates, both during the day and at night, suggesting some physiological purpose for the slower ascents. The whales also spent more time in dives to mid-water depths (100 to 600 m [328 to 1,969 ft]) during the day. At night, the whales spent more time in shallow (<100 m) dives. This diel variation (Diel means "in the course of the day"). Thus, a "diel variation" is a variation that occurs regularly every day or most days.) in behavior suggests that beaked whales may spend less time in surface waters during the day to avoid visually oriented predators, including sharks and killer whales.

Fahlman et al. (2009) modeled the effects of lung compression and collapse (pulmonary shunt) on the uptake and removal of oxygen, carbon dioxide, and nitrogen in blood and tissue, and on end-dive nitrogen concentrations for breath-holding marine mammals (e.g., elephant seals, Weddell seals, and beaked whales). Fahlman et al. (2009) suggested that repeated dives might result in tissue and blood levels of nitrogen sufficient to cause symptomatic bubble formation.

Based on the current knowledge of gas exchange and physiology of marine mammals, Hooker et al. (2009) developed a mathematical model to predict blood and tissue levels of nitrogen gas for three species of beaked whales: northern bottlenose, Cuvier's, and Blainville's beaked whales. Hooker et al. suggested that deep-diving marine mammals live with and manage high levels of nitrogen gas in their tissues and blood. Due to differences in dive behavior, predicted nitrogen levels were higher in Cuvier's beaked whales than in northern bottlenose whales and Blainville's beaked whales. Hooker et al. (2009) state that while the prevalence of Cuvier's beaked whale strandings after naval sonar exercises could be explained by a higher abundance of the species in the area, their results suggest that species differences in behavior and/or physiology may also play a role.

Moore et al. (2009) performed gross histologic and radiographic observations related to the presence of gas bubbles in the tissues and blood of seals and dolphins drowned in gillnets, set at a depth of approximately 80 m (263 ft). The majority (15 of 23) of the seals and dolphins had extensive bubble formation in multiple tissues and blood. In addition, computer tomography, which was performed on four randomly-selected marine mammals, identified gas bubbles in various tissues. Due to the good condition of the carcasses, absence of bacteria and autolytic (self-digestion) changes, the study concluded that peri- or post-mortem phase change of supersaturated blood and tissues was the most likely cause of the bubbles. Overall, Moore et al. (2009) found a high prevalence of vascular and interstitial bubbles in seals and dolphins drowned in gillnets set at a depth of approximately 80 m (263 ft). In contrast, a very low prevalence of bubble lesions was found for beach-stranded marine mammals in this study (one of 41) and in a

study by Jepson et al. (2005) (10 of 2,376). The results of the Moore et al. (2009) analyses support the modeling of simulated dive profiles by Zimmer and Tyack (2007), which suggest an increase in risk of bubble formation caused by repetitive dives with short to medium surface durations, without exceeding the depth of alveolar collapse, which is estimated to be about 80 m (263 ft) for dolphins.

Despite the increase in research and literature, there remains scientific disagreement and/or lack of scientific data regarding the evidence for gas bubble formation as a causal mechanism between certain types of acoustic exposures and stranding events. These issues include: 1) received acoustic exposure conditions; 2) pathological interpretation; 3) acoustic exposure conditions required to directly induce physiological trauma; 4) behavioral reactions caused by sound exposure such as atypical dive patterns; and 5) the extent of postmortem artifacts (Southall et al., 2007).

The underlying reasoning for beaked whale strandings predicated by by gas bubble formation is that beaked whales potentially have strong avoidance responses to MFA sonars because they sound similar to their main predator, the killer whale (Cox et al., 2006; Southall et al., 2007; Zimmer and Tyack, 2007; Baird et al., 2008; Hooker et al., 2009). Since SURTASS LFA sonar transmissions are lower in frequency (<500 Hz) and dissimilar in characteristics from those of marine mammal predators, the above scientific studies do not provide additional evidence that SURTASS LFA sonar has caused behavioral reactions, specifically avoidance responses, in beaked whales. Thus, SURTASS LFA sonar transmissions are not expected to cause gas bubble formation or beaked whale strandings.

3.8.3 Auditory Effects on Marine Mammals

Marine species are affected by the presence of intense underwater sounds in their environment (Richardson et al., 1995; Salvi et al., 1986). Exposure to intense sounds may lead to auditory effects that are either temporary or permanent in duration. Auditory effects include permanent threshold shift (PTS), which is a severe hearing effect that occurs when sound intensity is very high or of such long duration that the result is a permanent and unrecoverable hearing loss on the part of the listener and constitutes an injury. Temporary threshold shift (TTS) is a lesser hearing effect caused by underwater sounds of sufficient loudness that cause a transient condition in which an animal's hearing is impaired for a period of time. Since hearing is not permanently or irrevocably damaged, TTS is not considered an injury (Richardson et al., 1995; Southall et al., 2007). Although during a period of TTS, animals may be at some disadvantage in terms of detecting predators or prey.

For the purposes of the SURTASS LFA sonar analyses presented here, all marine mammals exposed to LFA sonar at ≥180 dB SPL RL are evaluated as if they are injured (under the MMPA, Level A harassment), including PTS and TTS effects. Even though actual injury would not occur unless animals were exposed to sound at a level greater than this value (Southall et al., 2007), the analysis herein will continue to define the injury level associated with LFA sonar as ≥180 dB

SPL RL. This should be viewed as a conservative value, used to maintain consistency in the analytical methodologies utilized throughout the environmental compliance process for SURTASS LFA sonar. Thus relative to SURTASS LFA sonar, all marine mammals exposed to underwater sound ≥180 dB SPL RL and experiencing either PTS and TTS, are evaluated as if they are injured (Level A harassment under the MMPA). For military readiness activities, such as the employment of SURTASS LFA sonar, any act that disturbs or is likely to disturb a marine mammal by causing disruption of natural behavioral patterns to a point where the patterns are abandoned or significantly altered may affect but is not likely to adversely affect that species (under MMPA, Level B harassment).

A study of TTS in harbor porpoises used a seismic airgun as a stimulus (Lucke et al., 2009). Airguns produce an impulsive signal and have a broad frequency range but also have substantial energy in the low frequency region. A small airgun was used in proximity to the animals (between 14 to 150 m), a context that is likely to enhance behavioral responsiveness. The harbor porpoises showed a behavioral response at a RL of 174 dB re 1 μPa (peak-to-peak), which is equivalent to an SEL of 145 decibels relative to one micro Pascal squared per second (dB re 1 μPa^2-sec) (Lucke et al., 2009). Harbor porpoise hearing was tested at a frequency of 4 kHz and TTS was detected at a RL of 199.7 dB re 1 μPa (peak-to-peak), which is equivalent to an SEL of 164.3 dB re 1 μPa^2-sec (Lucke et al., 2009). These are the lowest received sound levels that produce TTS yet reported. These data are intriguing and clearly indicate a need for additional research. Unfortunately, only one individual was tested in this study. The applicability of these results to SURTASS LFA sonar is uncertain, given the large differences in source characteristics between airguns and LFA sonar. Furthermore, LFA sonar typically operates in water deeper and further offshore than most harbor porpoise habitats. Harbor porpoises do not occur in only of the SURTASS LFA sonar mission areas proposed for use of SURTASS LFA sonar during 2014 to 2015. Nevertheless, this study indicates that further study of TTS in porpoises is warranted. Ideally, additional harbor porpoise individuals as well as additional high-frequency hearing species would be tested. If this type of results are confirmed for harbor porpoise or found in other HF hearing species, then the analyses for those species would merit revision.

In a study on the effects of noise level and duration of TTS in a bottlenose dolphin, Mooney et al. (2009) exposed a bottlenose dolphin to octave-band noise (4 to 8 kHz) of varying durations (2 to 30 minutes) and SPL RLs (130 to 178 dB re 1 μPa). The results of the Mooney et al. study indicated that shorter-duration sound exposures often require greater sound energy to induce TTS than longer-duration exposures and also supported the trend that longer-duration exposures often induce greater amounts of TTS, which concurrently require longer recovery times.

In a controlled exposure experiment, Mooney et al. (2009a) demonstrated that MFA sonar could induce temporary hearing loss in a bottlenose dolphin (*Tursiops truncatus*). Temporary hearing loss was induced by repeated exposure to an SEL of 214 dB re 1 μPa^2-sec. Subtle behavioral alterations were also associated with the sonar exposures. At least with one odontocete species

(common bottlenose dolphin), sonar can induce both TTS and mild behavioral effects; but exposures must be prolonged with high exposure levels to generate these effects. The RL used in the Mooney et al. (2009a) experiment was an SPL of 203 dB, which equates to the RL approximately 40 m (131 ft) from an MFA sonar operated at an SPL of 235 dB (SL). Mooney et al. (2009a) concluded that in order to receive an SEL of near 214 dB, an animal would have to remain in proximity of the moving sonar, which is transmitting for 0.5 sec every 24 sec over an approximately 2 to 2.5 min period, an unlikely situation.

SELs necessary for TTS onset for pinnipeds in water have been measured for harbor seals, California sea lions, and northern elephant seals. As reported by Southall et al. (2007), Kastak et al. (2005) presented comparative analysis of underwater TTS for pinnipeds. This indicated that in harbor seals, a TTS of ~6 dB occurred with a 25-min exposure to 2.5 kHz octave-band noise of 152 dB SPL (183 dB SEL); a California sea lion showed TTS-onset under the same conditions at 174 dB SPL (206 dB SEL); and a northern elephant seal under the same conditions experienced TTS-onset at 172 dB SPL (204 dB SEL). Finneran et al. (2003) exposed two California sea lions to single underwater pulses from an arc-gap transducer and found no measurable TTS following exposures of up to 183 dB SPL (215 dB SEL).

Animals suffering from TTS over longer periods of time, such as hours to days, may be considered to have a change in a biologically significant behavior, as they may be prevented from detecting sounds that are biologically relevant, including communication sounds, sounds of prey, or sounds of predators. As noted by Mooney et al. (2009), shorter duration sound exposures can require greater sound energy to induce TTS than longer duration exposures, and longer duration exposures can induce greater amounts of TTS. In assessing the potential for LFA sonar transmissions to cause TTS, the much shorter length of the LFA signal (1 min) versus the above studies (2 to 30 min) must be considered.

These scientific conclusions supports the assumptions and findings of the SURTASS LFA documents (DoN, 2001, 2007, 2012) that the likelihood that SURTASS LFA sonar may cause TTS at a SPL of 180 dB RL in marine mammals is negligible and very few animals are likely at all to be affected by TTS. Due to the length of the LFA sonar signal (~60 sec), the threshold used for marine mammal injury is the conservative SPL of 180 dB RL, which is lower than the estimated thresholds calculated by Southall et al. (2007) (and adjusted for the length of the LFA signal) of a sound exposure level (SEL) of 197 decibels relative to 1 microPascal squared per second (dB re 1 μPa^2-sec) RL for all cetaceans and an SEL of 185 dB RL for pinnipeds in water. Further, mitigation measures, such as defined mitigation zones and sonar shutdown protocols (NOAA, 2007 and 2012), are employed such that there is little potential for a marine mammal to incur TTS or PTS. Thus, the probability of SURTASS LFA sonar transmissions with mitigation measures applied, including LFA sonar shutdown, causing TTS or PTS in marine mammals is considered negligible.

3.8.4 Behavioral Change in Marine Mammals

Intense sound can lead to disruption of natural behavioral patterns that can have biologically significant effects. The National Research Council (NRC, 2005) discussed biologically significant behaviors and possible effects and stated that an action or activity becomes biologically significant to an individual animal when it affects the ability of the animal to grow, survive, and reproduce. These are the effects on individuals that can have population-level consequences and affect the viability of the species (NRC, 2005). For military readiness activities, such as the use of SURTASS LFA sonar, Level B "harassment" under the MMPA is defined as any act that disturbs or is likely to disturb a marine mammal by causing disruption of natural behavioral patterns to a point where the patterns are abandoned or significantly altered. Behaviors include migration, surfacing, nursing, breeding, feeding, and sheltering.

The results of the Low Frequency Sound Scientific Research Program (LFS SRP) in 1997 to 1998 confirmed that some portion of the total number of whales exposed to LFA sonar responded behaviorally by changing their vocal activity, moving away from the source vessel, or both; but the responses were short-lived (Clark et al., 2001). In the LFS SRP LFA sonar playback experiment (Phase II), migrating gray whales avoided exposure to LFA signals (source levels of 170 and 178 dB SPL) when the source was placed in the center of their migration corridor. Responses were similar for the 170-dB SL LFA stimuli and for the 170-dB SL 1/3rd-octave, band-limited noise with timing and frequency band similar to the LFA stimulus. However, during the LFA sonar playback experiments, in all cases, whales resumed their normal activities within tens of minutes after the initial exposure to the LFA signal (Clark et al., 2001). Essentially, the whales made minor course changes to go around the source. When the source was relocated within the outer portion of the migration corridor (twice the distance offshore), and the SL was increased to reproduce the same sound field for the central corridor playback condition, the gray whales showed little to no response to the LFA sonar source. This result stresses the importance of context in interpreting the animals' behavioral responses to underwater sounds and demonstrates that RL is not necessarily a good predictor of behavioral impact. The LFS SRP also conducted field tests to examine the effects of LFA sonar transmissions on foraging fin and blue whales off San Nicolas Island, California (Phase I). Overall, whale encounter rates and dive behavior appeared to be more strongly linked to changes in prey abundance associated with oceanographic parameters rather than LFA sound transmissions (Croll et al., 2001).

In the final phase of the LFS SRP (Phase III), the effect of LFA sonar on humpback whales during the winter mating season was investigated. Both Miller et al. (2000) and Fristrup et al. (2003) published results from tests conducted with male humpback singers off Hawaii during which they evaluated variation in song length as a function of exposure to LFA sounds. In spite of methodological and sample size differences, the results of the Miller et al. (2000) and Fristrup et al. (2003) analyses were generally in agreement, and both studies indicated that humpback whales might lengthen their songs in response to LF broadcasts. The Fristrup et al. (2003) results

also provided a detailed picture of short-term response as compared to behavioral variation observed in the absence of the stimuli. These responses were relatively brief in duration, with all observed effects occurring within 2 hrs of the last LFA source transmission. It should be noted that these effects were not obvious to the acoustic observers on the scene, but were revealed by careful, complex post-test statistical analyses (Fristrup et al., 2003). Aside from the delayed responses, other measures failed to indicate cumulative effects from LFA broadcasts, with song-length response being dependent solely on the most recent LFA transmission, and not the immediate transmission history. The modeled seasonal factors (changes in density of whales sighted near shore) and diurnal factors (changes in surface social activities) did not show trends that could be plausibly explained by cumulative exposure. Increases in song length from early morning to afternoon were the same on days with and without LFA transmissions, and the fraction of variation in song length that could be attributed to LFA broadcast was small (<10%). Fristrup et al. (2003) found high levels of natural variability in humpback song length and interpreted the whales' responses to LFA broadcasts to indicate that exposure to LFA sonar would not impose a risk of dramatic changes in humpback whale singing behavior that would have demographic consequences.

Southall et al. (2007) reviewed the relatively extensive behavioral observations of low frequency cetaceans exposed to non-pulse sources. While there are clearly major areas of uncertainty, Southall et al. concluded that the literature indicated that there were no (or very limited) responses to RLs of 90 dB to 120 dB SPL with an increasing probability of avoidance and other behavioral effects in the 120 to 160 dB SPL (RL) range.

A recent study by Risch et al. (2012) documented reduction in humpback whale vocalization concurrent with transmissions of the Ocean Acoustic Waveguide Remote Sensing (OAWRS) system, at distances of 200 km from the source. OAWRS is a low-frequency sensor designed to monitor fish populations and marine life over shelf-scale areas. The recorded OAWRS signal consisted of three one-second long frequency modulated upsweeps, with 0.5 seconds between each pulse in the signal. The pulses had a bandwidth of about 50 Hz and were centered on 415, 735, and 950 Hz. The interval between pulse trains was apparently variable. Signal received levels in the study region ranged from 88 to 110 dB RL (Risch et al., 2012). The OAWRS source appears to have affected more whales than the Phase III of the LFS SRP, even though OAWRS had a lower received level than the LFA signal. This strongly suggests that other acoustic characteristics may be responsible for the difference in observed behavioral responses. For example, Risch et al. (2012) note that (1) the duration and frequency range of the OAWRS signals are similar to those of natural humpback whale song components, (2) the duty cycle of the OAWRS signal was, at times, much greater and more variable that that of the LFS SRP. It may be that the greater predictability of the LFA signal mitigated the response, just as bowhead whales had a lesser response to predictable vessels (Richardson et al., 1985); (3) the bandwidth of the OAWRS signal was much greater than that of the LFS SRP. In summary, the OAWRS experiment appeared to produce a greater response with a lower sound source level. However

those experimental signals, in duration and frequency, were more similar to biological sound, more varied in their production rate, and greater in bandwidth. Risch et al. (2012) stated that due to differences in behavioral context, location, and proximity to the source, it is difficult to compare their findings directly to the Phase III of the LFS SRP. These observations are consistent with the importance of considering context in predicting and observing the level and type of behavioral response to anthropogenic signals (Ellison et al., 2012).

Studies similar to those of the LFS SRP have also researched the potential effects of mid-frequency sonar on marine mammal species (Tyack et al., 2011; Southall et al., 2012). Two Cuvier's beaked whale were shown to demonstrate behavioral responses to a 30-min playback (one 1.6-sec simulated MFA sonar signal repeated every 25 sec) at received levels of 89-127 dB (DeRuiter et al., 2013). Distant sonar signals with received levels of exercises 78-106 dB did not elicit similar responses, suggesting that reactions may be context dependent (Ellison et al., 2012). Context was shown to be a significant factor in determining whether blue whales responded to mid-frequency sonar (Goldbogen et al., 2013). Animals were classified as deep-diving foragers, shallow-diving foragers, or non-foraging, with shallow-diving foragers showing no response to controlled exposure experimental conditions. (Goldbogen et al., 2013) suggest that a combination of behavioral state and received sound level may influence behavioral response. It is clear that more research is needed to understand the complex interactions that may result in behavioral responses.

3.8.5 Masking in Marine Mammals

The obscuring of sounds of interest by interfering sounds, generally at similar frequencies, is referred to as masking (Fletcher, 1929; Richardson et al., 1995). In humans, masking has been measured as an increase in detection threshold of the sound of interest in the presence of a masking sound (compared to the detection threshold when there is no masker). Two types of masking have been described: energetic masking and informational masking (Pollack, 1975, Watson, 2005, Kidd et al., 2007). The definitions of energetic and informational masking and their physiological mechanisms, however, continue to be debated. Energetic masking is thought to result from an interfering sound(s) within the same critical band(s) as the signal of interest. It is usually ascribed to peripheral acoustic processing; i.e., the ear itself. A definition for informational masking has been even less forthcoming, and as a default position, informational masking has often been taken to mean masking that is greater than would be predicted by energetic masking alone (Kidd et al., 2007). Informational masking is associated with uncertainty of the signal of interest (Watson, 2005) and is generally assumed to occur as a result of central neural processing that includes analytic (e.g., auditory stream segregation and discrimination) and attentive components (e.g., distraction) (Kidd et al., 2007). As a general statement, the more similar the characteristics (i.e., frequency band, duration) of a masking sound are to the sound of interest, the greater the potential for masking.

Cerchio et al. (2014) used passive acoustic monitoring to document the presence of singing humpback whales off the coast of Northern Angola, and opportunistically test for the effect of seismic survey activity in the vicinity on the number of singing whales. The authors observed that humpback whale males increasingly stopped vocal displays on Angolan breeding grounds as received seismic airgun levels increased. The authors observed that the whales ceased to sing or moved to other areas to sing when seismic surveys occurred in relatively close proximity to the breeding grounds.

Acoustic masking from low frequency ocean noise is increasingly being considered as a threat, especially to low frequency hearing specialists such as baleen whales (Clark et al., 2009). Most underwater low frequency anthropogenic noise is generated by commercial shipping, which has contributed to the increase in oceanic background noise over the past 150 years (Parks et al., 2007). Shipping noise is primarily in the 20 to 200 Hz frequency band and is increasing yearly (Ross, 2005). Andrew et al. (2002) demonstrated an increase in oceanic ambient noise of 10 dB SPL since 1963 in the 20 to 80 Hz frequency band as sampled on the continental slope off Point Sur, California, and they ascribed this increase to increased commercial shipping. McDonald et al. (2006) compared data sets from 1964 to 1966 and 2003 to 2004 for continuous measurements west of San Nicolas Island, California, and found an increase in ambient noise levels of 10 to 12 dB SPL in the 30 to 50 Hz band. This increase in LF background noise is likely having a widespread impact on marine mammal low frequency hearing specialists by reducing their access to acoustic information essential for conspecific communication and other biologically important activities, such as navigation and prey/predator detection. Clark et al. (2009) considered this long-term, large-scale increase in low frequency background noise a chronic impact that results in a reduction in communication space, and the loss of acoustic habitat.

Parks et al. (2007, 2010) provided evidence of behavioral changes in the acoustic behaviors of the endangered North Atlantic right whale, and the South Atlantic right whale, and suggested that these were correlated to increased underwater noise levels. The studies indicated that right whales might shift the frequency band of their calls to compensate for increased in-band background noise (Parks et al., 2007) and increase the amplitude of their calls with increased background noise (Parks et al., 2010). The significance of their result is the indication of potential species-wide behavioral change in response to gradual, chronic increases in underwater ambient noise. Di Iorio and Clark (2010) showed that blue whale calling rates vary in association with seismic sparker survey activity, with whales calling more on days with survey than on days without surveys. They suggested that the whales called more during seismic survey periods as a way to compensate for the elevated noise conditions.

Changes in behavior are not limited to low frequency species. Holt et al. (2009) measured killer whale call source levels and background noise levels in the 1 to 40 kHz band. The whales increased their call source levels by 1 dB for every 1 dB increase in background noise level. A

similar rate of increase in vocalization activity was reported for St. Lawrence River belugas in response to passing vessels (Scheifele et al., 2005).

3.8.5.1 SURTASS LFA Sonar Potential for Masking

Masking effects from SURTASS LFA sonar signals will be limited for a number of reasons. First, the bandwidth of any LFA sonar transmitted signal is limited (30 Hz), and the instantaneous bandwidth at any given time of the signal is small, on the order of ≤10 Hz. Therefore, within the frequency range in which masking is possible, the effect will be limited because animals that use this frequency range typically use signals with greater bandwidths. Thus, only a portion of frequency band for the animal's signal is likely to be masked by the LFA sonar transmissions. Furthermore, when LFA is in operation, the LFA source is active only 7.5 to 10% of the time (based on historical LFA operational parameters), which means that for 90 to 92.5% of the time there is no risk that an animal's signal will be masked by LFA sonar. Therefore, within the area in which energetic masking is possible, any effect of LFA sonar transmissions will be minimal because of the limited bandwidth and intermittent nature of the signal, and the fact that animals that use this frequency region typically produce signals with greater bandwidth that are repeated for many hours.

Hildebrand (2005) provided a comparison of anthropogenic underwater sound sources by their annual energy output. On an annual basis, four LFA sonar systems were estimated to have a total energy output of 6.8×10^{11} Joules/yr. Seismic airgun arrays and mid-frequency military sonars were two orders of magnitude greater, with an estimated annual output of 3.9 and 2.6×10^{13} Joules/year, respectively. Super tankers were greater at 3.7×10^{12} Joules/yr. Hildebrand (2005) concluded that anthropogenic sources most likely to contribute to increased underwater noise in order of importance are: commercial shipping, offshore oil and gas exploration and drilling, and naval and other uses of sonar. The use of LFA sonar is not scheduled to increase beyond the originally analyzed four systems during the remaining period of the five-year regulation under the MMPA and the reasonably foreseeable future. The percentage of the total anthropogenic acoustic energy budget added by each LFA source is estimated to be 0.21% per system (or less), when other man-made sources are considered (Hildebrand, 2005). When combined with the naturally occurring and other man-made sources of noise in the oceans, the intermittent LFA signals barely contribute a measurable portion of the total acoustic energy.

The recent research provide additional support to the conclusion that broadband LF shipping noise is likely to be more detrimental to marine mammals than low duty-cycle SURTASS LFA sonar (Andrew et al., 2002; McDonald et al., 2006; Parks et al., 2007; Clark et al., 2009). Therefore, any masking in marine mammals due to narrowband, intermittent (low duty cycle) LFA sonar signal transmissions are expected to be minimal and unlikely.

3.8.6 Stranding

Strandings of both sea turtles and marine mammals occur globally and year-round. The vast majority of sea turtle strandings, however, occur typically when sea turtles are reproductively

active in the vicinity of nesting or rookery sites or when turtles are exposed to a sudden drop in ambient water temperature at the onset of winter, leading to "cold-stunning" (thermal shock) and death (Spotila, 2004). While stranding of sea turtles can be caused by anthropogenic activities, especially from gear entanglement due to fishing, ingestion of plastics or oil due to pollution, or boat/ship strikes, little is known of strandings associated with exposure to anthropogenic sound. However, searches of online databases have shown no sea turtle stranding or mortality events in any of the mission areas in which SURTASS LFA sonar has operated over the last annual period.

Marine mammals strand for a variety of reasons, including both natural and anthropogenic factors. Strandings of individual marine mammals occur routinely around the world, but multiple marine mammals stranding at the same time and place, or mass strandings, occur only rarely globally. Mass strandings typically involve pelagic odontocete marine mammal species that occur infrequently in coastal waters and are usually typified by highly developed social bonds. Marine mammal strandings and mortality events are natural events that have been recorded historically from as early as 350 B.C.

The use of SURTASS LFA sonar was not associated with any of the reported 27 mass stranding events that occurred globally between 2006 through 2012, nor does any evidence exist to indicate that LFA sonar transmissions resulted in any difference in the stranding rates of marine mammals in Japanese coastal waters adjacent to LFA sonar operating areas (DoN, 2012). The use of SURTASS LFA sonar was not associated with any of the 11 known mass strandings that occurred from 2013 through the present, as only two strandings, and one of those of a single marine mammal even occurred over the last year in the potential mission areas for SURTASS LFA; in January 2013, one young humpback whale stranded in Hawaii due to a fishing gear entanglement, and in March 2014, 24 *Kogia* spp. stranded in Surigao del Norte, Philippines after dynamite fishing occurred in the area, with at least two of the animals dying.

As has been reported previously (DoN, 2001, 2007, 2012), the employment of SURTASS LFA sonar is not expected to result in any sonar-induced strandings of sea turtles or marine mammals. Given the large number of natural and other anthropogenic factors that can result in marine mammal and sea turtle strandings, the high occurrence of marine mammal strandings, and the more than 12 years of LFA sonar operations without any associated stranding events, the likelihood of SURTASS LFA sonar transmissions causing marine mammals or sea turtles to strand is negligible. In summary, from the commencement of SURTASS LFA sonar use from 2002 through the present, neither LFA sonar nor operation of T-AGOS vessels has been associated with any mass or individual strandings of marine animals.

3.9 The Navy's Approach to Assessing Behavioral Response of Marine Mammals

To analyze the possibility for Level B / behavioral harrassment effects of simultaneous, or near-simultaneous, mid-frequency active (MFA) and LFA sonar transmissions occurring, the Navy used two separate methodologies, a parametric analysis and an Acoustic Integration Model© (AIM) analysis, which use the previously established risk continuum for SURTASS LFA sonar.

The risk continuum methodology for SURTASS LFA sonar was applied to facilitate a complex analytic process with two dissimilar sonar systems.

The risk continuum for SURTASS LFA sonar was initially developed for determining the risk from SURTASS LFA sonar. An exposure of 165 dB SPL (re 1 µPa) returns an associated risk of 0.5 (50%) from the risk continuum function; whereas 150 and 180 dB SPL (re 1 µPa) return 0.025 (2.5%) and 0.95 (95%) risk, respectively (Figure 7).

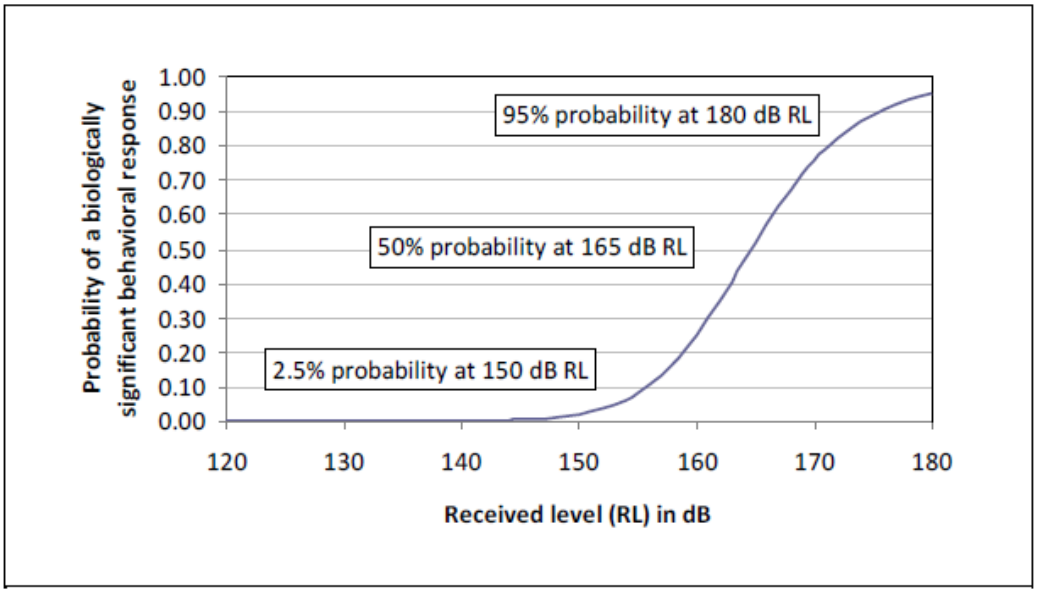

Figure 7. SURTASS LFA Sonar Risk Continuum Function

3.9.1 Parametric Analysis

Parametric analysis is a methodology to describe and examine the relationship between different parameters (e.g., in this case acoustic transmission loss as a function of range and depth) and the variable (e.g., potential acoustic effect on marine mammals) that it/they influence or affect. Parametric analysis is derived from "dimensional analysis," which is defined as:

"...the mathematics of dimensions and quantities and provides procedural techniques whereby the variables that are assumed to be significant in a problem can be formed into dimensionless parameters, the number of parameters being less than the number of variables." (Avallone and Baumeister, 1987)

The advantage of this type of analysis is the reduction of a large number of variables into a smaller, more manageable, number of parameters. This kind of analysis has been in use for over 100 years and is well accepted in the scientific community. One example is the use of a properly scaled ship model to identify the force needed to propel the actual full-size ship through the water, including the size of the engines needed to do so. One of the key inputs is the ratio of

inertia and viscous forces using the "Reynolds Number," a key dimensionless number used in naval architecture, aeronautics, and anywhere fluid flow is important.

This analysis identified appropriate metrics for each of the important parameters (e.g., difference in source level [SL], distance between sources, different propagation conditions, Level B harassment criteria, etc.). Then, using such metrics, the risk for multiple animal depths and a variety of sonar separation ranges in static conditions (i.e., a series of "snapshots" of single ping risk for each source, and for the combined sources, with the source vessels in specific locations, was examined. The analysis assumed a convergence zone (CZ) propagation condition (where sound waves in the ocean refract downward and then rise back to the surface at regular intervals known as convergence zones) because it is the most probable sound propagation path that would be encountered with concurrent SURTASS LFA and MFA sonar operations. Details of this analysis are provided in the Final SEIS/SOEIS for SURTASS LFA Sonar (June 2012), along with discussions of other propagation conditions (i.e., bottom bounce, surface duct).

3.9.2 Use of The Acoustic Integration Model (AIM) to Estimate Marine Mammal Exposures to Low-frequency Active Sonar

The model analysis presented here is an attempt to create a simulation of a representative concurrent operation with one LFA sonar and one MFA sonar. Actual waypoints representing plausible ship courses are input into the model, as are the source characteristics of each vessel (Table 13). Each modeled vessel produces a sonar ping according to the programmed sonar plan for the vessel. A population of representative marine mammals is placed in the simulation around the vessels. These simulated animals, referred to as "animats," are programmed to move in four dimensions, with movement parameters derived from actual animals. The acoustic propagation from the ships to each animat is modeled with the Ocean and Atmospheric Master Library (OAML)-approved Parabolic Equation (PE) model (Zingareli et al., 1999) for the LFA ship and the ray-based BELLHOP[6] model (Porter, 1992) for the MFA source (because BELLHOP is better suited to the acoustic parameters of MF sources than PE). The received level (RL) at each animat can therefore be predicted. These predicted RLs are then analyzed using the standard methods as described in Subchapter 4.4.1 of the SURTASS LFA Sonar Final SEIS. One additional calculation is needed to sum MFA and LFA transmissions that arrive simultaneously, which is discussed below.

[6] BELLHOP computes underwater acoustic transmission paths via beam (ray) tracing. Ray tracing is a method for calculating the path of sonar beams through water with regions of varying propagation conditions, absorption characteristics, and reflecting surfaces. Under these circumstances, sonar beam may bend, change direction, or reflect off the water surface or seafloor, complicating analysis. Ray tracing solves the problem by repeatedly advancing idealized narrow beams called rays through the water by discrete amounts. Simple problems can be analyzed by propagating a few rays using simple mathematics. More detailed analyses can be performed by using a computer to propagate many rays.

Table 13. Source Characteristics used for AIM modeling

SOURCE	LOW-FREQUENCY ACTIVE SONAR	MID-FREQUENCY ACTIVE SONAR
Source Level	Typical Operational	Typical Operational
Frequency	250 Hz	3500 Hz
Duty Cycle	60 second transmission every ten minutes	1 second transmission every 30 seconds
Beam Pattern	Normal LFA beam pattern	Normal omni-directional transmission, vertically beamformed

The concern for underwater acoustic impacts to marine mammals has been growing since the 1990s. Because of the complexity of underwater acoustic propagation, acoustic exposure of marine animals is a function of the animal's depth as much as its range from the source. Therefore, the accurate prediction of acoustic exposure of free-ranging animals requires the consideration of animal movement as well as physical environmental conditions. The Acoustic Integration Model (AIM) was developed to address this requirement. The AIM also seeks to address: 1) changing and variable acoustic thresholds; 2) the scarcity of data on marine mammal densities, distribution and their behavioral responses to underwater sound; 3) constantly improving and expanding environmental data bases and propagation model capabilities; and 4) the requirement from both federal regulators and the public to use the best available science for any impact analysis process. AIM was first applied to the U.S. Navy's SURTASS LFA Sonar EIS/OEIS (DoN, 2001), which was the first EIS prepared for a Navy operational system. Since then it has been used for other acoustic sources, including seismic profilers, underwater explosives, over-water sonic booms, and numerous active sonar applications. Today it is an open architecture coalition of candidate models and databases. The component of AIM that remains actively involved in all AIM executions is the animat movement engine, which creates the sound sources and animats of interest, moves them in 3D in the ocean volume, and facilitates tracking the estimated sound exposure on each modeled marine mammal.

Because the exact underwater positions of sources and receivers cannot be known, multiple runs of realistic predictions are used to provide statistical validity. The movement and/or behavioral patterns of sources and receivers can be known, and these data are incorporated into the model. Accurate representation of the movements of sources and receivers is necessary for realistic predictions. Each source and/or receiver is modeled via the animat concept. Each animat has parameters that control its speed and direction in three dimensions. Thus, it is possible to recreate the type of diving pattern that an animal shows in the real world. Furthermore, the movement of the animat can be programmed to respond to environmental factors, such as water depth and sound level. In this way, species that normally inhabit specific environments can be constrained in the model to stay within that habitat.

Once the behavior of the animats has been programmed, the model is run. AIM proceeds forward in time, with all features following the same master clock; the source produces a sound, the transmitted sound level at all ranges and depths is calculated using the propagation loss model, the range and depth of each animat at that time is noted and the respective RL for that animat is noted and retained with that animat's record. Then each animat and the source move ahead in time to the next source transmission, and the process is repeated. This continues until all source transmissions have been completed. After all the programmed runs are complete, each animat's full record of exposure levels is analyzed and a risk assessment is assigned, both to individual animals, as well as the resident population.

3.9.2.1 Model Scenarios

The ship movement scenarios selected were designed to address both intentional (clearing) and incidental (closing or parallel courses) interactions between LFA and MFA vessels. The clearing exercise scenario was designed as a possible sweep of an MFA ship around an LFA ship to detect any nearby submarines. The parallel course scenario is set up so that the LFA and MFA source ships start at approximately two convergence zones apart, with animals between them. The overtaking scenario starts with the MFA source ship approximately two convergence zones behind the LFA source ship, and then overtaking the LFA ship because of its greater speed. This scenario places the source ships much closer than would ever occur in actual LFA/MFA concurrent operations, but attempts to place an upper bound on potential risk.

To estimate the acoustic exposure that an animal is likely to receive while the sources are transmitting, the movement of animals and the acoustic fields to which they would be exposed are modeled. The sound fields around each source are estimated based on details of the proposed acoustic sources using the Navy's standard PE model 5.0 for low-frequency sources (SURTASS LFA sonar) to a range of 150 km (81 nmi), and BELLHOP for mid-frequency sources (AN/SQS-53C) to a range of 100 km (54 nmi). AIM is used to simulate the acoustic exposure for each marine mammal species from the nominal transmissions of the MFA and LFA acoustic sources. Analyses were performed using generic animal species behavior, and each model run involved two 5-hour simulations (one for LFA and one for MFA), with animal 3D movement replicated.

To estimate the risk of harassment from each acoustic source, the individual acoustic exposures an animal receives were converted to single ping equivalent (SPE), using established SURTASS LFA sonar procedures (i.e., 5LogN, where N = number of exposures). This SPE is input into the SURTASS LFA sonar risk continuum to estimate Level B/ behavioral harassment. The SPE RLs are then evaluated for each source three ways: 1) separately; 2) additive (i.e., the two separate values added together); and 3) combined by summing the pressure of the two waveforms, a procedure that accounts for difference in frequency between the two transmissions.

Three nominal operational scenarios were analyzed:

- A "clearing" exercise scenario, analyzed for both convergence zone and surface duct underwater sound propagation; A "clearing" scenario consists of an MFA vessel "clearing" all sectors around the LFA vessel, starting in the rear port quadrant, moving forward, then starboard, then aft, to check all quadrants for possible submarines;
- A "parallel courses" exercise scenario, with the LFA and MFA vessels two convergence zones apart, and the animals between the vessels, analyzed for both convergence zone and surface duct underwater sound propagation; and
- An "overtaking" exercise scenario, where the MFA vessel starts two convergence zones behind the LFA vessel, and by its greater speed, overtakes and passes the LFA vessel, analyzed for both convergence zone and surface duct underwater sound propagation.

3.9.2.2 AIM Input Parameters

In the Navy's analysis, an approach similar to that used for estimating the potential environmental effects from real-world operations of SURTASS LFA sonar was used. Courses and speeds for both LFA and MFA vessels, and the LFA and MFA sonar acoustic characteristics were input into AIM. Each of three potential operational scenarios was then populated with marine mammals around the LFA and MFA vessels. Model input parameters include:

- Animat species was a generic baleen whale, based on blue and fin whale movement parameters.
- Animat density = 0.1 animats/sq km (this is the model density)
- Animal density = 0.001 animals/sq km (this is predicted density of real animals)
- MFA ship speed 18.5 km/hr (10 kt)
- LFA ship speed 6 km/hr (3.2 kt)
- Feller risk continuum curve parameters:
- Basement (B) = 120 dB (same as baseline LFA case)
- Transition Point (K) = 45 dB (same as baseline LFA case)
- Slope Parameter (A) = 10 (as in the single LFA, single MFA, or combined effects LFA and MFA analysis).

4 STATUS OF LISTED RESOURCES

This opinion examines the status of each ESA-listed species under NMFS jurisdiction that potentially would be affected by the proposed action (Table 14). Our next step is to determine if any of these species and their designated critical habitat are likely to be adversely affected by the proposed action.

For species that are likely to be adversely affected, we describe the level of risk that the listed species face, based on parameters considered in documents such as recovery plans, status reviews, and listing decisions. The species status section helps to inform the species' current "reproduction, numbers, or distribution" as described in 50 CFR § 402.02. The opinion also examines the condition of critical habitat throughout the action area, evaluates the conservation

value of the various watersheds and coastal and marine environments that make up the action area, and discusses the current function of the essential physical and biological features that provides the conservation value of the designated habitat.

Table 14. Listed Resouces under NMFS' jurisdiction that may occur in the Action Area for this Consultation

Species	ESA Status (E = Endangered)	Critical Habitat	Recovery Plan
Marine Mammals – Cetaceans			
Blue Whale (*Balaenoptera musculus*)	E - 35 FR 18319	-- --	07/1998
Fin Whale (*Balaenoptera physalus*)	E - 35 FR 18319	-- --	75 FR 47538
Western North Pacific Gray Whale (*Eschrichtius robustus*)	E - 35 FR 18319	-- --	-- --
Humpback Whale (*Megaptera novaeangliae*)	E - 35 FR 18319	-- --	55 FR 29646
North Pacific Right Whale (*Eubalaena japonica*)	E - 73 FR 12024	73 FR 19000	78 FR 34347
Sei Whale (*Balaenoptera borealis*)	E - 35 FR 18319	-- --	12/2011
Sperm Whale (*Physeter macrocephalus*)	E - 35 FR 18619	-- --	75 FR 81584
Main Hawaiian Islands Insular False Killer Whale (*Pseudorca crassidens*)[1]	E - 77 FR 70915	--.--	-- --
Marine Mammals – Pinnipeds			
Hawaiian Monk Seal (*Monachus schauinslandi*)	E - 41 FR 51611	53 FR 18988/ 76 FR 32026[2]	72 FR 46966
Fish			
Scalloped Hammerhead Shark – Indo-West Pacific DPS	T - 78 FR 20718	-- --	-- --
Sea Turtles			
Green Sea Turtle (*Chelonia mydas*)	E - 43 FR 32800	-- --	63 FR 28359
Hawksbill Sea Turtle (*Eretmochelys imbricate*)	E - 35 FR 8491	-- --	63 FR 28359
North Pacific Ocean Loggerhead Sea Turtle (*Caretta caretta*)[1]	E - 76 FR 58868	-- --	63 FR 28359
Olive Ridley Sea Turtle (*Lepidochelys olivacea*)	E - 61 FR 17	-- --	63 FR 28359
Leatherback Sea Turtle (*Dermochelys coriacea*)	E - 61 FR 17	-- --	63 FR 28359

[1] *Distinct Population Segment (DPS)*

[2] *A revision to Hawaiian Monk Seal Critical Habitat was proposed on June 2, 2011.*

4.1 Listed Resources Not Considered Further

We used two criteria to identify those endangered or threatened species or critical habitat that are not likely to be adversely affected by the proposed activities and therefore will not be considered further in this opinion. As we do for all ESA consultation we have applied these criteria in each of the previous programmatic and annual consultations on SURTASS LFA sonar activities. The first criterion is exposure or some reasonable expectation of a co-occurrence between one or more potential stressors associated with the Navy's activities and a particular listed species or designated critical habitat. If we conclude that a listed species or designated critical habitat is not

likely to be exposed to the proposed activities, we must also conclude that the critical habitat is not likely to be adversely affected by those activities. The second criterion is the probability of a response given exposure, which considers susceptibility. Species that may be exposed to sound transmissions from active sonar, for example, but are likely to be unaffected by the sonar (at sound pressure levels they are likely to be exposed to) are also not likely to be adversely affected by the sonar.

4.1.1 Scalloped Hammerhead Shark - Indo-West Pacific DPS

On July, 3, 2014 NMFS issued the final determination to list the Central and Southwest (SW) Atlantic Distinct Population Segment (DPS) and the Indo-West Pacific DPS of scalloped hammerhead shark (*Sphyrna lewini*) as threatened species under the Endangered Species Act (ESA). NMFS also issued a final determination to list the Eastern Atlantic DPS and Eastern Pacific DPS of scalloped hammerhead sharks as endangered species under the ESA. NMFS will consider critical habitat for the Central & SW Atlantic, Indo-West Pacific, and Eastern Pacific DPSs in a separate rulemaking.

The Proposed Rule to list the Central SW Atlantic DPS, Eastern Atlantic DPS, Indo-West Pacific DPS, and the Eastern Pacific DPS (78 FR 20718) and the Status Review Report (Miller *et al.*, 2014) provide detailed discussion of the status and threats to each DPS. As described in the Proposed Rule, the primary factors responsible for the decline of these four DPSs are overutilization, due to both catch and bycatch of these sharks in fisheries, and inadequate regulatory mechanisms for protecting these sharks, with illegal fishing identified as a significant problem.

For the Indo-West Pacific DPS, we identified the inadequacy of current regulatory mechanisms as a moderate risk, with illegal fishing significantly contributing to the DPS' risk of extinction. Multiple Regional Fishery Management Organizations (RFMO) cover the Indo-West Pacific DPS area with requirements of full utilization of any retained catches of sharks and regulations that onboard fins cannot weigh more than 5 percent of the weight of the sharks. These regulations are aimed at curbing the practice of shark finning, but do not prohibit the fishing of sharks. In addition, these regulations may not even be effective in stopping finning of scalloped hammerhead sharks, as a recent study found the scalloped hammerhead shark to have an average wet-fin-to-round-mass ratio of only 2.13 percent (n=81; Biery and Pauly, 2012). This ratio suggests that fishing vessels operating in these RFMO convention areas would be able to land more scalloped hammerhead shark fins than bodies and still pass inspection. There are no scalloped hammerhead-specific RFMO management measures in place for this region, even though this DPS is heavily fished. Consequently, this species has seen population declines off the coasts of South Africa and Australia, so much so that in 2012, New South Wales, Australia, listed it as an endangered species.

Few countries within this DPS' range have regulations aimed at controlling the exploitation of shark species. Oman, Seychelles, Australia, South Africa, Taiwan, and most recently India all

have measures to prevent the waste of shark parts and discourage finning. The Maldives have designated their waters as a shark sanctuary. A number of Pacific Island countries (including U.S. territories) have also created shark sanctuaries, prohibited shark fishing, or have strong management measures to control the exploitation of sharks in their respective waters, including Tokelau, Palau, Marshall Islands, American Samoa, CNMI, Cook Islands, and French Polynesia, although effective enforcement of these regulations is an issue for some of the countries. Additionally, many of the top shark fishing nations and world's exporters of fins are also located within the range of this DPS, and have little to no regulation (or enforcement) of their expansive shark fisheries. For example, off northern Madagascar, where there is an active artisanal fin fishery, sharks are an open access resource, with no restrictions on gear, established quotas, or fishing area closures (Robinson and Sauer, 2011). Indonesia, which is the top shark fishing nation in the world, does not currently have restrictions pertaining to shark fishing or finning. Indonesian small-scale fisheries, which account for around 90 percent of the total fisheries production, are not required to have fishing permits (Varkey *et al.,* 2010), nor are their vessels likely to have insulated fish holds or refrigeration units (Tull, 2009), increasing the incentive for shark finning by this sector (Lack and Sant, 2012). Ultimately, their fishing activities remain largely unreported (Varkey *et al.,* 2010), which suggests that the estimates of Indonesian shark catches are greatly underestimated. In fact, in Raja Ampat, an archipelago in Eastern Indonesia, Varkey *et al.* (2010) estimated that 44 percent of the total shark catch in 2006 was unreported (including small-scale and commercial fisheries' unreported catch and illegal, unregulated, and unreported (IUU) fishing). Although Indonesia adopted an FAO recommended shark conservation plan (National Plan of Action—Shark) in 2010, due to budget constraints, it can only focus its implementation of key conservation actions in one area, East Lombok (Satria *et al.,* 2011). Due to this historical and current absence of shark management measures, especially in the small-scale fisheries sector, many of the larger shark species in Indonesian waters have already been severely overfished (Field *et al.,* 2009).

In addition to the largely unregulated fishing of this DPS, illegal fishing, especially for shark fins, has been identified as a significant contributor to the extinction risk of this DPS. Scalloped hammerhead sharks are valued for their large fins, which fetch a high commercial value in the Asian shark fin trade (Abercrombie et al.,2005) and comprise the second most traded fin category in the Hong Kong market (Clarke et al., 2006). Due to this profit incentive, there have been many reports of finning and seizures of illegally gained shark fins throughout the range of this DPS, including inwaters of Australia (Field et al., 2009), Mozambique, South Africa, Bay of Bengal, Arabian Gulf, Palau, the Federated States of Micronesia (FSM) (Paul, 2009), and Somalia (HSTF, 2006). Agnew et al. (2009) provided regional estimates of illegal fishing (using FAO fishing areas as regions) and found the Western Central Pacific (Area 71) and Eastern Indian Ocean (Area 57) regions to have relatively high levels of illegal fishing (compared to the rest of the regions), with illegal and unreported catch constituting 34 and 32 percent of the region's catch, respectively.

Although the number of shark management and conservation measures for this DPS is on the rise, the ERA team noted that the current protections that they afford the Indo-West Pacific DPS may be minimal if illegal fishing is not controlled. We agree and conclude that the inadequacy of current regulatory mechanisms, in the form of ineffective enforcement of current regulations or lack of existing regulatory measures, in combination with illegal fishing, is contributing significantly to the risk of extinction of this DPS.

4.1.1.1 Potential Effects on Scalloped Hammerhead Sharks from SURTASS LFA Sonar

The cartilaginous fishes, or elasmobranchs, include sharks and rays and their relatives. Virtually nothing is known about effects of human-generated sound on cartilaginous fishes, but there is concern about potential effects since these animals are integral to the ecosystem in many parts of the marine environment. There are also some data that some species will swim towards low-frequency human-generated sounds that resemble the sounds produced by struggling prey.

There is also evidence that elasmobranchs can detect and respond to human-generated sounds. Myrberg and colleagues did experiments in which they played back sounds and attracted a number of different shark species to the sound source (e.g., Myrberg et al, 1969, 1972, 1976; Nelson and Johnson, 1972). The results of these studies showed that sharks were attracted to pulsed low-frequency sounds (below several hundred Hz), in the same frequency range of sounds that might be produced by struggling prey (or divers in the water). However, sharks are not known to be attracted by continuous signals or higher frequencies (which they cannot hear).

Popper (A. N. Popper et al., 2007)studied the effect of SURTASS LFA on hearing, the structure of the ear, and select non-auditory systems in the rainbow trout (Oncorhynchus mykiss) and channel catfish (Ictalurus punctatus) also (Halvorsen, Wysocki, & Popper, 2006b). The SURTASS LFA sonar study was conducted in an acoustic free-field environment that enabled the investigators to have a calibrated sound source and to monitor the sound field throughout the experiments. In brief, experimental fish were placed in a test tank, lowered to depth, and exposed to LFA sonar for 324 or 648 seconds, an exposure duration that is far greater than any fish in the wild would get since, in the wild, the sound source is on a vessel moving past the far slower swimming fish. For a single tone, the maximum RL was approximately 193 dB re 1 μPa at 196 Hz and the level was uniform within the test tank to within approximately ±3 dB.

The signals were produced by a single SURTASS LFA sonar transmitter giving an approximate source level of 215 dB. Following exposure, hearing was measured in the test animals. Animals were also sacrificed for examination of auditory and non-auditory tissues to determine any non-hearing effects. All results from experimental animals were compared to results obtained from baseline control and control animals.

A number of results came from this study. Most importantly, no fish died as a result of exposure to the experimental source signals. Fish all appeared healthy and active until they were sacrificed or returned to the fish farm from which they were purchased. In addition, the study employed the

expertise of an expert fish pathologist who used double-blind methods to analyze the tissues of the fish exposed to the sonar source, and compared these to control animals. The results clearly showed that there were no pathological effects from sound exposure including no effects on all major body tissues (brain, swim bladder, heart, liver, gonads, blood, etc.). There was no damage to the swim bladder and no bleeding as a result of LFA sonar exposure. Furthermore, there were no short- or long-term effects on ear tissue (Popper et al., 2007 for figures; also Kane et al., in prep.).

Moreover, behavior of caged fish after sound exposure was no different than that prior to tests. It is critical to note, however, that behavior of fish in a cage in no way suggests anything about how fish would respond to a comparable signal in the wild. Just as the behavior of humans exposed to a noxious stimulus might show different behavior if in a closed room as compared to being out-of-doors, it is likely that the behaviors shown by fish to stimuli will also differ, depending upon their environment.

The study also incorporated effects of sound exposure on hearing both immediately post exposure and for several days thereafter to determine if there were any long-term effects, or if hearing loss showed up at some point post exposure. Catfish and some specimens of rainbow trout showed 10-20 dB of hearing loss immediately after exposure to the LFA sonar when compared to baseline and control animals; however another group of rainbow trout showed no hearing loss. Recovery in trout took at least 48 hours, but studies could not be completed. The different results between rainbow trout groups is difficult to understand, but may be due to developmental or genetic differences in the various groups of fish. Catfish hearing returned to, or close to, normal within about 24 hours.

During the LFA sonar study on rainbow trout, Popper et al. (2007) found that some fish showed a hearing loss, but other animals, obtained a year later but from the same supplier and handled precisely as the fish used in the earlier part of the study, showed no hearing loss. The conclusion reached by Popper et al. (2007) was that the differences in responses may have been related to differences in genetic stock or some aspect of early development in the two groups of fish studied.

The idea of a developmental effect was strengthened by findings of Wysocki (Wysocki, Amoser, & Ladich, 2007) who found differences in hearing sensitivity of rainbow trout that were from the same genetic stock, but that were treated slightly differently in the egg stage. This is further supported by studies on hatchery-reared Chinook salmon (Oncorhynchus tshawytscha) which showed that some animals from the same stock and age class had statistical differences in their hearing capabilities that was statistically correlated with differences in otolith structure (Oxman et al., 2007). While a clear correlation could not be made between these differences in otolith structure and specific factors, there is strong reason to believe that the differences resulted from environmental effects during development.

The conclusion one must reach from these findings is that there is not only variation in effects of intense sound sources on different species, but that there may also be differences based on genetics or development. Indeed, one can go even further and suggest that there may ultimately be differences in effects of sound on fish (or lack of effects) that are related to fish age as well as development and genetics since it was shown by Popper et al. (2005) that identical seismic airgun exposures had very different effects on hearing in young-of-the-year northern pike and sexually mature animals.

Hearing range for the bull shark, Carcharhinus leucas, reportedly is 100-1400 Hz (Kritzler & Wood, 1961); the lemon shark, Negaprion brevirostris, hears from 10-640 Hz (Banner, 1967) (Banner, 1967; Nelson, 1967; Banner, 1972), and the hammerhead shark, Sphyrna lewini, from 250-750 Hz (Olla, 1962). Data from shark attraction experiments suggest hearing up to 1500 Hz in a number of species, although these data are not quantified and need to be repeated. Sharks are also of interest because of their low frequency sound detection ability, a capability that is particularly important for detecting sounds that are produced by potential prey ((D. Nelson & Gruber, 1963); Myrberg et al., 1976; Nelson and Johnson, 1976; Myrberg, 1978). There are hearing data on very few species, and it is not yet clear whether sharks and rays are sensitive to sound pressure or to particle velocity (or displacement), or to both. In general, sharks appear to only detect frequencies that are in a range that is similar to that of fish that are classified as hearing generalists, and hearing sensitivity (the lowest sound levels detectable) is probably poorer than hearing generalist fish (Banner, 1967; Nelson, 1967; Kelly and Nelson, 1975). The function of the lateral line system of sharks is likely, as in fish, to respond to low frequency hydrodynamic stimuli.

Data on shark hearing are very limited and in need of replication and expansion to include more species and more specimens. Some representative data indicate that hammerhead sharks are able to detect sounds below 750 Hz, with best sensitivity from 250 to 275 Hz (Olla, 1962). Kritzler and Wood (1961) reported that the bull shark responded to signals at frequencies between 100 and 1,400 Hz, with the band of greatest sensitivity occurring at 400 to 600 Hz. Lemon sharks responded to sounds varying in frequency from 10 to 640 Hz, with the greatest sensitivity at 40 Hz. However, the lowest frequency may not accurately represent the lower limit of lemon shark hearing due to limitations in the range of frequencies that could be produced in the test tank due to the nature of the tank acoustics. Moreover, lemon sharks may have responded at higher frequencies, but sounds of sufficiently high intensity that could not be produced to elicit attraction responses (Nelson, 1967). Banner (1972) reported that lemon sharks he studied responded to sounds varying from 10 to 1,000 Hz. In a conditioning experiment with horn sharks, Kelly and Nelson (1975) discovered the sharks responded to frequencies of 20 to 160 Hz. The lowest particle motion threshold was at 60 Hz. The most recent study was that of the little skate, Raja erinacea (Casper et al., 2003). Results suggest that this species is able to detect sounds from 100 to over 800 Hz, with best hearing up to and possibly slightly greater than 500 Hz. However, these authors, as several others working with elasmobranchs, report thresholds in

terms of pressure, whereas it is highly likely that all of these species are detecting particle motion (van den Berg and Schuijf, 1983), and so the thresholds are possibly quite different than those reported since particle motion was not calibrated.

Researchers doing field studies on shark behavior found that several shark species appear to exhibit withdrawal responses to broadband noise (500-4,000 Hz, although it is not clear that sharks heard the higher frequencies in this sound). The oceanic silky shark (Carcharhinus falciformis) and coastal lemon shark (Negaprion brevirostris) withdrew from an underwater speaker playing low frequency sounds (Myrberg et al., 1978; Klimley and Myrberg, 1979). Lemon sharks exhibited withdrawal responses to broadband noise raised 18 dB at an onset rate of 96 dB/sec to a peak amplitude of 123 dB RL from a continuous level just masking broadband noise (Klimley and Myrberg, 1979). Myrberg et al. (1978) reported that a silky shark withdrew 10 m (33 ft) from a speaker broadcasting a 150-600 Hz sound with a sudden onset and a peak sound pressure level of 154 dB SL. These sharks avoided a pulsed LF attractive sound when its sound level was abruptly increased by more than 20 dB. Other factors enhancing withdrawal were sudden changes in the spectral or temporal qualities of the transmitted sound. Klimley (unpublished data) also noted the increase in tolerance of lemon sharks during successive sound playback tests. Myrberg (1978) has also reported withdrawal response from the pelagic whitetip shark (Carcharhinus longimanus) during limited testing.

The effects of pulse intermittency and pulse-rate variability on the attraction of five species of reef sharks to low frequency pulsed sounds were studied at Eniwetok Atoll, Marshall Islands in 1971 (Nelson and Johnson, 1972). The species of shark tested were: gray reef, blacktip reef, silvertip, lemon, and reef white tip. Nelson and Johnson (1972) concluded from these tests that the attractive value of 25-500 Hz pulsed sounds is enhanced by intermittent presentation, and that such intermittency contributes more to attractiveness than does pulse-rate variability. All tested sharks exhibited habituation to the sounds during the course of the experiment.

One caveat with all data collected with sharks is that they are generally obtained from studies of a single animal, and it is well known that sound detection ability (both sensitivity and hearing bandwidth) varies considerably among different species, and even among members of the same species. Moreover, it is known that hearing ability changes with age, health, and many other variables. Thus, while the thresholds reported for sharks give an indication of the sounds they detect, it would be of great value to replicate these analyses using modern methods and several animals. A similar observation may be made for some fish studies, but generally those are done with several animals and are replicated far more than is possible with the larger and more difficult-to-handle sharks. But it is important to note that in virtually all fish studies there is some variation in hearing sensitivity among fish, reflecting the normal variation found in hearing in all vertebrates.

4.1.1.1.1 Non-auditory Injury

In the absence of published, peer-reviewed reports on the potential for low frequency underwater

sound to cause non-auditory injury to sharks, potential for non-auditory impacts to fish would be resonance of the swim bladder, although the preponderance of recent evidence suggests this is not the case for SURTASS LFA sonar (or for seismic air guns). Moreover, sharks do not have a swim bladder.

4.1.1.1.2 Permanent Loss of Hearing

Hearing capability in sharks is on a par with or poorer than that of hearing non-specialist bony fish, and there is no evidence that any shark is a hearing specialist. There are also no data on permanent hearing loss, including PTS, in sharks or on damage to the ears. Nevertheless, the utilization of the 180-dB criterion for analysis is also applied to sharks, and its conservativeness is emphasized. A very small fraction of any shark stock would be exposed to these levels, even in the absence of mitigation. While extrapolation from bony fish to sharks is something that should be done only with caution, since the ears and auditory systems are so different, the lack of substantive effect on non-specialist fish may also be the same for sharks.

4.1.1.2 Temporary Loss of Hearing

There are no scientific data on TTS in sharks. However, because sharks are considered hearing non-specialists and assuming they have similar hearing sensitivities as bony fish discussed previously, the potential for TTS to cause substantial deleterious effects on shark stocks due to SURTASS LFA sonar transmissions is probably very small. Moreover, because sharks are considered hearing non-specialists, the Hastings et al. (1996) suggestion supported by the Smith et al. (2004a, 2004b) study may potentially apply, indicating that RLs of 220 to 240 dB would be required to temporarily affect hearing capability in the form of TTS. However, without any additional studies on sharks this suggestion must be considered speculative, and probably very conservative.

At the same time, while it is likely that the 180-dB value is highly conservative, it must be noted that extrapolating from bony fish to sharks is difficult, especially since the ears of fish and sharks have some significant differences in terms of associated structures that might be involved in hearing, and in the structure of certain regions of the ear. In particular, the ear structure involved in shark hearing may be the macula neglecta, a sensory receptor that, while very large in sharks, is tiny or not present in other vertebrates (Corwin, 1981; Popper and Fay, 1997). Because the macula neglecta has a somewhat different mechanism of sound-induced stimulation than do the otolithic organs of fish ears (i.e., the ear organs of fish that were damaged in the Hastings et al. [1996] study), extrapolation on the effects of intense sounds must be provisional.

Due to the lack of more definitive data on shark stock distributions in the open ocean, it is not feasible to estimate the percentage of a stock that could be located in a SURTASS LFA sonar operations area at a potentially vulnerable depth, during a sound transmission. Therefore, it is our assumption that the stocks are evenly distributed. Further, the five SURTASS LFA sonar operational parameters provide additional support to the conclusion that there would be minimal impact on any substantial fraction of a shark stock through TTS.

4.1.1.2.1 Behavioral Change (Attraction/Repulsion)

Some sharks are attracted to pulsing LF sounds. It has been proposed that such sounds mimic the thrashing of struggling fish that are potential prey for the sharks (Nelson and Gruber, 1963; Nelson and Johnson, 1972, 1976). Since the structure of SURTASS LFA sonar signals is unlike sounds made by struggling marine animals, it is highly unlikely that this sound would be attractive to sharks.

Several shark species, including the oceanic silky shark (Carcharhinus falciformis) and coastal lemon shark (Negaprion brevirostris), have been observed withdrawing from pulsed LF sounds played from an underwater speaker (Myrberg et al., 1978; Klimley and Myrberg, 1979). Lemon sharks exhibited withdrawal responses to pulsed low to mid frequency sounds (500 to 4,000 Hz) raised 18 dB at an onset rate of 96 dB/sec to a peak amplitude of 123 dB RL from a continuous level, just masking broadband ambient noise (Klimley and Myrberg, 1979). Sharks withdrew from a normally attractive pulsed sound composed of frequencies of 150 to 300 Hz at RLs >111dB. The "pulsed" signals referred to was those signals used by the researchers (Nelson and Johnson, 1972). In their study, lemon sharks withdrew from artificial sounds which included 10 pulses/second (continuous), 10 pulses/second (intermittent, and 15 to 7.5 decreasing pulses/second (intermittent). Myrberg et al. (1978) utilized sounds that simulated orca screams and a pure tone.

In a more recent study, Myrberg (2001) stated that sharks have demonstrated highest sensitivity to LF sound (40 to 800 Hz). Free-ranging sharks are attracted to sounds possessing specific characteristics including irregular pulsed, broadband frequencies below 80 Hz and transmitted suddenly without an increase in intensity thus resembling a struggling fish.

These signals, some "pulsed," are substantially different from the LFA signals. Myrberg et al. (1978) reported that a silky shark withdrew 10 m (33 ft) from a speaker broadcasting a 150 to 600 Hz sound with a sudden onset and a peak SL of 154 dB. These sharks avoided a pulsed LF attractive sound when its SL was abruptly increased by more than 20 dB. Other factors enhancing withdrawal were sudden changes in the spectral or temporal qualities of the transmitted sound. These results do not rule out that such sounds may have been harmful to them after habituation; the tests were not designed to examine that point. Klimley (unpublished data) also noted the increase in tolerance of lemon sharks during successive sound playback tests. The pelagic whitetip (Carcharhinus longimanus) also showed a withdrawal response during limited tests (Myrberg et al., 1978).

Since the likelihood of a significant portion of any shark stock being in the vicinity of the SURTASS LFA sonar source at any one time is low, and given that the LFA signals are not "pulsed" or structured is like sounds made by struggling marine animals, this attraction or repulsion behavioral response is not considered an issue of concern.

4.1.1.3 Behavioral Change (Migration)

There is a body of scientific evidence that oceanic sharks make directional migrations. The most rigorous study demonstrating this phenomenon involved placing a miniature heading sensor to track scalloped hammerhead sharks (Sphyrna lewini) and tracking them (Klimley, 1993). The movements of these sharks between their daytime aggregations at a seamount and their nighttime feeding grounds at other surrounding seamounts were highly directional. Their paths generally coincided with magnetic ridges and valleys leading from a seamount, which may be characterized by a strong dipole field that could serve as a landmark. In addition, movements of the sharks often were along the edge of a magnetic lineation, oriented roughly in a north-south direction.

These results have led to the theory that sharks often migrate along magnetic "roads" that run north-south (coincident with magnetic lineations) and aggregate at "cities" that are seamounts and islands (with dipole fields) (Klimley, 1995).

In assessing the potential for SURTASS LFA sonar signals to affect shark migrations, it is noted that the SURTASS LFA sonar source frequency is between 100 and 500 Hz, a region of the acoustic spectrum where these species appear to be best able to hear sound. Furthermore, the LFA signal usually has no ramp-up, an acoustic property that has been shown to provoke withdrawal in an inshore species (Negapion brevirostris) (Klimley and Myrberg, 1979) and two pelagic species (Carcharhinus falciformis and C. longimanus) (Myrberg et al., 1978). These studies suggest that sharks can detect sounds with intensities below 180 dB RL. The issue is whether one or more SURTASS LFA sonar transmissions could possibly cause displacement of a shark from its migratory path, such that this activity might be disrupted to such an extent that the shark would not be able to reestablish its direction along the path.

The sharks are believed to be migrating along the edges of the magnetic lineations, where the gradients are greatest, moving back and forth across the gradient (estimated travel +/- 0.5 km [0.27 nm] either side) at an approximate speed of 1 m/sec (Klimley, pers. comm., 2000). Given that the maximum SURTASS LFA sonar signal length is 100 sec, a shark that was annoyed and moved away from the sound would travel approximately 100 m (328 ft) during that time. In the worst case, the ship would be positioned so that the shark's movement would be away from the gradient, and the shark would be at its maximum distance from the gradient at the time of the transmission. Assuming 100 m (328 ft) maximum displacement in this case, it would be likely that the shark would be able to eventually reestablish its direction along the path. Thus, the conclusion here is that it would be unlikely that significant impacts to shark migration would occur due to SURTASS LFA sonar operations in the open ocean.

4.1.1.4 Masking

Sharks use hearing to detect prey (Banner, 1972; Myrberg et al., 1972; Nelson and Johnson, 1972; Myrberg et al., 1976; Nelson and Johnson, 1976), and this detection ability may potentially be affected by masking. By way of example, Nelson and Johnson (1970) measured a lemon

shark's hearing sensitivity to a 300 Hz, 130 dB SL in two different sea states (sea states 1 and 2) and two different levels of vessel traffic (light and heavy). The shark's auditory threshold was decreased by 2 dB for sea state 2 versus sea state 1, a level of difference that is probably not significant since it is certainly within the variation of the hearing ability of the animal. The difference caused by light versus heavy vessel traffic was 18 dB (measured in sea state 1). This represented differences in masking ranges (distance from animal that a sound or sounds would be masked) (due to sea state alone) of 45 m (148 ft) for sea state 2 versus 1; and 110 m (360 ft) for heavy versus light boat/ship traffic. Thus, it can be concluded that the masking range for sharks can be elevated by sea state and vessel traffic.

As in bony fish, masking effects would be most significant for sharks with critical bandwidths at the same frequencies as the SURTASS LFA sonar, assuming that masking mechanisms in sharks are similar to that in mammals. However, at an estimated 7.5 percent duty cycle and an average 60-second transmission window, any masking would probably be temporary since the intermittent nature of the signal reduces the potential impact. Long-term effects of masking sounds on hearing and potential injury to shark hearing by intense sounds have not been studied.

In summary, masking effects are not expected to be significant because the SURTASS LFA sonar bandwidth is very limited (approximately 30 Hz), signals do not remain at a single frequency for more than ten seconds, and the system is usually off over 90 percent of the time.

4.1.1.5 Conclusions

To understand the potential effects of SURTASS LFA sonar on the scalloped hammerhead shark, it is important to note the following operational parameters of the sonar:

- Small number of SURTASS LFA sonar systems to be deployed;
- Geographic restrictions imposed on the systems' employment;
- Narrow bandwidth of SURTASS LFA ·sonar active signal (approximately 30 Hz) ;
- Slowly moving ship, coupled with low system duty cycle, would mean that fish would spend less time in the LFA mitigation zone (180-dB SPL sound field); therefore, with a ship speed of less than 9 . 3 km/hr (5 kt), the potential for animals being in the sonar transmit beam during the estimated 7.5 to 20% of the time the sonar is actually transmitting is very low; and
- Small size of the LFA mitigation zone (180-dB SPL sound field) relative to fisheries provinces and open ocean areas.

Due to the lack of definitive stock or abundance data on the scalloped hammerhead shark Indo-West Pacific DPS in the ocean, it is not feasible to estimate the percentage of the DPS that could be located in a SURTASS LFA sonar operations area at a potentially vulnerable depth during an LFA sound transmission. The sparse data on hearing in the scalloped hammerhead shark by Olla (1962) indicate that this shark has best sensitivity in the range from 600 to 800 Hz. There are only limited new data on the potential effects of low frequency underwater sound on sharks,

rays, and skates using particle motion. Casper et al .(2003) showed that the little skate is able to detect sounds from 100 to over 800 Hz, with best hearing up to and possibly slightly greater than 500 Hz. More recent studies reported similar thresholds and hearing ranges for the nurse shark , the yellow stingray (Casper and Mann, 2006), the horn shark and the white-spotted bamboo shark (Casper and Mann, 2007) . These are consistent with elasmobranch species being able to detect sounds up to 1000 Hz, with u sable hearing limited to about 500 Hz.

In the absence of published, peer-reviewed reports on the potential for low frequency underwater sound to cause nonauditory injury to elasmobranches (sharks, rays, and skates), information regarding bony fish non-auditory injury as presented in the Navy's 2012 SEIS/SOEIS on SURTASS LFA sonar employment is relied on in this Opinion. To synopsize that information, studies of the effect of SURTASS LFA sonar on non- auditory tissue of bony fishes showed that while the sonar signal was very intense, no effect on non-auditory tissues resulted from exposure to sonar. In all fish, the swim bladder was fully intact after exposure and there was no damage to tissues either at the gross or cellular levels, and no fish mortalities r e sulted from an LFA sonar source at relative high levels (up to 193 dB re 1pPa [rms] RL) (Popper et al ., 2007; Kane et al., 2010). These studies provide the direct evidence that sounds from SURTASS LFA sonar do not kill or damage fish.

Some sharks in the SURTASS LFA sonar operations area could possibly be affected by LF sounds, but only if they were very close to the sound source. However, a negligible portion of any shark stock would be exposed to levels at or above 180 dB RL on an annual basis due to the small size of the LFA mitigation zone (180-dB sound field) relative to the open ocean areas inhabited by shark stocks.

Despite the ability of sharks to detect LF sound and the possibility of affecting sharks that are migrating or aggregating at seamounts/islands, the potential for the SURTASS LFA sonar to affect shark stocks would not be significant.

Therefore, based on the information above, although the employment of SURTASS LFA sonar may affect the Indo-West Pacific DPS of scalloped hammerhead sharks, it is not likely to adversely affect, and therefore is not likely to jeopardize its continued existence of, that DPS. We concur with the Navy's determination that SURTASS LFA sonar operation, training and testing activities may affect, but is not likely to adversely affect scalloped hammerhead sharks and do not anticipate effects rising to the level of take during the remaining period of the MMPA rule, annual LOAs or in the reasonably foreseeable future. This species will not be considered further in this opinion.

4.1.1.6 Scalloped Hammerhead – Indo-West Pacific DPS Critical Habitat

NMFS has not designated critical habitat for the scalloped hammerhead shark- Indo-West Pacific DPS.

4.1.2 North Pacific Right Whale Critical Habitat

In July 2006, NMFS designated two areas as critical habitat for right whales in the North Pacific (71 FR 38277). The designated areas encompass about 36,750 square miles of marine habitat, which include feeding areas within the Gulf of Alaska and the Bering Sea that support the species. The primary constituent element identified for this critical habitat is the presence of large copepods and oceanographic factors that concentrate copepods and provide foraging habitat for North Pacific right whales. The Bering Sea is part of the polar non-operational area for SURTASS LFA sonar. During 2014 to 2015, SURTASS LFA sonar will not operate in the Gulf of Alaska. Also, the Gulf of Alaska critical habitat has been designated as OBIA #5 for SURTASS LFA sonar. During 2014 to 2015, the nearest geographic region in which SURTASS LFA sonar would operate is approximately 2,900 km (1,566 nmi) south of the Gulf of Alaska critical habitat for the North Pacific right whale and even further for away for the Bering Sea critical habitat. At such great distances, we expect any SURTASS LFA sonar signals that may propagate within proximity of North Pacific right whale critical habitat would be of low signal strength; that is, due to transmission loss, any SURTASS LFA sonar signals would be reduced to negligible sound levels. We conclude that the SURTASS LFA sonar training, testing, and operations the U.S. Navy proposes to conduct in the action area on an annual basis consistent with the MMPA letters of authorization, cumulatively over five-year period of the MMPA rule (August 15, 2012 through August 14, 2017), and ongoing for the reasonably foreseeable future is not expected to affect the conservation value of North Pacific right whale critical habitat. As a result, this critical habitat will not be considered further in this opinion.

4.1.3 Hawaiian Monk Seal Critical Habitat

Critical habitat that has been designated for Hawaiian monk seals occurs in the action area for this consultation. Critical habitat was originally designated on 30 April 1986 (51 FR 16047) and was extended on 26 May 1988 (53 FR 18988; CFR § 226.201). Designated critical habitat includes all beach areas, sand spits and islets, including all beach crest vegetation to its deepest extent inland, lagoon waters, inner reef waters, and ocean waters out to a depth of 20 fathoms (37 m) around the following land areas: Kure Atoll, Midway Islands, except Sand Island and its harbor, Pearl and Hermes Reef, Lisianski Island, Laysan Island, Maro Reef, Gardner Pinnacles, French Frigate Shoals, Necker Island, Nihoa Island. The marine component of this habitat was designated primarily as feeding areas for Hawaiian monk seals, while terrestrial habitat serves as pupping and nursing habitat for mothers and pups. Because the marine critical habitat is located within the Papahānaumokuākea Marine National Monument, fishing is forbidden within critical habitat.

On 2 June 2011, NMFS proposed to extend critical habitat in the Northwestern Hawaiian Islands (NWHI) to Sand Island (Midway) and ocean waters out to a depth of 500 m (76 FR 32026). Additionally, the areas around the MHIs: [Kaula Island, Niihau, Kauai, Oahu, Maui Nui (including Kahoolawe, Lanai, Maui, and Molokai)], and Hawaii (except areas designated for military use as specified in the proposed rule) are proposed for designation to a depth of 500 m

and inland to a distance of 5 m from shore. As in the current critical habitat designation the marine component of this habitat is proposed primarily as feeding areas for juvenile and adult Hawaiian monk seals, while terrestrial habitat serves as pupping and nursing habitat for mothers and pups. Foraging habitat may range from barrier reefs, leeward slopes of reefs and islands, submarine ridges, nearby seamounts, submerged reefs and banks, and deep coral beds. Preferred foraging habitat of adult monk seals is characterized by sand terraces and talus slopes. These habitats provide substrate and materials for preferred benthic and cryptic prey species to hide. Prey resources may include a variety of species including some benthic and offshore teleosts, cephalopods, and crustaceans.

Due to the operational restrictions for SURTASS LFA sonar, sound levels above 180 dB will not be generated within 22 km (12 nmi) of any coastline or land area. Terrestrial areas that serve as pupping and nursing habitat for mothers and pups would not be exposed to sonar signals as the the SURTASS LFA array is deployed below the surface in the water column. The operational restrictions on the use of SURTASS LFA sonar would prevent the highest sound levels from LFA sonar signals from penetrating coastal waters with appreciable signal strengths; however, prey resources for Hawaiian monk seals occur in waters around submerged land areas such as barrier reefs and deep coral beds.

Studies using active acoustics indicate that sound sources could have indirect, adverse effects on marine animals by reducing the abundance or availability of prey or changing the structure or composition of the fish community. Hawaiian monk seals could forage in some portion of the Action Area. The indirect effects of LFA sonar could occur if fish or invertebrates experience lethal or sub-lethal damage, stress responses, or alterations in their behavior or distribution in response to acoustic energy produced by operation of SURTASS LFA. Because fish and invertebrate species are prey for Hawaiian monk seals, such effects might have adverse consequences for individuals foraging in the Action Area.

A number of investigators have suggested that fish exposed to high intensity sounds could show a range of non-auditory injuries, extending from the cellular level to gross damage of the swim bladder and circulatory system (A. N. Popper & Hastings, 2009a). However, the bulk of the data suggesting such injuries come from studies that tested the effects of explosives on fish (M. Hastings & Carlson, 2008; A. N. Popper & Hastings, 2009a, 2009b; Yelverton, Richmond, Hicks, Saunders, & Fletcher, 1975). There is less evidence for such damage (albeit, from fewer studies) when fish are exposed to sounds similar to those produced by sonars, pile driving, shipping noise, and other anthropogenic sources.

Studies estimating the effects of sound on terrestrial mammals suggest that lungs and other organs are potentially damaged by sound (Dodd, Mundie, Lagutchik, & Morris, 1997; Fletcher, Yelverton, & Richmond, 1976; Yang, Wang, Tang, & Ying, 1996); see also (Donald Henderson, 2008) for review of noise standards for humans). There is also some evidence, in “gray”

literature reports (i.e., non-peer-reviewed), that high sound pressure levels may cause tearing or rupturing of the swim bladder of some (but not all) fish species (Gaspin, 1975; Yelverton et al., 1975). Most recently, similar results have been observed in fish exposed to the impulsive sounds from pile driving when fish are at an undetermined range but very close to the pile driving source (Abbott & Bing-Sawyer, 2002; CalTrans, 2004). However, such studies have yet to be repeated under controlled experimental conditions and none have received scientific peer review (A. N. Popper & Hastings, 2009b).

The only studies that examined the effects of sound on non-auditory tissues of fish have been the recent work using SURTASS LFA sonar (undertaken by the U.S. Navy) and seismic airguns, both of which are reviewed below (Kane et al., 2010; Arthur N. Popper, 2005; A. N. Popper et al., 2007). The significant point from these studies is that neither source, despite being very intense, had any effect on non-auditory tissues. In all fish, the swim bladder was intact after exposure, and in the one study that involved an expert fish pathologist (to ensure that the non-auditory tissues of the fish sacrificed were examined properly), there was no damage to tissues either at the gross or cellular levels (Kane et al., 2010; A. N. Popper et al., 2007). These studies provide the first direct evidence that sounds, including seismic airguns and SURTASS LFA sonar may be of concern but that does not necessarily mean that they kill or damage fish. However, it must be cautioned, as done by Hastings et al. (1996), McCauley et al. (2003), Popper et al. (2007), and Kastelein et al. (2008) (among others) that all studies to date have been done with a very limited number of species, and that extrapolation among species and to other sound sources (or even to other levels or durations of the same sound sources), must be done with extreme caution, at least until there are more data upon which to base any extrapolations.

Recent studies using high intensity seismic airguns as well as LFA sonar and mid-frequency active (MFA) sonars have found no mortality resulting from exposure (M. Hastings & Carlson, 2008; Kane et al., 2010; McCauley et al., 2000; McCauley et al., 2003; Arthur N. Popper, 2005; A. N. Popper et al., 2007). In contrast, one report by Turnpenny et al. (1994) suggested that sound exposure could produce substantial damage in caged fish. However, reviews by subject matter experts found problems with the Turnpenny et al. report and concluded that it did not appear to reflect the best available science on this issue.

A range of invertebrates are reported to be sensitive to low-frequency (10–150 Hz) hydroacoustic disturbances induced by sound waves or other sources – e.g., jelly fish, crustaceans, arrow worms, octopus, and squid (Western Australian Department of Industry and Resources, 2002). This sensitivity overlaps the frequency range of SURTASS LFA sonar (100-500 Hz), indicating that invertebrates could likely perceive SURTASS LFA sonar activity (Western Australian Department of Industry and Resources, 2002). Available studies report responses to airgun shots as being limited to transient alarm responses such as tail-flicks (lobsters) or siphon closing (ascidians) (Western Australian Department of Industry and Resources, 2002), although mortality of giant squid in the Bay of Biscay may possibly have been

linked to seismic airgun activity in the area (Guerra, Gonzalez, & Rocha, 2004). McCauley *et al*. (2000b; 2000a) examined the effect of marine seismic surveys on captive squid and cuttlefish and reported a strong startle response or directed movement away from airguns during sudden, nearby start-ups at received levels of 174 dB re 1 μPa mean squared pressure. Alarm responses in squid were detected during gradual ramp-up of airguns once levels exceeded 156-161 dB re 1 μPa mean squared pressure. Squid in these trials appeared to make use of the sound shadow measured near the water surface. These responses for captive squid suggest that behavioral changes and avoidance of operating airguns would likely occur. The authors concluded squid significantly alter their behavior at an estimated distance of 2–5 km (1.1–2.7 nm) from an approaching large seismic source. A more recent study by André et al (2011) exposed four species of squid and octopus in a tank to two hours of intense sounds. The authors reported tissue degeneration cause by the sounds. However, this study had substantial design and control problems and the results could have been attributed to those problems. Based on a review of airgun impacts on invertebrates by Christian and Bocking (2010), studies to date have not revealed any consistent evidence of serious pathological or physiological effects on invertebrates.

These studies indicate that exposure to low frequency sound has the potential to affect fish and invertebrates. Fish appear to exhibit startle responses and avoidance of sounds, including seismic airguns and SURTASS LFA, recovering or habituating after a short time period. Squid also appear to exhibit alarm responses and avoidance of sound sources. Limited data on the physiological effects of sound on fish and invertebrates indicate these effects are short-term and most apparent after exposure at very close range. Although, disturbance of these prey species has the potential to negatively affect Hawaiian monk seals foraging in the Action Area, we expect such responses would have temporary effects on the feeding ability of the seals in the immediate survey area.

Based on the above, the proposed operation of SURTASS LFA sonar is not likely to adversely affect prey species of the Hawaiian monk seals. We conclude that the SURTASS LFA sonar training, testing, and operations the U.S. Navy proposes to conduct in the action area on an annual basis consistent with the MMPA letters of authorization, cumulatively over five-year period of the MMPA rule (August 15, 2012 through August 14, 2017), and ongoing for the reasonably foreseeable future is not likely to adversely affect the conservation value of the critical habitat that has been designated for Hawaiian monk seals. As a result, critical habitat of Hawaiian monk seals will not be considered further in this opinion.

4.2 Climate Change

We primarily discuss climate change as a threat common to all species addressed in this opinion, rather than in each of the species-specific narratives. As we better understand responses to climate change, we will address these effects in relevant species-specific sections.

In general, based on forecasts made by the Intergovernmental Panel on Climate Change (IPCC), climate change is projected to have substantial direct and indirect effects on individuals, populations, species, and the structure and function of marine, coastal, and terrestrial ecosystems in the near future (IPCC, 2000, 2001a, 2001b, 2002). From 1906 to 2006, global surface temperatures have risen 0.74º C and continue to rise at an accelerating pace; 11 of the 12 warmest years on record since 1850 have occurred since 1995 (Poloczanska, Limpus, & Hays, 2009). Furthermore, the Northern Hemisphere (where a greater proportion of ESA-listed species occur) is warming faster than the Southern Hemisphere, although land temperatures are rising more rapidly than over the oceans (Poloczanska et al., 2009). The direct effects of climate change will result in increases in atmospheric temperatures, changes in sea surface temperatures, patterns of precipitation, and sea level. Oceanographic models project a weakening of the thermohaline circulation, resulting in a reduction of heat transport into high latitudes of Europe, an increase in the mass of the Antarctic ice sheet, and a decrease in the Greenland ice sheet, although the magnitude of these changes remain unknown. Species that are shorter-lived, of larger body size, or generalist in nature are liable to be better able to adapt to climate change over the long term versus those that are longer-lived, smaller-sized, or rely upon specialized habitats (Brashares, 2003; Cardillo, 2003; Cardillo, Mace, Jones, & Bielby, 2005; Issac, 2009; Purvis, Gittleman, Cowlishaw, & Mace, 2000). Climate change is most likely to have its most pronounced affects on species whose populations are already in tenuous positions (Isaac, 2008). As such, we expect the risk of extinction to listed species to rise as climate shift associated with global warming increases.

Some indirect effects of climate change would result from changes in the distribution of temperatures suitable for whale calving and rearing, the distribution and abundance of prey, and abundance of competitors or predators. For species that undergo long migrations, individual movements are usually associated with prey availability or habitat suitability. If either is disrupted, the timing of migration can change or negatively impact population sustainability (Mark P. Simmonds & Eliott., 2009). Climate change can influence reproductive success by altering prey availability, as evidenced by low-success of northern elephant seals during El Niño periods, when cooler, more productive waters are associated with higher first year pup survival (Clive R. McMahon & Burton., 2005). Reduced prey availability resulting from increased sea surface temperatures has also been suggested to explain reductions in Antarctic fur seal pup and harbor porpoise survival (Forcada, Trathan, Reid, & Murphy., 2005; Macleod, Santos, Reid, Scott, & Pierce., 2007). Polygamous marine mammal mating systems can also be perturbated by rainfall levels, with the most competitive grey seal males being more successful in wetter years than in drier ones (Twiss, Thomas, Poland, Graves, & Pomeroy., 2007). For marine mammals considered in this Opinion, available data suggest sperm whale females have lower rates of conception following periods of unusually warm sea surface temperature (Hal. Whitehead, 1997). Marine mammals with restricted distributions linked to water temperature may be particularly exposed to range restriction (Issac, 2009; J.A. Learmonth et al., 2006). MacLeod (2009) estimated that, based upon expected shifts in water temperature, 88 percent of cetaceans

would be affected by climate change, 47 percent would be negatively affected, and 21 percent would be put at risk of extinction. Of greatest concern are cetaceans with ranges limited to non-tropical waters and preferences for continental shelf habitats (Macleod, 2009). Variations in the recruitment of krill and the reproductive success of krill predators correlate to variations in sea-surface temperatures and the extent of sea-ice coverage during winter months. Although the IPCC (2001b) did not detect significant changes in the extent of Antarctic sea-ice using satellite measurements, Curran et al (2003) analyzed ice-core samples from 1841-1995 and concluded that Antarctic sea ice cover had declined by about 20 percent since the 1950s.

Roughly 50 percent of the Earth's marine mammal biomass occurs in the Southern Ocean, where all baleen whales feed largely on a single krill species, *Euphausia superba*, and feed virtually nowhere else (Boyd, 2002). Atkinson et al. (2004) linked sea ice loss to severe decreases in krill populations over the past several decades in some areas of the Antarctic. Reid and Croxall (2001) analyzed a 23-year time series of the reproductive performance of predators (Antarctic fur seals, gentoo penguins, macaroni penguins, and black-browed albatrosses) that depend on krill for prey and concluded that these populations experienced increases in the 1980s followed by significant declines in the 1990s; overall an increase in the frequency of years with reduced reproductive success occurred. These declines resulted, at least in part, from changes in the structure of the krill population, particularly reduced recruitment into older krill age classes, which lowered the number of predators krill could sustain. The authors concluded that the biomass of krill within the largest size class was sufficient to support predator demand in the 1980s but not in the 1990s. By 2055, severe reductions in fisheries catch due to climate change have been suggested for the Indo-Pacific, Red Sea, Mediterranean Sea, Antarctic, and tropical areas worldwide while increased catches are expected in the Arctic, North Pacific, North Atlantic, and northern portions of the Southern Ocean (Cheung et al., 2010).

Climate change has been linked to changing ocean currents as well. Rising carbon dioxide levels have been identified as a reason for a poleward shift in the Eastern Australian Current, shifting warm waters into the Tasman Sea and altering biotic features of the area (Poloczanska et al., 2009). Similarly, the Kuroshio Current in the western North Pacific (an important foraging area for juvenile sea turtles) has shifted southward as a result of altered long-term wind patterns over the Pacific Ocean (Poloczanska et al., 2009).

Climate-mediated changes in the distribution and abundance of keystone prey species like krill and climate-mediated changes in the distribution of cephalopod populations worldwide are likely to affect marine mammal populations as they re-distribute throughout the world's oceans in search of prey. If sea ice extent decreases, then larval krill may not be able to survive without access to under ice algae to feed upon. This may be a cause of decreased krill abundance in the northern western Antarctic Peninsula during the last decade (Fraser & Hofmann, 2003). Melt waters have also reduced surface water salinities, shifting primary production along the Antarctic Peninsula (Moline, Claustre, Frazer, Schofields, & Vernet, 2004). Blue whales, as predators that

specialize in eating krill, are likely to change their distribution in response to changes in the distribution of krill (P. J. Clapham, Young, & Brownell Jr., 1999; P. Payne, Nicholas, O'Brien, & Powers, 1986; P. M. Payne et al., 1990). If they did not change their distribution or could not find the biomass of krill necessary to sustain their population numbers, their populations (and other large whales with similar life histories, such as humpback whales) would likely experience declines similar to those observed in other krill predators, including dramatic declines in population size and increased year-to year variation in population size and demographics. These outcomes would dramatically increase the extinction probability of baleen whales. Edwards et al. (2007) found a 70 percent decrease in one zooplankton species in the North Sea and an overall reduction in plankton biomass as warm-water species invade formerly cold-water areas. Productivity may increase in other areas, though, providing more resources for local species (C. J. Brown et al., 2009). In addition, reductions in sea ice may alleviate "choke points" that allow some marine mammals to exploit additional habitats (Higdon & Ferguson, 2009).

The indirect effects of climate change would result from changes in the distribution of temperatures suitable for reproduction, the distribution and abundance of prey and the abundance of competitors or predators. For species that undergo long migrations, individual movements are usually associated with prey availability or habitat suitability. If either is disrupted by changing ocean temperature regimes, the timing of migration can change or negatively impact population sustainability (Mark P. Simmonds & Eliott., 2009). For sea turtles, warming ocean temperatures may extend their suitable habitat poleward (Poloczanska et al., 2009). Seagrass habitats have declined by 29 percent in the last 130 years and 19 percent of coral reefs have been lost due to human degradation, reducing lower latitude habitat for some sea turtle species (Poloczanska et al., 2009). Primary production is estimated to have declined by 6 percent between the early 1980s and 2010, making foraging more difficult for marine species (Hoegh-Guldberg & Bruno, 2010).

Foraging is not the only potential aspect that climate change could influence. Acevedo-Whitehouse and Duffus (2009) proposed that the rapidity of environmental changes, such as those resulting from global warming, can harm immuno-competence and reproductive parameters in wildlife to the detriment of population viability and persistence. An example of this is the altered sex ratios observed in sea turtle populations worldwide (M. Fuentes, M. Hamann, & C. J. Limpus, 2009; A.D. Mazaris, Kallimanis, Sgardelis, & Pantis, 2008; Reina, Spotila, Paladino, & Dunham, 2008; R.A. Robinson et al., 2008). This does not yet appear to have affected population viabilities through reduced reproductive success, although average nesting and emergence dates have changed over the past several decades by days to weeks in some locations (Poloczanska et al., 2009). However, such a fundamental shift in population demographics causes a fundamental instability in population viability. Altered ranges can also result in the spread of novel diseases to new areas via shifts in host ranges (Mark P. Simmonds & Eliott., 2009). It has also been suggested that increases in harmful algal blooms could be a result from increases in sea surface temperature (Mark P. Simmonds & Eliott., 2009).

Changes in global climatic patterns will likely have profound effects on the coastlines of every continent by increasing sea levels and the intensity, if not the frequency, of hurricanes and tropical storms (Wilkinson & Souter, 2008). A half degree Celsius increase in temperatures during hurricane season from 1965 to 2005 correlated with a 40 percent increase in cyclone activity in the Atlantic Ocean. Sea levels have risen an average of 1.7 mm/year over the 20th century and 3.3 mm/year between 1993 and 2006 due to glacial melting and thermal expansion of ocean water; this rate will likely increase, which is supported by the latest data from 2009 (Arndt et al., 2010; Hoegh-Guldberg & Bruno, 2010; Wilkinson & Souter, 2008). Based on computer models, these phenomena would inundate nesting beaches of sea turtles, change patterns of coastal erosion and sand accretion that are necessary to maintain those beaches, and would increase the number of turtle nests destroyed by tropical storms and hurricanes (Wilkinson & Souter, 2008), although other areas might experience less frequent tropical activity and a subsequent reduction in tropical cyclone impacts to sea turtle nests (Fuentes & Abbs, 2010). The loss of nesting beaches, by itself, would have catastrophic effects on sea turtle populations globally if the turtles are unable to colonize new beaches t or if the beaches do not provide the habitat attributes (sand depth, temperatures regimes, and refuge) necessary for egg survival. In some areas, increases in sea level alone may be sufficient to inundate sea turtle nests and reduce hatching success (Caut, Guirlet, & Girondot, 2009). Storms may also cause direct harm to sea turtles, causing “mass” strandings and mortality (Poloczanska et al., 2009). Increasing temperatures in sea turtle nests alters sex ratios, reduces incubation times (producing smaller hatchlings), and reduces nesting success due to exceeded thermal tolerances (M. M. P. B. Fuentes, M. Hamann, & C. J. Limpus, 2009; Fuentes, Limpus, & Hamann, 2010; M. M. P. B. Fuentes, J. A. Maynard, et al., 2009). Smaller individuals likely experience increased predation (M. M. P. B. Fuentes, M. Hamann, et al., 2009). Taken together, the body of literature on climate change supports widespread and significant negative consequences to sea turtle species.

Periodic weather patterns such as El Niño, La Niña, and the Pacific decadal oscillation (PDO) can fundamentally change oceanographic conditions in the northeastern Pacific and the biology that is based upon it (P.R. Mundy & Cooney, 2005; P.R. Mundy & Olsson, 2005; P.J. Stabeno et al., 2004). Roughly every 3 to 7 years, El Niño can influence the northeastern Pacific (JOI/USSSP, 2003; P.J. Stabeno et al., 2004). Typical changes include increased winter air temperature, precipitation, sea level, and downwelling-favorable conditions (Royer & Weingartner, 1999; Whitney, Mackas, Welch, & Robert, 1999). La Niña events tend to swing these conditions in the negative direction (P.J. Stabeno et al., 2004). However, sea surface temperatures (SSTs) can take 1 year to change following an El Niño event or change to varying degrees (K. M. Bailey et al., 1995; Brodeur, Frost, Hare, Francis, & Jr., 1996; Freeland, 1990; T.C. Royer, 2005). The 1982/1983 El Niño and other downwelling events are generally regarded to have reduced food supplies for marine mammals along the US west coast (Feldkamp, DeLong, & Antonelis, 1991; Hayward, 2000; Le Boeuf & Crocker, 2005). Marine mammal distribution and social organization (group size) is also believed to have shifted northward in response to persistent or extralimital prey occurrence in more northerly waters during El Niño events (S.R.

Benson, Croll, Marinovic, Chavez, & Harvey, 2002; Danil & Chivers, 2005; D. Lusseau et al., 2004; Norman, C. E. Bowlby, et al., 2004; S.H. Shane, 1994, 1995). Low reproductive success and body condition in humpback whales have also been suggested to have resulted from the 1997/1998 El Niño (S. Cerchio, Jacobsen, Cholewiak, & Falcone, 2005). El Niño events in the winters of 1952-1953, 1957-1958, 1965-1966, and 1982-1983 were associated with strong downwelling anomalies, which reduce primary productivity as fewer nutrients are available for phytoplankton growth (K. M. Bailey et al., 1995; A. Thomas & Strub, 2001; Wheeler & Hill, 1999). Plankton diversity also shifts, as smaller plankton are better able to cope with reduced nutrient availability (Corwith & Wheeler, 2002; Sherr, Sherr, & Wheeler, 2005).

The Pacific decadal oscillation is the leading mode of variability in the North Pacific and operates over longer periods than either El Niño or La Niña and is capable of altering SST, surface winds, and sea level pressure (N. Mantua, 2002; N.J. Mantua & S.R. Hare, 2002; P.J. Stabeno et al., 2004). Unlike El Niño and La Niña events, Pacific decadal oscillation events can persist for 20-30 years, are more prominent outside the tropics, and mechanisms controlling them are relatively unknown (S.R. Hare & Mantua, 2000; N.J. Mantua & S.R. Hare, 2002; Minobe, 1997, 1999). During positive Pacific decadal oscillations, the northeastern Pacific experiences above-average SSTs while the central and western Pacific Ocean undergoes below-normal SSTs (P.R. Mundy & Olsson, 2005; T.C. Royer, 2005). Warm Pacific decadal oscillation regimes, as with El Niño events, tend to decrease productivity along the US west coast (Childers, Whitledge, & Stockwell, 2005; S.R. Hare, Mantua, & Francis, 1999). However, during the 1977 warm phase of the Pacific decadal oscillation, euphausiid biomass remained the same and copepod abundance actually increased in the Pacific northwest; zooplankton biomass doubled in offshore waters of the Gulf of Alaska (Brodeur, Frost, Hare, Francis, & W.J. Ingraham, 1996; Brodeur & Ware, 1992; Francis & Hare, 1997; MacCall et al., 2005; McFarlane & Beamish, 1992). Opposite SST regimes occur during negative Pacific decadal oscillations (P.R. Mundy & Olsson, 2005). Positive Pacific decadal oscillations occurred from 1925-1946 and 1977-1999. Negative Pacific decadal oscillations occurred from 1890-1924, 1947-1976, and 1999-present (Childers et al., 2005; N. J. Mantua, Hare, Zhang, Wallace, & Francis, 1997; Minobe, 1997).

The potential for invasive species to spread under the influence of climactic change is also a concern. If water temperatures warm in marine ecosystems, native species may shift poleward to cooler habitats, opening ecological niches that can be occupied by invasive species introduced via ships ballast water or other sources (Philippart et al., 2011; Ruiz, Fofonoff, & Hines, 1999). A similar observation of "Caribbean creep" has been observed, with warmer waters facilitating the range expansion of warmer-water species into more northerly regions (Canning-Clode, Fowler, Byers, Carlton, & Ruiz, 2011). Although these expansions may be temporary, they can include harmful algal bloom species whose presence even temporarily can cause major morbidity and mortality issues to a variety of endemic species (Hallegraeff, 2010). Moore et al. (2011) estimated that the impacts of a dinoflagellate establishment would likely intensify with a warming climate, resulting in roughly 13 more days of potential bloom conditions per year by

the end of the 21st century. Invasive species that are better adapted to warmer water temperatures can also outcompete native species that are physiologically geared towards lower water temperatures; such a situation currently occurs along central and northern California, where the Mediterranean blue mussel has established and is displacing a native mussel competitor (Lockwood & Somero, 2011).

Sperm whales, whose diets can be dominated by cephalopods, would have to re-distribute following changes in the distribution and abundance of their prey. This statement assumes that projected changes in global climate would only affect the distribution of cephalopod populations, but would not reduce the number or density of cephalopod populations. If, however, cephalopod populations collapse or decline dramatically, sperm whale populations are likely to collapse or decline dramatically as well.

Changes in global climatic patterns are also projected to have profound effect on the coastlines of every continent by increasing sea levels and increasing the intensity, if not the frequency, of hurricanes and tropical storms. Based on computer models, these phenomena would inundate nesting beaches of sea turtles, change patterns of coastal erosion and sand accretion that are necessary to maintain those beaches, and would increase the number of turtle nests that are destroyed by tropical storms and hurricanes. Further, the combination of increasing sea levels, changes in patterns of coastal erosion and accretion, and changes in rainfall patterns are likely to affect coastal estuaries, submerged aquatic vegetation, and reef ecosystems that provide foraging and rearing habitat for several species of sea turtles. Finally, changes in ocean currents associated with climate change projections would affect the migratory patterns of sea turtles. The loss of nesting beaches, by itself, would have catastrophic effect on sea turtles populations globally if they are unable to colonize any new beaches that do not provide the sand depths, grain patterns, elevations above high tides, or temperature regimes necessary to allow turtle eggs to survive. When combined with changes in coastal habitats and ocean currents, the future climates that are forecast place sea turtles at substantially greater risk of extinction than they already face.

4.3 Species Considered Further in this Opinion

The rest of this section consists of narratives for each of the threatened and endangered species that occur in the action area and that may be adversely affected by SURTASS LFA sonar activities. We present a summary of information on the distribution and population structure of each species to provide a foundation for the exposure analyses that appear later in this opinion. Then we summarize information on the threats to the species and the species' status given those threats to provide points of reference for the jeopardy determinations we make later in this opinion. That is, we rely on a species' status and trend to determine whether or not an action's direct or indirect effects are likely to increase the species' probability of becoming extinct.

After the *Status* subsection of each narrative, we present information on the diving and social behavior of the different species because that behavior helps determine whether aerial and ship board surveys are likely to detect each species.

More detailed background information on the status of these species and critical habitat can be found in a number of published documents including status reviews, recovery plans for the blue whale (NMFS, 1998b), fin whale (NMFS, 2010b), sei whale (NMFS, 1998a), humpback whale (NMFS, 1991a), right whale (NMFS, 2004d), sperm whale (NMFS, 2010c), a status report on large whales prepared by Perry et al. (1999), and a recovery plan for the Hawaiian Monk seal (NMFS, 2007d). Published documents for listed sea turtles include recovery plans for Pacific green turtles (NMFS & USFWS, 1998b) and the status review and recovery plan for the leatherback sea turtle (NMFS & USFWS, 1998a, 2007d). Richardson et al. (1995) and Tyack (2000) provide detailed analyses of the functional aspects of cetacean communication and their responses to active sonar. Finally, Croll et al. (1999), NRC (2000, 2003a, 2005), and Richardson and Wursig (1995) provide information on the potential and probable effects of active sonar on the marine animals considered in this opinion.

4.3.1 Blue Whale

The blue whale, *Balaenoptera musculus* (Linnæus, 1758), is a cosmopolitan species of baleen whale. It is the largest animal ever known to have lived on Earth: adults in the Antarctic have reached a maximum body length of about 33 m and can weigh more than 150,000 kg. The largest blue whales reported from the North Pacific are a female that measured 26.8 m (88 ft) taken at Port Hobron in 1932 (Randall R. Reeves, Leatherwood, Karl, & Yohe, 1985) and a 27.1 m (89 ft) female taken by Japanese pelagic whaling operations in 1959 (NMFS, 1998b).

As is true of other baleen whale species, female blue whales are somewhat larger than males. Blue whales are identified by the following characteristics: a long-body and comparatively slender shape; a broad, flat "rostrum" when viewed from above; a proportionately smaller dorsal fin than other baleen whales; and a mottled gray color pattern that appears light blue when seen through the water.

4.3.1.1 Distribution

Blue whales are distributed in subpolar to tropical continental shelf and deeper waters of all oceans and migrate between higher latitudes in summer and lower latitudes in winter (Richard Sears & Perrin, 2009). Blue whales are found along the coastal shelves of North America and South America (R. Clarke, 1980; Donovan, 1984; Dale W. Rice, 1998). In the western North Atlantic Ocean, blue whales are found from the Arctic to at least the mid-latitude waters of the North Atlantic (CETAP, 1982; Gagnon & Clark, 1993; Wenzel, Mattila, & Clapham, 1988; Yochem & Leatherwood, 1985). Blue whales have been observed frequently off eastern Canada, particularly in waters off Newfoundland, during the winter. In the summer months, they have been observed in Davis Strait (A. W. Mansfield, 1985), the Gulf of St. Lawrence (from the north shore of the St. Lawrence River estuary to the Strait of Belle Isle), and off eastern Nova Scotia (Richard Sears, 1983; Richard. Sears, 1983). In the eastern North Atlantic Ocean, blue whales have been observed off the Azores Islands, although Reiner et al. (1996) do not consider them common in that area.

In 1992, the Navy conducted an extensive acoustic survey of the North Atlantic Ocean using the Integrated Underwater Surveillance System's fixed acoustic array system (C.W. Clark, 1995). Concentrations of blue whale sounds were detected in the Grand Banks off Newfoundland and west of the British Isles. In the lower latitudes, one blue whale was tracked acoustically for 43 days, during which time the animal traveled 1400 nautical miles around the western North Atlantic from waters northeast of Bermuda to the southwest and west of Bermuda (Gagnon & Clark, 1993).

In the North Pacific Ocean, blue whales have been recorded off the island of Oahu in the Main Hawaiian Islands and off Midway Island in the western edge of the Hawaiian Archipelago (Barlow, 2006; Northrop, Cummings, & Norrison, 1971; P.O. Thompson & Friedl, 1982), although blue whales are rarely sighted in Hawaiian waters and have not been reported to strand in the Hawaiian Islands.

In the eastern tropical Pacific Ocean, the Costa Rica Dome appears to be an important foraging area for blue whales based on the high density of prey (euphausiids) available in the area and the number of blue whales that appear to (Reilly & Thayer, 1990). Blue whales have been sighted in the Dome region in every season of the year, although their numbers appear to be highest from June through November. Blue whales have also been reported year-round in the northern Indian Ocean, with sightings in the Gulf of Aden, Persian Gulf, Arabian Sea, and across the Bay of Bengal to Burma and the Strait of Malacca (Mizroch, Rice, & Breiwick, 1984). The migratory movements of these whales are unknown.

Blue whales in the eastern Pacific winter from California south while blue whales in the western Pacific overwinter in the Sea of Japan, the East China, Yellow, and Philippine Seas and less frequently in the central North Pacific. Blue whales occur in summer foraging the Chukchi Sea, the Sea of Okhotsk, around the Aleutian Islands, and the Gulf of Alaska. Nishiwaki (1966), reported that blue whales occur in the Aleutian Islands and in the Gulf of Alaska. An array of hydrophones, deployed in October 1999, detected two blue whale call types in the Gulf of Alaska (K. M. Stafford, 2003). Fifteen blue whale sightings off British Columbia and in the Gulf of Alaska have been made since 1997 (John Calambokidis, Barlow, Ford, Chandler, & Douglas., 2009). Three of these photographically verified sightings were in the northern Gulf of Alaska within 71 nm of each other and were less than 100 nm offshore (John Calambokidis, Barlow, Ford, Chandler, & Douglas., 2009).

Blue whales occur, at least seasonally, in the open ocean waters from Japan northward to Kamchatka, Russia but have been observed close to shore feeding and possibly breed in some areas (M. Nishiwaki, 1966); (T.A. Jefferson, Webber, & Pitman, 2008). Blue whales occur rarely in the central North Pacific, with few sightings and acoustic detections having been made (J. V. Carretta et al., 2013). No recent sightings of blue whales have been made around Hawaii in recent years (Barlow, 2006); (Mobley Jr, 2006). Further evidence of their occurrence in the area

exists in acoustic recordings. Stafford et al. (2001) studied the geographic variation of blue whale calls in the North Pacific Ocean using hydrophones off the Kamchatka Peninsula and along the western Aleutian Islands chain and found that all recorded blue whale calls were of north-central and north-west Pacific blue whales. Blue whales occur rarely in the central North Pacific, with few sightings and acoustic detections having been made (J. V. Carretta et al., 2013). There have been no recent sightings of blue whales around Hawaii in recent years (Barlow, 2006). Evidence of blue whale occurrence in the Hawaii region exists in acoustic recordings (K.M. Stafford et al., 2001).

Blue whales appear to migrate to waters offshore of Washington, Oregon, and northern California to forage. That seasonal migration brings blue whales to waters off the U.S. Pacific Northwest (with some individuals continuing north to the Gulf of Alaska) during the warm, summer season with a southward migration to waters off California, south to Central America, during the winter season (John Calambokidis, Barlow, Ford, Chandler, & Douglas., 2009; E. J. Gregr, Nichol, Ford, Ellis, & Trites, 2000; Mate, Lagerquist, & Calambokidis., 1998).

4.3.1.2 Population Structure

For this and all subsequent species, the term “population” refers to groups of individuals whose patterns of increase or decrease in abundance over time are determined by internal dynamics (births resulting from sexual interactions between individuals in the group and deaths of those individuals) rather than external dynamics (immigration or emigration). This definition is a reformulation of definitions articulated by Futuyma (1986) and Wells and Richmond (1995) and is more restrictive than those uses of ‘population’ that refer to groups of individuals that co-occur in space and time but do not have internal dynamics that determine whether the size of the group increases or decreases over time (see review by Wells & Richmond, 1995). The definition we apply is important to section 7 consultations because such concepts as ‘population decline,’ ‘population collapse,’ ‘population extinction,’ and ‘population recovery’ apply to the restrictive definition of ‘population’ but do not explicitly apply to alternative definitions. As a result, we do not treat the different whale “stocks” recognized by the International Whaling Commission or other authorities as populations unless those distinctions were clearly based on demographic criteria. We do, however, acknowledge those “stock” distinctions in these narratives.

At least three subspecies of blue whales have been identified based on body size and geographic distribution (*B. musculus intermedia*, which occurs in the higher latitudes of the Southern Oceans, *B. m. musculus*, which occurs in the Northern Hemisphere, and *B. m. brevicauda* which occurs in the mid-latitude waters of the southern Indian Ocean and north of the Antarctic convergence), but this consultation will treat them as a single entity. Readers who are interested in these subspecies will find more information in Gilpatrick et al. (1997), Kato et al. (1995), Omura et al. (1970), and Ichihara (1966).

In addition to these subspecies, the IWC’s Scientific Committee has formally recognized one blue whale population in the North Pacific (Donovan, 1991), although there is increasing

evidence that there may be more than one blue whale population in the Pacific Ocean Gilpatrick et al. (1997), Barlow et al. (1995), Mizroch et al. (1984), Ohsumi and Masaki (1972). For example, studies of the blue whales that winter off Baja California and in the Gulf of California suggest that these whales are morphologically distinct from blue whales of the western and central North Pacific (Gilpatrick et al., 1997), although these differences might result from differences in the productivity of their foraging areas more than genetic differences (Barlow et al., 1997a; J. Calambokidis et al., 1990; R. Sears, 1987). A population of blue whales that has distinct vocalizations inhabits the northeast Pacific from the Gulf of Alaska to waters off Central America (E. J. Gregr et al., 2000; Mate et al., 1998; K. M. Stafford, 2003). Although Stafford et al. (2001) showed that recordings made near Kaneohe, Hawaii from August 1992 through April 1993 consisted of approximately 30% of the northwest Pacific blue whale call type and 70% of northeast Pacific call type, stock structure of blue whales in the North Pacific Ocean remains uncertain.

4.3.1.3 Natural Threats

Natural causes of mortality in blue whales are largely unknown but probably include predation and disease (not necessarily in that order of importance). Blue whales are known to become infected with the nematode *Carricauda boopis* (Baylis, 1928), which are believed to have caused fin whales to die as a result of renal failure (R.H. Lambertsen, 1986); see additional discussion under fin whales). Killer whales and sharks are also known to attack, injure, and kill very young or sick fin and humpback whales and probably hunt blue whales as well (S. L. Perry et al., 1999).

4.3.1.4 Anthropogenic Threats

Two human activities are known to threaten blue whales; whaling and shipping. Historically, whaling represented the greatest threat to every population of blue whales and was ultimately responsible for listing blue whales as an endangered species. As early as the mid-seventeenth century, the Japanese were capturing blue, fin, and other large whales using a fairly primitive open-water netting technique (Tonnessen & Johnsen, 1982). In 1864, explosive harpoons and steam-powered catcher boats were introduced in Norway, allowing the large-scale exploitation of previously unobtainable whale species.

From 1889 to 1965, whalers killed about 5,761 blue whales in the North Pacific Ocean (P. S. Hill, Laake, & Mitchell., 1999). From 1915 to 1965, the number of blue whales captured declined continuously (Mizroch et al., 1984). Evidence of a population decline was seen in the catch data from Japan. In 1912, whalers captured 236 blue whales; in 1913, 58 blue whales; in 1914, 123 blue whales; from 1915 to 1965, the number of blue whales captured declined continuously (Mizroch et al., 1984). In the eastern North Pacific, whalers killed 239 blue whales off the California coast in 1926. And, in the late 1950s and early 1960s, Japanese whalers killed 70 blue whales per year off the Aleutian Islands (Mizroch et al., 1984).

Although the IWC banned commercial whaling in the North Pacific in 1966, Soviet whaling fleets continued to hunt blue whales in the North Pacific for several years after the ban. Surveys

conducted in these former-whaling areas in the 1980s and 1990s failed to find any blue whales (K. A. Forney & Brownell Jr., 1996). By 1967, Soviet scientists wrote that blue whales in the North Pacific Ocean (including the eastern Bering Sea and Prince William Sound) had been so overharvested by Soviet whaling fleets that some scientists concluded that any additional harvests were certain to cause the species to become extinct in the North Pacific (Latishev, 2007). As its legacy, whaling has reduced blue whales to a fraction of their historic population size and, as a result, makes it easier for other human activities to push blue whales closer to extinction. Otherwise, whaling currently does not threaten blue whale populations.

In 1980, 1986, 1987, and 1993, ship strikes have been implicated in the deaths of blue whales off California (Barlow, 1997). More recently, Berman-Kowalewski et al. (2010) reported that between 1988 and 2007, 21 blue whale deaths were reported along the California coast, typically one or two cases annually. In addition, several photo-identified blue whales from California waters were observed with large scars on their dorsal areas that may have been caused by ship strikes. Studies have shown that blue whales respond to approaching ships in a variety of ways, depending on the behavior of the animals at the time of approach as well as the speed and direction of the approaching vessel. While feeding, blue whales react less rapidly and with less obvious avoidance behavior than do whales that are not feeding (Richard Sears, 1983). Within the St. Lawrence Estuary, blue whales are believed to be affected by large amounts of recreational and commercial vessel traffic. Blue whales in the St. Lawrence appeared more likely to react to these vessels when boats made fast, erratic approaches or sudden changes in direction or speed (Peggy L. Edds & Macfarlane, 1987).

Although commercial fisheries using large gill nets or other large set gears poses some entanglement risk to marine mammals, there is little direct evidence of blue whale mortality from fishing gears. Therefore it is difficult to estimate the numbers of blue whales killed or injured by gear entanglements. The offshore drift gillnet fishery is the only fishery that is likely to take blue whales from this stock, but no fishery mortalities or serious injuries have been observed. In addition, the injury or mortality of large whales due to interactions or entanglements in fisheries may go unobserved because large whales swim away with a portion of the net or gear. Fishermen have reported that large whales tend to swim through their nets without becoming entangled and cause little damage to nets (James V. Carretta et al., 2008).

4.3.1.5 Status and Trends

Blue whales (including all subspecies) were originally listed as endangered in 1970 (35 FR 18319), and this status continues since the inception of the ESA in 1973. Blue whales are listed as endangered on the IUCN Red List of Threatened Animals (IUCN, 2010). They are also protected by the Convention on International Trade in Endangered Species of wild flora and fauna and the MMPA.

It is difficult to assess the current status of blue whales because (1) there is no general agreement on the size of the blue whale population prior to whaling and (2) estimates of the current size of

the different blue whale populations vary widely. We may never know the size of the blue whale population prior to whaling, although some authors have concluded that their population numbers about 200,000 animals before whaling. Similarly, estimates of the global abundance of blue whales are uncertain. Since the cessation of whaling, the global population of blue whales has been estimated to range from 11,200 to 13,000 animals (Maser, Mate, Franklin, & Dyrness, 1981). These estimates, however, are more than 20 years old. More recently, Jefferson et al. (2008) estimated the global population as 8,000 to 9,000 individuals.

Uncertainty surrounds estimates of blue whale abundance in the North Pacific Ocean as well. Wade and Gerrodette (1993) and Barlow et al. (1997a) estimated there were a minimum of 3,300 blue whales in the North Pacific Ocean in the 1990s, while Barlow (1995) estimated the abundance of blue whales off California at 2,200 individuals. Calambokidis et al. (2010) used photographic mark-recapture techniques to calculate an estimate of 2,497 (CV=0.24) whales.

The information available on the status and trend of blue whales does not allow us to reach any conclusions about the extinction risks facing blue whales as a species or particular populations of blue whales. With the limited data available on blue whales, we do not know whether these whales exist at population sizes large enough to avoid demographic phenomena that are known to increase the extinction probability of species that exist as “small” populations (that is, “small” populations experience phenomena such as demographic stochasticity, inbreeding depression, and Allee effects, among others, that cause their population size to become a threat in and of itself) or if blue whales are threatened more by exogenous threats such as anthropogenic activities (primarily whaling and ship strikes) or natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate).

4.3.1.6 Diving and Social Behavior

Blue whales spend more than 94 percent of their time underwater (Lagerquist, Stafford, & Mate, 2000). Generally, blue whales dive 5-20 times at 12-20 sec intervals before a deep dive of 3-30 min (D. Croll et al., 1999; Stephen Leatherwood, Caldwell, & Winn., 1976; Maser et al., 1981; Yochem & Leatherwood, 1985). Average foraging dives to 140 m deep and averaging for 4 to 15 mi (D. A. Croll, Acevedo-Gutierrez, Tershy, & Urban-Ramirez, 2001). Non-foraging dives are shallower and shorter, averaging 68 m and 4.9 min (D. A. Croll, A. Acevedo-Gutierrez, et al., 2001). However, dives of up to 300 m are known with the longest dive recorded at 36 min (J. Calambokidis et al., 2003). Nighttime dives are generally shallower (50 m).

Blue whales occur singly or in groups of two or three (Aguayo, 1974; Mackintosh, 1965; Takahisa. Nemoto, 1964; Pike & Macaskie, 1969; Ruud, 1956; Slijper, 1962). However, larger foraging aggregations, even with other species such as fin whales, are regularly reported (Fiedler et al., 1998; Schoenherr, 1991). Little is known of the mating behavior of blue whales.

4.3.1.7 Vocalization and Hearing

There is no direct measurement of the hearing sensitivity of blue whales (Darlene R. Ketten, 2000; Thewissen & Williams, 2002). In one of the few studies to date, no change in blue whale vocalization pattern or movements relative to an LFA sonar sound source was observed for received levels of 70 to 85 dB (Aburto, Rountry, & Danzer, 1997). Croll et al. (2001) studied the effects of anthropogenic low-frequency noise on the foraging ecology of blue and fin whales off San Nicolas Island, California and observed no responses or change in foraging behavior that could be attributed to the low-frequency sounds.

Blue whales produce prolonged low-frequency vocalizations that include moans in the range from 12.5-400 Hz, with dominant frequencies from 16-25 Hz, and songs that span frequencies from 16-60 Hz that last up to 36 sec repeated every 1 to 2 min (see Mark A. McDonald, Hildebrand, & Webb, 1995). Berchok et al. (2006) examined vocalizations of St. Lawrence blue whales and found mean peak frequencies ranging from 17.0-78.7 Hz. Reported source levels are 180-188 dB re: 1µPa, but may reach 195 dB re: 1µPa (Aburto et al., 1997; C. W. Clark & Gagnon, 2004; Darlene R. Ketten, 1998; Mark A. McDonald, Calambokidis, Teranishi, & Hildebrand, 2001). Samaran et al. (2010) estimated Antarctic blue whale calls in the Indian Ocean at 179 ± 5 dB re: 1 μPa_{rms} -1 m in the 17-30 Hz range and pygmy blue whale calls at 175± 1 dB re: 1 μPa_{rms} -1 m in the 17-50 Hz range.

As with other baleen whale vocalizations, blue whale vocalization function is unknown, although numerous hypotheses exist (maintaining spacing between individuals, recognition, socialization, navigation, contextual information transmission, and location of prey resources) (Edds-Walton, 1997; R. Payne & Webb., 1971; Paul O. Thompson, Findley, & Vidal., 1992). Intense bouts of long, patterned sounds are common from fall through spring in low latitudes, but these also occur less frequently while in summer high-latitude feeding areas. Short, rapid sequences of 30-90 Hz calls are associated with socialization and may be displays by males based upon call seasonality and structure. The low-frequency sounds produced by blue whales can, in theory, travel long distances, and it is possible that such long-distance communication occurs (Edds-Walton, 1997; R. Payne & Webb., 1971). The long-range sounds may also be used for echolocation in orientation or navigation (Peter L. Tyack, 1999).

Blue whales produce long, patterned hierarchically organized sequences of vocalizations that are characterized as songs. Blue whales produce songs throughout most of the year with a peak period of singing overlapping with the general period of functional breeding. Blue whales also produce a variety of transient sound (i.e., they do not occur in predictable patterns or have much interdependence of probability) in the 30 to 100 Hz band (sometimes referred to as “D” calls). These usually sweep down in frequency or are inflected (up-over-down), occur throughout the year, and are assumed to be associated with socializing when animals are in close proximity (Christopher W. Clark & Ellison, 2004; Mellinger & Clark, 2003).

Cetaceans have an auditory anatomy that follows the basic mammalian pattern, with some modifications to adapt to the demands of hearing in the sea. The typical mammalian ear is divided into the outer ear, middle ear, and inner ear. The outer ear is separated from the inner ear by the tympanic membrane, or eardrum. In terrestrial mammals, the outer ear, eardrum, and middle ear function to transmit airborne sound to the inner ear, where the sound is detected in a fluid. Since cetaceans already live in a fluid medium, they do not require this matching, and thus do not have an air-filled external ear canal. The inner ear is where sound energy is converted into neural signals that are transmitted to the central nervous system via the auditory nerve. Acoustic energy causes the basilar membrane in the cochlea to vibrate. Sensory cells at different positions along the basilar membrane are excited by different frequencies of sound (Peter L. Tyack, 1999). Baleen whales have inner ears that appear to be specialized for low-frequency hearing. In a study of the morphology of the mysticete auditory apparatus, Ketten (1997) hypothesized that large mysticetes have acute infrasonic hearing.

Direct studies of blue whale hearing have not been conducted, but it is assumed that blue whales can hear the same frequencies that they produce (low-frequency) and are likely most sensitive to this frequency range (D. R. Ketten, 1997; W. John Richardson, Charles R. Greene Jr., et al., 1995).

4.3.2 Fin Whale

The fin whale, *Balaenoptera physalus* (Linnæus, 1758), is a well-defined, cosmopolitan species of baleen whale (R. Gambell, 1985). Fin whales are the second-largest whale species by length. Fin whales are long-bodied and slender, with a prominent dorsal fin set about two-thirds of the way back on the body. The streamlined appearance can change during feeding when the pleated throat and chest area becomes distended by the influx of prey and seawater, giving the animal a tadpole-like appearance. The basic body color of the fin whale is dark gray dorsally and white ventrally, but the pigmentation pattern is complex. The lower jaw is gray or black on the left side and creamy white on the right side. This asymmetrical coloration extends to the baleen plates as well, and is reversed on the tongue. Individually distinctive features of pigmentation, along with dorsal fin shapes and body scars, have been used in photo-identification studies (Agler et al., 1990). Fin whales live 70-80 years (Kjeld, 1982).

4.3.2.1 Distribution

Fin whales are distributed widely in every ocean except the Arctic Ocean. They are primarilyfound in temperate and cool waters. Fin whales migrate seasonally between higher latitudes for foraging and lower latitudes for mating and calving. Specific breeding areas are unknown and mating is assumed to occur in pelagic waters, presumably some time during the winter when the whales are in mid-latitudes. Foraging grounds tend to be near coastal upwelling areas and data indicate that some whales remain year round at high latitudes (C. W. Clarke & Charif, 1998).

In the North Atlantic Ocean, fin whales occur in summer foraging areas from the coast of North America to the Arctic, around Greenland, Iceland, northern Norway, Jan Meyers, Spitsbergen, and the Barents Sea. In the western Atlantic, they winter from the edge of sea ice south to the Gulf of Mexico and the West Indies. In the eastern Atlantic, they winter from southern Norway, the Bay of Biscay, and Spain with some whales migrating into the Mediterranean Sea (R. Gambell, 1985).

In the Southern Hemisphere, fin whales are distributed broadly south of 50° S in the summer and migrate into the Atlantic, Indian, and Pacific Oceans in the winter, along the coast of South America (as far north as Peru and Brazil), Africa, and the islands in Oceania north of Australia and New Zealand (R. Gambell, 1985).

Fin whales are common off the Atlantic coast of the United States in waters immediately off the coast seaward to the continental shelf (about the 1,000-fathom contour). In this region, they tend to occur north of Cape Hatteras where they accounted for about 46 percent of the large whales observed in surveys conducted between 1978 and 1982. During the summer months, fin whales in this region tend to congregate in feeding areas between 41°20'N and 51°00'N, from shore seaward to the 1,000-fathom contour. This species preys opportunistically on both invertebrates and fish (Watkins, Moore, Sigujónsson, Wartzok, & di Sciara, 1984). They feed by filtering large volumes of water for the associated prey.

In the North Pacific Ocean, fin whales occur in summer foraging areas in the Chukchi Sea, the Sea of Okhotsk, around the Aleutian Islands, and the Gulf of Alaska; in the eastern Pacific, they occur south to California; in the western Pacific, they occur south to Japan (P. G. H. Evans, 1987). Fin whales in the eastern Pacific winter from California south; in the western Pacific, they winter from the Sea of Japan, the East China, Yellow, and Philippine Seas (R. Gambell, 1985). The overall distribution may be based on prey availability. Fin whales are larger and faster than humpback and right whales and are less concentrated in nearshore environments.

Fin whales also appear to migrate to waters offshore of Washington, Oregon, and northern California to forage. Most fin whales that occur in the action area for this consultation appear to migrate between summer foraging areas and winter rearing areas along the Pacific Coast of the United States, although Moore et al. (1998) recorded fin whale vocalizations in waters off Washington and Oregon throughout the year, with concentrations between September and February, which demonstrates that fin whales are likely to occur in the action area throughout the year. Fin whales are rare in inland waters and are not likely to be found in Puget Sound.

4.3.2.2 Population Structure

Fin whales have two recognized subspecies: *Balaoptera physalus physalus* occurs in the North Atlantic Ocean while *B. p. quoyi* (Fischer, 1829) occurs in the Southern Ocean. Globally, fin whales are sub-divided into three major groups: Atlantic, Pacific, and Antarctic. Within these major areas, different organizations use different population structure.

In the North Atlantic Ocean, the International Whaling Commission recognizes seven management units or "stocks" of fin whales: (1) Nova Scotia, (2) Newfoundland-Labrador, (3) West Greenland, (4) East Greenland-Iceland, (5) North Norway, (6) West Norway-Faroe Islands, and (7) British Isles-Spain-Portugal. In addition, the population of fin whales that resides in the Ligurian Sea, in the northwestern Mediterranean Sea, is believed to be genetically distinct from other fin whale populations.

In the North Pacific Ocean, the International Whaling Commission recognizes two "stocks": (1) East China Sea and (2) rest of the North Pacific (Donovan, 1991). However, Mizroch et al. (1984) concluded that there were five possible "stocks" of fin whales within the North Pacific based on histological analyses and tagging experiments: (1) East and West Pacific that intermingle around the Aleutian Islands; (2) East China Sea; (3) British Columbia; (4) Southern-Central California to Gulf of Alaska; and (5) Gulf of California. Based on genetic analyses, Bérubé et al. (1998) concluded that fin whales in the Sea of Cortez represent an isolated population that has very little genetic exchange with other populations in the North Pacific Ocean (although the geographic distribution of this population and other populations can overlap seasonally).

Regardless of how different authors structure the fin whale population, mark-recapture studies have demonstrated that individual fin whales migrate between management units (Mitchell, 1974; Sigurjonsson, Gunnlaugsson, & Payne., 1989), which suggests that these management units are not geographically isolated populations.

Mizroch et al. (1984) identified five fin whale "feeding aggregations" in the Pacific Ocean: (1) an eastern group that move along the Aleutians, (2) a western group that move along the Aleutians (Berzin & Rovnin, 1966a; Nasu, 1974); (3) an East China Sea group; (4) a group that moves north and south along the west coast of North America between California and the Gulf of Alaska (D. W. Rice, 1974); and (5) a group centered in the Sea of Cortez (Gulf of California).

Hatch (2004) reported that fin whale vocalizations among five regions of the eastern North Pacific were heterogeneous: the Gulf of Alaska, the northeast North Pacific (Washington and British Columbia), the southeast North Pacific (California and northern Baja California), the Gulf of California, and the eastern tropical Pacific.

Sighting data show no evidence of migration between the Sea of Cortez and adjacent areas in the Pacific, but seasonal changes in abundance in the Sea of Cortez suggests that these fin whales might not be isolated (Tershy, Urbán-R, Breese, Rojas-B, & Findley, 1993). Nevertheless, Bérubé et al. (2002) concluded that the Sea of Cortez fin whale population is genetically distinct from the oceanic population and has lower genetic diversity, which suggests that these fin whales might represent an isolated population.

4.3.2.3 Natural Threats

Natural sources and rates of mortality are largely unknown, but Aguilar and Lockyer (1987) suggested annual natural mortality rates might range from 0.04 to 0.06 for northeast Atlantic fin whales. The occurrence of the nematode *Crassicauda boopis* appears to increase the potential for kidney failure and may be preventing some fin whale populations from recovering (Richar H. Lambertsen, 1983). Adult fin whales engage in flight responses (up to 40 km/h) to evade killer whales, which involves high energetic output, but show little resistance if overtaken (Ford & Reeves, 2008). Killer whale or shark attacks may also result in serious injury or death in very young and sick individuals (S. L. Perry et al., 1999).

4.3.2.4 Anthropogenic Threats

Fin whales have undergone significant exploitation but are currently protected under the IWC. Fin whales are still hunted in subsistence fisheries off West Greenland. In 2004, five males and six females were killed, and two other fin whales were struck and lost. In 2003, two males and four females were landed and two others were struck and lost (IWC, 2005). Between 2003 and 2007, the IWC set a catch limit of up to 19 fin whales in this subsistence fishery. However, the scientific recommendation was to limit the number killed to four individuals until accurate populations could be produced (IWC, 2005). Japanese whalers plan to kill 50 whales per year starting in the 2007-2008 season and continuing for the next 12 years (IWC, 2006; S. Nishiwaki et al., 2006).

Fin whales experience significant injury and mortality from fishing gear and ship strikes (J. V. Carretta et al., 2007; Douglas et al., 2008; Lien, 1994; Perkins & Beamish, 1979; G. T. Waring, Josephson, Fairfield, & Maze-Foley, 2007). Between 1969 and 1990, 14 fin whales were captured in coastal fisheries off Newfoundland and Labrador; of these seven are known to have died because of capture (Lien, 1994; Perkins & Beamish, 1979). In 1999, one fin whale was reported killed in the Gulf of Alaska Pollock trawl fishery and one was killed the same year in the offshore drift gillnet fishery (Angliss & Outlaw, 2005; J. V. Carretta & Chivers., 2004). According to Waring et al. (2007), four fin whales in the western North Atlantic died or were seriously injured in fishing gear, while another five were killed or injured as a result of ship strikes between January 2000 and December 2004.

Jensen and Silber (2004) review of the NMFS' ship strike database revealed fin whales as the most frequently confirmed victims of ship strikes (26 percent of the recorded ship strikes [n = 75/292 records]), with most collisions occurring in waters off the U.S. east coast, followed by the west coast of the U.S. and Alaska/Hawaii. Between 1999-2005, there were 15 reports of fin whales strikes by vessels along the U.S. and Canadian Atlantic coasts (Cole, Hartley, & Merrick, 2005; M. Nelson, Garron, Merrick, Pace III, & Cole, 2007). Of these, 13 were confirmed, resulting in the deaths of 11 individuals. Five of seven fin whales stranded along Washington State and Oregon showed evidence of ship strike with incidence increasing since 2002 (Douglas et al., 2008). Similarly, 2.4 percent of living fin whales from the Mediterranean show ship strike

injury and 16 percent of stranded individuals were killed by vessel collision (Panigada et al., 2006). There are also numerous reports of ship strikes off the Atlantic coasts of France and England (A. S. Jensen & Silber, 2004).

Management measures aimed at reducing the risk of ships hitting right whales should also reduce the risk of collisions with fin whales. In the Bay of Fundy, recommendations for slower vessel speeds to avoid right whale ship strike appear to be largely ignored (Angelia S. M. Vanderlaan, Taggart, Serdynska, Kenney, & Brown, 2008). However, new rules for seasonal (June through December) slowing of vessel traffic to 10 knots and changing shipping lanes by less than one nautical mile to avoid the greatest concentrations of right whales are predicted to be capable of reducing ship strike mortality by 27 percent in the Bay of Fundy region.

The organochlorines DDE, DDT, and PCBs have been identified from fin whale blubber, but levels are lower than in toothed whales due to the lower level in the food chain that fin whales feed at (Alex Aguilar & Borrell, 1988; A. Borrell, 1993; A Borrell & Aguilar, 1987; Henry & Best, 1983; Marsili & Focardi, 1996). Females contained lower burdens than males, likely due to mobilization of contaminants during pregnancy and lactation (Alex Aguilar & Borrell, 1988; J. M. Gauthier, Metcalfe, & Sears, 1997). Contaminant levels increase steadily with age until sexual maturity, at which time levels begin to drop in females and continue to increase in males (Alex Aguilar & Borrell, 1988).

Climate change also presents a potential threat to fin whales, particularly in the Mediterranean Sea, where fin whales appear to rely exclusively upon northern krill as a prey source. These krill occupy the southern extent of their range and increases in water temperature could result in their decline and that of fin whales in the Mediterranean Sea (Gambaiani, Mayol, Isaac, & Simmonds, 2009).

4.3.2.5 Status and Trends

Fin whales were originally listed as endangered in 1970 (35 FR 18319), and this status continues since the inception of the ESA in 1973. Although fin whale population structure remains unclear, various abundance estimates are available. Pre-exploitation fin whale abundance is estimated at 464,000 individuals worldwide; the estimate for 1991 was roughly 25 percent of this (Braham, 1991). Historically, worldwide populations were severely depleted by commercial whaling, with more than 700,000 whales harvested in the twentieth century (Cherfas, 1989).

The global population estimate is roughly 140,000 whales (T.A. Jefferson et al., 2008). In the western North Atlantic, there is an estimated 2,269 whales (Gordon T. Waring, Josephson, Fairfield, & (Eds). 2009), while the population estimated for the central and eastern North Atlantic is 30,000 (IWC, 2009). The eastern North Pacific has an estimated 2,636 whales, and Hawaii has an estimated 174 fin whales (J. V. Carretta et al., 2009). The IWC (2009) estimates that 3,200 fin whales exist in West Greenland.

Based on ecological theory and demographic patterns derived from several hundred imperiled species and populations, fin whales appear to exist at population sizes that are large enough to avoid demographic phenomena that are known to increase the extinction probability of species that exist as "small" populations (that is, "small" populations experience phenomena such as demographic stochasticity, inbreeding depression, and Allee effects, among others, that cause their population size to become a threat in and of itself). As a result, we assume that fin whales are likely to be threatened more by exogenous threats such as anthropogenic activities (primarily whaling, entanglement, and ship strikes) or natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate) than endogenous threats caused by the small size of their population.

Nevertheless, based on the evidence available, the number of fin whales that are recorded to have been killed or injured in the past 20 years by human activities or natural phenomena, does not appear to be increasing the extinction probability of fin whales, although it may slow the rate at which they recover from population declines that were caused by commercial whaling.

4.3.2.6 Diving and Social Behavior

The amount of time fin whales spend at the surface varies. Some authors have reported that fin whales make 5-20 shallow dives, each of 13-20 s duration, followed by a deep dive of 1.5-15 min (R. Gambell, 1985; Lafortuna, Jahoda, Azzellino, Saibene, & Colombini., 2003; G. S. Stone, Katona, Mainwaring, Allen, & Corbett., 1992). Other authors have reported that the fin whale's most common dives last 2-6 min (James H. W. Hain, Ratnaswamy, Kenney, & Winn, 1992; Watkins, 1981b). Fin whales dive for a mean duration of 4.2 min at depths averaging 60 m (197 ft) (Donald A. Croll et al., 2001); (Panigada et al., 2006). Maximum dive depths have been recorded deeper than 360 m (1,181 ft) (Charif, Mellinger, Dunsmore, Fristrup, & Clark, 2002). Fin whales forage at dive depths between 100 and 200 m (328 to 656 ft), with foraging dives lasting from 3 to 10 min (A Aguilar, 2002; A. Aguilar & Lockyer, 1987). In waters off the U.S. Atlantic Coast, individuals or duos represented about 75 percent of sightings during the Cetacean and Turtle Assessment Program (James H. W. Hain et al., 1992).

Individuals or groups of less than five individuals represented about 90 percent of the observations. Barlow (2003) reported mean group sizes of 1.1–4.0 during surveys off California, Oregon, and Washington.

4.3.2.7 Vocalization and Hearing

There is no direct measurement of fin whale hearing sensitivity (Darlene R. Ketten, 2000; Thewissen & Williams, 2002). Fin whales produce a variety of LF sounds that range from 10 to 200 Hz (P.L. Edds, 1988; Paul O. Thompson et al., 1992; Watkins, 1981a; Watkins, Tyack, Moore, & Bird, 1987). Short sequences of rapid FM calls from 20 to 70 Hz are associated with animals in social groups (P.L. Edds, 1988; Mark A. McDonald et al., 1995; Watkins, 1981a). The most common fin whale vocalization is what is referred to as the "20-Hz signal", which is a low frequency (18 to 35 Hz) loud and long (0.5 to 1.5 sec) patterned sequence signal (C. W.

Clark, Borsani, & Notarbartolo-di-Sciara, 2002; B. Patterson & Hamilton, 1964; Watkins et al., 1987). The pulse patterns of the 20-Hz signal vary geographically and with seasons (C. W. Clark et al., 2002; Donald A. Croll et al., 2002). Regional differences in vocalization production and structure have been found between the Gulf of California and several Atlantic and Pacific Ocean regions. The 20-Hz signal is common from fall through spring in most regions, but also occurs to a lesser extent during the summer in high-latitude feeding areas (C. W. Clark et al., 2002; C. W. Clarke & Charif, 1998). In the Atlantic region, 20-Hz signals are produced regularly throughout the year. Atlantic fin whales also produce higher frequency downsweeps ranging from 100 to 30 Hz (Adam S. Frankel, 2009). Estimated source levels of the 20-Hz signal are as high as 180 to 190 dB re: 1 µPa @ 1 m (B. Patterson & Hamilton, 1964) (Charif et al., 2002; Donald A. Croll et al., 2002; Mark A. McDonald et al., 1995; Paul O. Thompson et al., 1992; Watkins et al., 1987). Croll et al.(2002) verified the earlier conclusion of Watkins et al. (1987) that the 20-Hz vocalizations are only produced by male fin whales and likely are male breeding displays.

Croll et al. (2001) studied the effects of anthropogenic low-frequency sound with received levels greater than 120 dB on the foraging ecology and vocalizations of blue and fin whales off San Nicolas Island, California. No obvious responses of either whale species was detected that could be attributable to the anthropogenic low-frequency sounds produced by SURTASS LFA sonar (Donald A. Croll et al., 2001).

Although their function is still in doubt, low-frequency fin whale vocalizations travel over long distances and may aid in long-distance communication (Edds-Walton, 1997; R. Payne & Webb., 1971). During the breeding season, fin whales produce pulses in a regular repeating pattern, which have been proposed to be mating displays similar to those of humpbacks (Donald A. Croll et al., 2002). These vocal bouts last for a day or longer (Peter L. Tyack, 1999).

A general description of the anatomy of the ear for cetaceans is provided in the description of the blue whale. Direct studies of fin whale hearing have not been conducted, but it is assumed that fin whales can hear the same frequencies that they produce (low) and are likely most sensitive to this frequency range (D. R. Ketten, 1997; W. John Richardson, Charles R. Greene Jr., et al., 1995).

4.3.3 Western North Pacific Gray Whale

Gray whales have a mottled gray body, with small eyes located just above the corners of the mouth. Their "pectoral fins" (flippers) are broad, paddle-shaped, and pointed at the tips. Lacking a dorsal fin, they instead have a "dorsal hump" located about two-thirds of the way back on the body, and a series of 8-14 small bumps, known as "knuckles," between the dorsal hump and the tail flukes. The tail flukes are more than 15 ft (3 m) wide, have S-shaped trailing edges, and a deep median notch. They are the only species in the family Eschrichtiidae. These large whales can grow to about 50 ft (15 m) long, and weigh approximately 80,000 lb (35,000 kg). Females are slightly larger than males.

4.3.3.1 Distribution

The western Pacific population of gray whales is distributed between the west central Sea of Okhotsk (from summer to fall) and the South China Sea (winter). Gray whales are the most coastal of the baleen whales and the western Pacific population is no exception: whales in this population generally forage in shallow, nearshore waters throughout the year except when crossing open-water passages.

Feeding grounds off northeastern Sakhalin Island, Russia appear to be important for this species during the summer. Their migratory route from these feeding grounds to winter rearing areas include regions off the eastern shore of Sakhalin Island in the Okhotsk Sea, along the eastern shore of mainland Russia near the La Perouse and Tatarskiy Straits (Berzin & Vladimirov, 1990), and off the eastern shore of the Korean peninsula (Roy C Andrews, 1914; Robert L. Brownell, Jr. & Chun, 1977). Data derived from whaling records suggest that gray whale numbers off Korea peaked in two pulses, one between December and January (probably during the southward migration) and the other between March and April (during the northward migration) (Roy C Andrews, 1914; Hidehiro Kato & Kasuya., 2002). In addition, gray whale have been occasionally sighted off the Pacific coast of southern Japan between 1959-1997 (H. Kato & Tokuhiro, 1997; Hideo. Omura, 1984).

Winter calving and mating areas for this population remain unknown. However, some investigators have suggested that the western population of gray whales calves off the southern end of the Korean Peninsula (Roy C Andrews, 1914), although the analyses that led to this conclusion are being debated (Dale W. Rice & Wolman, 1971). Historical records indicate that the western population of gray whales occurred as far south as the Yellow Sea, East China Sea, and South China Sea (David. Henderson, 1990; D. A. Henderson, 1984; Hidehiro Kato & Kasuya., 2002; Hideo Omura, 1988; Wang, 1984), so calving may occur in these areas. In addition, Omura (1974) suggested that an alternative or additional calving and mating area was in the Seto Inland Sea off southern Japan, although this suggestion has not been supported by empirical observation.

4.3.3.2 Population Structure

The gray whale population is divided into two different stocks. Gray whales in the western North Pacific Ocean are genetically distinct from those gray whales occurring in the eastern North Pacific Ocean (LeDuc et al., 2002). The western North Pacific stock was thought to be extinct, but a small group of less than 100 gray whales still remain (T.A. Jefferson et al., 2008). Current IWC abundance estimates report less than 121 animals in the western Pacific stock (IWC, 2009).

4.3.3.3 Natural Threats

Too little is known about the western Pacifc gray whale to determine what natural sources mayb be threatening the health and mortality of this species.

4.3.3.4 Anthropogenic Threats

The decline in the western Pacific gray whale population can be largely attributed to modern commercial whaling off Russia, Korea, and Japan between the 1890s and 1960s. This population has been legally protected under the International Convention for the Regulation of Whaling since 1946. The Republic of Korea and China, however, did not join the International Whaling Commission (IWC) until 1978 and 1980, respectively. Prior to their IWC membership, at least 67 gray whales were killed between 1948-1966 off the Republic of Korea, and the absence of catch reports from 1967 to 1980 does not necessarily indicate the absence of gray whale harvests by either of these countries during that fourteen year period (Robert L. Brownell, Jr. & Chun, 1977). The Democratic People's Republic of Korea, recognized for its long-term involvement in coastal and pelagic whaling operations, is not currently a member of the IWC and nothing is known about this country's whaling activities over the past fifty years.

Current threats to the western gray whale population include continued mortality from an undetermined level of hunting (R. L. Brownell, Jr., Weller, Wursig, Blokhin, & Burdin., 1999), and incidental catches in the extensive coastal net fisheries off southern China (J. Brownell, R. L., 2007). The substantial nearshore industrialization and shipping congestion throughout the migratory corridor(s) of this population also represent potential threats by increasing the likelihood of exposure to chemical pollution and ship strikes. Present and planned large-scale offshore gas and oil development in the South China Sea, and within 20 km of the only known feeding ground for western gray whales off northeast Sakhalin Island in the Okhotsk Sea, is of particular concern (R. L. Brownell, 2004; D. Weller, Burdin, Wursig, Taylor, & Robert L. Brownell, 2002). Activities related to oil and gas exploration, including high-intensity geophysical seismic surveying, drilling operations, increased ship and air traffic, and oil spills, all pose potential threats to gray whales. Disturbance from underwater industrial noise may displace whales from critical feeding, migratory, and breeding habitat (Bryant, Lafferty, & Lafferty., 1984). Physical habitat damage from drilling and dredging operations, combined with possible impacts of oil and chemical spills on benthic prey communities also warrants concern.

Despite international agreements that prohibit harvests of these whales, at least one western gray whale was illegally killed off Hokkaido, Japan, in 1996 (C. S. Baker, Dalebout, Lento, & Funahashi, 2002; R. L. Brownell, Jr. et al., 1999). Baker et al. (2002) report the sale of meat from seven gray whales, whose genetics apparently match the published sequence from Washington State, in Japan in 1999. Based on the results of their investigations, Baker et al. (2002) suggested that illegal hunting along the coast of Japan could be one of the factors inhibiting the recovery of this critically endangered population.

4.3.3.5 Listing Status and Trends

The North Atlantic population of gray whales became extinct as a result of whaling activity during the early 1900s. In the North Pacific, the IWC began management of commercial whaling for gray whales in 1969; gray whales were fully protected from commercial whaling in 1976 (K.

R. Allen, 1980). North Pacific gray whales were listed as endangered under the ESA in 1973. The eastern Pacific population of the gray whale was removed from the list of threatened and endangered species in 1994 (59 FR 31094). However, the western Pacific population of gray whales remains protected as an endangered species under the ESA. Critical habitat has not been designated for the western Pacific population of gray whales.

The IUCN lists this gray whale population as critically endangered. Gray whales are also protected by the Convention on International Trade in Endangered Species of wild flora and fauna and the MMPA. Gray whales were listed as critically endangered by IUCN in 2000.

The western Pacific population of gray whales was thought to be extinct as recently as 1972 (S. L. Bowen, 1974), but is known to survive today as a small remnant (Robert L. Brownell, Jr. & Chun, 1977; D. W. Weller et al., 1999). Aerial and ship-based sighting records in the Okhotsk Sea between 1979 and 1989 indicated that gray whales aggregated predominantly along the shallow-water shelf of northeastern Sakhalin Island, Russia, and were most common off the southern portion of Piltun Lagoon (Berzin & Vladimirov, 1989, 1990; Blokhin, 1985).

Photo-identification studies (1994-1995 and 1997-1999) on the primary feeding grounds off northeastern Sakhalin Island have identified a total of 88 individual whales (D. W. Weller et al., 2002). These photo-identification data indicate high levels of annual return and pronounced seasonal site fidelity. While new individuals continue to be identified annually, the rate at which this is occurring is near the asymptote. Only 20 previously unidentified whales (excluding calves) were photographed during 91 days of effort between 1998 and 1999. This finding suggests that a majority of the population had been identified between 1994 and 1997 (D. W. Weller et al., 2002). Between 1995 and 1999, 11 reproductive females and their 13 calves were observed (R. L. Brownell & Weller., 2002; D. W. Weller et al., 2002; D. W. Weller et al., 1999). Two calves were observed in 1995, 1997, and 1999, and seven calves were sighted in 1998. Crude birth rates ranged between a low of 2.8 percent in 1999 to a high of 13.0 percent in 1998. Of the 11 calves identified between 1995 and 1998, seven (63.3 percent) have not been resighted on the Sakhalin feeding grounds subsequent to their birth year.

Using the minimum estimate of 88 western gray whales (D. W. Weller et al., 2002; D. W. Weller et al., 1999) and using the proportion of immature from Rice and Wolman (Dale W. Rice & Wolman, 1971), Brownell et al. (1999) estimated the number of mature whales in the western subpopulation at between 34 and 49. After assuming that all males in the population reproduced but only 70 percent of the females reproduced, Brownell et al. (1999) estimated that 85 percent of the sexually mature animals are capable of reproduction. With this assumption, they concluded that the western population of gray whales may only contain 50 breeding adults, making them one of the most critically-endangered cetaceans in the world.

4.3.3.6 Diving and Social Behavior

Gray whales feed primarily on benthic amphipods (Dale W. Rice & Wolman, 1971) by sucking sediment filled with amphipods off the sea floor and expelling the sediment and water through their baleen plates (Nerini, 1984). In addition, gray whales will feed on other invertebrates that include mysiids, euphausiids, and pelagic crabs (Randall R. Reeves & Mitchell., 1988). Most feeding takes place on their northern feeding grounds, although Nerini (1984) reported evidence of extensive feeding during migration.

Gray whales are not deep divers. While foraging, they will generally remain in waters less than 80 m in depth. Their average dive times are between 4 and 5 minutes.

4.3.3.7 Vocalizations and Hearing

There are sparse data on the hearing sensitivity of gray whales. Dahlheim and Ljungblad (1990) suggest that free-ranging gray whales are most sensitive to tones between 800 and 1,500 Hz. Migrating gray whales showed avoidance responses at ranges of several hundred meters to LF playback source levels of 170 to 178 dB when the source was placed within their migration path at about 2 km (1.1 nmi) from shore. However, this response did not occur when the source was moved out of their migration path but occurred when the source level increased to duplicate the animals' received level within their migration corridor (Christopher W. Clark, Tyack, & Ellison., 1999).

Gray whales produce a variety of sounds from about 100 Hz, potentially up to 12 kHz (M. L. Jones & Swartz., 2009). The most common sounds recorded during foraging and breeding are knocks and pulses in frequencies from <100 Hz to 2 kHz, with most energy concentrated at 327 to 825 Hz (W. John Richardson, Charles R. Greene Jr., et al., 1995). Tonal moans are produced during migration in frequencies ranging between 100 and 200 Hz (M. L. Jones & Swartz., 2009). Combinations of clicks and grunts have also been recorded from migrating gray whales in frequencies ranging below 100 Hz to above 10 kHz (Adam S. Frankel, 2009). The seasonal variation in the sound production is correlated with the different ecological functions and behaviors of the gray whale. Whales make the least amount of sound when dispersed on the feeding grounds and are most vocal on the breeding-calving ground. The source levels for these sounds range between 167 and 188 dB (Adam S. Frankel, 2009).

Moore and Clarke (2002) reviewed information on how offshore oil and gas activities, commercial fishing and vessel traffic, and whale watching and scientific research affected gray whales. The underwater noise sources played during these experiments included helicopter over flights, drill ship operations, drilling and production platforms, a semi-submersible drilling rig, and tripping operations. Malme et al. (1983, 1984; 1989) also conducted experiments using air gun arrays and single air guns. The gray whales' responses to the noise playback experiments and air gun shots include changes in swimming speed and changes in direction (away from the sound sources) (Malme et al., 1984). Changes in feeding with a resumption of feeding after

exposure, changes in call rates and structure, and changes in surface behavior were also observed (Marilyn E. Dahlheim, 1987) (Malme et al., 1989; Sue E. Moore & Clark, 2002).

A general description of the anatomy of the ear for cetaceans is provided in the description of the blue whale.

4.3.4 Humpback Whale

Humpback whales (*Megaptera novaeangliae*) are distinguished from other whales in the same Family (Balaenopteridae) by extraordinarily long flippers (up to 5 m or about 1/3 total body length), a more robust body, fewer throat grooves (14-35), more variable dorsal fin, and utilization of very long (up to 30 min.), complex, repetitive vocalizations (songs) (R. S. Payne & McVay, 1971) during courtship. Their grayish-black baleen plates, approximately 270-440 on each side of the jaw, are intermediate in length (6570 cm) to those of other baleen whales. Humpbacks in different geographical areas vary somewhat in body length but maximum recorded size is 18m (Winn & Reichley, 1985).

The whales are generally dark on the back, but the flippers, sides and ventral surface of the body and flukes may have substantial areas of natural white pigmentation plus acquired scars (white or black). Researchers distinguish individual humpbacks by the apparently unique black and white patterns on the underside of the flukes as well as other individually variable features (Glockner & Venus, 1983; Katona & Whitehead, 1981; Kaufman & Osmond, 1987).

4.3.4.1 Distribution

Humpback whales are a cosmopolitan species that occur in the Atlantic, Indian, Pacific, and Southern oceans. Humpback whales migrate seasonally between warmer, tropical or sub-tropical waters in winter months (where they breed and give birth to calves, although feeding occasionally occurs) and cooler, temperate or sub-Arctic waters in summer months (where they feed). In both regions, humpback whales tend to occupy shallow, coastal waters. However, migrations are undertaken through deep, pelagic waters (Winn & Reichley, 1985).

In the North Pacific Ocean, the summer range of humpback whales includes coastal and inland waters from Point Conception, California, north to the Gulf of Alaska and the Bering Sea, and west along the Aleutian Islands to the Kamchatka Peninsula and into the Sea of Okhotsk (Tomlin 1967, Nemoto 1957, Johnson and Wolman 1984 as cited in NMFS, 1991a). These whales migrate to Hawaii, southern Japan, the Mariana Islands, and Mexico during the winter. In the North Pacific, there are breeding grounds around the Mariana Islands, Bonin, Ogasawara, Okinawa, Ryukyu Island, and Taiwan (P. J. Clapham et al., 2009). In the eastern North Pacific, breeding grounds occur around the Hawaiian Islands, off the tip of Baja California, and off the Revillagigedo Islands (P. J. Clapham et al., 2009). In the western North Pacific during winter and early spring, humpback whale distribution is centered along the Ogasawara Islands, Ryukyu Islands, Taiwan, the Philippines, and the Mariana Islands (John Calambokidis et al., 2008). The remainder of the year, humpback whales are largely absent from these regions as they move

northward to other regions of the North Pacific to feed, principally off Russia but also to the Bering Sea and the Gulf of Alaska (John Calambokidis et al., 2008).

The Central North Pacific stock of humpback whales is identified as individuals that migrate from summer/fall feeding grounds of northern British Columbia and southeast Alaska (Prince William Sound west to Kodiak), to winter/spring breeding and calving grounds of the Hawaiian Islands (J. V. Carretta et al., 2013). Some exchange between winter/spring areas has been documented, as well as movement between Japan and British Columbia, and Japan and the Kodiak Archipelago (John Calambokidis et al., 1997b). Acoustic surveys suggest a northbound migration heading of approximately magnetic north (10° true), with a "migration corridor" of 150° to 160°W (T. F. Norris, McDonald, & Barlow, 1999) and a winter presence in the Northwestern Hawaiian Islands (Lammers et al., 2011). Animals are cycling through the breeding grounds with an average residency of approximately 30 to 45 days.

4.3.4.2 Population Structure

Descriptions of the population structure of humpback whales differ depending on whether an author focuses on where humpback whales winter or where they feed. During winter months in northern or southern hemispheres, adult humpback whales migrate to specific areas in warmer, tropical waters to reproduce and give birth to calves. During summer months, humpback whales migrate to specific areas in northern temperate or sub-arctic waters to forage. In summer months, humpback whales from different "reproductive areas" will congregate to feed; in the winter months, whales will migrate from different foraging areas to a single wintering area. In either case, humpback whales appear to form "open" populations; that is, populations that are connected through the movement of individual animals.

In the North Pacific NMFS currently recognizes four stocks, likely corresponding to populations, of humpback whales in the North Pacific Ocean: two in the eastern North Pacific, one in the central North Pacific, and one in the western Pacific (P. S. Hill & DeMaster, 1998b) based on genetic and photo-identification studies. However, gene flow between them may exist. Humpback whales summer in coastal and inland waters from Point Conception, California, north to the Gulf of Alaska and the Bering Sea, and west along the Aleutian Islands to the Kamchatka Peninsula and into the Sea of Okhotsk (J. H. Johnson & Wolman, 1984; T. Nemoto, 1957; Tomilin, 1967). These whales migrate to Hawaii, southern Japan, the Mariana Islands, and Mexico during winter. However, more northerly penetrations in Arctic waters occur on occasion (Hashagen, Green, & Adams., 2009). The central North Pacific population winters in the waters around Hawaii while the eastern North Pacific population (also called the California-Oregon-Washington-Mexico stock) winters along Central America and Mexico. However, Calambokidis et al. (1997a) identified individuals from several populations wintering (and potentially breeding) in the areas of other populations, highlighting the potential fluidity of population structure. Herman (1979) presented extensive evidence that humpback whales associated with the main Hawaiian Islands immigrated there only in the past 200 years. Winn and Reichley (1985)

identified genetic exchange between the humpback whales that winter off Hawaii and Mexico (with further mixing on feeding areas in Alaska) and suggested that humpback whales that winter in Hawaii may have emigrated from Mexican wintering areas. A "population" of humpback whales winters in the South China Sea east through the Philippines, Ryukyu Retto, Ogasawara Gunto, Mariana Islands, and Marshall Islands, with occurrence in the Mariana Islands, at Guam, Rota, and Saipan from January-March (Darling & Cerchio, 1993; Eldredge, 1991, 2003; Dale W. Rice, 1998). During summer, whales from this population migrate to the Kuril Islands, Bering Sea, Aleutian Islands, Kodiak, Southeast Alaska, and British Columbia to feed (Angliss & Outlaw, 2008; John. Calambokidis, 1997; J. Calambokidis et al., 2001).

Separate feeding groups of humpback whales are thought to inhabit western U.S. and Canadian waters, with the boundary between them located roughly at the U.S./Canadian border. The southern feeding ground ranges between 32°-48°N, with limited interchange with areas north of Washington State (John Calambokidis, Steiger, Ellifrit, Troutman, & Bowlby, 2004; J. Calambokidis et al., 1996). Humpback whales feed along the coasts of Oregon and Washington from May-November, with peak numbers reported May-September, when they are the most commonly reported large cetacean in the region (John Calambokidis & Chandler., 2000; John Calambokidis et al., 2004; Dohl, 1983; G. A. Green et al., 1992). Off Washington State, humpback whales concentrate between Juan de Fuca Canyon and the outer edge of the shelf break in a region called "the Prairie," near Barkley and Nitinat canyons, in the Blanco upwelling zone, and near Swiftsure Bank (John Calambokidis et al., 2004). Humpback whales also tend to congregate near Heceta Bank off the coast of Oregon (G. A. Green et al., 1992). Additional data suggest that further subdivisions in feeding groups may exist, with up to six feeding groups present between Kamchatka and southern California (Witteveen, Worthy, Wynne, & Roth., 2009).

4.3.4.3 Natural Threats

Natural sources and rates of mortality of humpback whales are not well known. Based upon prevalence of tooth marks, attacks by killer whales appear to be highest among humpback whales migrating between Mexico and California, although populations throughout the Pacific Ocean appear to be targeted to some degree (Steiger et al., 2008). Juveniles appear to be the primary age group targeted. Humpback whales engage in grouping behavior, flailing tails, and rolling extensively to fight off attacks. Calves remain protected near mothers or within a group and lone calves have been known to be protected by presumably unrelated adults when confronted with attack (Ford & Reeves, 2008).

Parasites and biotoxins from red-tide blooms are other potential causes of mortality (S. L. Perry et al., 1999). The occurrence of the nematode *Crassicauda boopis* appears to increase the potential for kidney failure in humpback whales and may be preventing some populations from recovering (Richard H. Lambertsen, 1992). Studies of 14 humpback whales that stranded along

Cape Cod between November 1987 and January 1988 indicate they apparently died from a toxin produced by dinoflagellates during this period.

4.3.4.4 Anthropogenic Threats

Three human activities are known to threaten humpback whales: whaling, commercial fishing, and shipping. Historically, whaling represented the greatest threat to every population of whales and was ultimately responsible for listing several species as endangered.

Humpback whales are also killed or injured during interactions with commercial fishing gear. Like fin whales, humpback whales have been entangled by fishing gear off Newfoundland and Labrador, Canada. A total of 595 humpback whales were reported captured in coastal fisheries in those two provinces between 1969 and 1990, of which 94 died (Lien, 1994; Perkins & Beamish, 1979). Along the Atlantic coast of the U.S. and the Maritime Provinces of Canada, there were 160 reports of humpback whales being entangled in fishing gear between 1999 and 2005 (Cole et al., 2005; M. Nelson et al., 2007). Of these, 95 entangled humpback whales were confirmed, with 11 whales sustaining injuries and nine dying of their wounds. NMFS estimates that between 2002 and 2006, there were incidental serious injuries to 0.2 humpback annually in the Bering Sea/Aleutian Islands sablefish longline fishery. This estimation is not considered reliable. Observers have not been assigned to a number of fisheries known to interact with the Central and Western North Pacific stocks of humpback whale. In addition, the Canadian observation program is also limited and uncertain (Angliss & Allen, 2009).

More humpback whales are killed in collisions with ships than any other whale species except fin whales (A.S. Jensen & Silber, 2003). Along the Pacific coast, a humpback whale is known to be killed about every other year by ship strikes (Barlow et al., 1997a). Of 123 humpback whales that stranded along the Atlantic coast of the U.S. between 1975 and 1996, 10 (8.1 percent) showed evidence of collisions with ships (Laist, Knowlton, Mead, Collet, & Podesta, 2001). Between 1999 and 2005, there were 18 reports of humpback whales being struck by vessels along the Atlantic coast of the U.S. and the Maritime Provinces of Canada (Cole et al., 2005; M. Nelson et al., 2007). Of these reports, 13 were confirmed as ship strikes and in seven cases, ship strike was determined to be the cause of death. In the Bay of Fundy, recommendations for slower vessel speeds to avoid right whale ship strike appear to be largely ignored (Angelia S. M. Vanderlaan et al., 2008). However, new rules for seasonal (June through December) slowing of vessel traffic to 10 knots and changing shipping lanes by less than one nautical mile to avoid the greatest concentrations of right whales are expected to reduce the chance of humpback whales being hit by ships by 9 percent.

Organochlorines, including PCB and DDT, have been identified from humpback whale blubber (J. M. Gauthier et al., 1997). Higher PCB levels have been observed in Atlantic waters versus Pacific waters along the United States and levels tend to increase with individual age (Elfes et al., 2010). Although humpback whales in the Gulf of Maine and off Southern California tend to have the highest PCB concentrations, overall levels are on par with other baleen whales, which

are generally lower than odontocete cetaceans (Elfes et al., 2010). As with blue whales, these contaminants are transferred to young through the placenta, leaving newborns with contaminant loads equal to that of mothers before bioaccumulating additional contaminants during life and passing the additional burden to the next generation (Metcalfe, Koenig, Metcalfe, Paterson, & Sears, 2004). Contaminant levels are relatively high in humpback whales as compared to blue whales. Humpback whales feed higher on the food chain, where prey carries higher contaminant loads than the krill that blue whales feed on.

4.3.4.5 Status and Trends

Humpback whales were originally listed as endangered in 1970 (35 FR 18319), and this status remains under the ESA. Recent research conducted by the SLASH consortium of scientists in the North Pacific Ocean has shown that humpback whale movement patterns between feeding areas in high latitudes and wintering grounds in lower latitudes are extremely complex but are indicative of a high level of population structure (John Calambokidis et al., 2008).

In the North Pacific the pre-exploitation population size may have been as many as 15,000 humpback whales (John Calambokidis, Barlow, Ford, Chandler, & Douglas., 2009; D. W. Rice, 1978). It is estimated that 15,000 humpback whales resided in the North Pacific in 1905 (D. W. Rice, 1978). However, from 1905 to 1965, nearly 28,000 humpback whales were harvested in whaling operations, reducing the number of all North Pacific humpback whale to roughly 1,000 (S. L. Perry et al., 1999). Population estimates have risen over time from 1,407-2,100 in the 1980s to 6,010 in 1997 (C. Scott Baker, 1985; C. Scott Baker & Herman, 1987; John Calambokidis et al., 1997a; Darling & Morowitz, 1986).

Based on surveys between 2004 and 2006, Calambokidis et al. (2008) estimated that the number of humpback whales in the North Pacific consisted of about 18,302 whales, not counting calves. Calambokidis et al. (2008) also derived an average abundance for the Asian wintering grounds as 1,107 humpback whales and for the Central North Pacific stock of humpback whales as10,103 individuals, which is a much higher estimate than former surveys and research provided. Tentative estimates of the eastern North Pacific stock suggest an increase of 6-7 percent annually, but fluctuations have included negative growth in the recent past (Angliss & Outlaw, 2005).

4.3.4.6 Diving and Social Behavior

Maximum diving depths are approximately 170 m, with a very deep dive (240 m) recorded off Bermuda (Hamilton, Stone, & Martin., 1997). Dives can last for up to 21 min, although feeding dives ranged from 2.1-5.1 min in the north Atlantic (Dolphin, 1987). In southeast Alaska, average dive times were 2.8 min for feeding whales, 3.0 min for non-feeding whales, and 4.3 min for resting whales (Dolphin, 1987). The deepest recorded humpback dive was 240 m (790 ft), with most dives between 60 and 120 m (197 to 394 ft) (Hamilton et al., 1997). Because most humpback prey is likely found within 300 m of the surface, most humpback dives are probably relatively shallow. In Alaska, capelin are the primary prey of humpback and are found primarily

between 92 and 120 m; depths to which humpbacks apparently dive for foraging (Witteveen, Foy, Wynne, & Tremblay, 2008).

During the feeding season, humpback whales form small groups that occasionally aggregate on concentrations of food that may be stable for long-periods of times. Humpbacks use a wide variety of behaviors to feed on various small, schooling prey including krill and fish (James H. W. Hain, Carter, Kraus, Mayo, & Winn., 1982; J. H. W. Hain et al., 1995; Jurasz & Jurasz, 1979; Weinrich et al., 1992). There is good evidence of some territoriality on feeding and calving areas (P. J. Clapham, 1994, 1996; Peter. Tyack, 1981). Humpback whales are generally believed to fast while migrating and on breeding grounds, but some individuals apparently feed while in low-latitude waters normally believed to be used exclusively for reproduction and calf-rearing (Danilewicz, Tavares, Moreno, Ott, & Trigo., 2009; Pinto De Sa Alves, Andriolo, Zerbini, Pizzorno, & Clapham., 2009). Some individuals, such as juveniles, may not undertake migrations at all (Findlay & Best, 1995).

Humpback whales feed on pelagic schooling euphausiids and small fish including capelin, herring and mackerel. Like other large mysticetes, they are a “lunge feeder” taking advantage of dense prey patches and engulfing as much food as possible in a single gulp. They also blow nets, or curtains, of bubbles around or below prey patches to concentrate the prey in one area, then lunge with open mouths through the middle. Dives appear to be closely correlated with the depths of prey patches, which vary from location to location. In the north Pacific (southeast Alaska), most dives were of fairly short duration (<4 min) with the deepest dive to 148 m (Dolphin, 1987), while whales observed feeding on Stellwagen Bank in the North Atlantic dove to <40 m (J. H. W. Hain et al., 1995). Hamilton et al. (1997) tracked one possibly feeding whale near Bermuda to 240 m depth.

4.3.4.7 Vocalization and Hearing

Humpback whale vocalization is much better understood than is hearing. Different sounds are produced that correspond to different functions: feeding, breeding, and other social calls (Dunlop, Cato, & Noad, 2008). Males sing complex sounds while in low-latitude breeding areas in a frequency range of 20 Hz to 4 kHz with estimated source levels from 144-174 dB (Whitlow W. L. Au et al., 2006; W.W.L Au, Popper, & Fay, 2000; L. N. Frazer & Mercado III, 2000; W. John Richardson, Charles R. Greene Jr., et al., 1995; Winn, Perkins, & Poulter, 1970). Males also produce sounds associated with aggression, which are generally characterized as frequencies between 50 Hz to 10 kHz and having most energy below 3 kHz (Silber, 1986; Peter Tyack, 1983). Such sounds can be heard up to 9 km away (Peter Tyack, 1983). Other social sounds from 50 Hz to 10 kHz (most energy below 3 kHz) are also produced in breeding areas (W. John Richardson, Charles R. Greene Jr., et al., 1995; Peter Tyack, 1983). While in northern feeding areas, both sexes vocalize in grunts (25 Hz to 1.9 kHz), pulses (25-89 Hz), and songs (ranging from 30 Hz to 8 kHz but dominant frequencies of 120 Hz to 4 kHz) which can be very loud (175-192 dB re: 1 μPa at 1 m); (W.W.L Au et al., 2000; Christine. Erbe, 2002; K. Payne, 1985; W.

John Richardson, Charles R. Greene Jr., et al., 1995; Paul O. Thompson, Cummings, & Ha., 1986). However, humpbacks tend to be less vocal in northern feeding areas than in southern breeding areas (W. John Richardson, Charles R. Greene Jr., et al., 1995).

A general description of the anatomy of the ear for cetaceans is provided in the description of the blue whale. Direct studies of humpback whale hearing have not been conducted, but it is assumed that blue whales can hear the same frequencies that they produce (low-frequency) and are likely most sensitive to this frequency range (D. R. Ketten, 1997; W. John Richardson, Charles R. Greene Jr., et al., 1995).

4.3.5 North Pacific Right Whale

Right whales are large baleen whales. Adults are generally between 13.7 and 16.7 m (45 and 55 feet) in length and can weigh up to 70 tons (140,000 lbs; 63,502 kg). Females are larger than males. Calves are 3.9-4.6 m (13-15 feet) in length at birth.

Distinguishing features for right whales include a stocky body, generally black coloration (although some individuals have white patches on their undersides), lack of a dorsal fin, a large head (about 1/4 of the body length), strongly bowed margin of the lower lip, and callosities (raised patches of roughened skin) on the head region. Two rows of long (up to eight feet in length) dark baleen plates hang from the upper jaw, with about 225 plates on each side. The tail is broad, deeply notched, and all black with a smooth trailing edge.

4.3.5.1 Distribution

Little is known of the distribution of right whales in the North Pacific and very few of these animals have been seen in the past 20 years. Historical whaling records indicate that right whales ranged across the North Pacific north of 30° N latitude and occasionally as far south as 20° N, with a bimodal distribution longitudinally favoring the eastern and western North Pacific and occurring infrequently in the central North Pacific (Edward J. Gregr & Coyle., 2009; Josephson, Smith, & Reeves, 2008; Maury, 1853; J.E. Scarff, 1986; J. E. Scarff, 1991; Townsend, 1935). North Pacific right whales summered in the North Pacific and southern Bering Sea from April or May to September, with a peak in sightings in coastal waters of Alaska in June and July (Klumov, 1962; Maury, 1852; H. Omura, 1958; H. Omura, Ohsumi, Nemoto, & Kasuya, 1969; Townsend, 1935). North Pacific right whale summer range extended north of the Bering Strait (H. Omura et al., 1969). However, they were particularly abundant in the Gulf of Alaska from 145° to 151°W, and apparently concentrated in the Gulf of Alaska, especially south of Kodiak Islands and in eastern Aleutian Islands and southern Bering Sea waters (Berzin & Rovnin, 1966b; Braham & Rice, 1984).

Current information on the seasonal distribution of right whales is spotty. In the eastern North Pacific, this includes sightings over the middle shelf of the Bering Sea, Bristol Bay, Aleutian and Pribilof Islands (P. C. Goddard & D. J. Rugh, 1998; P. S. Hill & DeMaster, 1998a; Perryman, LeDuc, & R. L. Brownell, 1999; P. M. Wade et al., 2006; J. M. Waite, Wynne, & Mellinger,

2003). Some more southerly records also record occurrence along Hawaii, California, Washington, and British Columbia (L. M. Herman, Baker, Forestell, & Antinoja, 1980; J.E. Scarff, 1986). However, records from Mexico and California may suggest historical wintering grounds in offshore southern North Pacific latitudes (J. Brownell, Robert L., Clapham, Miyashita, & Kasuya., 2001; Edward J. Gregr & Coyle., 2009).

4.3.5.2 Population Structure

North Pacific right whales appear to exist as a single population that occurs in the North Pacific Ocean.

4.3.5.3 Natural Threats

Right whales have been subjects of killer whale attacks and, because of their robust size and slow swimming speed, tend to fight killer whales when confronted (Ford & Reeves, 2008). Similarly, mortality or debilitation from disease and red tide events are not known, but have the potential to be significant problems in the recovery of right whales because of their small population size.

4.3.5.4 Anthropogenic Threats

Whaling for North Pacific right whales was discontinued in 1966 with the IWC whaling moratorium. However, North Pacific right whales remain at high risk of extinction. Demographic stressors include but are not limited to the following: (1) life history characteristics such as slow growth rate, long calving intervals, and longevity; (2) distorted age structure of the population and reduced reproductive success; (3) strong depensatory or Allee effects; (4) habitat specificity or site fidelity; and (5) habitat sensitivity. The proximity of the known right whale habitats to shipping lanes (e.g. Unimak Pass) suggests that collisions with vessels may also represent a threat to North Pacific right whales (Elvin & Hogart, 2008).

Climate change may have a dramatic affect on survival of North Pacific right whales. Right whale life history characteristics make them very slow to adapt to rapid changes in their habitat (see Reynolds, DeMaster, & Silber, 2002). They are also feeding specialists that require exceptionally high densities of their prey (see M. F. Baumgartner, Cole, Clapham, & Mate, 2003; M. F. Baumgartner & Mate, 2003). Zooplankton abundance and density in the Bering Sea has been shown to be highly variable, affected by climate, weather, and ocean processes and in particular ice extent (Baier & Napp, 2003; Napp & G.L. Hunt, 2001). The largest concentrations of copepods occurred in years with the greatest southern extent of sea ice (Baier & Napp, 2003). It is possible that changes in ice extent, density, and persistence may alter the dynamics of the Bering Sea shelf zooplankton community and in turn affect the foraging behavior and success of right whales. No data are available for the western North Pacific.

4.3.5.5 Status and Trends

The Northern right whale was originally listed as endangered in 1970 (35 FR 18319), and this status remained since the inception of the ESA in 1973. The early listing included both the North

Atlantic and the North Pacific populations, although subsequent genetic studies conducted by Rosenbaum (2000) resulted in strong evidence that the North Atlantic and North Pacific right whales are separate species. Following a comprehensive status review, NMFS concluded that Northern right whales are indeed two separate species. In March 2008, NMFS published a final rule listing North Pacific and North Atlantic right whales as separate species (73 FR 12024).

Very little is known about right whales in the eastern North Pacific, which were severely depleted by commercial whaling in the 1800s (R. L. J. Brownell, Clapham, Miyashita, & Kasuya, 2001). At least 11,500 individuals were taken by American whalers in the early- to mid-19th century, but harvesting continued into the 20th century (Peter B. Best, 1987). Illegal Soviet whaling took 372 individuals between 1963 and 1967 (J. Brownell, Robert L. et al., 2001). In the last several decades there have been markedly fewer sightings due to a drastic reduction in number, caused by illegal Soviet whaling in the 1960s (Doroshenko, 2000). The current population size of right whales in the North Pacific is likely fewer than 1,000 animals compared to possibly 11,000 individuals or more prior to exploitation (NMFS, 2006d) (NMFS, 1991b). Wade et al. (2011) used mark-recapture and genotyping methodologies that produced estimates of 31 and 28 individuals, respectively, for individuals in the Bering Sea (likely representing all individuals from the eastern North Pacific).

Abundance estimates and other vital rate indices in both the eastern and western North Pacific are not well established. Where such estimates exist, they have very wide confidence limits. Previous estimates of the size of the right whale population in the Pacific Ocean range from a low of 100-200 to a high of 220-500 (Berzin & Yablokov, 1978; Braham & Rice, 1984). Although Hill and DeMaster (1998a) argued that it is not possible to reliably estimate the population size or trends of right whales in the North Pacific, Reeves et al. (2003) and Brownell et al. (2001) concluded that North Pacific right whales in the eastern Pacific Ocean exist as a small population of individuals while the western population of right whales probably consists of several hundred animals, although Clapham et al. (2005) placed this population at likely under 100 individuals (Wade et al. (2010) estimated 25-38 individuals). Brownell et al. (2001) reviewed sighting records and also estimated that the abundance of right whales in the western North Pacific was likely in the low hundreds.

Scientists participating in a recent study utilizing acoustic detection and satellite tracking identified 17 right whales (10 males and 7 females) in the Bearing Sea, which is almost threefold the number seen in any previous year in the last four decades (P. M. Wade et al., 2006). These sightings increased the number of individual North Pacific right whales identified in the genetic catalog for the eastern Bering Sea to 23. Amidst the uncertainty of the eastern North Pacific right whale's future, the discovery of females and calves gives hope that this endangered population may still possess the capacity to recover (P. M. Wade et al., 2006). Available age composition of the North Pacific right whale population indicates most individuals are adults (R. D. Kenney, 2002). Length measurements for two whales observed off California suggest at least one of these

whales was not yet sexually mature and two calves have been observed in the Bering Sea (J. V. Carretta, Lynn, & LeDuc, 1994; P. M. Wade et al., 2006). However, to date, there is no evidence of reproductive success (i.e., young reared to independence) in the eastern North Pacific. No data are available for the western North Pacific.

4.3.5.6 Diving and Social Behavior

Right whales dive as deep as 306 meters (Mate, Nieukirk, Mesecar, & Martin., 1992). In the Great South Channel, average diving durations were almost 2 minutes with depths averaging 7.3 meters and reaching a maximum of 85.3 meters (Winn, Goodyear, Kenney, & Petricig, 1995). In the U.S. Outer Continental Shelf the average diving durations were about 7 minutes (CETAP, 1982).

In the North Pacific Ocean, most recent sightings have been of single animals or pairs; however, groups numbering six to ten individuals have been sighted in the northeastern Pacific Ocean (P. D. Goddard & D. J. Rugh, 1998).

4.3.5.7 Vocalizations and Hearing

A general description of the anatomy of the ear for cetaceans is provided in the description of the blue whale above. Limited data indicate that northern right whales produce moans of less than 400 Hz in frequency (Spero, 1981; T. J. Thompson, Winn, & Perkins., 1979; Watkins & Schevill, 1972). Right whales appear to use low frequency sounds as contact calls while summering in the Bay of Fundy (Spero, 1981). A general description of the anatomy of the ear for cetaceans is provided in the description of the blue whale.

4.3.6 Sei Whale

Sei whales are members of the baleen whale family and are considered one of the "great whales" or rorquals. These large animals can reach lengths of about 40-60 ft (12-18 m) and weigh 100,000 lbs (45,000 kg). Females may be slightly longer than males. Sei whales have a long, sleek body that is dark bluish-gray to black in color and pale underneath. The body is often covered in oval-shaped scars (probably caused from cookie-cutter shark and lamprey bites) and sometimes has subtle "mottling". This species has an erect "falcate", "dorsal" fin located far down (about two-thirds) the animals back. The Sei is regarded as the fastest swimmer among the great whales, reaching bursts of speed in excess of 20 knots. . Sei whales have an estimated lifespan of 50-70 years.

4.3.6.1 Distribution

The sei whale occurs in all oceans of the world except the Arctic. The migratory pattern of this species is thought to encompass long distances from high-latitude feeding areas in summer to low-latitude breeding areas in winter; however, the location of winter areas remains largely unknown (S. L. Perry et al., 1999). Sei whales are often associated with deeper waters and areas along continental shelf edges (James H. W. Hain, Hyman, Kenney, & Winn, 1985). This general offshore pattern is disrupted during occasional incursions into shallower inshore waters (Gordon

T. Waring, Pace, Quintal, Fairfield, & Maze-Foley, 2004). The species appears to lack a well-defined social structure and individuals are usually found alone or in small groups of up to six whales (S. L. Perry et al., 1999). When on feeding grounds, larger groupings have been observed (Ray Gambell, 1985). Sei whales appear to prefer to forage in regions of steep bathymetric relief, such as continental shelf breaks, canyons, or basins situated between banks and ledges (P. B. Best & Lockyer, 2002; Edward J. Gregr & Trites, 2001; Robert D. Kenney & Winn, 1987), where local hydrographic features appear to help concentrate zooplankton, especially copepods. In their foraging areas, sei whales appear to associate with oceanic frontal systems (Horwood, 1987). In the north Pacific, sei whales are found feeding particularly along the cold eastern currents (S. L. Perry et al., 1999).

In the western Atlantic Ocean, sei whales occur from Nova Scotia and Labrador in the summer months and migrate south to Florida, the Gulf of Mexico, and the northern Caribbean (Ray Gambell, 1985). In the eastern Atlantic Ocean, sei whales occur in the Norwegian Sea (as far north as Finnmark in northeastern Norway), occasionally occurring as far north as Spitsbergen Island, and migrate south to Spain, Portugal, and northwest Africa (Ray Gambell, 1985).

In the North Pacific Ocean, sei whales occur from the Bering Sea south to California (on the east) and the coasts of Japan and Korea (on the west). During the winter, sei whales are found from 20°-23°N (Ray Gambell, 1985; Y. Masaki, 1977).

Sei whales occur throughout the Southern Ocean during the summer months, although they do not migrate as far south to feed as blue or fin whales. During the austral winter, sei whales occur off Brazil and the western and eastern coasts of Southern Africa and Australia.

4.3.6.2 Population Structure

The population structure of sei whales is not well defined but is presumed to be discrete by ocean basin (north and south), except for sei whales in the Southern Ocean, which may form a ubiquitous population or several discrete ones. Two subspecies of sei whales are recognized, *B. b. borealis* in the Northern Hemisphere and *B. b. schlegellii* in the Southern Hemisphere.

For the North Pacific mark-recapture, catch distribution, and morphological research indicate more than one population may exist – one between 155°-175° W, and another east of 155° W (Y. Masaki, 1976; Y. Masaki, 1977). Sei whales have been reported primarily south of the Aleutian Islands, in Shelikof Strait and waters surrounding Kodiak Island, in the Gulf of Alaska, and inside waters of southeast Alaska and south to California to the east and Japan and Korea to the west (S. Leatherwood, Reeves, Perrin, & Evans, 1982; Nasu, 1974). Sightings have also occurred in Hawaiian waters (M. A. Smultea, Jefferson, & Zoidis, 2010). Sei whales have been occasionally reported from the Bering Sea and in low numbers on the central Bering Sea shelf (P. S. Hill & DeMaster, 1998b). Whaling data suggest that sei whales do not venture north of about 55°N (E. J. Gregr et al., 2000). Masaki (1977) reported sei whales concentrating in the northern and western Bering Sea from July-September, although other researchers question these

observations because no other surveys have reported sei whales in the northern and western Bering Sea. Harwood (1987) evaluated Japanese sighting data and concluded that sei whales rarely occur in the Bering Sea. Harwood (1987) reported that 75-85 percent of the North Pacific population resides east of 180°. During winter, sei whales are found from 20°-23° N (Ray Gambell, 1985; Y. Masaki, 1977). Considering the many British Columbia whaling catches in the early to mid 1900s, sei whales have clearly utilized this area in the past (E. J. Gregr et al., 2000; Pike & Macaskie, 1969).

In the early to mid-1900s, sei whales were hunted off the coast of British Columbia (E. J. Gregr et al., 2000; Pike & Macaskie, 1969). Masaki (1977) presented sightings data on sei whales in the North Pacific from the mid-1960s to the early 1970s. More recently, sei whales have become known for an irruptive migratory habit in which they appear in an area then disappear for time periods that can extend to decades. Based on a sei whale that stranded near Port Angeles and the sei whales observed by Forney and her co-workers (Karin A. Forney, 2007), we know that these whales still occur in waters off Washington, Oregon, and northern California.

4.3.6.3 Natural Threats

Andrews (1916) suggested that killer whales attacked sei whales less frequently than fin and blue whales in the same areas. Sei whales engage in a flight responses to evade killer whales, which involves high energetic output, but show little resistance if overtaken (Ford & Reeves, 2008). Endoparasitic helminths (worms) are commonly found in sei whales and can result in pathogenic effects when infestations occur in the liver and kidneys (D. W. Rice, 1977).

4.3.6.4 Anthropogenic Threats

Human activities known to threaten sei whales include whaling, commercial fishing, and maritime vessel traffic. Historically, whaling represented the greatest threat to every population of sei whales and was ultimately responsible for listing sei whales as an endangered species. Sei whales are thought to not be widely hunted, although harvest for scientific whaling or illegal harvesting may occur in some areas.

Sei whales, because of their offshore distribution and relative scarcity in U.S. Atlantic and Pacific waters, probably have a lower incidence of entrapment and entanglement than fin whales. Data on entanglement and entrapment in non-U.S. waters are not reported systematically. Heyning and Lewis (1990) made a crude estimate of about 73 rorquals killed/year in the southern California offshore drift gillnet fishery during the 1980s. Some of these may have been fin whales instead of sei whales. Some balaenopterids, particularly fin whales, may also be taken in the drift gillnet fisheries for sharks and swordfish along the Pacific coast of Baja California, Mexico (Barlow et al., 1997a). Heyning and Lewis (1990) suggested that most whales killed by offshore fishing gear do not drift far enough to strand on beaches or to be detected floating in the nearshore corridor where most whale-watching and other types of boat traffic occur. Thus, the small amount of documentation may not mean that entanglement in fishing gear is an insignificant cause of mortality. Observer coverage in the Pacific offshore fisheries has been too

low for any confident assessment of species-specific entanglement rates (Barlow et al., 1997a). The offshore drift gillnet fishery is the only fishery that is likely to take sei whales from this stock, but no fishery mortalities or serious injuries to sei whales have been observed. Sei whales, like other large whales, may break through or carry away fishing gear. Whales carrying gear may die later, become debilitated or seriously injured, or have normal functions impaired, but with no evidence recorded.

Sei whales are occasionally killed in collisions with vessels. Of three sei whales that stranded along the U.S. Atlantic coast between 1975 and 1996, two showed evidence of collisions (Laist et al., 2001). Between 1999 and 2005, there were three reports of sei whales being struck by vessels along the U.S. Atlantic coast and Canada's Maritime Provinces (Cole et al., 2005; M. Nelson et al., 2007). Two of these ship strikes were reported as having resulted in death. One sei whale was killed in a collision with a vessel off the coast of Washington in 2003 (Gordon T. Waring et al., 2009). New rules for seasonal (June through December) slowing of vessel traffic in the Bay of Fundy to 10 knots and changing shipping lanes by less than one nautical mile to avoid the greatest concentrations of right whales are predicted to reduce sei whale ship strike mortality by 17 percent.

Sei whales are known to accumulate DDT, DDE, and PCBs (A. Borrell, 1993; A Borrell & Aguilar, 1987; Henry & Best, 1983). Males carry larger burdens than females, as gestation and lactation transfer these toxins from mother to offspring.

4.3.6.5 Status and Trends

The sei whale was originally listed as endangered in 1970 (35 FR 18319), and this status remained since the inception of the ESA in 1973.

Ohsumi and Fukuda (1975) estimated that sei whales in the North Pacific numbered about 49,000 whales in 1963, had been reduced to 37,000-38,000 whales by 1967, and reduced again to 20,600-23,700 whales by 1973. From 1910-1975, approximately 74,215 sei whales were caught in the entire North Pacific Ocean (M. B. Harwood & Hembree., 1987; S. L. Perry et al., 1999). From the early 1900s, Japanese whaling operations consisted of a large proportion of sei whales: 300-600 sei whales were killed per year from 1911-1955. The sei whale catch peaked in 1959, when 1,340 sei whales were killed. In 1971, after a decade of high sei whale catch numbers, sei whales were scarce in Japanese waters. Japanese and Soviet catches of sei whales in the North Pacific and Bering Sea increased from 260 whales in 1962 to over 4,500 in 1968-1969, after which the sei whale population declined rapidly (Mizroch et al., 1984). When commercial whaling for sei whales ended in 1974, the population in the North Pacific had been reduced to 7,260-12,620 animals (Tillman, 1977). There have been no direct estimates of sei whale populations for the eastern Pacific Ocean (or the entire Pacific). Between 1991 and 2001, during aerial surveys, there were two confirmed sightings of sei whales along the U.S. Pacific coast.

Sei whales are known to occur in the Gulf of Alaska and as far north as the Bering Sea in the north Pacific. However, their distribution is poorly understood. The only stock estimate for U.S. waters is for the eastern north Pacific stock offshore California, Oregon and Washington (J. V. Carretta et al., 2009); abundance in Alaskan waters is unknown and they have not been sighted during recent surveys (Rone et al., 2010; Janice M. Waite, Wynne, & Mellinger., 2003).

4.3.6.6 Diving and Social Behavior

Generally, sei whales make 5-20 shallow dives of 20-30 sec duration followed by a deep dive of up to 15 min (Ray Gambell, 1985). When a sei whale begins a dive it usually submerges by sinking quietly below the surface, often remaining only a few meters deep, leaving a series of swirls or tracks as it move its flukes. The depths of sei whale dives have not been studied; however the composition of their diet suggests that they do not perform dives in excess of 300 meters. Sei whales are usually found in small groups of up to 6 individuals, but they commonly form larger groupings when they are on feeding grounds (Ray Gambell, 1985).

Little is known about the actual social system of these animals. Groups of 2-5 individuals are typically observed, but sometimes thousands may gather if food is abundant. However, these large aggregations may not be dependent on food supply alone, as they often occur during times of migration. Norwegian workers call the times of great sei whale abundance "invasion years." During mating season, males and females may form a social unit, but strong data on this issue are lacking.

4.3.6.7 Vocalization and Hearing

There is no direct measurement of the hearing sensitivity of sei whales (Darlene R. Ketten, 2000; Thewissen & Williams, 2002). Sei whale vocalizations are the least studied of all the rorquals. Although data on sei whale vocal behavior is limited records off the Antarctic Peninsula indicate that sei whales produce sounds in the 100-600 Hz range with 1.5 s duration and tonal and upsweep calls in the 200-600 Hz range of 1-3 s durations (Mark A. McDonald et al., 2005). Rankin and Barlow (2007) recorded sei whale vocalizations in Hawaii and reported that all vocalizations were downsweeps, ranging from on average from 100.3 to 446 Hz for “high frequency” calls and from 39.4 to 21.0 Hz for “low frequency” calls. Differences may exist in vocalizations between ocean basins (C. H. Rankin et al., 2009). In another study, McDonald et al. (2005) recorded sei whales in Antarctica with an average frequency of 433 Hz. Vocalizations from the North Atlantic consisted of paired sequences (0.5-0.8 sec, separated by 0.4-1.0 sec) of 10-20 short (4 msec) FM sweeps between 1.5-3.5 kHz (W. John Richardson, Charles R. Greene Jr., et al., 1995).

A general description of the anatomy of the ear for cetaceans is provided in the description of the blue whale.

4.3.7 Sperm Whale

Sperm whales (*Physeter macrocephalus*) are the largest of the odontocetes (toothed whales) and the most sexually dimorphic cetaceans, with males considerably larger than females. Adult females may grow to lengths of 36 feet (11 m) and weigh 15 tons (13,607 kg). Adult males, however, reach about 52 feet (16 m) and may weigh as much as 45 tons (40,823 kg).

The sperm whale is distinguished by its extremely large head, which takes up to 25 to 35 percent of its total body length. It is the only living cetacean that has a single blowhole asymmetrically situated on the left side of the head near the tip.

Sperm whales are mostly dark gray, but oftentimes the interior of the mouth is bright white, and some whales have white patches on the belly. Their flippers are paddle-shaped and small compared to the size of the body, and their flukes are very triangular in shape. They have small dorsal fins that are low, thick, and usually rounded.

4.3.7.1 Distribution

Sperm whales are distributed in all of the world's oceans, from equatorial to polar waters, and are highly migratory. Mature males range between 70º N in the North Atlantic and 70º S in the Southern Ocean (S. L. Perry et al., 1999; Randall R. Reeves & Whitehead, 1997), whereas mature females and immature individuals of both sexes are seldom found higher than 50º N or S (Randall R. Reeves & Whitehead, 1997). In winter, sperm whales migrate closer to equatorial waters (Kasuya & Miyashita, 1988; Gordon T. Waring, 1993) where adult males join them to breed.

Sperm whales are found throughout the North Pacific and are distributed broadly in tropical and temperate waters to the Bering Sea as far north as Cape Navarin in summer, and occur south of 40° N in winter (Gosho, Rice, & Breiwick., 1984; Miyashita, Kato, & Kasuya, 1995 as cited in Carretta et al. 2005; D. W. Rice, 1974). Sperm whales are found year-round in Californian and Hawaiian waters (Barlow, 1995; Dohl, 1983; K. A. Forney, R. L. Brownell, & Fiedler., 1995; E. W. Shallenberger, 1981). They are seen in every season except winter (December-February) in Washington and Oregon (G. A. Green et al., 1992).

Movement patterns of Pacific female and immature male groups appear to follow prey distribution and, although not random, movements are difficult to anticipate and are likely associated with feeding success, perception of the environment, and memory of optimal foraging areas (Hal. Whitehead, 2008). However, no sperm whale in the Pacific has been known to travel to points over 5,000 km apart and only rarely have been known to move over 4,000 km within a time frame of several years. This means that although sperm whales do not appear to cross from eastern to western sides of the Pacific (or vice-versa), significant mixing occurs that can maintain genetic exchange. Movements of several hundred miles are common, (i.e. between the Galapagos Islands and the Pacific coastal Americas). Movements appear to be group or clan specific, with some groups traveling straighter courses than others over the course of several

days. However, general transit speed averages about 4 km/h. Sperm whales in the Caribbean region appear to be much more restricted in their movements, with individuals repeatedly sighted within less than 160 km of previous sightings.

Gaskin (1973) proposed a northward population shift of sperm whales off New Zealand in the austral autumn based on reduction of available food species and probable temperature tolerances of calves.

4.3.7.2 Population Structure

There is no clear understanding of the global population structure of sperm whales (Dufault, Whitehead, & Dillon, 1999). Recent ocean-wide genetic studies indicate low, but statistically significant, genetic diversity and no clear geographic structure, but strong differentiation between social groups (T. Lyrholm & Gyllensten, 1998; Thomas Lyrholm, Leimar, & Gyllensten., 1996; Thomas Lyrholm, Leimar, Johanneson, & Gyllensten, 1999). The IWC currently recognizes four sperm whale stocks: North Atlantic, North Pacific, northern Indian Ocean, and Southern Hemisphere (Dufault et al., 1999; Randall R. Reeves & Whitehead, 1997). The NMFS recognizes six stocks under the MMPA- three in the Atlantic/Gulf of Mexico and three in the Pacific (Alaska, California-Oregon-Washington, and Hawaii; (Simona L. Perry, Demaster, & Silber., 1999; Gordon T. Waring et al., 2004). Genetic studies indicate that movements of both sexes through expanses of ocean basins are common, and that males, but not females, often breed in different ocean basins than the ones in which they were born (Hal. Whitehead, 2003). Sperm whale populations appear to be structured socially, at the level of the clan, rather than geographically (Hal. Whitehead, 2003, 2008).

4.3.7.3 Natural Threats

Sperm whales are known to be occasionally predated upon by killer whales (Thomas A. Jefferson, Stacey, & Baird., 1991; R. L. Pitman, Ballance, Mesnick, & Chivers, 2001) by pilot whales (Arnbom, Papastavrou, Weilgart, & Whitehead, 1987; Palacios & Mate, 1996; Dale W. Rice, 1989; D. W. Weller et al., 1996; Hal Whitehead, Christal, & Dufault., 1997) and large sharks (P. B. Best, Canham, & Macleod, 1984) and harassed by pilot whales (Arnbom et al., 1987; Palacios & Mate, 1996; Dale W. Rice, 1989; D. W. Weller et al., 1996; Hal Whitehead et al., 1997). Strandings are also relatively common events, with one to dozens of individuals generally beaching themselves and dying during any single event. Although several hypotheses, such as navigation errors, illness, and anthropogenic stressors, have been proposed (J. C. Goold, Whitehead, & Reid, 2002; Andrew J. Wright, 2005), direct widespread causes remain unclear. Calcivirus and papillomavirus are known pathogens of this species (R. H. Lambertsen, Kohn, Sundberg, & Buergelt, 1987; Smith & Latham, 1978).

4.3.7.4 Anthropogenic Threats

Sperm whales historically faced severe depletion from commercial whaling operations. From 1800 to 1900, the IWC estimated that nearly 250,000 sperm whales were killed by whalers, with another 700,000 from 1910 to 1982 (IWC Statistics 1959-1983). However, other estimates have

included 436,000 individuals killed between 1800-1987 (J. V. Carretta et al., 2005). However, all of these estimates are likely underestimates due to illegal killings and inaccurate reporting by Soviet whaling fleets between 1947 and 1973. In the Southern Hemisphere, these whalers killed an estimated 100,000 whales that they did not report to the IWC (A. V. Yablokov, Zemsky, Mikhalev, Tormosov, & Berzin, 1998), with smaller harvests in the Northern Hemisphere, primarily the North Pacific, that extirpated sperm whales from large areas (A. V. Yablokov, 2000). Additionally, Soviet whalers disproportionately killed adult females in any reproductive condition (pregnant or lactating) as well as immature sperm whales of either gender.

Following a moratorium on whaling by the IWC, significant whaling pressures on sperm whales were eliminated. However, sperm whales are known to have become entangled in commercial fishing gear and 17 individuals are known to have been struck by vessels (A. S. Jensen & Silber, 2004). Whale-watching vessels are known to influence sperm whale behavior (C. Richter, Dawson, & Slooten, 2006).

In U.S. waters in the Pacific, sperm whales have been incidentally taken only in drift gillnet operations, which killed or seriously injured an average of nine sperm whales per year from 1991-1995 (Barlow et al., 1997a).

Interactions between sperm whales and longline fisheries in the Gulf of Alaska have been reported since 1995 and are increasing in frequency (P. S. Hill & DeMaster, 1998b; P. S. Hill et al., 1999; Dale W. Rice, 1989). Between 2002 and 2006, there were three observed serious injuries (considered mortalities) to sperm whales in the Gulf of Alaska from the sablefish longline fishery (Angliss & Outlaw, 2008). Sperm whales have also been observed in Gulf of Alaska feeding off longline gear (for sablefish and halibut) at 38 of the surveyed stations (Angliss & Outlaw, 2008). Recent findings suggest sperm whales in Alaska may have learned that fishing vessel propeller cavitations (as gear is retrieved) are an indicator that longline gear with fish is present as a predation opportunity (Thode, Straley, Tiemann, Folkert, & O'Connell, 2007).

Contaminants have been identified in sperm whales, but vary widely in concentration based upon life history and geographic location, with northern hemisphere individuals generally carrying higher burdens (K. Evans, Hindell, & Hince, 2004). Contaminants include dieldrin, chlordane, DDT, DDE, PCBs, HCB and HCHs in a variety of body tissues (Alex Aguilar, 1983; K. Evans et al., 2004), as well as several heavy metals (Law, Stringer, Allchin, & Jones, 1996). However, unlike other marine mammals, females appear to bioaccumulate toxins at greater levels than males, which may be related to possible dietary differences between females who remain at relatively low latitudes compared to more migratory males (Alex Aguilar, 1983; Wise et al., 2009). Chromium levels from sperm whales skin samples worldwide have varied from undetectable to 122.6 μg Cr/g tissue, with the mean (8.8 μg Cr/g tissue) resembling levels found

in human lung tissue with chromium-induced cancer (Wise et al., 2009). Older or larger individuals did not appear to accumulate chromium at higher levels.

4.3.7.5 Status and Trends

Sperm whales were originally listed as endangered in 1970 (35 FR 18319), and this status remained with the inception of the ESA in 1973. Although population structure of sperm whales is unknown, several studies and estimates of abundance are available. Sperm whale populations probably are undergoing the dynamics of small population sizes, which is a threat in and of itself. In particular, the loss of sperm whales to directed Soviet whaling likely inhibits recovery due to the loss of adult females and their calves, leaving sizeable gaps in demographic and age structuring (H. Whitehead & Mesnick, 2003).

There are approximately 76,803 sperm whales in the eastern tropical Pacific, eastern North Pacific, Hawaii, and western North Pacific (H. Whitehead, 2002). Minimum estimates in the eastern North Pacific are 1,719 individuals and 5,531 in the Hawaiian Islands (J. V. Carretta et al., 2007). The tropical Pacific is home to approximately 26,053 sperm whales and the western North Pacific has approximately 29,674 (H. Whitehead, 2002). There was a dramatic decline in the number of females around the Galapagos Islands during 1985-1999 versus 1978-1992 levels, likely due to migration to nearshore waters of South and Central America (H. Whitehead & Mesnick, 2003).

Hill and DeMaster (1999) concluded that about 258,000 sperm whales were harvested in the North Pacific between 1947-1987. Although the IWC protected sperm whales from commercial harvest in 1981, Japanese whalers continued to hunt sperm whales in the North Pacific until 1988 (Randall R. Reeves & Whitehead, 1997). In 2000, the Japanese Whaling Association announced plans to kill 10 sperm whales in the Pacific Ocean for research. Although consequences of these deaths are unclear, the paucity of population data, uncertainly regarding recovery from whaling, and re-establishment of active programs for whale harvesting pose risks for the recovery and survival of this species. Sperm whales are also hunted for subsistence purposes by whalers from Lamalera, Indonesia, where a traditional whaling industry has been reported to kill up to 56 sperm whales per year.

4.3.7.6 Diving and Social Behavior

Sperm whales are probably the deepest and longest diving mammalian species, with dives to 3 km down and durations in excess of 2 hours (M. R. Clarke, 1976; Watkins, 1985; Watkins, Daher, Fristrup, Howald, & Disciara, 1993). However, dives are generally shorter (25- 45 min) and shallower (400-1,000 m). Dives are separated by 8-11 min rests at the surface (J. C. D. Gordon, 1987; Watwood, Miller, Johnson, Madsen, & Tyack, 2006) (Jochens et al., 2006; Papastavrou, Smith, & Whitehead, 1989). Sperm whales typically travel ~3 km horizontally and 0.5 km vertically during a foraging dive (Hal. Whitehead, 2003). Differences in night and day diving patterns are not known for this species, but, like most diving air-breathers for which there

are data (rorquals, fur seals, and chinstrap penguins), sperm whales probably make relatively shallow dives at night when prey are closer to the surface.

Davis et al. (2007) report that dive-depths (100-500 m) of sperm whales in the Gulf of California overlapped with depth distributions (200-400 m) of jumbo squid, based on data from satellite-linked dive recorders placed on both species, particularly during daytime hours. Their research also showed that sperm whales foraged throughout a 24-hour period, and that they rarely dove to the sea floor bottom (>1000 m). The most consistent sperm whale dive type is U-shaped, during which the whale makes a rapid descent to the bottom of the dive, forages at various velocities while at depth (likely while chasing prey) and then ascends rapidly to the surface. There is some evidence that male sperm whales, feeding at higher latitudes during summer months, may forage at several depths including <200 m, and utilize different strategies depending on position in the water column (Teloni, Zimmer, Wahlberg, & Madsen., 2007).

Local information is inconsistent regarding sperm whale tendencies. Gregr and Trites (2001) reported that female sperm whales off British Columbia were relatively unaffected by the surrounding oceanography. However, Tynan et al. (2005) reported increased sperm whale densities with strong turbulence associated topographic features along the continental slope near Heceta Bank. Two noteworthy strandings in the region include an infamous incident (well publicized by the media) of attempts to dispose of a decomposed sperm whale carcass on an Oregon beach by using explosives. In addition, a mass stranding of 47 individuals in Oregon occurred during June 1979 (Norman, Bowlby, et al., 2004; D. W. Rice, Wolman, Mate, & Harvey, 1986).

Stable, long-term associations among females form the core of sperm whale societies (Christal, Whitehead, & Lettevall, 1998). Up to about a dozen females usually live in such groups, accompanied by their female and young male offspring. Young individuals are subject to alloparental care by members of either sex and may be suckled by non-maternal individuals (Gero, Engelhaupt, Rendell, & Whitehead., 2009). Group sizes may be smaller overall in the Caribbean Sea (6-12 individuals) versus the Pacific (25-30 individuals) (Jaquet & Gendron, 2009). Males start leaving these family groups at about 6 years of age, after which they live in "bachelor schools," but this may occur more than a decade later (Pinela et al., 2009). The cohesion among males within a bachelor school declines with age. During their breeding prime and old age, male sperm whales are essentially solitary (Christal & Whitehead, 1997).

4.3.7.7 Vocalization and Hearing

Sound production and reception by sperm whales are better understood than in most cetaceans. Sperm whales produce broad-band clicks in the frequency range of 100 Hz to 20 kHz that can be extremely loud for a biological source (200-236 dB re: 1μPa), although lower source level energy has been suggested at around 171 dB re: 1 μPa (John C. Goold & Jones, 1995; Madsen et al., 2003; L. S. Weilgart & Whitehead, 1997; L. S. Weilgart, Whitehead, Carler, & Clark., 1993). Most of the energy in sperm whale clicks is concentrated at around 2-4 kHz and 10-16 kHz (John

C. Goold & Jones, 1995; NMFS, 2006b; L. S. Weilgart et al., 1993). The highly asymmetric head anatomy of sperm whales is likely an adaptation to produce the unique clicks recorded from these animals (Cranford, 1992; K. S. Norris & Harvey., 1972). These long, repeated clicks are associated with feeding and echolocation (John C. Goold & Jones, 1995; Linda S. Weilgart & Whitehead, 1993; L. S. Weilgart & Whitehead, 1997). However, clicks are also used in short patterns (codas) during social behavior and intra-group interactions (L. S. Weilgart et al., 1993). They may also aid in intra-specific communication. Another class of sound, "squeals", are produced with frequencies of 100 Hz to 20 kHz (e.g., C. R. Weir, Frantzis, Alexiadou, & Goold, 2007).

Our understanding of sperm whale hearing stems largely from the sounds they produce. The only direct measurement of hearing was from a young stranded individual from which auditory evoked potentials were recorded (Carder & Ridgway., 1990). From this whale, responses support a hearing range of 2.5-60 kHz. However, behavioral responses of adult, free-ranging individuals also provide insight into hearing range; sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses made by echosounders and submarine sonar (Watkins, 1985; Watkins & Schevill, 1975). They also stop vocalizing for brief periods when codas are being produced by other individuals, perhaps because they can hear better when not vocalizing themselves (John C. Goold & Jones, 1995). Because they spend large amounts of time at depth and use low-frequency sound, sperm whales are likely to be susceptible to low frequency sound in the ocean (Donald A. Croll, Tershy, Acevedo, & Levin, 1999).

4.3.8 Main Hawaiian Islands Insular Distinct Population Segment (DPS) of False Killer Whale

The Main Hawaiian Islands (MHI) Insular DPS of false killer whales *(Pseudorca crassidens*) are large members of the dolphin family. Females reach lengths of 16.6 ft (5.06 m), while males are almost 20 ft (6.1 m). In adulthood, MHI Insular false killer whales can weigh approximately 1,500 kg (3,300 pounds). They have a rounded head which lacks a distinctive beak. Their dorsal fin is tall and their flippers (pectoral fins) have a distinctive hump or bulge in the middle of the front edge. MHI Insular false killer whales have dark coloration except for some lighter patches near the throat and middle chest. Their body shape is more slender than other large delphinids.

4.3.8.1 Distribution

MHI Insular false killer whales principally occur near the waters of the main Hawaiian Islands. In the MHI, false killer whales are typically found in waters ranging from 11 to 23 km (6 to 12 nmi) from shore and in waters averaging 623 m (2,044 ft) in depth, which is shallower than the water depths as which false killer whales occur in other areas of the world, including false killer whales of the Hawaiian pelagic and NWHI stocks ((Robin W Baird et al., 2011)Baird et al., 2011). Distribution of MHI Insular false killer whales has been assessed using data from visual surveys, photo identification studies, and satellite tag. Tagging data indicate that MHI Insular false killer whales move rapidly among the MHI, covering a total range of 82,800 km^2 (24,141 nmi^2), and a broad range of water depths, from shallow (<50 m) to very deep (>4,000 m) (Robin

W Baird et al., 2012; Robin W Baird et al., 2010). Additionally, photo identification studies also have documented that false killer whales in the MHI Insular DPS regularly use waters on both the leeward and windward sides of the MHI (Robin W Baird, 2009; Robin W Baird et al., 2012; Robin W Baird et al., 2010); (K.A. Forney, Baird, Oleson, & Center, 2010); (Oleson et al., 2010). Three high-use areas were identified: (1) off the north half of Hawai‘i Island, (2) north of Maui and Moloka‘i and (3) southwest of Lana‘i (R.W. Baird et al., 2012). However, note that limitations in the sampling, “suggest the range of the population is likely underestimated, and there are probably other high-use areas that have not been identified.”

Some individual false killer whales tagged off the island of Hawaii have remained around that island for extended periods (days to weeks), but individuals from all tagged groups eventually were found broadly distributed throughout the MHI (Robin W. Baird, 2009; K. A. Forney, Baird, & Oleson, 2010). It has been hypothesized that interisland movements may depend on the density and movement patterns of their prey species (Robin W. Baird, 2009).

4.3.8.2 Population Structure

NMFS currently recognizes three stocks of false killer whales in Hawaiian waters: the MHI Insular (which is listed under the ESA as eandangered), Hawaii pelagic, and the Northwest Hawaiian Islands stocks (Bradford, Forney, Oleson, & Barlow, 2012; J. V. Carretta et al., 2011) (77 FR 70915). NMFS considers all false killer whales found within 40 km (22 nm) of the Main Hawaiian Islands as belonging to the MHI Insular stock and all false killer whales beyond 140 km (76 nm) as to belonging to the Pelagic Stock (77 FR 70915, November 15, 2012). The animals belonging to the Northwest Hawaiian Islands stock are insular to the Northwest Hawaii Islands (Bradford et al., 2012), however, this stock was identified by animals encountered off Kauai. It has been previously recognized that the ranges for the two stocks (pelagic and insular) overlap by 100 km (Bradford et al., 2012; J. V. Carretta et al., 2011), but there is in addition, also overlap between all three stocks given these presently identified ranges.

The MHI Insular DPS is considered resident to the MHIs and is genetically and behaviorally distinct compared to other stocks ((Oleson et al., 2010). Genetic data suggest little immigration into the Hawaiian Insular false killer whale population (R.W. Baird et al., 2012). Additional data are being collected to identify whether other false killer whale groups in the Hawaiian Islands should also be considered part of the MHI Insular DPS.

4.3.8.3 Threats to Species

Natural Threats. Reduced genetic diversity may be a natural but partially anthropogenically induced factor leading to the decline of the MHI Insular false killer whales (Wearmouth & Sims, 2008). Only a single instance of depredation on false killer whales has been documented, where killer whales attacked, killed, and consumed a false killer whale calf off New Zealand (Heithaus, 2001; Visser et al., 2010).

Various parasites have been documented in tissues of false killer whales, including nematodes (e.g., *Anisakis simplex, Stenurus globicephalae*), trematodes (e.g., *Nasitrema globicephalae*), acanthocephalans (e.g., *Bolbosoma capitatum*), amphipods (e.g., *Isocyamus delphinii, Syncyamus aequus, Syncyamus pseudorcae*) and crustaceans (e.g., *Xenobalanus globicipitus*) (Andrade, Pinedo, & Barrreto., 2001; Hernandez-Garcia, 2002; Morimitsu et al., 1987; Odell, Asper, Baucom, & Cornell, 1980; Sedlak-Weinstein, 1991; Stacey, Leatherwood, & Baird, 1994; Zylber, Failla, & Bas, 2002). Parasitic infections have risen to levels thought to contribute to the deaths of some false killer whales, but these deaths were associated wtih stranded individuals with unknown other health issues that allowed for unhealthy levels of parasitism to develop (Andrade et al., 2001; Hernandez-Garcia, 2002; Morimitsu et al., 1987; Odell et al., 1980; Sedlak-Weinstein, 1991; Stacey et al., 1994; Zylber et al., 2002). For example, following two mass stranding events that occurred along the coasts of Japan and Florida, tissues of false killer whales were examined for parasites (Morimitsu et al., 1987; Odell et al., 1980). In both studies, evidence of parasitic infections (e.g., *Stenurus globicephalus, Nasitrema gondo*) was noted in the lungs, pterygoid sinus complexes, and tympanic cavities of the whales, and the authors postulated that these infections most likely contributed to the demise of the animals. Currently, no information is available on parasitism in MHI Insular false killer whales.

Anthropogenic Threats. Several threats have been identified that may have or continue to lead to the decline of MHI Insular false killer whales. These include competition with fisheries for prey, bioaccumulation of contaminants, live captures for aquaria, and injury from longline fisheries (Wearmouth & Sims, 2008). False killer whales in Hawaiian waters have been seen to take catches from longline and trolling lines (Nitta & Henderson, 1993; E. Shallenberger, Commission, States, & Corporation, 1981). Interactions with longline and troll fishery operations appear to result in disfigurement to dorsal fins, with roughly 4 percent of the population showing this injury, as well as entanglement and hooking (Robin W. Baird & Antoinette M. Gorgone, 2005; Karin A. Forney & Kobayashi., 2007; M. L. McCracken & K. A. Forney, 2010; Nitta & Henderson, 1993; E. Shallenberger et al., 1981; Zimmerman, 1983a). Carretta et al. (2009) estimated that 7.4 individuals per year are killed or seriously injured during the course of fishing operations in the Hawaiian EEZ. In this area, false killer whales are the most frequently hooked or entangled cetacean species, with most interactions occurring in tuna-targeting longline operations (Karin A. Forney & Kobayashi., 2007; M. L. McCracken & K. A. Forney, 2010). In total, 31 observations of serious injury or mortality have been documented from 1994-2008, which has led to an estimated 13 false killer whales killed or seriously injured throughout the Hawaiian longline fishery (Karin A. Forney & Kobayashi., 2007; M. L. McCracken & K. A. Forney, 2010). It is noteworthy that most interactions occurred well beyond the range known for Hawaiian Island Insular False killer whales (0.6 were estimated to have been killed or seriously injured from 2003-2008) (M. L. McCracken & K. A. Forney, 2010). In addition, false killer whales have been reported preying on catches from shortline fisheries off northern Maui, with deliberate shootings of the animals having occurred in some cases (Nitta & Henderson, 1993; NMFS, 2009b; Schlais, 1985; TEC, 2009).

Overfishing of some pelagic fishes, including bigeye and yellowfin tuna, may be adversely affecting MHI Insular false killer whales. Catch weights for mahi mahi have also declined since 1987 (NMFS, 2009d). These changes may limit the prey quantity or quality available for false killer whales.

Bioaccumulation, particularly of organic contaminants, may be more of a concern for false killer whales than for many other cetaceans due to the high trophic level at which false killer whales feed. The only available study of false killer whale contaminant burden found PCBs and DDT present, with adult females carrying lower burdens than subadults or adult males (likely due to contaminants being unloaded into fetuses and milk during pregnancy and lactation) (A. Aguilar & Borrell, 1994; Krahn et al., 2009; Ylitalo et al., 2009). PCB levels were high enough that biological effects would be experienced in other mammals (Kannan, Blankenship, Jones, & Giesy, 2000). Persistent organic pollutant levels are similar between false killer whales sampled in Taiwan and Japan but smaller (some much smaller) than samples from British Columbia (Chou, Chen, & Li., 2004; Haraguchi, Hisamichi, & Endo., 2006; Ylitalo et al., 2009). Although these pollutants are believed to typically be sequestered in blubber, individuals undergoing metabolic stress mobilize fat tissue, resulting in pollutants being mobilized into other body tissues (A. Aguilar, Forcada, Arderiu, Gazo, & Silvani, 1999). False killer whales from Australia and Japan have been found to have relatively high body burdens of mercury, lead, and cadmium (Endo et al., 2010; Kemper, Gibbs, Obendorf, Marvanek, & Lenghaus., 1994).

4.3.8.4 Status

Reeves et al. (2009) summarized information on false killer whale sightings near Hawaii between 1989 and 2007, based on various survey methods, and suggested that the insular stock of false killer whales may have declined during the last two decades. More recently, Baird (2009) reviewed trends in sighting rates of false killer whales from aerial surveys conducted using consistent methodology around the MHI between 1994 and 2003 (Mobley Jr, 2001; Mobley Jr., 2003, 2004, 2005). Sighting rates during these surveys exhibited a statistically significant decline that could not be attributed to any weather or methodological changes. Reanalysis of previously published abundance estimates for the insular stock has led to them generally being discounted (77 FR 70915). The best estimate from 2000 to 2004 abundance data was that there were 162 animals in the insular population (77 FR 70915). The most recent and best estimate for the MHIH insular stock, however, derived from from 2006 to 2009 sighting survey data and open population models is 151 animals, suggesting that the decline continues, even if at a lower rate than prior to 2000 (77 FR 70915).

The recent Status Review of MHI Insular false killer whales (Oleson et al., 2010) presented a quantitative analysis of extinction risk using a Population Viability Analysis (PVA). The modeling exercise was conducted to evaluate the probability of actual or near extinction, defined as fewer than 20 animals, given measured, estimated, or inferred information on population size and trends, and varying impacts of catastrophes, environmental stochasticity and Allee effects. A

variety of alternative scenarios were evaluated, with all plausible models indicating the probability of decline to fewer than 20 animals within 75 years as greater than 20 percent. Though causation was not evaluated, all models indicated current declines at an average rate of -9 percent since 1989 (95 percent probability intervals -5 percent to -12.5 percent) (Oleson et al., 2010).

4.3.8.5 Diving and Social Behavior

False killer whale group sizes can vary widely. False killer whale groups can consist of multiple dispersed subgroups, and total group size may be underestimated if encounter duration is insufficient (R. W. Baird, Gorgone, et al., 2008). Recent line-transect surveys around the Hawaiian Island have determined that subgroups consist of three animals that are part of larger cluster groups (Bradford et al., 2012). It has been proposed that groups seen during surveys are a part of larger aggregations maintaining acoustic contact (Robin W. Baird et al., 2010). Indeed, larger dispersed aggregations of false killer whales have been noted during surveys (Robin W. Baird, 2009; J. V. Carretta et al., 2007; Randall R. Reeves, Leatherwood, & Baird., 2009; Paul R. Wade & Gerrodette, 1993) moving in a coordinated fashion (R. W. Baird, Gorgone, et al., 2008). MHI Insular false killer whales form strong long-term bonds (R. W. Baird, Gorgone, et al., 2008)

Diving is not well-known in false killer whales, but individuals are believed capable of reaching 500 m in depth and possibly 700 m (Cummings & Fish, 1971; Wearmouth & Sims, 2008). However, most dives are significantly shallower. Hawaiian false killer whales occasionally dive to 150 m (apart from the possible 700 m dive), with frequent dives to 5-20 m during daytime and 30-40 m during nighttime, with durations for nighttime dives running 6-7 minutes (Wearmouth & Sims, 2008). These dive depths may be associated with the waters depths at which common prey occur, such as mahi mahi, which occur most prevalently in the top 100 m of the water column, while other prey, such as tuna and swordfish, may occur down to several hundred meters (Boggs, 1992; Carey & Robinson, 1981).

4.3.8.6 Vocalization and Hearing

Yuen et al. (2005) measured both behavioral and auditory evoked potential (AEP) audiograms of a false killer whale to assess the correspondence of auditory thresholds collected by behavioral and electrophysiological techniques. Behavioral audiograms using 3-s pure-tone stimuli from 4 kHz to 45 kHz indicated the frequencies of best sensitivity were between 16 kHz and 24 kHz and peak sensitivity at 20 kHz, while the AEP audiograms produced thresholds that were also consistent over time, with the range of best hearing sensitivity from 16 kHz to 22.5 kHz and peak sensitivity at 22.5 kHz (Yuen et al., 2005).

False killer whales produce a wide variety of sounds from 4 to 130 kHz, with dominant frequencies between 25 to 30 kHz and 95 to 130 kHz (Busnel & Dziedzic, 1968; Kamminga & Van Velden, 1987); (Murray, Mercado, & Roitblat, 1998; Jeanette A Thomas & Turl, 1990). Most signal types vary among whistles, burst-pulse sounds and click trains (Murray et al., 1998).

Whistles generally range between 4.7 and 6.1 kHz. False killer whales echolocate highly directional clicks ranging between 20 and 60 kHz and 100 and 130 kHz (Kamminga & Van Velden, 1987); (Jeanette A Thomas & Turl, 1990).

Kloepper et al. (2012) investigated the signals from an echolocating false killer whale and found that the majority of clicks had a single-lobed structure with peak energy between 20 and 80 kHz false rather than dual-lobed clicks, as has been demonstrated in the bottlenose dolphin. Navy researchers measured the hearing of a false killer whale and demonstrated the ability of this species to change its hearing during echolocation (Navy, 2012b). The researchers discovered at least three mechanisms of automatic gain control in odontocete echolocation, suggesting that echolocation and hearing are a very dynamic process (Navy, 2012b). For instance, false killer whales change the focus of the echolocation beam based on the difficulty of the task and the distance to the target. The echo from an outgoing signal can change by as much as 40 dB,while the departing and returning signal are the same strength entering the brain (Navy, 2012b). The Navy demonstrated that with a warning signal, the false killer whale can adjust hearing by 15 dB prior to sound exposure (Navy, 2012b)

4.3.9 Hawaiian Monk Seal

Hawaiian monk seals are silvery-grey colored backs with lighter creamy coloration on their underside; newborns are black. Additional light patches and red and green tinged coloration from attached algae are common. The back of the animals may become darker with age, especially in males.

4.3.9.1 Distribution

Hawaiian monk seals are found primarily in the Northwest Hawaiian Islands, which extend more than 2,000 km miles northwest of the main Hawaiian Islands. Major breeding subpopulations occur at French Frigate Shoals, Laysan Island, Lisianski Island, Pearl and Hermes Reef, Midway Atoll, and Kure Atoll (J.V. Carretta, Barlow, Forney, Muto, & Baker, 2001). Smaller groups are found at Nihoa and Necker Islands, seals have been observed at Gardner Pinnacles, Maro Reef, and Johnston Atoll, and several dozen seals are distributed throughout main Hawaiian Islands (J.V. Carretta et al., 2001; NMFS, 2007c). Midway was an important breeding rookery at one time, but is no longer used (Randall R. Reeves, 1992). Reported sightings on each of the eight main Hawaiian Islands have become increasingly common, and births have been reported on all of the main Hawaiian Islands except Lanai and Hawaii. Virtually all terrestrial substrates, including emergent reefs and shipwrecks, are used by monk seals. Sandy beaches with shallow protected water near shore are the primary haul-out areas, for pupping, nursing, and resting, although pups are born on a variety of substrates (Gilmartin, 1983).

4.3.9.2 Population Structure

However, all Hawaiian monk seals represent a single population, with genetic connectivity high enough to maintain population-level genetic differentiation (Schultz, Baker, Toonen, Harting, & Bowen, 2011). Hawaiian monk seal distribution, destinations, routes, food sources, and causes of

movements when not traveling between islands are not well known. Approximately 10-15 percent of Hawaiian monk seals migrate among the breeding populations (A. M. Johnson & Kridler, 1983). Inter-island movement appears to be more likely when the islands are close together. For example, movement between Kure Atoll, Midway Atoll, Pearl and Hermes Reef appear to be fairly common, while movement between French Frigate Shoals and Kure Atoll (a distance of 2,000 km) is not known to occur. The western subpopulations (Pearl and Hermes Reef, Midway Islands, and Kure Atoll) exhibit a higher degree of migration compared to the more isolated subpopulations at Laysan, Lisianski, and French Frigate Shoals (NMFS, 2007c).

4.3.9.3 Status and Trends

The Hawaiian monk seal was listed as endangered under the ESA on November 23, 1976 (41 FR 51611). Hawaiian monk seals are considered one of the most endangered groups of pinnipeds on the planet because all of their populations are either extinct (Caribbean monk seal) or close to extinction (Mediterranean and Hawaiian monk seals). Two periods of anthropogenic decline have been reported; the first decline occurred in the 1800s when sealers, crews of wrecked vessels, and guano and feather hunters nearly hunted monk seals to extinction (Dill & Bryan, 1912; Kenyon & Rice, 1959).

Following the initial collapse, expeditions to the northwest Hawaiian Islands reported increasing seal numbers and partial recovery to slightly more than 1,000 individuals (A. M. Bailey, 1952; D.W. Rice, 1960). However, a second decline occurred from the late 1950s to the mid-1970s; the population declined by roughly 50 percent by the 1980s (NMFS, 1991a). The total population in the French Frigate Shoals, Laysan Island, Lisianski Island, Pearl and Hermes Reef, Kure Atoll, and Midway, Necker, and Nihoa was estimated to be 1,501 in 1984, 1,976 seals in 1986, and 1,580 in 1992 (Ragen, 1993). For the years 1985 to 1993 the mean beach counts declined by approximately 5 percent per year. This downward trend is expected to continue, mainly because of poor pup and juvenile survival in recent years.

The best estimate of the total population of the species is 1,202 seals and the minimum population size estimate for the Hawaiian monk seal is 1,176 seals (NMFS, 2007c). Data collected in 2008 suggest that the species population is now 1,146 (NMFS, 2009a). A log-linear regression of estimated abundance from 1998 to 2006 suggests the population has declined on average -3.9 percent per year, and models predict that the total population of the species will fall below 1,000 monk seals within 5 years (NMFS, 2007c). Trends in abundance vary considerably among the six main subpopulations.

A recent five-year status review conducted by NMFS recommends that the Hawaiian monk seals' endangered status should remain the same (72 FR 46966, August 2007). The population dynamics at the different subpopulations have varied considerably, and current demographic variability among the island populations probably reflects a combination of different histories of human disturbance and management (Mitchell P. Craig & Timothy J. Ragen, 1999; Gerrodette & DeMaster, 1990), and varying environmental conditions (J. D. Baker, Polovina, & Howell, 2007;

J. D. Baker & Thompson, 2007; M. P. Craig & T. J. Ragen, 1999; J. F. Polovina et al., 1994). The current status of the Hawaiian monk seal is dire, due to low juvenile survival and the number of aging breeding females in the population. Consequently, NMFS is currently exploring development of a captive care program for juvenile Hawaiian monk seals to enhance their potential for survival and recovery (NMFS, 2007c).

The total of mean, non-pup, beach counts at the main reproductive subpopulations in 2005 was approximately 67 percent lower than in 1958 (Scott R. Benson, Forney, Harvey, Carretta, & Dutton, 2007). A log-linear regression of estimated abundance from 1998 (the first year for which a reliable total abundance estimate was obtained) to 2006 estimates that abundance declined by 3.9 percent annually(NMFS, 2007c).

Trends vary among the six main subpopulations. Non-pup beach counts at French Frigate Shoals have decreased by 73 percent from 1989 to 2005 (Scott R. Benson et al., 2007). At one time, French Frigate Shoals accounted for over 50 percent of the total non-pup beach counts among the northwest Hawaiian Islands subpopulations; however, that proportion has dropped to 25 percent, although French Frigate Shoals still maintains the single largest subpopulation (NMFS, 2007c). The annual number of births has dropped from a high of 127 in 1988 to 39 in 2006, and survival from weaning to age two has declined from a high near 90 percent in the mid-1980s to a low of 8 percent in 1997 (NMFS, 2007c). Shark predation and prey availability are two potentially responsible factors.

Populations at Laysan and Lisianski Islands declined sharply after the late 1950s. In 1994, 21 adult male Hawaiian monk seals were relocated from Laysan Island to the main Hawaiian Islands in an attempt to equalize the sex ratio at Laysan Island, and beach counts increased from 1995 to 2000, but have declined in the following years, while the Lisianski subpopulation has remained relatively stable, yet low, since the 1970s. Marine debris and low fecundity are factors that might contribute to the lack of subpopulation growth at Lisianski Island. And while the decline in abundance in Laysan may be related to female mortality caused by male aggression, juvenile survival is relatively good for most cohorts, and the lack of recovery on Laysan is not understood (NMFS, 2007c).

Until recently, the three westernmost subpopulations, Kure, Midway and Pearl and Hermes Reef exhibited substantial growth. Beach counts on Kure increased 5 percent per year from 1983 to 2000, declined in 2000-2001, and are now slowly increasing. At Midway, beach counts increased from 1995 to 2000, and have since declined. The subpopulation at Pearl and Hermes Reef increased after the mid-1970s. Prior to 1999, beach count increases of up to 7 percent per year were observed. This is the highest estimate of the maximum net productivity rate observed for this species. Since 2000, low juvenile survival, thought to be due largely to food limitation, has been widespread with rare exceptions in the northwest Hawaiian Islands, resulting in the

population decline (Scott R. Benson et al., 2007), and several recent cohorts at the three westernmost sites indicate a drop in juvenile survival (NMFS, 2007c).

The decrease in survival rates of immature animals, including a decline in survival from birth to weaning, and survival from weaning to age 2 years has contributed to a dramatically imbalanced age structure for all six of the main northwest Hawaiian Islands subpopulations (NMFS, 2007c, 2009a). Although studies show that the relationship between size of pups and first year survival vary between subpopulations and over time, site-specific analyses do support girth and year as predictors of first-year survival at each location. When conditions for survival are worse, the relationship between size and survival strengthens. The simplest explanation for this is food limitation (J. D. Baker, 2008).

Sightings and births are increasing in the main Hawaiian Islands, although systematic surveys were not conducted before 2000, and counts do not represent total abundance, as they do not account for seals in the water, and not every seal on land is detected. In 2000, the count in the main Hawaiian Islands was 45 seals, and in 2001, 52 were counted. In 2005, the total number of unique seals identified was 77, based on non-systematic sightings. Annual births have increased since the mid-1990s. Although this could be a positive indication for the survival of the species, the increased chance of contraction of diseases such as leptospirosis and toxoplasmosis from wild and domestic animals, and increased interactions with humans, including fishermen, boaters, and divers raise conservation concerns which do not apply to the northwest Hawaiian Islands (NMFS, 2007c). The only available estimate of abundance in the main Hawaiian Islands is 152 individuals, with an annual population growth rate of 7 percent (J. D. Baker, Harting, Wurth, & Johanos, 2011). Survival to one year of age is 77 percent in the NHI, versus 42-57 percent in the northwest Hawaiian Islands (J. D. Baker et al., 2011).

4.3.9.4 Diving and Social Behavior

Dive depths appear to differ slightly between rookeries as well as between age and sex classes. Stewart et al. (2006) found that throughout the six northwest Hawaiian Islands breeding colonies, most dives were less than 150 meters, but found some dives exceeding 550 meters. At Pearl and Hermes Reef, most dives reach 8-40 meters, with some dives to three- to four-fold greater depths (Harington, Ross, Mathewes, Stewart, & Beattie, 2004). At Kure Atoll, males tended to dive deeper than females (Harington et al., 2004). However, at Laysan Island, this trend was reversed and dives were much deeper (800 to 1,150 feet; Harington et al., 2004). Most dives at French Frigate Shoals were to depths of 4-40 meters, but some dives exceeded 500 meters (Abernathy, 1999). Parrish et al. (2002) noted a tendency towards night diving at French Frigate Shoals.

Hawaiian monk seals tend to dive within the water column, rather than to the sea floor, regardless of site (Pullin & Stewart, 2006). Some work using Crittercams on seals at French Frigate Shoals indicates that most time spent underwater was for resting and socializing, not feeding. Despite the reef fishes of the coral shallows, adult seals forage on the slopes of the atoll and neighboring banks (Frank A. Parrish, Craig, Ragen, Marshall, & Buhleier, 2000). This is

corroborated by the comparison between the diet composition of tagged seals and the composition of fish in each of four ecological zones (defined by depth) (F.A. Parrish & Abernathy, 2006). Foraging has been shown to vary by age, with older juveniles (years 2 and 3) focusing on shallow atoll depths (10-30 meters) and yearlings feeding in sand fields at 50-100 meters. It is possible that the shift in foraging behavior with age is dependent on the physical strength to flip small rocks to find prey, rather than increasing dive duration or depth (F.A. Parrish & Abernathy, 2006).

4.3.9.5 Vocalizations and Hearing

Only one audiogram has been recorded for the Hawaiian monk seal, which indicated relatively poor hearing sensitivity, a narrow range of best hearing sensitivity (12 to 28 kHz), and a relatively low upper frequency limit (Jeanette A. Thomas, Moore, Withrow, & Stoermer, 1990a); (David Kastak & Schusterman, 1996, 1998). Above 30 kHz, high-frequency hearing sensitivity dropped markedly (Jeanette A. Thomas et al., 1990a). However, the audiogram was obtained from a single, untrained seal whose hearing curve suggested that its responses may have been affected by disease or age (R.R. Reeves, Read, & Notarbartolo-di-Sciara, 2001). No underwater sound production has been reported for this species. Recorded in-air vocalizations of Hawaiian monk seals consist of a variety of sounds, including a liquid bubble sound (100 to 400 Hz), a guttural expiration (about 800 Hz) produced during short-distance agonistic encounters, a roar (<800 Hz) for long-distance threats, a belch-cough made by males when patrolling (<1 kHz), and sneeze/snorts/coughs of variable frequencies that are <4 kHz (E. H. Miller & Job, 1992).

4.3.10 Green Turtle

Green turtles are the largest of all the hard-shelled sea turtles with a comparatively small head. While hatchlings are just 2 inches (50 mm) long, adults can grow to more than 3 feet (0.91 m) long and weigh 300-350 pounds (136-159 kg).

Adult green turtles are unique among sea turtles in that they are herbivorous, feeding primarily on seagrasses and algae. This diet is thought to give them greenish colored fat, from which they take their name. A green turtle's carapace (top shell) is smooth and can be shades of black, gray, green, brown, and yellow. Their plastron (bottom shell) is yellowish white.

Scientists estimate green turtles reach sexual maturity anywhere between 20 and 50 years, at which time females begin returning to their natal beaches (i.e., the same beaches where they were born) every 2-4 years to lay eggs.

The nesting season varies depending on location. In the southeastern U.S., females generally nest between June and September, while peak nesting occurs in June and July. During the nesting season, females nest at approximately two week intervals, laying an average of five clutches. In Florida, green turtle nests contain an average of 135 eggs, which will incubate for approximately 2 months before hatching.

4.3.10.1 Distribution

Green turtles are found in the Pacific Ocean, Atlantic Ocean, Indian Ocean, Caribbean Sea, and Mediterranean Sea, primarily in tropical or, to a lesser extent, subtropical waters. These regions can be further divided into nesting aggregations within the eastern, central, and western Pacific Ocean; the western, northern, and eastern Indian Ocean; Mediterranean Sea; and eastern, southern, and western Atlantic Ocean, including the Caribbean Sea.

Green turtles are highly mobile and undertake complex movements through geographically disparate habitats during their lifetimes (Musick & Limpus, 1997; P. Plotkin, 2003). The periodic migration between nesting sites and foraging areas by adults is a prominent feature of their life history. After departing as hatchlings and residing in a variety of marine habitats for 40 or more years (C. Limpus & Chaloupka, 1997), green sea turtles make their way back to the same beach from which they hatched (A. Carr, Carr, & Meylan, 1978; A. B. Meylan, Bowen, & Avise, 1990). At approximately 20-25 cm carapace length, juveniles leave pelagic habitats and enter benthic foraging areas (Bjorndal, 1997). Green sea turtles spend the majority of their lives in coastal foraging grounds. These areas include both open coastline and protected bays and lagoons. While in these areas, green sea turtles rely on marine algae and seagrass as their primary dietary constituents, although some populations also forage heavily on invertebrates. There is some evidence that individuals move from shallow seagrass beds during the day to deeper areas at night (Hazel, 2009). However, avoidance of areas of greater than 10 m when moderate depths of 5-10 m with sea grass beds has been found, with speed and displacement from capture locations being similar at night as during the daytime (Jesse Senko et al., 2010).

Primary sites for green turtle nesting in the Atlantic/Caribbean include: (1) Yucatán Peninsula, Mexico; (2) Tortuguero, Costa Rica; (3) Aves Island, Venezuela; (4) Galibi Reserve, Suriname; (5) Isla Trindade, Brazil; (6) Ascension Island, United Kingdom; (7) Bioko Island, Equatorial Guinea; and (8) Bijagos Achipelago, Guinea-Bissau (NMFS & USFWS, 2007a). Nesting at all of these sites was considered to be stable or increasing with the exception of Bioko Island and the Bijagos Archipelago where the lack of sufficient data precludes a meaningful trend assessment for either site (NMFS & USFWS, 2007a). Seminoff (2004) reviewed green sea turtle nesting data for eight sites in the western, eastern, and central Atlantic. Seminoff (2004) concluded that all sites in the central and western Atlantic showed increased nesting, with the exception of nesting at Aves Island, Venezuela, while both sites in the eastern Atlantic demonstrated decreased nesting. These sites are not inclusive of all green sea turtle nesting in the Atlantic. However, other sites are not believed to support nesting levels high enough that would change the overall status of the species in the Atlantic (NMFS & USFWS, 2007a).

By far, the most important nesting concentration for green sea turtles in the western Atlantic is in Tortuguero, Costa Rica (NMFS & USFWS, 2007a). Nesting in the area has increased considerably since the 1970s and nest count data from 1999-2003 suggest nesting by 17,402-37,290 females per year (NMFS & USFWS, 2007a). The number of females nesting per year on

beaches in the Yucatán, at Aves Island, Galibi Reserve, and Isla Trindade number in the hundreds to low thousands, depending on the site (NMFS & USFWS, 2007a).

The vast majority of green sea turtle nesting within the southeastern US occurs in Florida (S. A. Johnson & Ehrhart, 1994; A.B. Meylan, Schroeder, & Mosier, 1995). Green sea turtle nesting in Florida has been increasing since 1989 (Florida Fish and Wildlife Conservation Commission, Florida Marine Research Institute Index Nesting Beach Survey Database). Since establishment of index beaches in 1989, the pattern of green turtle nesting shows biennial peaks in abundance with a generally positive trend during the ten years of regular monitoring. This is perhaps due to increased protective legislation throughout the Caribbean (A.B. Meylan et al., 1995). A total statewide average (all beaches, including index beaches) of 5,039 green turtle nests were laid annually in Florida between 2001 and 2006, with a low of 581 in 2001 and a high of 9,644 in 2005 (NMFS & USFWS, 2007a). Data from index nesting beaches substantiate the dramatic increase in nesting. In 2007, there were 9,455 green turtle nests found just on index nesting beaches, the highest since index beach monitoring began in 1989. The number fell back to 6,385 in 2008, further dropping under 3,000 in 2009, but that consecutive drop was a temporary deviation from the normal biennial nesting cycle for green turtles, as 2010 saw an increase back to 8,426 nests on the index nesting beaches (FWC Index Nesting Beach Survey Database). Occasional nesting has been documented along the Gulf coast of Florida (A.B. Meylan et al., 1995). More recently, green turtle nesting occurred on Bald Head Island, North Carolina; just east of the mouth of the Cape Fear River; on Onslow Island; and on Cape Hatteras National Seashore. In 2010, a total of 18 nests were found in North Carolina, 6 nests in South Carolina, and 6 nests in Georgia (nesting databases maintained on www.seaturtle.org). Increased nesting has also been observed along the Atlantic coast of Florida, on beaches where only loggerhead nesting was observed in the past (P. C. H. Pritchard, 1997). Recent modeling by Chaloupka et al. (2008) using data sets of 25 years or more has resulted in an estimate of the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9 percent, and the Tortuguero, Costa Rica, population growing at 4.9 percent.

There are no reliable estimates of the number of immature green sea turtles that inhabit coastal areas of the southeastern US. However, information on incidental captures of immature green sea turtles at the St. Lucie Power Plant in St. Lucie County, Florida, shows that the annual number of immature green sea turtles captured by their offshore cooling water intake structures has increased significantly. Green sea turtle annual captures averaged 19 for 1977-1986, 178 for 1987-1996, and 262 for 1997-2001 (Florida Power and Light Company St. Lucie Plant, 2002). More recent unpublished data shows 101 captures in 2007, 299 in 2008, 38 in 2009 (power output was cut—and cooling water intake concomitantly reduced—for part of that year) and 413 in 2010. Ehrhart et al. (2007) documented a significant increase in in-water abundance of green turtles in the Indian River Lagoon area.

Green turtles are thought to be declining throughout the Pacific Ocean, with the exception of Hawaii, from a combination of overexploitation and habitat loss (K. L. Eckert, 1993; Jeffrey A. Seminoff, Resendiz, & Nichols, 2002). In the western Pacific, the only major (>2,000 nesting females) populations of green turtles occur in Australia and Malaysia, with smaller colonies throughout the area. Indonesian nesting is widely distributed, but has experienced large declines over the past 50 years. Hawaii green turtles are genetically distinct and geographically isolated, and the population appears to be increasing in size despite the prevalence of fibropapillomatosis and spirochidiasis (Aguirre, Spraker, Balazs, & Zimmerman, 1998).

The East Island nesting beach in Hawaii is showing a 5.7 percent annual growth rate over >25 years (M. Chaloupka et al., 2008). In the Eastern Pacific, mitochondrial DNA analysis has indicated three key nesting populations: Michoacán, Mexico; Galapagos Islands, Ecuador; and Islas Revillagigedos, Mexico (P. H. Dutton, 2003). The number of nesting females per year exceeds 1,000 females at each site (NMFS & USFWS, 2007a). However, historically, >20,000 females per year are believed to have nested in Michoacán alone (Clifton, Cornejo, & Felger, 1982; NMFS & USFWS, 2007a). Thus, the current number of nesting females is still far below historical levels. Datasets over 25 years in Chichi-jima, Japan; Heron Island, Australia; and Raine Island, Australia, show increases in abundance (M. Chaloupka et al., 2008).

4.3.10.2 Habitat

Green turtles appear to prefer waters that usually remain around 20º C in the coldest month, but may occur considerably north of these regions during warm-water events, such as El Niño. Stinson (1984) found green turtles to appear most frequently in US coastal waters with temperatures exceeding 18º C. Further, green sea turtles seem to occur preferentially in drift lines or surface current convergences, probably because of the prevalence of cover and higher prey densities that associate with flotsam. For example, in the western Atlantic Ocean, drift lines commonly containing floating *Sargassum* spp. are capable of providing juveniles with shelter (NMFS & USFWS, 1998b). Underwater resting sites include coral recesses, the underside of ledges, and sand bottom areas that are relatively free of strong currents and disturbance. Available information indicates that green turtle resting areas are near feeding areas (Bjorndal & Bolten, 2000). Strong site fidelity appears to be a characteristic of juveniles green sea turtles along the Pacific Baja coast (Senko, Lopez-Castro, Koch, & Nichols, 2010).

4.3.10.3 Population Structure

The population dynamics of green sea turtles and all of the other sea turtles we consider here are usually described based on the distribution and habit of nesting females, rather than their male counterparts. The spatial structure of male sea turtles and their fidelity to specific coastal areas is unknown; however, we describe sea turtle populations based on the nesting beaches that female sea turtles return to when they mature. Because the patterns of increase or decrease in the abundance of sea turtle nests over time are determined by internal dynamics rather than external

dynamics, we make inferences about the growth or decline of sea turtle populations based on the status and trend of their nests.

Primary nesting aggregations of green turtles (i.e. sites with greater than 500 nesting females per year) include: Ascension Island (south Atlantic Ocean), Australia, Brazil, Comoros Islands, Costa Rica, Ecuador (Galapagos Archipelago), Equatorial Guinea (Bioko Island), Guinea-Bissau (Bijagos Archipelago), Iles Eparses Islands (Tromelin Island, Europa Island), Indonesia, Malaysia, Myanmar, Oman, Philippines, Saudi Arabia, Seychelles Islands, Suriname, and United States (Florida; NMFS & USFWS, 1998c; Jeffrey A. Seminoff et al., 2002).

Smaller nesting aggregations include: Angola, Bangladesh, Bikar Atoll, Brazil, Chagos Archipelago, China, Costa Rica, Cuba, Cyprus, Democratic Republic of Yemen, Dominican Republic, d'Entrecasteaux Reef, French Guiana, Ghana, Guyana, India, Iran, Japan, Kenya, Madagascar, Maldives Islands, Mayotte Archipelago, Mexico, Micronesia, Pakistan, Palmerston Atoll, Papua New Guinea, Primieras Islands, Sao Tome é Principe, Sierra Leone, Solomon Islands, Somalia, Sri Lanka, Taiwan, Tanzania, Thailand, Turkey, Scilly Atoll, United States (Hawaii), Venezuela, and Vietnam (Florida; NMFS & USFWS, 1998c; Jeffrey A. Seminoff et al., 2002).

Molecular genetics techniques have helped researchers gain insight into the distribution and ecology of migrating and nesting green turtles. In the Pacific Ocean, green sea turtles group into two distinct regional clades: (1) western Pacific and South Pacific islands, and (2) eastern Pacific and central Pacific, including the rookery at French Frigate Shoals, Hawaii. In the eastern Pacific, greens forage coastally from San Diego Bay, California in the north to Mejillones, Chile in the South. Based on mtDNA analyses, green turtles found on foraging grounds along Chile's coast originate from the Galapagos nesting beaches, while those greens foraging in the Gulf of California originate primarily from the Michoacan nesting stock. Green turtles foraging in San Diego Bay and along the Pacific coast of Baja California originate primarily from rookeries of the Islas Revillagigedos (P. Dutton, 2003).

4.3.10.4 Natural Threats

The various habitat types green sea turtles occupy throughout their lives exposes these sea turtles to a wide variety of natural threats. The beaches on which green sea turtles nest and the nests themselves are threatened by hurricanes and tropical storms as well as the storm surges, sand accretion, and rainfall that are associated with hurricanes. Hatchlings are hunted by predators like herons, gulls, dogfish, and sharks. Larger green sea turtles, including adults, are also killed by sharks and other large, marine predators.

Green turtles in the northwest Hawaiian Islands are afflicted with a tumor disease, fibropapilloma, which is of an unknown etiology and often fatal, as well as spirochidiasis, both of which are the major causes of strandings of this species. The presence of fibropapillomatosis among stranded turtles has increased significantly over the past 17 years, ranging from 47-69

percent during the past decade (Murakawa, Balazs, Ellis, Hau, & Eames, 2000). Preliminary evidence suggests an association between the distribution of fibropapillomatosis in the Hawaiian Islands and the distribution of toxic benthic dinoflagellates (*Prorocentrum* spp.) known to produce a tumor promoter, okadaic acid (Landsberg et al., 1999). Green turtles captured off Molokai from 1982-96 showed a massive increase in the disease over this period, peaking at 61 percent prevalence in 1995 (Balazs, Puleloa, Medeiros, Murakawa, & Ellis, 1998). Fibropapillomatosis is considered to decrease growth rates in afflicted turtles and may inhibit the growth rate of Hawaiian green turtle populations (Balazs et al., 1998).

4.3.10.5 Anthropogenic Threats

Three human activities are known to threaten green sea turtles: overharvests of individual animals, incidental capture in commercial fisheries, and human development of coastlines. Historically, the primary cause of the global decline of green sea turtles populations were the number of eggs and adults captured and killed on nesting beaches in combination with the number of juveniles and adults captured and killed in coastal feeding areas. Some populations of green sea turtles still lose large numbers of eggs, juveniles, and adults to subsistence hunters, local communities that have a tradition of harvesting sea turtles, and poachers in search of turtle eggs and meat.

Directed harvests of eggs and other life stages of green sea turtles were identified as a “major problem” in American Samoa, Guam, Palau, Commonwealth of the Northern Mariana Islands, Federated States of Micronesia, Republic of the Marshall Islands, and the Unincorporated Islands (Wake, Johnston, Kingman, Palmyra, Jarvis, Howland, Baker, and Midway). In the Atlantic, green sea turtles are captured and killed in turtle fisheries in Colombia, Grenada, the Lesser Antilles, Nicaragua, St. Vincent and the Grenadines (Brautigam & Eckert, 2006); the turtle fishery along the Caribbean coast of Nicaragua, by itself, has captured more than 11,000 green sea turtles each year (Brautigam & Eckert, 2006; Lagueux, 1998).

Severe overharvests have resulted from a number of factors in modern times: (1) the loss of traditional restrictions limiting the number of turtles taken by island residents; (2) modernized hunting gear; (3) easier boat access to remote islands; (4) extensive commercial exploitation for turtle products in both domestic markets and international trade; (5) loss of the spiritual significance of turtles; (6) inadequate regulations; and (7) lack of enforcement (NMFS & USFWS, 1998c).

Green turtles are also captured and killed in commercial fisheries. Gillnets account for the highest number of green sea turtles that are captured and killed, but they are also captured and killed in trawls, traps and pots, longlines, and dredges. Along the Atlantic coast of the U.S., NMFS estimated that almost 19,000 green sea turtles are captured in shrimp trawl fisheries each year in the Gulf of Mexico, with 514 of those sea turtles dying as a result of their capture. Each year, several hundred green sea turtles are captured in herring fisheries; mackerel, squid, and butterfish fisheries; monkfish fisheries; pound net fisheries, summer flounder and scup fisheries;

Atlantic pelagic longline fisheries; and gillnet fisheries in Pamlico Sound. Although most of these turtles are released alive, these fisheries are expected to kill almost 100 green sea turtles each year; the health effects of being captured on the sea turtles that survive remain unknown.

Green turtles are also threatened by domestic or domesticated animals which prey on their nests; artificial lighting that disorients adult female and hatchling sea turtles, which can dramatically increase the mortality rates of hatchling sea turtles; beach replenishment; ingestion and entanglement in marine debris; and environmental contaminants.

4.3.10.6 Status and Trends

Green turtles are listed as threatened under the ESA, except for breeding populations found in Florida and the Pacific coast of Mexico, which are listed as endangered. The green turtle is protected under CITES and is listed as endangered by the IUCN. Seminoff (2002) estimates using a conservative approach that the global green turtle population has declined by 34 to 58 percent over the last three generations (approximately 150 years). Actual declines may be closer to 70 to 80 percent. Causes for this decline include harvest of eggs, subadults and adults, incidental capture by fisheries, loss of habitat, and disease.

While some nesting populations of green turtles appear to be stable or increasing in the Atlantic Ocean (e.g. Bujigos Archipelago (Guinea-Bissau), Ascension Island, Tortuguero (Costa Rica), Yucatan Peninsula (Mexico), and Florida), declines of over 50 percent have been documented in the eastern (Bioko Island, Equatorial Guinea) and western Atlantic (Aves Island, Venezuela). Nesting populations in Turkey (Mediterranean Sea) have declined between 42 and 88 percent since the late 1970s. Population trend variations also appear in the Indian Ocean. Declines greater than 50 percent have been documented at Sharma (Republic of Yemen) and Assumption and Aldabra (Seychelles), while no changes have occurred at Karan Island (Saudi Arabia) or at Ras al Hadd (Oman). The number of females nesting annually in the Indian Ocean has increased at the Comoros Islands, Tromelin and maybe Europa Island (Iles Esparses; J. A. Seminoff, 2004).

Green turtles are thought to be declining throughout the Pacific Ocean, with the exception of Hawaii, as a direct consequence of a historical combination of overexploitation and habitat loss (K.L. Eckert, 1993; J. A. Seminoff, 2004). They are also thought to be declining in the Atlantic Ocean. However, like several of the species we have already discussed, the information available on the status and trend of green sea turtles do not allow us to make a definitive statement about the global extinction risks facing these sea turtles or risks facing particular populations (nesting aggregations) of these turtles. With the limited data available on green sea turtles, we do not know whether green sea turtles exist at population sizes large enough to avoid demographic phenomena that are known to increase the extinction probability of species that exist as “small” populations (that is, “small” populations experience phenomena such as demographic stochasticity, inbreeding depression and Allee effects, among others, that cause their population size to become a threat in and of itself) or if green sea turtles are threatened more by exogenous

threats such as anthropogenic activities (entanglement, habitat loss, overharvests, etc.) or natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate). Nevertheless, with the exception of the Hawaiian nesting aggregations, we assume that green sea turtles are endangered because of both anthropogenic and natural threats as well as changes in their population dynamics.

A recovery plan for the U.S Population of Atlantic Green Turtles was written in 1991 (NMFS & USFWS, 1991). A recovery plan for the U.S. Pacific Populations of the Green Turtle was written in 1998 (NMFS & USFWS, 1998c).

4.3.10.7 Diving Behavior

Based on the behavior of post-hatchlings and juvenile green turtles raised in captivity, it is presumed that those in pelagic habitats live and feed at or near the ocean surface, and that their dives do not normally exceed several meters in depth (NMFS & USFWS, 1998c). The maximum recorded dive depth for an adult green turtle was 110 meters (Berkson, 1967) (M. E. Lutcavage & Lutz, 1997), while subadults routinely dive 20 meters for 9-23 minutes, with a maximum recorded dive of 66 minutes (Brill et al. 1995 in M. E. Lutcavage & Lutz, 1997).

4.3.10.8 Vocalizations and Hearing

Sea turtles do not appear to use sound for communication, and there are no published recordings of green sea turtle vocalizations. Nesting leatherback turtles have been recorded producing sounds (sighs, grunts or belch-like sounds) up to 1,200 Hz with maximum energy from 300 to 500 Hz (Cook & Forrest, 2005), however these sounds appeared to be associated with breathing (Mrosovsky, 1972);(Cook & Forrest, 2005).

Several studies have been conducted to measure green turtle hearing sensitivity, each using a slightly different methodology. Ridgway et al. (1969) studied the auditory evoked potentials of three green sea turtles (in air and through direct mechanical stimulation of the inner ear) and concluded that their maximum sensitivity occurred from 300 to 400 Hz with rapid declines for tones at lower and higher frequencies. They reported an upper limit for cochlear potentials without injury of 2000 Hz and a practical limit of about 1000 Hz.

Bartol and Ketten (2006) measured auditory brainstem responses (short latency auditory evoked potentials) to aerial tones in partially submerged green turtles and documented hearing sensitivity between 100 and 800 Hz, with maximum sensitivity between 600 and 700 Hz in Atlantic juvenile greens, and100 and 500 Hz with maximum sensitivity between 200 and 400 Hz in Pacific subadult greens (S.M. Bartol & Ketten, 2006).

The Navy’s Supplemental EIS referred to studies conducted by Streeter and colleagues on a female green sea turtle that had been trained to respond to acoustic signals (Navy, 2007c). Those authors established that this sea turtle had a hearing range of at least 100 to 500 Hz (the maximum frequency that could be used in the study, as opposed to what may be a wider hearing

range) with hearing thresholds of 120-130 dB received level. We could not be certain whether and to what degree the results of this study might apply to other sea turtles, particularly sea turtles in the wild. First, only one individual was studied and that individual was an older individual that had lived its entire life in captivity. Second, the study was conducted at the New England Aquarium, which is a relatively noisy environment. As a result, the thresholds reported may have been masked by the background noise and the “absolute thresholds” (the lowest detectable signal within a noisy environment) may be several dB lower than the reported results.

Dow Piniak et al. (2012) recorded auditory evoked potential in response to both aerial and underwater acoustic stimuli. Green turtles detected acoustic stimuli in both media, responding to underwater signals between 50 and 1,600 Hz (turtles completely submerged) and aerial signals between 50 and 800 Hz, with maximum sensitivity between 200 and 400 Hz underwater and 300 and 400 Hz in air (Dow Piniak et al., 2012; Piniak, 2012).

4.3.11 Hawksbill Turtle

The hawksbill turtle is small to medium-sized compared to other sea turtle species. Adults weigh 100-150 lbs (45 to 68 kg) on average, but can grow as large as 200 lbs (91 kg). Hatchlings weigh about 0.5 oz (14 g). The carapace (top shell) of an adult ranges from 25 to 35 inches (63 to 90 cm) in length and has a "tortoiseshell" coloring, ranging from dark to golden brown, with streaks of orange, red, and/or black. The shells of hatchlings are 1-2 inches (about 42 mm) long and are mostly brown and somewhat heart-shaped. The plastron (bottom shell) is clear yellow. The rear edge of the carapace is almost always serrated, except in older adults, and has overlapping "scutes".

Male hawksbills mature when they are about 27 inches (69 cm) long. Females mature at about 31 inches (78 cm). The ages at which turtles reach these lengths are unknown. Female hawksbills return to their natal beaches every 2-3 years to nest at night approximately every 14-16 days during the nesting season. A female hawksbill generally lays 3-5 nests per season, which contain an average of 130 eggs. Hawksbill turtles usually nest high up on the beach under or in the beach/dune vegetation on both calm and turbulent beaches. They commonly nest on pocket beaches, with little or no sand.

4.3.11.1 Distribution

Hawksbill turtles occur in tropical and subtropical seas of the Atlantic, Pacific, and Indian Oceans. The species is widely distributed in the Caribbean Sea and western Atlantic Ocean, with individuals from several life history stages occurring regularly along southern Florida and the northern Gulf of Mexico (especially Texas); in the Greater and Lesser Antilles; and along the Central American mainland south to Brazil. Within the United States, hawksbills are most common in Puerto Rico and its associated islands, and in the U.S. Virgin Islands.

In the continental U.S., hawksbill sea turtles have been reported in every state on the coast of the Gulf of Mexico and along the coast of the Atlantic Ocean from Florida to Massachusetts, except

for Connecticut; however, sightings of hawksbill sea turtles north of Florida are rare. The only states where hawksbill sea turtles occur with any regularity are Florida (particularly in the Florida Keys and the reefs off Palm Beach County on Florida's Atlantic coast, where the warm waters of the Gulf Stream pass close to shore) and Texas. In both of these states, most sightings are of post-hatchlings and juveniles that are believed to have originated from nesting beaches in Mexico.

Within United States territories and U.S. dependencies in the Caribbean Region, hawksbill sea turtles nest principally in Puerto Rico and the U.S. Virgin Islands, particularly on Mona Island and Buck Island. They also nest on other beaches on St. Croix, Culebra Island, Vieques Island, mainland Puerto Rico, St. John, and St. Thomas. Within the continental United States, hawksbill sea turtles nest only on beaches along the southeast coast of Florida and in the Florida Keys.

The largest populations live in the Caribbean Sea, the Seychelles, Indonesia, and Australia. There are no hawksbills in the Mediterranean Sea (James R. Spotila, 2004). Some adults make long oceanic migrations between feeding and nesting areas but juveniles typically occur on shallow reefs (Bjorndal, 1997). Hawksbills were once thought to be non-migratory residents of reefs adjacent to their nesting beaches, but recent tagging, telemetry, and genetic studies confirm that hawksbills migrate hundreds to thousands of kilometers between feeding and nesting grounds (P. Plotkin, 2003). While the migratory habits of hawksbills are still largely unknown, females appear to migrate between nearshore foraging grounds and their natal beach. No apparent patterns have emerged to explain why some females migrate short distances, while others migrate greater distances, bypassing reefs close to their nesting beaches (P. Plotkin, 2003) (James R. Spotila, 2004).

Hawksbill sea turtles occupy different habitats depending on their life history stage. After entering the sea, hawksbill sea turtles occupy pelagic waters and occupy weed-lines that accumulate at convergence points. When they grow to about 20-25 cm carapace length, hawksbill sea turtles reenter coastal waters where they inhabit and forage in coral reefs as juveniles, subadults and adults. Hawksbill sea turtles also occur around rocky outcrops and high energy shoals, where sponges grow and provide forage, and they are known to inhabit mangrove-fringed bays and estuaries, particularly along the eastern shore of continents where coral reefs are absent (H. Hildebrand, 1987).

4.3.11.2 Population Structure

Hawksbill sea turtles, like other sea turtles, are divided into regional groupings that represent major oceans or seas: the Atlantic Ocean, Pacific Ocean, Indian Ocean, Caribbean Sea and Mediterranean Sea. In these regions, the population structure of hawksbill turtles is usually based on the distribution of their nesting aggregations. Although there is a lack of data to determine good population estimates, the best estimate of the number of annual nesting females worldwide is 21,212 to 28,138 turtles, which represents about 83 nesting areas (NMFS & USFWS, 2007b). The largest nesting populations in the Pacific Ocean occurs in eastern Australia with some 6,500

females nesting per year, in the Atlantic Ocean Yucatan Peninsula, Mexico and Cuba have 534 to 891 and 400 to 833 females nesting, respectively, and in the Indian Ocean, about 2,000 females nest in western Australia and 1,000 nest in Madagascar annually (NMFS & USFWS, 2007b). Although very few hawksbills nest in U.S. waters, nesting does occur on four Puerto Rico locations (120 to 200 female turtles annually), U.S. Virgin Islands (56 to 222 females annually), Hawaii (5 to 10 females annually), and fewer than 10 females annually in the north Pacific U.S. territories (James R. Spotila, 2004) (NMFS & USFWS, 2007b).

4.3.11.3 Natural Threats

The various habitat types hawksbill sea turtles occupy throughout their lives exposes these sea turtles to a wide variety of natural threats. The beaches on which hawksbill sea turtles nest and the nests themselves are threatened by hurricanes and tropical storms as well as the storm surges, sand accretion, and rainfall that are associated with hurricanes. Hatchlings are hunted by predators like herons, gulls, dogfish, and sharks. Adult hawksbill sea turtles are also killed by sharks and other large, marine predators.

4.3.11.4 Anthropogenic Threats

Three human activities are known to threaten hawkbill sea turtles: overharvests of individual animals, incidental capture in commercial fisheries, and human development of coastlines. Historically, the primary cause of the global decline of hawkbill sea turtle populations was overharvest by humans for subsistence and commercial purposes. In the Atlantic, hawksbill sea turtles are still captured and killed in turtle fisheries in Colombia, Grenada, the Lesser Antilles, Nicaragua, St. Vincent and the Grenadines (Brautigam & Eckert, 2006).

For centuries, hawksbill sea turtles have been captured for their shells, which have commercial value, rather than food (the meat of hawksbill sea turtles is considered to have a bad taste and can be toxic to humans) (NMFS & USFWS, 1993, 1998d). Until recently, tens of thousands of hawksbills were captured and killed each year to meet demand for jewelry, ornamentation, and whole stuffed turtles (Milliken and Tokunga 1987 cited in K.L. Eckert, 1993). In 1988, Japan's imports from Jamaica, Haiti, and Cuba represented some 13,383 hawksbills: it is extremely unlikely that this volume could have originated solely from local waters (Greenpeace 1989 cited in K.L. Eckert, 1993).

Although Japan banned the importation of turtle shell in 1994, domestic harvests of eggs and turtles continue in the United States, its territories, and dependencies, particularly in the Caribbean and Pacific Island territories. Large numbers of nesting and foraging hawksbill sea turtles are captured and killed for trade in Micronesia, the Mexican Pacific coast, southeast Asia and Indonesia (NMFS & USFWS, 1993, 1998d). In addition to the demand for the hawksbill's shell, there is a demand for other products including leather, oil, perfume, and cosmetics. Before the U.S. certified Japan under the Pelly Amendment, Japan had been importing about 20 metric tons of hawksbill shell per year, representing approximately 19,000 turtles.

The second most important threat to hawksbill sea turtles is the loss of nesting habitat caused by the expansion of resident human populations in coastal areas of the world and increased destruction or modification of coastal ecosystems to support tourism. Hawksbill sea turtles are also captured and killed in commercial fisheries. Along the Atlantic coast of the U.S., NMFS estimated that about 650 hawksbill sea turtles are captured in shrimp trawl fisheries each year in the Gulf of Mexico, with most of those sea turtles dying as a result of their capture. Each year, about 35 hawksbill sea turtles are captured in Atlantic pelagic longline fisheries. Although most of these turtles are released alive, these fisheries are expected to kill about 50 hawksbill sea turtles each year; the health effects of being captured on the sea turtles that survive remain unknown.

Like green sea turtles, hawksbill sea turtles are threatened by domestic or domesticated animals that prey on their nests; artificial lighting that disorients adult female and hatchling sea turtles, which can dramatically increase the mortality rates of hatchling sea turtles; beach replenishment; ingestion and entanglement in marine debris; and environmental contaminants.

Oil spills are a risk for all sea turtles. Several aspects of sea turtles life histories put them at risk, including the lack of avoidance behavior of oiled waters and indiscriminate feeding in convergence zones. Sea turtles are air breathers and all must come to the surface frequently to take a breath of air. In a large oil spill, these animals may be exposed to volatile chemicals during inhalation.

Additionally, sea turtles may experience oiling impacts on nesting beaches when they come ashore to lay their eggs, and their eggs may be exposed during incubation potentially resulting in increased egg mortality and/or possibly developmental defects in hatchlings. Hatchlings emerging from their nests may encounter oil on the beach and in the water as they begin their lives at sea.

Oil and other chemicals on skin and body may result in skin and eye irritation, burns to mucous membranes of eyes and mouth, and increased susceptibility to infection.

Inhalation of volatile organics from oil or dispersants may result in respiratory irritation, tissue injury, and pneumonia. Ingestion of oil or dispersants may result in gastrointestinal inflammation, ulcers, bleeding, diarrhea, and maldigestion. Absorption of inhaled and ingested chemicals may damage organs such as the liver or kidney, result in anemia and immune suppression, or lead to reproductive failure or death.

4.3.11.5 Status and Trends

Hawksbill sea turtles were listed as endangered under the ESA in 1970. Under Appendix I of the Convention on International Trade in Endangered Species of Wild Fauna and Flora, hawksbill sea turtles are identified as “most endangered.”

Hawksbill sea turtles are solitary nesters, which makes it difficult to estimate the size of their populations. There are no global estimates of the number of hawksbill sea turtles, but a minimum of 15,000 to 25,000 females are thought to nest annually in more than 60 geopolitical entities (Groombridge & Luxmoore, 1989). Moderate populations appear to persist around the Solomon Islands, northern Australia, Palau, Persian Gule islands, Oman, and parts of the Seychelles. In a more recent review, Groombridge and Luxmoore (1989) list Papua New Guinea, Queensland, and Western Australia as likely to host 500-1,000 nesting females per year, while Indonesia and the Seychelles may support >1,000 nesting females. The largest known nesting colony in the world is located on Milman Island, Queensland, Australia where Loop (1995) tagged 365 hawksbills nesting within an 11 week period. With the exception of Mexico, and possibly Cuba, nearly all Wider Caribbean countries are estimated to receive <100 nesting females per year.

Of the 65 geopolitical units on which hawksbill sea turtles nest and where hawksbill nesting densities can be estimated, 38 geopolitical units have hawksbill populations that are suspected or known to be declining. Another 18 geopolitical units have experienced well-substantiated declines (NMFS & USFWS, 1995). The largest remaining nesting concentrations occur on remote oceanic islands off Australia (Torres Strait) and the Indian Ocean (Seychelles).

Hawksbill sea turtles, like green sea turtles, are thought to be declining globally as a direct consequence of a historical combination of overexploitation and habitat loss. However, like several of the species we have already discussed, the information available on the status and trend of hawksbill sea turtles does not allow us to make definitive statements about the global extinction risks facing these sea turtles or the risks facing particular populations (nesting aggregations) of these turtles. However, the limited data available suggests that several hawksbill sea turtles populations exist at sizes small enough to be classified as “small” populations (that is, populations that exhibit population dynamics that increase the extinction probabilities of the species or several of its populations) while others are large enough to avoid these problems. Exogenous threats such as overharvests and entanglement in fishing gear only increase their probabilities of becoming extinct in the foreseeable future.

4.3.11.6 Diving Behavior

The duration of foraging dives in hawksbill sea turtles commonly depends on the size of the turtle: larger turtles diving deeper and longer. At a study site in the northern Caribbean, foraging dives were made only during the day and dive durations ranged from 19-26 minutes in duration at depths of 8-10 m. At night, resting dives ranged from 35-47 minutes in duration (vanDam & Diez, 1997).

4.3.11.7 Vocalizations and Hearing

Sea turtles do not appear to use sound for communication, and there are no published recordings of hawksbill sea turtle vocalizations. Nesting leatherback turtles have been recorded producing sounds (sighs, grunts or belch-like sounds) up to 1,200 Hz with maximum energy from 300 to 500 Hz (Cook & Forrest, 2005), however these sounds appeared to be associated with breathing

(Mrosovsky, 1972);(Cook & Forrest, 2005).

Recent research measureing hatchling hawksbill sea turtle auditory evoked potentials has shown that aerial and underwater acoustic stimuli elicited auditory evoked potential responses between and 50 and 1,600 Hz (underwater fully submerged and in air), with maximum sensitivity between 200 and 400 Hz in hatchling hawksbill (Dow Piniak et al., 2012; Piniak, 2012).

4.3.12 Loggerhead Turtle

Loggerheads were named for their relatively large heads, which support powerful jaws and enable them to feed on hard-shelled prey, such as whelks and conch. The carapace (top shell) is slightly heart-shaped and reddish-brown in adults and sub-adults, while the plastron (bottom shell) is generally a pale yellowish color. The neck and flippers are usually dull brown to reddish brown on top and medium to pale yellow on the sides and bottom. Mean straight carapace length of adults in the southeastern U.S. is approximately 36 in (92 cm); corresponding weight is about 250 lbs (113 kg).

Loggerheads reach sexual maturity at around 35 years of age. In the southeastern U.S., mating occurs in late March to early June and females lay eggs between late April and early September. Females lay three to five nests, and sometimes more, during a single nesting season. The eggs incubate approximately two months before hatching sometime between late June and mid-November.

4.3.12.1 Distribution

Loggerheads are circumglobal occurring throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian oceans. Loggerheads are the most abundant species of sea turtle found in US coastal waters.

Atlanic Ocean. Western Atlantic nesting locations include The Bahamas, Brazil, and numerous locations from the Yucatán Peninsula to North Carolina (Addison, 1997; Addison & Morford, 1996; Marcovaldi & Chaloupka, 2007). This group comprises five nesting subpopulations: Northern, Southern, Dry Tortugas, Florida Panhandle, and Yucatán. Additional nesting occurs on Cay Sal Bank (Bahamas), Cuba, the Bahamian Archipelago, Quintana Roo (Yucatan Peninsula), Colombia, Brazil, Caribbean Central America, Venezuela, and the eastern Caribbean Islands. Genetic studies indicate that, although females routinely return to natal beaches, males may breed with females from multiple populations and facilitate gene flow Bowen et al. (2005). In the eastern Atlantic, we know of five rookeries from Cape Verde, Greece, Libya, Turkey, and the western Africa coast.

Indian Ocean. Loggerhead sea turtles nest along the Indian Ocean in Oman, Yemen, Sri Lanka, Madagascar, South Africa, and possibly Mozambique.

Pacific Ocean. Pacific Ocean rookeries are limited to the western portion of the basin. These sites include Australia, New Caledonia, New Zealand, Indonesia, Japan, and the Solomon Islands.

4.3.12.2 Population Sturcture

Population structure in the Pacific is comprised of a northwestern Pacific nesting aggregation in Japan and a smaller southwestern nesting aggregation in Australia and New Caledonia (NMFS, 2006e). Genetics of Japanese nesters suggest that this subpopulation is comprised of genetically distinct nesting colonies (H. Hatase et al., 2002). Almost all loggerheads in the North Pacific seem to stem from Japanese nesting beaches (B.W Bowen et al., 1995; Resendiz, Resendiz, Nichols, Seminoff, & Kamezaki, 1998). The fidelity of nesting females to their nesting beach allowed differentiation of these subpopulations and the loss of nesting at a beach means a significant loss of diversity and the beach is unlikely to be recolonized (NMFS, 2006e).

Loggerhead nesting is confined to lower latitudes temperate and subtropic zones but absent from tropical areas (NMFS and USFWS, 1991b; NRC, 1990; B.E. Witherington, Herren, & Bresette, 2006b). The life cycle of loggerhead sea turtles can be divided into seven stages: eggs and hatchlings, small juveniles, large juveniles, subadults, novice breeders, first year emigrants, and mature breeders (Crouse, Crowder, & Caswell, 1987). Hatchling loggerheads migrate to the ocean (to which they are drawn by near ultraviolet light Kawamura, Naohara, Tanaka, Nishi, & Anraku, 2009), where they are generally believed to lead a pelagic existence for as long as 7-12 years (NMFS, 2005a). Loggerheads in the Mediterranean, similar to those in the Atlantic, grow at roughly 11.8 cm/yr for the first six months and slow to roughly 3.6 cm/yr at age 2.5-3.5. As adults, individuals may experience a secondary growth pulse associated with shifting into neritic habitats, although growth is generally monotypic (declines with age Casale, D'Astore, & Argano, 2009; Casale, Mazaris, Freggi, Vallini, & Argano, 2009). Individually-based variables likely have a high impact on individual-to-individual growth rates (Casale, Mazaris, et al., 2009). At 15-38 years, loggerhead sea turtles become sexually mature, although the age at which they reach maturity varies widely among populations (Casale, Mazaris, et al., 2009; N. B. Frazer & Ehrhart, 1985; N. B. Frazer, Limpus, & Greene, 1994; NMFS, 2001; B.E. Witherington et al., 2006b). However, based on new data from tag returns, strandings, and nesting surveys, NMFS (2001) estimated ages of maturity ranging from 20-38 years and benthic immature stage lasting from 14-32 years.

Loggerhead mating likely occurs along migration routes to nesting beaches, as well as in offshore from nesting beaches several weeks prior to the onset of nesting (C. K. Dodd, 1988; NMFS and USFWS, 1998d). Females usually breed every 2-3 years, but can vary from 1-7 years (C. K. Dodd, 1988; T. H. Richardson, Richardson, Ruckdeshel, & Dix, 1978). Females lay an average of 4.1 nests per season (Murphy & Hopkins, 1984) , although recent satellite telemetry from nesting females along southwest Florida support 5.4 nests per female per season, with increasing numbers of eggs per nest during the course of the season (Tucker, 2009). The authors

suggest that this finding warrants revision of the number of females nesting in the region. The western Atlantic breeding season is March-August. Nesting sites appear to be related to beaches with relatively high exposure to wind or wind-generated waves (Santana Garcon, Grech, Moloney, & Hamann, 2010).

The Japanese rookeries are the most significant nesting sites for loggerheads in the North Pacific, with nesting occurring on the Japanese mainland, except for Hokkaido, as well as the Ryukyu Islands to the south (Kamezaki, 1989; Kamezaki et al., 2003; Sea Turtle Association of Japan, 2010; Uchida & Nishiwaki, 1995). Nesting generally occurs through summer and fall (April-August, peaking in July), with females returning every two to three years (Iwamoto, Ishii, Nakashima, Takeshita, & Itoh, 1985). Nesting females lay at least three nests of 60-115 eggs per nest each season, with roughly two weeks between nests (K. L. Eckert, 1993; Iwamoto et al., 1985; Nishimura, 1994). Between nests, females appear to swim offshore into the Kuroshio Current, possibly to speed egg development (NMFS and USFWS, 1998; Sato et al., 1998).

Nesting in the Gulf of Mexico does occur, although primarily in Florida, with rare nests occuring along North and South Padre Island in Texas (C. K. J. Dodd, 1988; H. H. Hildebrand, 1983).

4.3.12.3 Migration and Movement

Loggerhead hatchlings migrate offshore and become associated with *Sargassum* spp. habitats, driftlines, and other convergence zones (A. F. Carr, 1986). After 14-32 years of age, they shift to a benthic habitat, where immature individuals forage in the open ocean and coastal areas along continental shelves, bays, lagoons, and estuaries (B.W. Bowen et al., 2004; NMFS, 2001). Adult loggerheads make lengthy migrations from nesting beaches to foraging grounds (TEWG, 1998b). In the Gulf of Mexico, larger females tend to disperse more broadly after nesting than smaller individuals, which tend to stay closer the nesting location (Girard, Tucker, & Calmettes, 2009). In the North Atlantic, loggerheads travel north during spring and summer as water temperatures warm and return south in fall and winter, but occur offshore year-round assuming adequate temperature. For immature individuals, this movement occurs in two patterns: a north-south movement over the continental shelf with migration south of Cape Hatteras in winter and movement north along Virginia for summer foraging, and a not-so-seasonal oceanic dispersal into the Gulf Stream as far north as the 10-15˚ C isotherm (K. L. Mansfield, Saba, Keinath, & Musick, 2009). Wallace et al. (2009) suggested differences in growth rate based upon these foraging strategies. There is conflicting evidence that immature loggerheads roam the oceans in currents and eddies and mix from different natal origins or distribute on a latitudinal basis that corresponds with their natal beaches (Monzon-Arguello et al., 2009; B. P. Wallace et al., 2009). McCarthy et al. (2010) found that movement patterns of loggerhead sea turtles were more convoluted when sea surface temperatures were higher, ocean depths shallower, ocean currents stronger, and chlorophyll a levels lower.

Individuals in the western Pacific also show wide-ranging movements. Loggerheads hatched on beaches in the southwest Pacific travel have been found to range widely in the southern portion

of the basin, with individuals from populations nesting in Australia found as far east as Peruvian coast foraging areas still in the juvenile stage (Boyle et al., 2009). Individuals hatched along Japanese coasts have been found to migrate to waters off Baja California via the North Pacific Subtropical Gyre (and the Kuroshio Extension) to feed for several years before migrating back to western Pacific waters to breed (B.W Bowen et al., 1995; Nichols, 2005; J. Polovina et al., 2006; J.J. Polovina, Kobayashi, Parker, Seki, & Balazs, 2000; Resendiz et al., 1998). Adult loggerheads also reside in oceanic waters off Japan (Hatase, Matsuzawa, Sakamoto, Baba, & Miyawaki, 2002). Habitat use off Japan may further be partitioned by sex and size (H. Hatase, Y. Matsuzawa, et al., 2002; Hatase & Sakamoto, 2004; H. Hatase, N. Takai, et al., 2002). Loggerheads returning to Japanese waters seem to migrate along nutrient-rich oceanic fronts (Kobayashi et al., 2008; Nichols, Resendiz, Seminoff, & Resendiz, 2000; J.J. Polovina et al., 2000). Individuals bycaught and satellite tracked in Hawaii longline fisheries show individual movement north and south within a thermal range of 15-25º C, or 28-40º N, with juveniles following the 17-20º C isotherm (Kobayashi et al., 2008; Nichols et al., 2000; J. J. Polovina et al., 2004). The Transition Zone Chlorophyll Front and Kuroshio Extension Current are likely important foraging areas for juvenile loggerheads (J. J. Polovina et al., 2004). The Kuroshio Current off Japan may be significant for juvenile and adult loggerheads as a wintering areas for those individuals not migrating south (H. Hatase, N. Takai, et al., 2002).

Sighting and stranding records support loggerhead sea turtles to be common, year-round residents of the Gulf of Mexico, although their abundance is much greater in the northeastern region versus the northwestern (R.W. Davis, Evans, & Würsig, 2000; T.H. Fritts, Hoffman, & McGehee, 1983; Landry & Costa, 1999). Loggerheads may occur in both offshore habitats (particularly around oil platforms and reefs, where prey and shelter are available; (R.W. Davis et al., 2000; T.H. Fritts et al., 1983; Gitschlag & Herczeg, 1994; Lohoefener, Hoggard, Mullin, Roden, & Rogers, 1990; Rosman, Boland, Martin, & Chandler, 1987), as well as shallow bays and sounds (which may be important developmental habitat for late juveniles in the eastern Gulf of Mexico; (R.W. Davis et al., 2000; Lohoefener et al., 1990; USAF, 1996). Offshore abundance in continental slope waters increases during the winter in the eastern Gulf of Mexico, as cooler inshore waters force individuals into warmer offshore areas (R.W. Davis et al., 2000).

4.3.12.4 Natural Threats

Sea turtles face predation primarily by sharks and to a lesser extent by killer whales. All sea turtles except leatherbacks can undergo “cold stunning” if water temperatures drop below a threshold level, which can pose lethal effects. In January 2010, an unusually large cold-stunning event occurred throughout the southeast US, with well over 3,000 sea turtles (mostly greens but also hundreds of loggerheads) found cold-stunned. Most survived, but several hundred were found dead or died after being discovered in a cold-stunned state. Eggs are commonly eaten by raccoons and ghost crabs along the eastern US (Barton & Roth, 2008). In the water, hatchlings are hunted by herons, gulls, dogfish, and sharks. Heavy loads of barnacles are associated with unhealthy or dead stranded loggerheads (Deem et al., 2009).

4.3.12.5 Anthropogenic Threats

Anthropogenic threats impacting loggerhead nesting habitat are numerous: coastal development and construction, placement of erosion control structures, beachfront lighting, vehicular and pedestrian traffic, sand extraction, beach erosion, beach nourishment, beach pollution, removal of native vegetation, and planting of non-native vegetation (R. M. Baldwin, 1992; Margaritoulis et al., 2003; A. D. Mazaris, Matsinos, & Pantis, 2009; USFWS, 1998). Surprisingly, beach nourishment also hampers nesting success, but only in the first year post-nourishment before hatching success increases (Brock, Reece, & Ehrhart, 2009). Loggerhead sea turtles face numerous threats in the marine environment as well, including oil and gas exploration, marine pollution, trawl, purse seine, hook and line, gill net, pound net, longline, and trap fisheries, underwater explosions, dredging, offshore artificial lighting, power plant entrapment, entanglement in debris, ingestion of marine debris, marina and dock construction and operation, boat collisions, and poaching. At least in the Mediterannean Sea, Anthorpogenic threats appear to disproportionally impact larger (more fecund) loggerheads (Bellido et al., 2010).

The major factors inhibiting their recovery include mortalities caused by fishery interactions and degradation of the beaches on which they nest. Shrimp trawl fisheries account for the highest number of captured and killed loggerhead sea turtles. Along the Atlantic coast of the US, the NMFS estimated that shrimp trawls capture almost 163,000 loggerhead sea turtles each year in the Gulf of Mexico, of which 3,948 die. However, more recent estimates from suggest interactions and mortality has decreased from pre-regulatory periods, with a conservative estimate of 26,500 loggerheads captured annually in US Atlantic fisheries causing mortality to 1,400 individuals per year (Finkbeiner et al., 2011a). Pacific bycatch is much less, with about 400 individuals bycaught annually in US fisheries resulting in at least 20 mortalities (Finkbeiner et al., 2011a). Each year, various fisheries capture about 2,000 loggerhead sea turtles in Pamlico Sound, of which almost 700 die. As a result of the 2006 and 2007 tri-national fishermen's exchanges in 2007 a prominent Baja California Sur fleet retired its bottom-set longlines (S. Hoyt Peckham et al., 2008) (S. H. Peckham et al., 2011). Prior to this closure, the longline fleet interacted with an estimated 1,160-2,174 loggerheads annually, with nearly all (89 percent) of the takes resulting in mortalities (S. Hoyt Peckham et al., 2008). Offshore longline tuna and swordfish longline fisheries are also a serious concern for the survival and recovery of loggerhead sea turtles and appear to affect the largest individuals more than younger age classes (R. Aguilar, Mas, & Pastor, 1995; A.B. Bolten, K.A. Bjorndal, & Martins, 1994; Carruthers, Schneider, & Neilson, 2009; Howell, Kobayashi, Parker, Balazs, & Polovina, 2008; Marshall, Moss, & Guzman, 2009; Petersen, Honig, Ryan, Nel, & Underhill, 2009; Tomás, Gozalbes, Raga, & Godley, 2008). Deliberate hunting of loggerheads for their meat, shells, and eggs has declined from previous exploitation levels, but still exists and hampers recovery efforts (Lino, Gonçalves, & Cozens, 2010). In the Pacific, loggerhead turtles are captured, injured, or killed in numerous Pacific fisheries including:

- Japanese longline fisheries in the western Pacific Ocean and South China Seas

- direct harvest and commercial fisheries off Baja California, Mexico
- commercial and artisanal swordfish fisheries off Chile, Columbia, Ecuador, and Peru
- purse seine fisheries for tuna in the eastern tropical Pacific Ocean
- California/Oregon drift gillnet fisheries (NMFS, 2006e)

Wallace et al. (2010) estimated that between 1990 and 2008, at least 85,000 sea turtles were captured as bycatch in fisheries worldwide. This estimate is likely at least two orders of magnitude low, resulting in a likely bycatch of nearly half a million sea turtles annually (Bryan P. Wallace et al., 2010); many of these are expected to be loggerhead sea turtles.

Marine debris ingestion can be a widespread issue for loggerhead sea turtles. More than one-third of loggerheads found stranded or bycaught had injected marine debris in a Mediterranean study, with possible mortality resulting in some cases (Lazar & Gračan, 2010).

Climate change may also have significant implications on loggerhead populations worldwide. In addition to potential loss of nesting habitat due to sea level rise, loggerhead sea turtles are very sensitive to temperature as a determinant of sex while incubating. Ambient temperature increase by just 1º-2º C can potentially change hatchling sex ratios to all or nearly all female in tropical and subtropical areas (L. A. Hawkes, A. Broderick, M. H. Godfrey, & B. J. Godley, 2007). Over time, this can reduce genetic diversity, or even population viability, if males become a small proportion of populations (Hulin, Delmas, Girondot, Godfrey, & Guillon, 2009). Sea surface temperatures on loggerhead foraging grounds correlate to the timing of nesting, with higher temperatures leading to earlier nesting (Antonios D. Mazaris, Kallimanis, Tzanopoulos, Sgardelis, & Pantis, 2009; Schofield et al., 2009). Increasing ocean temperatures may also lead to reduced primary productivity and eventual food availability. This has been proposed as partial support for reduced nesting abundance for loggerhead sea turtles in Japan; a finding that could have broader implications for other populations in the future if individuals do not shift feeding habitat (Chaloupka, Kamezaki, & Limpus, 2008). Warmer temperatures may also decrease the energy needs of a developing embryo (K. A. Reid, Margaritoulis, & Speakman, 2009).

Tissues taken from loggerheads sometimes contain very high levels of organochlorines chlorobiphenyl, chlordanes, lindane, endrin, endosulfan, dieldrin, PFOS, PFOA, DDT, and PCB (Alava et al., 2006; S. Corsolini, A. Aurigi, & S. Focardi, 2000; Gardner, Pier, Wesselman, & Juarez, 2003; Keller, Kannan, Taniyasu, Day, et al., 2005; Keller, Kucklick, Harms, & McClellan-Green, 2004; Keller, Kucklick, & McClellan-Green, 2004; McKenzie, Godley, Furness, & Wells, 1999; Monagas, Oros, Anana, & Gonzalez-Diaz, 2008; Oros, Gonzalez-Diaz, & Monagas, 2009; Perugini et al., 2006; M. J. Rybitski, R. C. Hale, & J. A. Musick, 1995; M. Storelli, M. G. Barone, & G. O. Marcotrigiano, 2007). It appears that levels of organochlorines have the potential to suppress the immune system of loggerhead sea turtles and may affect

metabolic regulation (Keller, Kucklick, Stamper, Harms, & McClellan-Green, 2004; J. M. Keller, P. D. McClellan-Green, J. R. Kucklick, D. E. Keil, & M. M. Peden-Adams, 2006; Oros et al., 2009). These contaminants could cause deficiencies in endocrine, developmental, and reproductive health (M. Storelli et al., 2007). It is likely that the omnivorous nature of loggerheads makes them more prone to bioaccumulating toxins than other sea turtle species (B. J. Godley, Thompson, & Furness, 1999; McKenzie et al., 1999).

Heavy metals, including arsenic, barium, cadmium, chromium, iron, lead, nickel, selenium, silver, copper, zinc, and manganese, have also been found in a variety of tissues in levels that increase with turtle size (Anan, Kunito, Watanabe, Sakai, & Tanabe, 2001; Fujihara, Kunito, Kubota, & Tanabe, 2003; Garcia-Fernandez et al., 2009; S. C. Gardner, S. L. Fitzgerald, B. A. Vargas, & L. M. Rodriguez, 2006; B. J. Godley et al., 1999; Saeki, Sakakibara, Sakai, Kunito, & Tanabe, 2000; M. Storelli, M. G. Barone, A. Storelli, & G. O. Marcotrigiano, 2008). These metals likely originate from plants and seem to have high transfer coefficients (Anan et al., 2001; Celik et al., 2006; Talavera-Saenz, Gardner, Rodriquez, & Vargas, 2007).

Loggerhead sea turtles have higher mercury levels than any other sea turtle studied, but concentrations are an order of magnitude less than many toothed whales (B. J. Godley et al., 1999; Pugh & Becker, 2001). Arsenic occurs at levels several fold more concentrated in loggerhead sea turtles than marine mammals or seabirds.

Also of concern is the spread of antimicrobial agents from human society into the marine environment. Loggerhead sea turtles may harbor antibiotic-resistant bacteria, which may have developed and thrived as a result of high use and discharge of antimicrobial agents into freshwater and marine ecosystems (Foti et al., 2009).

4.3.12.6 Status and Trends

Loggerhead sea turtles were initially listed as threatened under the ESA of 1973 on July 28, 1978 (43 FR 32800). On 22 September 2011 NMFS issued a final determination of nine distinct population segments (DPSs) of loggerhead turtles as endangered or threatened: northwest Atlantic Ocean, South Atlantic Ocean, southeast Indo-Pacific Ocean and southwest Indian Ocean as threatened as well as northeast Atlantic Ocean, Mediterranean Sea, North Indian Ocean, North Pacific Ocean, and South Pacific Ocean as endangered (76 FR 58868).

Atlantic Ocean. In the eastern Atlantic, the Cape Verde Islands support the only known loggerhead nesting assemblage, which is of at least intermediate size (J. Fretey, 2001b); 1,071 nests were observed in 2009 (Lino et al., 2010). In 2000, researchers tagged over 1,000 nesting females (Erhart, D.A. Bagley, & Redfoot, 2003). Annual data from monitoring projects in Cyprus, Greece, Israel, Tunisia, and Turkey reveal total annual nesting in the Mediterranean ranging of 3,375-7,085 nests per season (Margaritoulis et al., 2003). Libya and the West African coast host genetically-unique breeding populations of loggerhead sea turtles as well (Hutchinson & Dutton, 2007). A recently discovered nesting site along the southern Italian shores of the

Ionian Sea found particularly high genetic diversity amongst nesting females (Garofalo, Mingozzi, Mico, & Novelletto, 2009). Nesting at Dalyan Beach, Turkey does not have an apparent trend, with between 50 and 286 nests laid annually for the past 19 years (Turkozan & Yilmaz, 2008).

The greatest concentration of loggerheads occurs in the Atlantic Ocean and the adjacent Caribbean Sea, primarily on the Atlantic coast of Florida, with other major nesting areas located on the Yucatán Peninsula of Mexico, Columbia, Cuba, South Africa (EuroTurtle 2006 as cited in LGL Ltd., 2007; M.R. Márquez, 1990).

Among the five subpopulations, loggerhead females lay 53,000-92,000 nests per year in the southeastern US and the Gulf of Mexico, and the total number of nesting females is 32,000-56,000. All of these are currently in decline or data are insufficient to access trends (NMFS, 2001; TEWG, 1998a). Loggerheads from western North Atlantic nesting aggregations may or may not feed in the same regions from which they hatch. Loggerhead sea turtles from the northern nesting aggregation, which represents about 9 percent of the loggerhead nests in the western North Atlantic, comprise 25-59 percent of individuals foraging from Georgia up to the northeast US (Bass, Epperly, Braun, Owens, & Patterson, 1998; Norrgard, 1995; Rankin-Baransky, 1997; C. J. Sears, 1994; C. J. Sears et al., 1995). Loggerheads associated with the South Florida nesting aggregation occur in higher frequencies in the Gulf of Mexico (where they represent ~10 percent of the loggerhead captures) and the Mediterranean Sea (where they represent ~45 percent of loggerhead sea turtles captured). About 4,000 nests per year are laid along the Brazilian coast (L. M. Ehrhart, Bagley, & Redfoot, 2003).

The northern recovery unit along Georgia, South Carolina, and North Carolina has a forty-year time-series trend showing an overall decline in nesting, but the shorter comprehensive survey data (20 years) indicate a stable population (GDNR, NCWRC, and SCDNR nesting data located at www.seaturtle.org). NMFS scientists have estimated that the northern subpopulation produces 65 percent males (NMFS, 2001).

The peninsular Florida recovery unit is the largest loggerhead nesting assemblage in the northwest Atlantic. A near-complete nest census (all beaches including index nesting beaches) undertaken from 1989 to 2007 showed a mean of 64,513 loggerhead nests per year, representing approximately 15,735 nesting females annually (NMFS & USFWS, 2008). The statewide estimated total for 2010 was 73,702 (FWRI nesting database). An analysis of index nesting beach data shows a 26 percent nesting decline between 1989 and 2008, and a mean annual rate of decline of 1.6 percent despite a large increase in nesting for 2008, to 38,643 nests (FWRI nesting database)(NMFS & USFWS, 2008; B. Witherington, Kubilis, Brost, & Meylan, 2009). In 2009, nesting levels, while still higher than the lows of 2004, 2006, and 2007, dropped below 2008 levels to approximately 32,717 nests, but in 2010 a large increase was seen, with 47,880 nests on the index nesting beaches (FWRI nesting database). The 2010 index nesting number is the

largest since 2000. With the addition of data through 2010, the nesting trend for the nowrthwestern Atlantic DPS is slightly negative and not statistically different from zero (no trend)(NMFS and USFWS, 2010). Preliminary, unofficial reports indicate that 2011 nesting may be a high nesting year on par with 2010.

Because of its size, the south Florida subpopulation of loggerheads may be critical to the survival of the species in the Atlantic, and in the past it was considered second in size only to the Oman nesting aggregation (NMFS, 2006e; NMFS and USFWS, 1991b). The South Florida population increased at ~5.3 percent per year from 1978-1990, and was initially increasing at 3.9-4.2 percent after 1990. An analysis of nesting data from 1989-2005, a period of more consistent and accurate surveys than in previous years, showed a detectable trend and, more recently (1998-2005), has shown evidence of a declining trend of approximately 22.3 percent (FFWCC, 2007a, 2007b; B. Witherington et al., 2009). This is likely due to a decline in the number of nesting females within the population (B. Witherington et al., 2009). Nesting data from the Archie Carr Refuge (one of the most important nesting locations in southeast Florida) over the last 6 years shows nests declined from approximately 17,629 in 1998 to 7,599 in 2004, also suggesting a decrease in population size[7]. Loggerhead nesting is thought to consist of just 60 nesting females in the Caribbean and Gulf of Mexico (NMFS, 2006c). Based upon the small sizes of almost all nesting aggregations in the Atlantic, the large numbers of individuals killed in fisheries, and the decline of the only large nesting aggregation, we suspect that the extinction probabilities of loggerhead sea turtle populations in the Atlantic are only slightly lower than those of populations in the Pacific.

Zurita et al. (2003) found a statistically significant increase in the number of nests on seven of the beaches on Quintana Roo, Mexico, from 1987-2001, where survey effort was consistent during the period. However, nesting has declined since 2001, and the previously reported increasing trend appears to have been temporary (NMFS & USFWS, 2008).

Mediterranean Sea. Nesting in the Mediterranean is confined almost exclusively to the eastern basin. The highest level of nesting in the Mediterranean occurs in Greece, with an average of 3,050 nests per year.

Pacific Ocean. Abundance has declined dramatically over the past 10-20 years, although loggerheads range widely from Alaska to Chile (NMFS and USFWS, 1998d). Pacific nesting is limited to two major locations, Australia and Japan, although low level nesting may occur outside of Japan in areas surrounding the South China Sea (S. K. F. Chan et al., 2007; Kamezaki et al., 2003). Eastern Australia supported one of the major global loggerhead nesting assemblages until recently (C. J. Limpus, 1985). Now, less than 500 females nest annually, an 86

[7] While this is a long period of decline relative to the past observed nesting pattern at this location, aberrant ocean surface temperatures complicate the analysis and interpretation of these data. Although caution is warranted in interpreting the decreasing nesting trend given inherent annual fluctuations in nesting and the short time period over which the decline has been noted, the recent nesting decline at this nesting beach is reason for concern.

percent reduction in the size of the annual nesting population in 23 years (C. J. Limpus & Limpus, 2003). The status of loggerhead nesting colonies in southern Japan and the surrounding region is uncertain, but approximately 1,000 female loggerhead turtles may nest there; a 50-90 percent decline compared to historical estimates (A. B. Bolten, Wetherall, Balazs, & Pooley, 1996; Dodd Jr., 1988b; Kamezaki et al., 2003; STAJ, 2002). Nine major nesting beaches (greater than 100 nests per season) and six "submajor" beaches (10–100 nests per season) exists, including Yakushima island where 40 percent of nesting occurs (Kamezaki et al., 2003). Nesting declined from an initial peak of approximately 6,638 nests in 1990–1991, followed by a steep decline to a low of 2,064 nests in 1997 (Conant et al., 2009). During the past decade, nesting increased gradually to 5,167 nests in 2005 (Conant et al., 2009), declined and then rose again to a record high of 11,082 nests in 2008, and then 7,495 and 10,121 nests in 2009 and 2010, respectively (STAJ 2008, 2009, 2010).

In addition, loggerheads uncommonly occur in US Pacific waters, and there have been no documented strandings of loggerheads on the Hawaiian Islands in nearly 20 years (1982-1999 stranding data). There are very few records of loggerheads nesting on any of the many islands of the central Pacific, and the species is considered rare or vagrant in this region (USFWS, 1998). Overall, Gilman (2009) estimated that the number of loggerheads nesting the Pacific has declined by 80 percent in the past 20 years.

Indian Ocean. The largest known nesting aggregation occurs on Masirah and Kuria Muria Islands in Oman (J. P. Ross & Barwani, 1982). Extrapolations resulting from partial surveys and tagging in 1977-1978 provided broad estimates of 19,000-60,000 females nesting annually at Masirah Island, while a more recent partial survey in 1991 provided an estimate of 23,000 nesting females (R. M. Baldwin, 1992; J. P. Ross, 1979, 1998; J. P. Ross & Barwani, 1982). Over 3,000 nests per year have been recorded on the Al-Halaniyat Islands, while along the Oman mainland of the Arabian Sea, about 2,000 nests are deposited per year (R. V. Salm, 1991; R. V. Salm, Jensen, & Papastavrou, 1993). Based upon genetic analyses, additional populations nest in Yemen, Sri Lanka, and Madagascar (Hutchinson and Dutton 2007). In the southwestern Indian Ocean, the highest concentration of nesting occurs on the coast of Tongaland, South Africa (R. Baldwin, Hughes, & Prince, 2003). The total number of females nesting annually in South Africa is estimated to be between 500-2,000 (R. Baldwin et al., 2003). In the Eastern Indian Ocean, all known nesting sites are found in Western Australia (Dodd Jr., 1988a). An estimated 800-1,500 loggerheads nest annually on Dirk Hartog Island beaches along Western Australia (R. Baldwin et al., 2003).

4.3.12.7 Diving Behavior

Studies of loggerhead diving behavior indicate varying mean depths and surface intervals, depending on whether they were located in shallow coastal areas (short surface intervals) or in deeper, offshore areas (longer surface intervals). The maximum recorded dive depth for a post-nesting female was 211-233 meters, while mean dive depths for both a post-nesting female and a

subadult were 9-22 meters. Routine dive times for a post-nesting female were between 15 and 30 minutes, and for a subadult, between 19 and 30 minutes (Sakamot et al. 1990 cited in M. E. Lutcavage & Lutz, 1997). Two loggerheads tagged by Hawaii-based longline observers in the North Pacific and attached with satellite-linked dive recorders were tracked for about 5 months. Analysis of the dive data indicates that most of the dives were very shallow - 70 percent of the dives were no deeper than 5 meters. In addition, the loggerheads spent approximately 40 percent of their time in the top meter and nearly all of their time at depths shallower than 100 meters. On 5 percent of the days, the turtles dove deeper than 100 meters; the deepest daily dive recorded was 178 meters (J. J. Polovina, Howell, Parker, & Balazs, 2003).

Polovina et al. (2004) reported that tagged turtles spent 40 percent of their time at the surface and 90 percent of their time at depths shallower than 40 meters. On only five percent of recorded dive days loggerheads dove to depths greater than 100 meters at least once. In the areas that the loggerheads were diving, there was a shallow thermocline at 50 meters. There were also several strong surface temperature fronts the turtles were associated with, one of 20°C at 28°N latitude and another of 17°C at 32°N latitude.

4.3.12.8 Vocalizations and Hearing

Sea turtles do not appear to use sound for communication, and there are no published recordings of loggerhead sea turtle vocalizations. Nesting leatherback turtles have been recorded producing sounds (sighs, grunts or belch-like sounds) up to 1,200 Hz with maximum energy from 300 to 500 Hz (Cook & Forrest, 2005), however these sounds appeared to be associated with breathing (Cook & Forrest, 2005; Mrosovsky, 1972)

Two studies have been conducted to measure loggerhead sea turtle hearing sensitivity, each using a slightly different methodology. Vibratory stimuli delivered directly to the tympanun produced auditory brainstem responses in loggerheads between 250 and 750 Hz (Bartol et al. 1999). Underwater tones elicited behavioral responses to frequencies between 50 and 800 Hz and auditory evoked potential responses between 100 and 1,131 Hz in one adult loggerhead ((Martin et al., 2012). The lowest threshold recorded in this study was 98 dB re: 1 µPa at 100 Hz.

4.3.13 Olive Ridley Turtle

The olive ridley was named for the olive color of its heart-shaped shell and is one of the smallest of the sea turtles, with adults reaching 2 to 2.5 feet in length and weighing 80 to 110 pounds. The species may be identified by the uniquely high and variable numbers of vertebral and costal scutes. Although some individuals have only five pairs of costals, in nearly all cases some division of costal scutes occurs, so that as many as six to nine pairs may be present. In addition, the vertebral scutes also show frequent division, as do the scales on the dorsal surface of the head.

4.3.13.1 Distribution

Olive ridleys are globally distributed in tropical regions (>20º C) of the Pacific (southern California to Peru, and rarely in the Gulf of Alaska Hodge & Wing, 2000), Indian (eastern Africa and the Bay of Bengal), and Atlantic oceans (Grand Banks to Uruguay and Mauritania to South Africa Foley, Dutton, Singel, Redlow, & Teas, 2003; J. Fretey, 1999; J. Fretey et al., 2005; Stokes & Epperly, 2006). Olive ridleys are uncommon in the western Pacific and western Indian Oceans, and most of the North Atlantic (J.R. Spotila, 2004).

4.3.13.2 Population Structure

Population designations are poorly known. Populations likely correspond somewhat to nesting beach location. Most olive ridleys nest synchronously in huge events called "arribadas", with hundreds to thousands of females nesting over the course of three to seven days; other individuals nest alone, out of sequence with the arribada (M. L. Aprill, 1994; H. Kalb & Owens, 1994).

Atlantic Ocean. Olive ridley distribution in the western North Atlantic occurs mostly along the northern coast of South America and adjacent waters. In the Caribbean, non-nesting individuals occur regularly near Isla Margarita, Trinidad, and Curacao, but are rare further west, such as in Puerto Rico, the Dominican Republic, and Cuba. In rare cases, olive ridleys are known to occur as far north as Puerto Rico, the Dominican Republic, and Cuba and as far south as Brazil (Moncada-G. 2000 as cited in NMFS, 2004a). Regular nesting occurs only in Guyana, Suriname, and French Guiana, with most foraging grounds likely nearby (Reichart 1989 as cited in LGL Ltd., 2007). Nesting occurs along the north coast of Venezuela (Sternberg, 1981). Olive ridleys likely occur in low numbers along western Africa.

Pacific Ocean. Typical distribution is from Peru to California, with rare Alaskan sightings. Peak arribada nesting in the eastern Pacific occurs at several beaches in Mexico, Nicaragua, Costa Rica, and Panama (NMFS and USFWS, 2007f). In Peru, they can be found along the entire coast but are most common in the north, although they are rare in the Galápagos (Kelez, Velez-Zuzzo, Angulo, & Manrique, 2009; Zárate, Parra, Robles, Dutton, & Seminoff, 2010). Olive ridley sea turtles were the most commonly sighted sea turtle during regional seismic surveys funded by the NSF (Hauser, Holst, & Moulton, 2008; Holst & Smultea, 2008; Holst, Smultea, Koski, & Haley, 2005; M. Smultea & Holst, 2003). Tagged Costa Rican nesters have been recovered as far south as Peru, as far north as Oaxaca, Mexico, and offshore to a distance of 2,000 km. Olive ridleys are the most common sea turtle in oceanic waters of the eastern tropical Pacific but move into nearshore waters prior to breeding (R.L. Pitman, 1990). This species frequently basks at the surface, is accompanied by seabirds, and associates with floating debris, from logs to plastic debris to dead whales (Arenas & Hall, 1991a; Pitman 1992 as cited in NMFS, 2004a).

Eastern Pacific nests are most concentrated in southern Mexico and northern Costa Rica, with secondary nesting as far north as southern Baja California (T. H. Fritts, Stinson, & Márquez, 1982) and as far south as Peru (C. H. Brown & Brown, 1982; Kelez et al., 2009). Nesting occurs

year-round, but tends to peak from September through December (NMFS & USFWS, 1998e). Most females lay two clutches of 100-107 eggs with an inter-nesting period of 1–2 months and incubation lasting 50-60 days (K. L. Eckert, 1993; NMFS & USFWS, 1998e; P. T. Plotkin, Byles, & Owens, 1994a). Internesting females tend to stay within 5 km of shore (H. Kalb & Owens, 1994).

Southern Hemisphere. Distribution is poorly known, but nesting colonies occur in the Philippines, Papua New Guinea, and northern Australia (Euroturtle, 2009; Spring, 1982). Solitary nesting beaches occur in Australia, Brunei, Malaysia, Indonesia, and Vietnam (J. R. Spotila, 2004). Olive ridleys have been sighted in Fiji, Vanuatu, French Polynesia, the Solomon and Marshall islands, and Palau (SPREP, 2007). The occurrence of olive ridleys in Tonga and Kiribati is suspected but unconfirmed (SPREP, 2007).

Little is known about olive ridley growth or reproduction. However, some beaches, such as Ostional Beach on the Pacific coast of Costa Rica, is known to have extremely low hatching success, particularly at the onset of the dry season onward, at least partly due to the high temperatures of nests (Valverde et al. 2010).

Olive ridleys are highly migratory and may spend most of their non-breeding life cycle in deep-ocean waters, but occupy the continental shelf region during the breeding season (Arenas & Hall, 1991b; Beavers & Cassano, 1996; S. E. Cornelius & Robinson, 1986; R. L. Pitman, 1991, 1993; P. T. Plotkin, 1994; P. T. Plotkin et al., 1994a; P. T. Plotkin, Byles, Rostal, & Owens, 1995). Reproductively active males and females migrate toward the coast and aggregate at nearshore breeding grounds near nesting beaches (S. E. Cornelius, 1986; Hughes & Richard, 1974; H. Kalb, Valverde, & Owens, 1995; P. T. Plotkin, Byles, Rostal, & Owens, 1991; P. T. Plotkin, Owens, Byles, & Patterson, 1996; P. T. Plotkin, Rostal, Byles, & Owens, 1997; P. C. H. Pritchard, 1969). Other males and females may not migrate to nearshore breeding aggregations at all (Kopitsky, Pitman, & Plotkin, 2000; R. L. Pitman, 1991). Some males appear to remain in oceanic waters, are non-aggregated, and mate opportunistically as they intercept females en route to near shore breeding grounds and nesting beaches (Kopitsky et al., 2000; P. T. Plotkin, 1994; P. T. Plotkin, Byles, & Owens, 1994b; P. T. Plotkin et al., 1996). Their migratory pathways vary annually (P. T. Plotkin, 1994), there is no spatial and temporal overlap in migratory pathways among groups or cohorts of turtles (P. T. Plotkin et al., 1994a; P. T. Plotkin et al., 1995), and no apparent migration corridors exist. Olive ridleys may use water temperature more than any other environmental cue during migrations (J.R. Spotila, 2004). Post-nesting migration routes from Costa Rica traverse more than 3,000 km out into the central Pacific (P. Plotkin, Byles, & Owens, 1993; P. T. Plotkin et al., 1994a). Olive ridleys from different populations may occupy different oceanic habitats (J. J. Polovina et al., 2004; J. J. Polovina et al., 2003). Unlike other marine turtles that migrate from a breeding ground to a single feeding area, where they reside until the next breeding season, olive ridleys are nomadic migrants that swim hundreds to thousands of kilometers over vast oceanic areas (P. T. Plotkin, 1994; P. T. Plotkin et al., 1994a; P. T. Plotkin

et al., 1995). Olive ridleys may associate with flotsam, which could provide food, shelter, and/or orientation cues (Arenas & Hall, 1991b). In the oceanic eastern tropical Pacific, olive ridley sea turtles are far more common than any other cheloniid (R.L. Pitman, 1990).

4.3.13.3 Natural Threats

Sea turtles face predation primarily by sharks and to a lesser extent by killer whales. Natural predators of olive ridleys also include crabs, garrabos, iguanas, crocodiles, black vultures, coyotes, raccoons, and coatis (M.L. Aprill, 1994). All sea turtles except leatherbacks can undergo "cold stunning" if water temperatures drop below a threshold level, which can pose lethal effects.

4.3.13.4 Anthropogenic Threats

Collection of eggs as well as adult turtles has historically led to species decline (NMFS & USFWS, 2007e). Harvests remain a concern for olive ridley recovery. In some locations, takes are now regulated or banned (with varying compliance), while harvests remain uncontrolled in other areas. Adult harvests are now largely banned, except along African coasts.

High levels of adult mortality due to harvesting are believed to be the reason why rapid and large nesting population declines occurred in Mexico (S. E. Cornelius et al., 2007). The nationwide ban on commercial sea turtles harvest in Mexico, enacted in 1990, has greatly aided olive ridley conservation, but the population is still seriously decremented and threatened with extinction (Groombridge, 1982). Several solitary and arribada nesting beaches experience (although banned) egg harvesting, which is causing declines (S. E. Cornelius et al., 2007). Approximately 300,000-600,000 eggs were seized each year from 1995-1998 (Trinidad & Wilson, 2000).

In India, uncontrolled mechanized fishing in areas of high sea turtle concentration, primarily illegally operated trawl fisheries, has resulted in large-scale mortality of adult olive ridley turtles during the last two decades. Since 1993, more than 50,000 olive ridleys have stranded along the coast, at least partially because of near-shore shrimp fishing (Shanker & Mohanty, 1999). In 2008, several hundred olive ridleys stranded dead along Orissa beaches coincident with trawl fisheries operating in the area (Das, 2008). Fishing in coastal waters off Gahirmatha was restricted in 1993 and completely banned in 1997 with the formation of a marine sanctuary around the rookery. However, mortality due to shrimp trawling reached a record high of 13,575 ridleys during the 1997 to 1998 season and none of the approximately 3,000 trawlers operating off the Orissa coast use turtle excluder devices in their nets despite mandatory requirements passed in 1997 (Pandav & Choudhury, 1999). Shrimp trawls off of Central America are estimated capture over 60,000 sea turtles annually, most of which are olive ridleys (Arauz 1996 as cited in NMFS and USFWS, 2007f). Olive ridleys in the eastern Pacific are also incidentally caught by purse seine fisheries and gillnet fisheries (J. Frazier et al., 2007). Wallace et al. (2010) estimated that between 1990 and 2008, at least 85,000 sea turtles were captured as bycatch in fisheries worldwide. This estimate is likely at least two orders of magnitude low, resulting in a

likely bycatch of nearly half a million sea turtles annually (Bryan P. Wallace et al., 2010); many of these turtles are expected to be olive ridley sea turtles.

There are additional impacts to the nesting and marine environment that affect olive ridleys. Structural impacts to nesting habitat include the construction of buildings and pilings, beach armoring and renourishment, and sand extraction (Bouchard et al., 1998; M. E. Lutcavage, Plotkin, Witherington, & Lutz, 1997). The presence of lights on or adjacent to nesting beaches alters the behavior of nesting adults and is often fatal to emerging hatchlings as they are attracted to light sources and drawn away from the water, with up to 50 percent of some olive ridley hatchlings disoriented upon emergence in some years (Karnad, Isvaran, Kar, & Shanker, 2009; B. E. Witherington, 1992b; B. E. Witherington & Bjorndal, 1991b). At sea, there are numerous potential threats including marine pollution, oil and gas exploration, lost and discarded fishing gear, changes in prey abundance and distribution due to commercial fishing, habitat alteration and destruction caused by fishing gear and practices, agricultural runoff, and sewage discharge (J. Frazier et al., 2007; M. E. Lutcavage et al., 1997).

Olive ridley tissues have been found to contain the organochlorines chlordanes, lindane, endrin, endosulfan, dieldrin, DDT, and PCB (Gardner et al., 2003). These contaminants have the potential to cause deficiencies in endocrine, developmental, and reproductive health (M. Storelli et al., 2007), and are known to depress immune function in loggerhead sea turtles (J. M. Keller et al., 2006). Heavy metals, including cadmium, iron, nickel, copper, zinc, and manganese, have been found in a variety of tissues in levels that increase with turtle size (S. C. Gardner, S. L. Fitzgerald, B. A. Vargas, & L. M. Rodriguez, 2006). Females from sexual maturity through reproductive life should have lower levels of contaminants than males because females offload contaminants to their eggs. Newly emerged hatchlings have higher concentrations than are present when laid, suggesting that metals may be accumulated during incubation from surrounding sands (G. Sahoo, R. K. Sahoo, & P. Mohanty-Hejmadi, 1996).

4.3.13.5 Status and Trends

Except for the Mexico breeding stock, olive ridley sea turtles were listed as threatened under the ESA on July 28, 1978 (43 FR 32800). The olive ridley is the most abundant sea turtle in the world (P. C. H. Pritchard, 1997). Worldwide, abundance of nesting female olive ridleys is estimated at two million (J.R. Spotila, 2004).

Pacific Ocean. The eastern Pacific population is believed to number roughly 1.39 million (Eguchi, Gerrodette, Pitman, Seminoff, & Dutton, in preperation). Abundance estimates in recent years indicate that the Mismaloya and Moro Ayuta nesting populations appear to be stable and the nesting population at La Escobilla is increasing, although less than historical levels, which was roughly 10 million adults prior to 1950 (Cliffton, Cornejo, & Felger, 1982; NMFS & USFWS, 2007e). By 1969, after years of adult harvest, the estimate was just over one million (Cliffton et al., 1982). Olive ridley nesting at La Escobilla rebounded from approximately 50,000 nests in 1988 to over 700,000 nests in 1994, and more than a million nests by 2000 (M. R.

Márquez, Carrasco, Jimenez, S., & Bravo-G, 2005; M. R. Márquez, Peñaflores, & Vasconcelos, 1996). The largest known arribadas in the eastern Pacific are on the coast of Costa Rica (~475,000-650,000 females estimated nesting annually) and in southern Mexico (~800,000 nests per year at La Escobilla, in Oaxaca, Mexico). Along Costa Rica, 25,000-50,000 olive ridleys nest at Playa Nancite and 450,000-600,000 turtles nest at Playa Ostional annually (NMFS & USFWS, 1998e). At a nesting site in Costa Rica, an estimated 0.2 percent of 11.5 million eggs laid during a single arribada produced hatchlings (NMFS & USFWS, 1998e). Two of the five arribada beaches in Nicaragua have available estimates – Chacocente at over 42,000 nests and La Flor at 1,300 to 9,000 turtles per arribada (NMFS, 2004a, 2004b). Analysis of bycatch data off Costa Rica suggest a female-biased sex ration of roughly two females for every male (Arauz, 2001).

4.3.13.6 Diving Behavior

Diving behavior remains somewhat of a mystery, but several studies have highlighted general insights. The average dive length for an adult female and male were reported to be 54.3 and 28.5 min, respectively (Plotkin 1994 in Lutcavage and Lutz 1997, as cited in NMFS and USFWS, 2007f). McMahon et al. (2007) reported a maximum dive duration of 200 min (± 20 min) in northern Australia. In the eastern tropical Pacific, diving rate is greater during daytime than at night (Beavers & Cassano, 1996; Parker, Dutton, Kopitsky, & Pitman, 2003). During nighttime however, dives are longer (up to 95 min). In the eastern tropical Pacific, at least 25 percent of olive ridley total dive time is spent in the permanent thermocline, located at 20–100 m (Parker et al., 2003).

Olive ridleys can dive and feed at considerable depths (80–300 m), although ~90 percent of their time is spent at depths <100 m (J. J. Polovina et al., 2003). At least 25 percent of their total dive time is spent in the permanent thermocline, located at 20–100 m (Parker et al., 2003). In the North Pacific Ocean, two olive ridleys tagged with satellite-linked depth recorders spent about 20 percent of their time in the top meter and about 10 percent of their time deeper than 100 m; 70 percent of the dives were no deeper than 5 m (J. J. Polovina et al., 2003).

4.3.13.7 Vocalization and Hearing

Sea turtles do not appear to use sound for communication, and there are no published recordings of olive ridley sea turtle vocalizations. Nesting leatherback turtles have been recorded producing sounds (sighs, grunts or belch-like sounds) up to 1,200 Hz with maximum energy from 300 to 500 Hz (Cook & Forrest, 2005), however these sounds appeared to be associated with breathing (Mrosovsky, 1972);(Cook & Forrest, 2005)

There is no information on olive ridley sea turtle hearing. However we assume that their hearing sensitivities are similar to those reported for green, hawksbill, leatherback and loggerhead sea turtles with best hearing sensitivity at low frequencies (below 400 Hz) and and upper range of hearing not likely to exceed 2000 Hz.

4.3.14 Leatherback Turtle

The leatherback is the largest turtle and the largest living reptile in the world. Mature males and females can be as long as six and a half feet (2 m) and weigh almost 2000 lbs. (900 kg). The leatherback is the only sea turtle that lacks a hard, bony shell. A leatherback's carapace is approximately 1.5 inches (4 cm) thick and consists of leathery, oil saturated connective tissue overlaying loosely interlocking dermal bones. The carapace has seven longitudinal ridges and tapers to a blunt point. Adult leatherbacks are primarily black with a pinkish white mottled ventral surface and pale white and pink spotting on the top of the head. The front flippers lack claws and scales and are proportionally longer than in other sea turtles; back flippers are paddle-shaped. The ridged carapace and large flippers are characteristics that make the leatherback uniquely equipped for long distance foraging migrations.

Female leatherbacks lay clutches of approximately 100 eggs on sandy, tropical beaches. Females nest several times during a nesting season, typically at 8-12 day intervals. After 60-65 days, leatherback hatchlings with white striping along the ridges of their backs and on the margins of the flippers emerge from the nest. Leatherback hatchlings are approximately 50-77 cm (2-3 inches) in length, with fore flippers as long as their bodies, and weigh approximately 40-50 grams (1.4-1.8 ounces).

Leatherbacks lack the crushing chewing plates characteristic of sea turtles that feed on hard-bodied prey (P. C. H. Pritchard, 1971) Instead, they have pointed tooth-like cusps and sharp edged jaws that are perfectly adapted for a diet of soft-bodied pelagic (open ocean) prey, such as jellyfish and salps. A leatherback's mouth and throat also have backward-pointing spines that help retain such gelatinous prey

4.3.14.1 Distribution

Leatherback turtles are widely distributed throughout the oceans of the world. The species is found in four main regions of the world: the Pacific, Atlantic, and Indian Oceans, and the Caribbean Sea. Leatherbacks also occur in the Mediterranean Sea, although they are not known to nest there. The four main regional areas may further be divided into nesting aggregations. Leatherback turtles are found on the western and eastern coasts of the Pacific Ocean, with nesting aggregations in Mexico and Costa Rica (eastern Pacific) and Malaysia, Indonesia, Australia, the Solomon Islands, Papua New Guinea, Thailand, and Fiji (western Pacific). In the Atlantic Ocean, leatherback nesting aggregations have been documented in Gabon, Sao Tome and Principe, French Guiana, Suriname, and Florida. In the Caribbean, leatherbacks nest in the U.S. Virgin Islands and Puerto Rico. In the Indian Ocean, leatherback nesting aggregations are reported in India and Sri Lanka.

Leatherback sea turtles are highly migratory, exploiting convergence zones and upwelling areas in the open ocean, along continental margins, and in archipelagic waters (S. A. Eckert, 1998, 1999; Morreale, Standora, Paladino, & Spotila, 1994). In a single year, a leatherback may swim more than 10,000 kilometers (S. A. Eckert, 1998). In the North Atlantic Ocean, leatherback

turtles regularly occur in deep waters (>328 ft), and an aerial survey study in the north Atlantic sighted leatherback turtles in water depths ranging from 3 to 13,618 ft, with a median sighting depth of 131.6 ft (CETAP, 1982). This same study found leatherbacks in waters ranging from 7 to 27.2°C. In the Pacific Ocean, leatherback turtles have the most extensive range of any living reptile and have been reported in all pelagic waters of the Pacific between 71°N and 47 °S latitude and in all other major pelagic ocean habitats (NMFS & USFWS, 1998a). Leatherback turtles lead a completely pelagic existence, foraging widely in temperate waters except during the nesting season, when gravid females return to tropical beaches to lay eggs. Males are rarely observed near nesting areas, and it has been hypothesized that leatherback sea turtles probably mate outside of tropical waters, before females swim to their nesting beaches (K. L. Eckert & Eckert, 1988).

Leatherback turtles are uncommon in the insular Pacific Ocean, but individual leatherback turtles are sometimes encountered in deep water and prominent archipelagoes. To a large extent, the oceanic distribution of leatherback turtles may reflect the distribution and abundance of their macroplanktonic prey, which includes medusae, siphonophores, and salpae in temperate and boreal latitudes (NMFS & USFWS, 1998a). There is little information available on their diet in subarctic waters.

4.3.14.2 Population Structure

Leatherback turtles are widely distributed throughout the oceans of the world. The species is divided into four main populations in the Pacific, Atlantic, and Indian Oceans, and the Caribbean Sea. Leatherbacks also occur in the Mediterranean Sea, although they are not known to nest there. The four main populations are further divided into nesting aggregations. Leatherback turtles are found on the western and eastern coasts of the Pacific Ocean, with nesting aggregations in Mexico and Costa Rica (eastern Pacific) and Malaysia, Indonesia, Australia, the Solomon Islands, Papua New Guinea, Thailand, and Fiji (western Pacific). In the Atlantic Ocean, leatherback nesting aggregations have been documented in Gabon, Sao Tome and Principe, French Guiana, Suriname, and Florida. In the Caribbean, leatherbacks nest in the U.S. Virgin Islands and Puerto Rico. In the Indian Ocean, leatherback nesting aggregations are reported in India, Sri Lanka, and the Andaman and Nicobar Islands.

4.3.14.3 Natural Threats

The various habitat types leatherback sea turtles occupy throughout their lives exposes these sea turtles to a wide variety of natural threats. The beaches on which leatherback sea turtles nest and the nests themselves are threatened by hurricanes and tropical storms as well as the storm surges, sand accretion, and rainfall that are associated with hurricanes. Hatchlings are hunted by predators like herons, gulls, dogfish, and sharks. Larger leatherback sea turtles, including adults, are also killed by sharks and other large, marine predators.

4.3.14.4 Anthropogenic Threats

Leatherback sea turtles are endangered by several human activities, including fisheries interactions, entanglement in fishing gear (e.g., gillnets, longlines, lobster pots, weirs), direct harvest, egg collection, the destruction and degradation of nesting and coastal habitat, boat collisions, and ingestion of marine debris (NMFS & USFWS, 2007c).

The foremost threat is the number of leatherback turtles killed or injured in fisheries. Spotila (2000) concluded that a conservative estimate of annual leatherback fishery-related mortality (from longlines, trawls and gillnets) in the Pacific Ocean during the 1990s is 1,500 animals. He estimates that this represented about a 23 percent mortality rate (or 33 percent if most mortality was focused on the East Pacific population). Spotila (2000) asserts that most of the mortality associated with the Playa Grande nesting site was fishery related.

Leatherback sea turtles are exposed to commercial fisheries in many areas of the Atlantic Ocean. For example, leatherback entanglements in fishing gear are common in Canadian waters where Goff and Lien (1988) reported that 14 of 20 leatherbacks encountered off the coast of Newfoundland and Labrador were entangled in fishing gear including salmon net, herring net, gillnet, trawl line and crab pot line. Leatherbacks are reported taken by the many other nations that participate in Atlantic pelagic longline fisheries (Petersen et al., 2009), including Taiwan, Brazil, Trinidad, Morocco, Cyprus, Venezuela, Korea, Mexico, Cuba, U.K., Bermuda, People's Republic of China, Grenada, Canada, Belize, France, and Ireland.

In the Pacific Ocean, between 1,000 and 1,300 leatherback sea turtles are estimated to have been captured and killed in longline fisheries in 2000 (Lewison, Freeman, & Crowder, 2004). Shallow-set longline fisheries based out of Hawaii are estimated to have captured and killed several hundred leatherback sea turtles before they were closed in 2001. When they were re-opened in 2004, with substantial modifications to protect sea turtles, these fisheries were estimated to have captured and killed about 1 or 2 leatherback sea turtles each year. Between 2004 and 2008, shallow-set fisheries based out of Hawaii are estimated to have captured about 19 leatherback sea turtles, killing about 5 of these sea turtles. Leatherback sea turtles have also been and are expected to continue to be captured and killed in the deep-set based longline fisheries based out of Hawaii and American Samoa.

Shrimp trawls in the Gulf of Mexico capture the largest number of leatherback sea turtles: each year, they have been estimated to capture about 3,000 leatherback sea turtles with 80 of those sea turtles dying as a result. Along the Atlantic coast of the U.S., NMFS estimated that about 800 leatherback sea turtles are captured in pelagic longline fisheries, bottom longline and drift gillnet fisheries for sharks as well as lobster, deep-sea red crab, Jonah crab, dolphin fish and wahoo, and Pamlico Sound gillnet fisheries. Although most of these turtles are released alive, these fisheries combined kill about 300 leatherback sea turtles each year; the health effects of being captured on the sea turtles that survive remain unknown.

Leatherback sea turtles are known to drown in fish nets set in coastal waters of Sao Tome, West Africa (Castroviejo, Juste, Delval, Castelo, & Gil, 1994; Graff, 1995). Gillnets are one of the suspected causes for the decline in the leatherback turtle population in French Guiana (Chevalier, Desbois, & Girondot, 1999), and gillnets targeting green and hawksbill turtles in the waters of coastal Nicaragua also incidentally catch leatherback turtles (Lagueux, 1998). Observers on shrimp trawlers operating in the northeastern region of Venezuela documented the capture of six leatherbacks from 13,600 trawls (Marcano & Alió-M, 2000). An estimated 1,000 mature female leatherback turtles are caught annually off of Trinidad and Tobago with mortality estimated to be between 50-95 percent (S.A. Eckert & Lien, 1999). However, many of the turtles do not die as a result of drowning, but rather because the fishermen butcher them in order to get them out of their nets (NMFS, 2001). There are known to be many sizeable populations of leatherbacks nesting in West Africa, possibly as many as 20,000 females nesting annually (J. Fretey, 2001a). In Ghana, nearly two thirds of the leatherback turtles that come up to nest on the beach are killed by local fishermen.

On some beaches, nearly 100 percent of the eggs laid have been harvested. Eckert (1997) and Spotila et al. (1996) note that adult mortality has also increased significantly, particularly as a result of driftnet and longline fisheries. Like green and hawksbill sea turtles, leatherback sea turtles are threatened by domestic or domesticated animals that prey on their nests; artificial lighting that disorients adult female and hatchling sea turtles, which can dramatically increase the mortality rates of hatchling sea turtles; beach replenishment; ingestion and entanglement in marine debris; and environmental contaminants.

4.3.14.5 Status and Trends

The leatherback turtles are listed as endangered under the ESA throughout the species' global range. Increases in the number of nesting females have been noted at some sites in the Atlantic Ocean, but these are far outweighed by local extinctions, especially of island populations, and the demise of populations throughout the Pacific, such as in Malaysia and Mexico. Spotila et al. (1996) estimated the global population of female leatherback turtles to be only 34,500 (confidence limits: 26,200 to 42,900) nesting females; however, the eastern Pacific population has continued to decline since that estimate, leading some researchers to conclude that the leatherback is now on the verge of extinction in the Pacific Ocean (e.g., Spotila et al., 1996; Spotila et al., 2000).

Globally, leatherback turtle populations have been decimated worldwide. In 1980, the global leatherback population was estimated at approximately 115,000 adult females (P.C.H. Pritchard, 1982). By 1995, this global population (of adult females) is estimated to have declined to 34,500 (Spotila et al., 1996). Populations have declined in Mexico, Costa Rica, Malaysia, India, Sri Lanka, Thailand, Trinidad, Tobago, and Papua New Guinea. Throughout the Pacific, leatherbacks are seriously declining at all major nesting beaches.

In the Atlantic and Caribbean, the largest nesting assemblages of leatherbacks are found in the U.S. Virgin Islands, Puerto Rico, and Florida. Since the early 1980s, nesting data has been collected at these locations. Populations in the eastern Atlantic (*i.e.* off Africa) and Caribbean appear to be stable; however, information regarding the status of the entire leatherback population in the Atlantic is lacking and it is certain that some nesting populations (*e.g.*, St. John and St. Thomas, U.S. Virgin Islands) have been extirpated (NMFS & USFWS, 1995). Data collected in southeast Florida clearly indicate increasing numbers of nests for the past twenty years (9.1-11.5 percent increase), although it is critical to note that there was also an increase in the survey area in Florida over time (NMFS, 2001). However, the largest leatherback rookery in the western North Atlantic remains along the northern coast of South America in French Guiana and Suriname. Recent information suggests that Western Atlantic populations declined from 18,800 nesting females in 1996 (Spotila et al., 1996) to 15,000 nesting females by 2000 (NMFS, 2001). The nesting population of leatherback turtles in the Suriname-French Guiana trans-boundary region has been declining since 1992 (Chevalier & Girondot, 1998). Poaching and fishing gear interactions are believed to be the major contributors to the decline of leatherbacks in the area.

Leatherback sea turtles appear to be in a critical state of decline in the North Pacific Ocean. The leatherback population that nests along the east Pacific Ocean was estimated to be over 91,000 adults in 1980 (Spotila et al., 1996), but is now estimated to number less than 3,000 total adult and subadult animals (Spotila et al., 2000). Leatherback turtles have experienced major declines at all major Pacific basin rookeries. At Mexiquillo, Michoacan, Mexico, Sarti et al. (1996) reported an average annual decline in nesting of about 23 percent between 1984 and 1996. The number of females nesting on the Pacific coast of Mexico during the 1995-1996 season was estimated at fewer than 1,000. Less than 700 females are estimated for Central America (Spotila et al., 2000). In the western Pacific, the decline is equally severe. Current nestings at Terengganu, Malaysia represent 1 percent of the levels recorded in the 1950s (E. H. Chan & Liew, 1996).

While Spotila et al. (1996) indicated that turtles may have been shifting their nesting from French Guiana to Suriname due to beach erosion, analyses show that the overall area trend in number of nests has been negative since 1987 at a rate of 15.0 -17.3 percent per year (NMFS, 2001). If turtles are not nesting elsewhere, it appears that the Western Atlantic portion of the population is being subjected to mortality beyond sustainable levels, resulting in a continued decline in numbers of nesting females.

Based on published estimates of nesting female abundance, leatherback populations are declining at all major Pacific basin nesting beaches, particularly in the last two decades (NMFS & USFWS, 1998a; Spotila et al., 1996; Spotila et al., 2000). Declines in nesting populations have been documented through systematic beach counts or surveys in Malaysia (Rantau Abang, Terengganu), Mexico and Costa Rica. In other leatherback nesting areas, such as Papua New

Guinea, Indonesia, and the Solomon Islands, there have been no systematic consistent nesting surveys, so it is difficult to assess the status and trends of leatherback turtles at these beaches. In all areas where leatherback nesting has been documented, however, current nesting populations are reported by scientists, government officials, and local observers to be well below abundance levels of several decades ago. The collapse of these nesting populations was most likely precipitated by a tremendous overharvest of eggs coupled with incidental mortality from fishing (Scott A. Eckert & Sarti, 1997; Sarti et al., 1996).

Based on recent modeling efforts, some authors concluded that leatherback turtle populations cannot withstand more than a 1 percent human-related mortality level which translates to 150 nesting females (Spotila et al., 1996). As noted previously, there are many human-related sources of mortality to leatherbacks; every year, 1,800 leatherback turtles are expected to be captured or killed as a result of federally-managed activities in the U.S. (this total includes both lethal and non-lethal take). An unknown number of leatherbacks are captured or killed in fisheries managed by states. Spotila et al. (Spotila et al., 1996) recommended not only reducing fishery-related mortalities, but also advocated protecting eggs and hatchlings. Zug and Parham (Zug & Parham, 1996) point out that a combination of the loss of long-lived adults in fishery-related mortalities and a lack of recruitment stemming from elimination of annual influxes of hatchlings because of intense egg harvesting has caused the sharp decline in leatherback populations.

For several years, NMFS' biological opinions have established that leatherback populations currently face high probabilities of extinction as a result of both environmental and demographic stochasticity. Demographic stochasticity, which is chance variation in the birth or death of an individual of the population, is facilitated by the increases in mortality rates of leatherback populations resulting from the premature deaths of individual sea turtles associated with human activities (either removal of eggs or adult females that are killed on nesting beaches or that die as a result of being captured in fisheries) or incidental capture and mortality of individuals in various fisheries.

In the Pacific Ocean, leatherback sea turtles are critically endangered as a direct consequence of a historical combination of overexploitation and habitat loss. The information available suggests that leatherback sea turtles have high probabilities of becoming extinct in the Pacific Ocean unless they are protected from the combined threats of entanglements in fishing gear, overharvests, and loss of their nesting habitat. The limited data available suggests that leatherback sea turtles exist at population sizes small enough to be classified as "small" populations (that is, populations that exhibit population dynamics that increase the extinction probabilities of the species or several of its populations) as evidenced by biases in the male to female ratios in the Pacific. The status of leatherback sea turtles in the Atlantic Ocean remains uncertain.

4.3.14.6 Diving Behavior

The maximum dive depths for post-nesting female leatherback turtles in the Caribbean have been recorded at 475 meters and over 1,000 meters, with routine dives recorded at between 50 and 84 meters. The maximum dive length recorded for such female leatherback turtles was 37.4 minutes, while routine dives ranged from 4 -14.5 minutes (M. E. Lutcavage & Lutz, 1997). Leatherback turtles also appear to spend almost the entire portion of each dive traveling to and from maximum depth, suggesting that maximum exploitation of the water column is of paramount importance to the leatherback (S. A. Eckert, Eckert, & Richardson, 1989).

A total of six adult female leatherback turtles from Playa Grande, Costa Rica were monitored at sea during their internesting intervals and during the 1995 through 1998 nesting seasons. The turtles dived continuously for the majority of their time at sea, spending 57 - 68 percent of their time submerged. Mean dive depth was 19±1 meters and the mean dive duration was 7.4± 0.6 minutes (Southwood et al., 1999). Similarly, Eckert (1999) placed transmitters on nine leatherback females nesting at Mexiquillo Beach and recorded dive behavior during the nesting season. The majority of the dives were less than 150 meters depth, although maximum depths ranged from 132 meters to over 750 meters. Although the dive durations varied between individuals, the majority of them made a large proportion of very short dives (less than two minutes), although Eckert (1999) speculates that these short duration dives most likely represent just surfacing activity after each dive. Excluding these short dives, five of the turtles had dive durations greater than 24 minutes, while three others had dive durations between 12 - 16 minutes.

Migrating leatherback turtles also spend a majority of time at sea submerged, and they display a pattern of continual diving (Southwood et al., 1999; Standora, Spotila, Keinath, & Shoop, 1984). Based on depth profiles of four leatherbacks tagged and tracked from Monterey Bay, California in 2000 and 2001, using satellite-linked dive recorders, most of the dives were to depths of less than 100 meters and most of the time was spent shallower than 80 meters. Based on preliminary analyses of the data, 75-90 percent of the time the leatherback turtles were at depths less than 80 meters.

4.3.14.7 Vocalizations and Hearing

Sea turtles do not appear to use sound for communication. Nesting leatherback turtles have been recorded producing sounds (sighs, grunts or belch-like sounds) up to 1,200 Hz with maximum energy from 300 to 500 Hz (Cook & Forrest, 2005), however these sounds appeared to be associated with breathing (Mrosovsky, 1972);(Cook & Forrest, 2005).

Recent research measureing hatchling leatherback sea turtle auditory evoked potentials has shown that hatchling leatherbacks respond to tonal stimuli between 50 and 1,200 underwater (maximum sensitivity: 100-400 Hz) and 50 and 1,600 in air (maximum sensitivity: 50-400Hz) (Dow Piniak et al., 2012; Piniak, 2012).

There is no information on the vocalizations or hearing of leatherback sea turtles. However, we assume that their hearing sensitivities will be similar to those of green and loggerhead sea turtles: their best hearing sensitivity will be in the low frequency range: from 200 to 400 Hz with rapid declines for tones at lower and higher frequencies. Their hearing will probably have a practical upper limit of about 1000 Hz (Soraya Moein Bartol, Musick, & Lenhardt, 1999b).

These hearing sensitivities are similar to the hearing sensitivities reported for two terrestrial species: pond turtles (*Pseudemys scripta*) and wood turtles (*Chrysemys inscuplta*). Pond turtles are reported to have best hearing responsiveness between 200 and 700 Hz, with slow declines below 100 Hz and rapid declines above 700 Hz and almost no sensitivity above 3000 Hz (Wever & Vernon, 1956). Wood turtles are reported to have sensitivities up to about 500 Hz, followed by a rapid decline above 1000 Hz and almost no responses beyond 3000 or 4000 Hz (P. D. Patterson, 1966).

5 ENVIRONMENTAL BASELINE

The "environmental baseline" includes the past and present impacts of all Federal, state, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions that are contemporaneous with the consultation in process (50 CFR § 402.02). The environmental baseline includes a wide variety of natural phenomena and human activities in the action area.

5.1 Natural Phenomena

Natural phenomena include changes in oceanic temperature regimes and ambient noises in the ocean environment, which vary both in location and season as well as natural mortality.

5.1.1 Oceanic Temperature Regimes

Oceanographic conditions in the Atlantic and Pacific Oceans can be altered due to periodic shifts in atmospheric patterns caused by the Southern Oscillation in the Pacific Ocean, which lead to El Niño and La Niña events, the Pacific Decadal Oscillation (PDO), and the North Atlantic Oscillation (NAO). These climatic events can alter habitat conditions and prey distribution for listed species in the action area (Beamish et al., 1999; A. J. Benson & Trites, 2002; Francis & Hare, 1997; S.R. Hare et al., 1999; N. J. Mantua et al., 1997; P.R. Mundy & Cooney, 2005; P. J. Stabeno et al., 2004); (P.R. Mundy & Olsson, 2005). For example, decade-scale climatic regime shifts have been related to changes in zooplankton in the North Atlantic (Fromentin & Planque, 1996), and decadal trends in the NAO (Hurrell, 1995) can affect the position of the Gulf Stream (Taylor & Stephens, 1998) and other circulation patterns in the North Atlantic that act as migratory pathways for various marine species, especially fish.

The Pacific decadal oscillation is the leading mode of variability in the North Pacific and operates over longer periods than either El Niño or La Niña/Southern Oscillation (ENSO) events and is capable of altering sea surface temperature (SST), surface winds, and sea level pressure

(N. Mantua, 2002; N. J. Mantua & S. R. Hare, 2002; P. J. Stabeno et al., 2004). During positive Pacific decadal oscillations, the northeastern Pacific experiences above average SSTs while the central and western Pacific Ocean undergoes below-normal SSTs (P.R. Mundy & Olsson, 2005; T. C. Royer, 2005). Warm Pacific decadal oscillation regimes, as occurs in El Niño events, tends to decrease productivity along the U.S. west coast, as upwelling typically diminishes (Childers et al., 2005; S.R. Hare et al., 1999). Opposite SST regimes occur during negative Pacific decadal oscillations (P.R. Mundy & Olsson, 2005). Recent sampling of oceanographic conditions just south of Seward, Alaska has revealed anomalously cold conditions in the Gulf of Alaska from 2006-2009, suggesting a shift to a colder Pacific decadal oscillation phase. More research needs to be done to determine if the region is indeed shifting to a colder Pacific decadal oscillation phase in addition to what effects these phase shifts have on the dynamics of prey populations important to listed cetaceans throughout the Pacific action area. A shift to a colder Pacific decadal oscillation phase would be expected to impact prey populations over the five year duration of the proposed permit, although the magnitude of this effect is uncertain.

In addition to periodic variation in weather and climate patterns that affect oceanographic conditions in the action area, longer term trends in climate change and/or variability also have the potential to alter habitat conditions suitable for listed species in the action area on a much longer time scale. For example, from 1906-2006, global surface temperatures have risen 0.74º C and this trend is continuing at an accelerating pace. Twelve of the warmest years on record since 1850 have occurred since 1995 (Poloczanska et al., 2009). Possible effects of this trend in climate change and/or variability for listed marine species in the action area include the alteration of community composition and structure, changes to migration patterns or community structure, changes to species abundance, increased susceptibility to disease and contaminants, and altered timing of breeding and nesting (Kintisch, 2006; J. A. Learmonth et al., 2006; Macleod et al., 2005; C. R. McMahon & Hays, 2006; Robert A. Robinson et al., 2005). Climate change can influence reproductive success by altering prey availability, as evidenced by the low success of northern elephant seals during El Niño periods (Clive R. McMahon & Burton, 2005) as well as data suggesting that sperm whale females have lower rates of conception following periods of unusually warm sea surface temperature (Hal Whitehead et al., 1997). However, gaps in information and the complexity of climatic interactions complicate the ability to predict the effects that climate change and/or variability may have to these species from year to year in the action area (Kintisch, 2006; Mark P. Simmonds & Isaac, 2007).

5.1.2 Ambient Noise

Ambient noise is the sound in the environment caused by naturally-occurring physical and biological sources as well as anthropogenic sources. Ambient noise levels are higher in the northern hemisphere, where sources of anthropogenic sounds are more pervasive. However, even in relatively quiet regions in the southern hemisphere, ambient noise levels commonly vary by 20 dB and will vary by 30 dB with lower frequency because of biological sources and sea

surface noise (Cato & McCauley., 2001b). There are numerous ambient sources of noise that have low frequencies that are comparable to SURTASS LFA sonar, such as:

- *Wind and waves* are common and interrelated sources of ambient noise in all of the world's oceans. All other factors being equal, ambient noise levels tend to increase with increasing wind speed and wave height (W. John Richardson, Charles R. Greene Jr., et al., 1995). Noise generated by surface wave activity is one of the two primary contributors to ambient noise in the frequency range from 300 Hz to 5 kHz. The wind-generated noise level decreases smoothly with increasing acoustic frequency (i.e., there are no spikes at any given frequency).

- *Precipitation.* At some frequencies, rain and hail will increase ambient noise levels. Significant noise is produced by rain squalls over a range of frequencies from 500 Hz to 15 kHz. Large storms with heavy precipitation can generate noise at frequencies as low as 100 Hz and significantly affect ambient noise levels at a considerable distance from a storm's center. Lightning strikes associated with storms are loud, explosive events that deliver an average of 100 kilojoules per meter (kJ/m) of energy (Considine, 1995). Hill (1985) estimated the source level for cloud-to-water pulse to be 260.5 dB. It has been estimated that over the earth's oceans the frequency of lightning averages about 10 flashes per second, or 314 million strikes per year (Kraght, 1995).

- *Seismic Phenomena*. Underwater volcanic eruptions, earthquakes, and landslides produce sound in the frequency range of 1 to 100 Hz. In the Pacific Ocean, where the majority of seismic activity occurs in the world's oceans, about 10,000 natural, seismic phenomena occur each year (Fox, Matsumoto, & Lau, 2001). These phenomena produce sounds with source levels exceeding 255 dB re: 1 µPa at 1 m (R. D. Hill, 1985).

- *Shipping Noise*. The dominate source of anthropogenic sound in the oceans can be directly attributed to commercial shipping, with the propulsion of ships (primarily propeller cavitation) generating noise in the 20 to 200 Hz frequency band (P. L. Tyack, 2008; Wenz, 1962). As the number of ships plying the oceans around the world increases, so does the noise they generate (D. Ross, 2005). The number of commercial vessels traversing the world's oceans approximately doubled between 1965 and 2003, and the gross tonnage quadrupled, with a corresponding increase in horsepower (M.A. McDonald, Hildebrand, & Wiggins, 2006). Due to the increase in propeller-driven vessels, low-frequency ambient noise has increased 10 to 15 dB, at an average of approximately 3 dB/decade over the past 50 years (Andrew, Howe, Mercer, & Dzieciuch, 2002; Cato & McCauley., 2001a; Curtis, Howe, & Mercer, 1999; M.A. McDonald et al., 2006; Zakarauskas, Chapman, & Staal, 1990).

- *Biological Noise. S*ounds created by animals in the sea and may contribute significantly to ambient noise in many areas of the oceans (Curtis et al., 1999). Because of the habits, distribution, and acoustic characteristics of these sound producers, certain areas of the oceans are louder than others. Only three groups of marine animals are known to make sounds: crustaceans (such as snapping shrimp), fish, and marine mammals (Urick, 1983). The most widespread, broadband noises from animal sources (in shallow water) are those produced by croakers (representative of a variety of fish classified as drumfish) (100 Hz to 10 kHz) and snapping shrimp (500 Hz to 20 kHz). Sound-producing fishes and crustaceans are restricted almost entirely to bays, reefs, and other coastal waters, although there are some pelagic, sound-producing fish. In oceanic waters, whales and other marine mammals are principal contributors to biological noise. For example, dolphins produce whistles associated with certain behaviors, and the baleen whales are noted for their low frequency vocalizations.

5.1.3 Natural Mortality

Natural mortality rates in cetaceans, especially large whale species, are largely unknown. Although factors contributing to natural mortality cannot be quantified, there are a number of suspected causes, including parasites, predation, red tide toxins, and ice entrapment. For example, the giant spirurid nematode (*Crassicauda boopis*) has been attributed to congestive kidney failure and death in some large whale species (R.H. Lambertsen, 1986). A well-documented observation of killer whales attacking a blue whale off Baja, California proves that blue whales are at least occasionally vulnerable to these predators (Tarpy, 1979). Other stochastic events, such as fluctuations in weather and ocean temperature affecting prey availability, may also contribute to large whale natural mortality.

Sea turtles are also affected by disease and environmental factors. Turtles can be injured by predators such as birds, fish, and sharks (R. H. George, 1997). Hypothermic or cold stunning occurs when a turtle is exposed to cold water for a period of time. Cold stunned turtles often have decreased salt gland function which may lead to plasma electrolyte imbalance and a lowered immune response (R. H. George, 1997).

Steller sea lions shift diet composition in response to changes in prey availability of pollock (*Theragra chalcogramma*), hake (*Merluccius productus*), herring (*Clupea pallasi*) and salmon (*Oncorhynchus spp*.) (Sigler et al., 2009).

5.2 Human Activities

Human activities that have occurred and may continue in the action area include whaling, anthropogenic noise, shipping, seismic surveys for oil and gas development, Navy research and training, fisheries, scientific research, and commercial and private marine mammal watching. Each of these activities is discussed further below.

5.2.1 Whaling

Large whale populations in the proposed action areas have historically been impacted by commercial exploitation, mainly in the form of whaling. Prior to current prohibitions on whaling, such as the IWC's 1966 moratorium, most large whale species had been depleted to the extent it was necessary to list them as endangered under the Endangered Species Act of 1966. For example, from 1900 to 1965 nearly 30,000 humpback whales were captured and killed in the Pacific Ocean alone with an unknown number of additional whales captured and killed before 1900 (S. L. Perry et al., 1999). Sei whales are estimated to have been reduced to 20 percent (8,600 out of 42,000) of their pre-whaling abundance in the North Pacific (Tillman, 1977). In addition, 9,500 blue whales were reported killed by commercial whalers in the North Pacific between 1910-1965 (S. Ohsumi & Wada., 1972); 46,000 fin whales between 1947-1987 (Dale W. Rice, 1984); and 25,800 sperm whales (Barlow et al., 1997a). North Pacific right whales once numbered 11,000 animals but commercial whaling has now reduced their population to 29-100 animals (Wada, 1973).

Presently three types of whaling take place; (1) commercial whaling conducted either under objection or reservation to the moratorium; (2) aboriginal subsistence whaling is to support the needs of indigenous peoples; (3) whaling under special permit (IWC, 2012).

Norway and Iceland take whales commercially at present, either under objection to the moratorium decision or under reservation to it. These countries establish their own catch limits but must provide information on those catches and associated scientific data to the Commission. The Russian Federation has also registered an objection to the moratorium decision but does not exercise it. The moratorium is binding on all other members of the IWC. Norway takes North Atlantic common minke whales within its Exclusive Economic Zone, and Iceland takes North Atlantic common minke whales and also North Atlantic fin whales, again within its Exclusive Economic Zone (IWC, 2012).

Under current IWC regulations, aboriginal subsistence whaling is permitted for Denmark (Greenland, fin and minke whales), the Russian Federation (Siberia, gray and bowhead whales), St Vincent and the Grenadines (Bequia, humpback whales) and the USA (Alaska, bowhead and gray whales). It is the responsibility of national governments to provide the Commission with evidence of the cultural and subsistence needs of their people. The Scientific Committee provides scientific advice on safe catch limits for such stocks. Based on the information on need and scientific advice, the Commission then sets catch limits, recently in five-year blocks. It is the responsibility of national governments to provide the Commission with evidence of the cultural and subsistence needs of their people. The Scientific Committee provides scientific advice on safe catch limits for such stocks (IWC, 2012).

Scientific permit whaling has been done by Japan and Iceland. Japan has issued scientific permits in the Antarctic and in the western North Pacific every year in recent years. The 2004/05 Antarctic season was the final year of the 16-year 'JARPA' program, following a 2-year

feasibility study. After completion of the JARPA program, Japan initiated a JARPA II program, initially as a 2-year feasibility study, for 850±10 percent and 10 fin whales in the Antarctic. In 2009/10, the full JARPA II program commenced and the current permit has been for 850±10 percent Antarctic minke whales, 50 fin whales and 50 humpback whales annually. To date, Japan has refrained from taking humpback whales. After completion of a six year JARPN program in the North Pacific in 1999, Japan initiated a JARPN II program, initially as a 2-year feasibility study in 2000, for 100 common minke whales, 50 Bryde's whales and 10 sperm whales in the western North Pacific (IWC, 2012).

In Iceland, the stated overall objective of the research program was to increase understanding of the biology and feeding ecology of important cetacean species in Icelandic waters for improved management of living marine resources based on an ecosystem approach. While Iceland stated that its program was intended to strengthen the basis for conservation and sustainable use of cetaceans, it noted that it was equally intended to form a contribution to multi-species management of living resources in Icelandic waters. In practice, the Government of Iceland only issued permits for the common minke whale segment of the original proposal. A total of 200 common minke whales were caught from 2003-2007 as originally proposed, although the initial proposal expected 100 per year for two years. Again, as in the past, different views on the value of this research were expressed in the Scientific Committee (IWC, 2012).

5.2.2 Shipping

Surface shipping is the most widespread source of anthropogenic, low frequency (0 to 1,000 Hz) noise in the oceans (Mark P. Simmonds & Hutchinson, 1996; P. L. Tyack, 2008). The U.S. Navy estimated that the 60,000 vessels of the world's merchant fleet annually emit low frequency sound into the world's oceans for the equivalent of 21.9 million days, assuming that 80 percent of the merchant ships at sea at any one time (Navy, 2001b). Shipping noise centers in the 20 to 200 Hz frequency band, and was increasing yearly by about 5 dB per year until around 1980 (Chapman & Price., 2011). Since 1980 ocean noise has been increasing at about 0.2 dB per year (Chapman & Price., 2011). The National Research Council (NRC, 2003a) estimated that the background ocean noise level at 100 Hz has been increasing by about 1.5 dB per decade since the advent of propeller-driven ships. Due to the increase in the number of propeller-driven vessels, low-frequency ambient noise has increased 10 to 15 dB, at an average of approximately 3 dB/decade over the past 50 years (Andrew et al., 2002; Cato & McCauley., 2001a; Curtis et al., 1999; M.A. McDonald et al., 2006; Zakarauskas et al., 1990).

Collisions with commercial ships are an increasing threat to many large whale species, particularly as shipping lanes cross important large whale breeding and feeding habitats or migratory routes. The number of observed physical injuries to humpback whales as a result of ship collisions has increased in Hawaiian waters (Glockner-Ferrari, Ferrari, & McSweeney., 1987). On the Pacific coast, a humpback whale is probably killed about every other year by ship strikes (Barlow et al., 1997b). From 1996-2002, eight humpback whales were reported struck by

vessels in Alaskan waters. In 1996, a humpback whale calf was found stranded on Oahu with evidence of vessel collision (propeller cuts; NMFS unpublished data).

Based on the data available from Douglas et al. (2008), Jensen and Silber (2004), and Laist et al. (2001), there have been at least 25 incidents in which marine mammals are known to have been struck by ships in the Puget Sound region and southwestern British Columbia. The marine mammals that were involved in almost half of these incidents died as a result of the strike and they suffered serious injuries in four of those strikes.

Fin whales were struck most frequently, accounting for almost 30 percent of the total number of incidents and two-thirds of the incidents in which the whale died as a result of the collision. Northern resident killer whales were struck slightly less frequently, although a cluster of ship strikes in 2006 accounted for four of the six ship strikes involving this population of killer whales. Humpback whales were third in frequency, followed by southern resident killer whales, offshore killer whales, and blue whales. About two-thirds (17 out of the 25) of the incidents occurred in waters off British Columbia, although the locations were variable.

Historical records suggest that ship strikes fatal to whales first occurred late in the 1800s as ships began to reach speeds of 13-15 kn, remained infrequent until about 1950, and then increased during the 1950s-1970s as the number and speed of ships increased. Of 11 species known to be hit by ships, fin whales are struck most frequently; right whales, humpback whales, sperm whales, and gray whales are hit commonly (Laist et al., 2001; A. S. M. Vanderlaan & Taggart, 2007). In some areas, one-third of all fin whale and right whale strandings appear to involve ship strikes (Laist et al., 2001). All sizes and types of vessels can hit whales; most lethal or severe injuries are caused by ships 80 m or longer; whales usually are not seen beforehand or are seen too late to be avoided; and most lethal or severe injuries involve ships travelling 14 kt or faster (Laist et al., 2001) . Ship strikes can significantly affect small populations of whales, such as northern right whales in the western North Atlantic. In areas where special caution is needed to avoid such events, measures to reduce the vessel speed below 14 kt may be beneficial (Laist et al., 2001).

False killer whales in waters surrounding Hawaii (belonging to both insular and pelagic stocks) are known to ride the bow or stern wake of vessels and may come into proximity of propellers (Baird, pers. comm. as cited in (Oleson et al., 2010)). No ship-strike related injuries or deaths of false killer whales have been documented in Hawaiian waters, but Baird (2009 as cited in (Oleson et al., 2010)) reported a fresh head wound on one individual from the insular population photographed off Oahu in September 2009 that may have been caused by a propeller strike.

5.2.3 Fisheries

5.2.3.1 Cetaceans

Directed harvest has affected sei, blue, fin, humpback, sperm, and North Atlantic right whales. Commercial harvest of these large whale species no longer occurs in the United States, and the

IWC has moratoriums in place to protect these species from commercial whaling internationally. Nonetheless, historical whaling significantly reduced large whale abundance, and the effects of these reductions likely still persist. Conclusions based on historical whaling data suggest that the numbers of right whales in the western North Atlantic numbered in the hundreds before commercial exploitation (Randall R. Reeves & Mitchell., 1987). More recent analysis concluded that these numbers may have been closer to 1,000, and that the greatest population decline occurred in the early 1700s (Reeves et al. in (Reeves et al. in Breiwick, Reeves, & Mitchell., 1993). However, the authors caution that these estimates were based on incomplete records. Although extensively hunted historically, there has been little hunting of right whales in the 20th century. Hunting in the 19th and early 20th centuries, largely by Norwegian whaling operations, likely irreversibly damaged or extirpated this stock (S. G. Brown, 1976; Collett, 1909).

Historically, whaling represented the greatest threat to every population of sei whales and was ultimately responsible for listing sei whales as an endangered species. Sei whales are thought to not be widely hunted, although harvest for scientific whaling or illegal harvesting may occur in some areas.

Fin whales continue to be hunted in subsistence fisheries off West Greenland. Between 2003 and 2007, the IWC set a catch limit of up to 19 fin whales in this subsistence fishery. In the Antarctic Ocean, fin whales are hunted by Japanese whalers who have been allowed to kill up to 10 fin whales each year for the 2005-2006 and 2006-2007 seasons under an Antarctic Special Permit NMFS (IWC, 2006). The Japanese whalers plan to kill 50 whales per year starting in the 2007-2008 season and continuing for the next 12 years (IWC, 2006; S. Nishiwaki et al., 2006).

Sperm whales historically faced severe depletion from commercial whaling operations. From 1800 to 1900, the IWC estimated that nearly 250,000 sperm whales were killed by whalers, with another 700,000 from 1910 to 1982 (IWC Statistics 1959-1983). Others have estimated 436,000 individuals taken from 1800-1987 (James V Carretta, Chivers, & Danil, 2005).

Entrapment in commercial fishing gear continues to impact listed cetaceans in the action area. Robbins and Mattila (2001) studied entanglement-related scarring on 134 individual humpback whales and concluded that between 48 and 65 percent had experienced entanglements. An estimated 78 baleen whales were killed annually in the offshore southern California drift gillnet fishery during the 1980's (Heyning & Lewis, 1990) and 22 humpback whale entanglements were reported from 1996-2000 (Angliss & Lodge, 2004). More recent records show that during the period 2004-2008 there were 18 reported entanglements of humpback whales off the U.S. west coast (J. V. Carretta et al., 2011). Eleven were reported entangled at sea in trap/pot fishery gear off California and Oregon, including two animals later found dead (Northwest Regional Stranding Program, unpublished data).

In the Northeast Pacific, fishery-related minimum mortality and serious injury rate for humpbacks is 3.8 individuals per year based on observer and stranding data from Alaska and

Hawaii (B. M. Allen & Angliss, 2011). Fin and sei whales also interact with fishing gear although reported takes are much lower than those reported for humpbacks. According to the most recent stock assessment reports for the western North Atlantic region, the annual rate of serious injury and mortality of fin and sei whales from fishery interactions is 1.2 and 0.6 individuals per year, respectively (G.T. Waring, Josephson, Maze-Foley, & P.E.Rosel, 2011). During the period 2004-2008, there were 3 confirmed fin whale deaths and an additional 3 reports of fin whales sustaining serious injury as a result of entanglement while for sei whales, there was 1 confirmed mortality and 2 reports of serious injury as a result of entanglement (Glass, Cole, & Garron, 2010). In the Pacific, there was one observed fin whale mortality in the Bering Sea/Aleutian Island pollock trawl fishery between 2002 and 2006 (B. M. Allen & Angliss, 2011) while for the offshore drift gillnet fishery, there has been one fin whale death reported since 1990 (J. V. Carretta et al., 2011).

Based on photographs of Hawaiian insular false killer whales, Baird and Gorgone (2005)documented a high rate of dorsal fin disfigurements that were consistent with injuries from unidentified fishing line (3 out of 80 individuals or 3.75%, compared to 0–0.85% for other studied cetacean populations). Interactions with false killer whales have been reported for troll fisheries (Nitta & Henderson, 1993; E. Shallenberger et al., 1981; Zimmerman, 1983b), deep-set and shallow-set longline fisheries (Karin A Forney & Kobayashi, 2007; M. McCracken & K. Forney, 2010; Nitta & Henderson, 1993), and possibly shortline or kaka-line fisheries (anecdotal reports of ‘blackfish’ interactions that may have been false killer whales, as cited in {Baird, 2010 #86333}.

In addition to direct injury and/or mortality, entanglements also make listed species more vulnerable to additional dangers (e.g., predation and ship strikes) by restricting agility and swimming speed. Robbins and Mattila (2001) found that female humpbacks showing evidence of prior entanglements produced significantly fewer calves, suggesting entanglement may significantly reduce reproductive success. Also, many marine mammals that die from entanglement in commercial fishing gear tend to sink rather than strand ashore thus making it difficult to accurately determine the extent of such mortalities. This, in addition to a lack of observer coverage in the case of many fisheries operating in the action area, mean that many “takes” associated with commercial fisheries are likely being underreported for many of the listed species affected.

As part of an effort to reduce fishing gear entanglement by whales in the North Atlantic, NMFS developed the Atlantic Large Whale Take Reduction Plan. This plan has improved safety measures in fishing gear in order to reduce entanglements by whales. This plan also expanded restrictions on fishing grounds and prohibited gillnet fishing in restricted areas during the calving season.

5.2.3.2 Sea Turtles

Directed harvest of sea turtles and their eggs for food and other products has existed for years and was a significant factor causing the decline of Kemp's ridley, green, leatherback, hawksbill, and loggerhead turtles. At present, despite conservation efforts such as bans and moratoriums by the responsible governments, the harvest of turtles and their eggs still occurs throughout the action area. Countries including Mexico, Peru, and the Philippines have made attempts to reduce the threats to sea turtles, but illegal harvesting still occurs. In Vietnam and Fiji, harvest of turtle meat and eggs remains unregulated.

Bycatch, particularly by longline fisheries, is a major source of mortality for leatherback sea turtles (Crognale, Eckert, Levenson, & Harms, 2008; Fossette et al., 2009; Gless, Salmon, & Wyneken, 2008; Petersen et al., 2009). Wallace et al. (2010) estimated that between 1990 and 2008, at least 85,000 sea turtles were captured as bycatch in fisheries worldwide. This estimate is likely at least two orders of magnitude low, resulting in a likely bycatch of nearly half a million sea turtles annually (Bryan P. Wallace et al., 2010). Finkbeiner et al. (2011b) provides the following estimates for mean annual bycatch interactions and mortalities for U.S. Atlantic fisheries: Kemp's ridley – 98,300 interactions, 27,000 mortalities; loggerhead – 26,500 interactions, 1400 mortalities; green – 11,400 interactions, 300 mortalities; leatherback – 1400 interactions, 40 mortalities; hawksbill – less than 10 interactions and mortalities.

Foreign high-seas driftnet fishing in the North Pacific Ocean for squid, tuna and billfish ended with a United Nations moratorium in December, 1992. Except for observer data collected in 1990-1991, there is virtually no information on the incidental take of threatened and endangered species by the driftnet fisheries prior to the moratorium. The high seas squid driftnet fishery in the North Pacific was observed in Japan, Korea, and Taiwan, while the large-mesh fisheries targeting tuna and billfish were observed in the Japanese fleet (1990-91) and the Taiwanese fleet (1990). A combination of observer data and fleet effort statistics indicate that 4,373 turtles, mostly loggerheads and leatherbacks, were entangled by the combined fleets of Japan, Korea and Taiwan during June, 1990 through May, 1991, when all fleets were monitored. Of these incidental entanglements, an estimated 1,011 turtles were killed (77 percent survival rate).

Numerous longline fisheries occur in the Pacific Ocean. These fisheries have had significant impacts on threatened and endangered species. One of these, the Japanese tuna longliners in the Western Pacific Ocean and South China Sea has been estimated to capture 21,200 sea turtles, including green, leatherback turtle, loggerhead, olive ridley and hawksbill sea turtles each year. These interactions kill about 12,300 of these sea turtles each year.

Other fisheries include coastal fisheries off Japan, coastal setnet and gillnet fisheries off Taiwan, foreign and U.S. purse seine fisheries for tuna in the eastern tropical Pacific Ocean, and multi-gear fisheries in the North Pacific Ocean (Bering Sea, Aleutian Islands, and Gulf of Alaska). Several of these fisheries have been implicated in the declining trend of several threatened and endangered species.

Entrapment and entanglement in commercial fishing gear is one of the most frequently documented sources of human-caused mortality in large whale species and sea turtles. For example, in 1978, Nishimura and Nakahigashi (1990) estimated that 21,200 turtles, including greens, leatherback turtles, loggerheads, olive ridleys and hawksbills, were captured annually by Japanese tuna longliners in the Western Pacific and South China Sea, with a reported mortality of approximately 12,300 turtles per year. Using commercial tuna longline logbooks, research vessel data and questionnaires, Nishimura and Nakahigashi (1990) estimated that for every 10,000 hooks in the Western Pacific and South China Sea, one turtle is captured, with a mortality rate of 42 percent.

5.2.4 Contaminants

The accumulation of stable pollutants is a possible human-induced source of mortality in long-lived high trophic level animals (Gordon T. Waring et al., 2004), and some researchers have correlated contaminant exposure to possible adverse health effects in marine mammals. Contaminants may be introduced by rivers, coastal runoff, wind, ocean dumping, dumping of raw sewage by boats and various industrial activities, including offshore oil and gas or mineral exploitation. Due to their large amount of blubber and fat, marine mammals readily accumulate lipid-soluble contaminants (T. M. O'Hara & Rice, 1996).

In sea turtles, heavy metals have been found in a variety of tissues in levels that increase with turtle size (Anan et al., 2001; Barbieri, 2009; Fujihara et al., 2003; García-Fernández et al., 2009; S. C. Gardner et al., 2006; Brendan J. Godley, 1999; Sakai et al., 2000; M. M. Storelli, G. Barone, A. Storelli, & G. O. Marcotrigiano, 2008). Cadmium has been found in leatherbacks at the highest concentration compared to any other marine vertebrate (Caurant, Bustamante, Bordes, & Miramand, 1999; J. Gordon et al., 1998).

Newly emerged hatchlings have higher concentrations than are present when laid, suggesting that metals may be accumulated during incubation from surrounding sands (G. Sahoo, R. K. Sahoo, & P. Mohanty-Hejmadi, 1996). Arsenic has been found to be very high in green sea turtle eggs (Van de Merwe et al., 2009). Sea turtle tissues have been found to contain organochlorines (Alava et al., 2006; S. Corsolini, S. Aurigi, & S. Focardi, 2000; Keller, Kannan, Taniyasu, Yamashita, et al., 2005; J. M. Keller, J. R. Kucklick, C. A. Harms, et al., 2004; McKenzie et al., 1999; Monagas et al., 2008; Oros et al., 2009; M. J. Rybitski, R. C. Hale, & J. A. Musick, 1995; M. M. Storelli, G. Barone, & G. O. Marcotrigiano, 2007).

Concentrations of PCBs are reportedly equivalent to those in some marine mammals, with liver and adipose levels of at least one congener being exceptionally high (PCB 209: 500- 530 ng/g wet weight) (Davenport, Wrench, McEvoy, & Carnacho-Ibar, 1990; Oros et al., 2009). Levels of PCBs found in green sea turtle eggs are considered far higher than what is fit for human consumption (Van de Merwe et al., 2009).

Organochlorines have the potential to suppress the immune system of loggerhead sea turtles and may affect metabolic regulation (J. M. Keller, P. D. McClellan-Green, John R. Kucklick, Deborah E. Keil, & Margie M. Peden-Adams, 2006; Keller, Stamper, Kucklick, & McClellan-Green, 2004; Oros et al., 2009). These contaminants could cause deficiencies in endocrine, developmental, and reproductive health (M. M. Storelli et al., 2007), and are known to depress immune function in loggerhead sea turtles (J. M. Keller et al., 2006). Females from sexual maturity through reproductive life should have lower levels of contaminants than males because contaminants are shared with progeny through egg formation.

Exposure to sewage effluent may also result in green sea turtle eggs harboring antibiotic resistant strains of bacteria (Al-Bahry et al., 2009).

5.2.5 Pollution and Ocean Debris

Anthropogenic activities such as discharges from wastewater systems, dredging, ocean dumping and disposal, aquaculture, and additional impacts from coastal development are known to degrade coastal waters utilized by listed whales in the action area. Multiple municipal, industrial and household sources as well as atmospheric transport introduce various pollutants such as pesticides, hydrocarbons, organochlorides (e.g., DDT and PCBs), and other pollutants that may cause adverse health effects to listed whales (Garrett, 2004; Grant & Ross, 2002; Hartwell, 2004; Iwata, Tanabe, Sakai, & Tatsukawa, 1993; P. S. Ross, 2002). The accumulation of persistent pollutants through trophic transfer may cause mortality and sub-lethal effects including immune system abnormalities, endocrine disruption and reproductive effects (Krahn et al., 2007). Recent efforts have led to improvements in regional water quality in the action area, although the more persistent chemicals are still detected and are expected to endure for years (Grant & Ross, 2002).

Acute exposure to hydrocarbons from petroleum products released into the environment via oil spills and other discharges are known to cause behavioral changes in marine mammals (Grant & Ross, 2002) and may directly injure individuals through skin contact with oils (Geraci & Aubin, 1990), inhalation at the water's surface, and ingesting compounds while feeding (Matkin & Saulitis, 1997). The Exxon Valdez released an estimated 11 million gallons of Alaskan crude oil in 1989. The Alaska Department of Environmental Conservation estimated that 149 kilometers of shoreline was heavily oiled and 459 kilometers were at least lightly oiled. The Gulf of Mexico also represents an area of high-density offshore oil extraction with chronic, low-level spills and occasional massive spills (such as the Deep Horizon oil spill event in 2010, Ixtoc I oil well blowout and fire in the Bay of Campeche in 1979, and the explosion and destruction of a loaded supertanker, the Mega Borg, near Galveston in 1990). Experience gained during the Exxon Valdez spill indicates that large-scale spills can cause persistent negative effects on wildlife that can last for decades (Peterson et al., 2003). Matkin et al. (2008) utilized photo-identification methods to monitor two killer whale populations five years prior to and 16 years after the Exxon Valdez oil spill and noted that the two populations had not recovered from pre-spill numbers. The recent *Deepwater Horizon* oil spill in the northeastern Gulf of Mexico particularly affected

sea turtles. Kemp's ridley sea turtles were the predominant sea turtle species found injured and dead following spill. It is expected that marine mammals continue to feel the effects of these major oil spill events and will continue to be threatened by any future spills as oil and gas exploration and extraction expands throughout the action area.

Habitat in the action area may also be degraded by various sources of marine debris such as plastics, glass, metal, polystyrene foam, rubber, and derelict fishing gear. Marine debris is introduced into the marine environment through ocean dumping, littering, or hydrologic transport of these materials from land-based sources. Even natural phenomena, such as tsunamis and continental flooding, can cause large amounts of debris to enter the ocean environment. Listed whales and turtles may become entangled in marine debris or directly ingest it while feeding, potentially leading to digestive problems, injury, or even death. Recently in March of 2011, a significant amount of debris was scattered into the western Pacific as a result of an earthquake and tsunami occurring in Japan. The Japanese Ministry of the Environment estimated the total quantity of the disaster waste at 25 million tons[8]. Data obtained at Independent models run by NOAA and the University of Hawaii anticipate the debris passing close or washing ashore in the Northwestern Hawaiian Islands in winter 2012, approaching the West Coast of the U.S. in 2013, and circling back to Hawaii in 2014 to 2016 (NOAA, unpublished[9]). Thus, it is expected that marine mammals and sea turtles may be exposed to this marine debris over the course of the MMPA regulations although the risk of ingestion or entanglement and the resulting impacts are uncertain at the time of this consultation.

Coastal development can deter or interfere with nesting, affect nest success, and degrade foraging habitats for sea turtles. Many nesting beaches have already been significantly degraded or destroyed. Nesting habitat is threatened by rigid shoreline protection or "coastal armoring" such as sea walls, rock revetments, and sandbag installations. Many miles of once productive nesting beach have been permanently lost to this type of shoreline protection. Nesting habitat can be reduced by beach renourishment projects, which result in altered beach and sand characteristics, affecting nesting activity and nest success. Beach nourishment also hampers nesting success of loggerhead sea turtles, but only in the first year post-nourishment, after which hatching success increases (Brock et al., 2009). In some areas, timber and marine debris accumulation as well as sand mining reduce available nesting habitat (Bourgeois, Gilot-Fromont, Viallefont, Boussamba, & Deem, 2009). Because hawksbills prefer to nest under vegetation (Horrocks & Scott, 1991; Mortimer, 1982), they are particularly affected by beachfront development and clearing of dune vegetation (Mortimer & Donnelly, 2007).

The presence of lights on or adjacent to nesting beaches alters the behavior of nesting adults and is often fatal to emerging hatchlings as they are attracted to light sources and drawn away from the sea (Karnad et al., 2009; B. E. Witherington, 1992a; B. E. Witherington & Bjorndal, 1991a).

[8] www.env.go.jp/en/focus/110603.html .

[9] Data obtained at http://marinedebris.noaa.gov/info/japanfaqs.html.

Coasts can also be threatened by contamination from herbicides, pesticides, oil spills, and other chemicals, as well as structural degradation from excessive boat anchoring and dredging (Francour, Ganteaume, & Poulain, 1999; Lee Long, Coles, & McKenzie, 2000; Waycott, Longstaff, & Mellors, 2005).

At sea, there are numerous potential threats including marine pollution, oil and gas exploration, lost and discarded fishing gear, changes in prey abundance and distribution due to commercial fishing, habitat alteration and destruction caused by fishing gear and practices, agricultural runoff, and sewage discharge (J. Frazier et al., 2007; M. E. Lutcavage et al., 1997). Hawksbills are typically associated with coral reefs, which are among the world's most endangered marine ecosystems (Wilkinson, 2000).

Although climate change may expand foraging habitats into higher latitude waters and increasing ocean temperatures may also lead to reduced primary productivity and eventual food availability, climate change could reduce nesting habitat due to sea level rise, as well as affect egg development and nest success. Rising temperatures may increase feminization of leatherback nests (Hawkes, Broderick, Coyne, Godfrey, & Godley, 2007; James, Sherrill-mix, Martin, & Myers, 2006; C. R. McMahon & Hays, 2006; Mrosovsky, Hopkins-Murphy, & Richardson, 1984). Hawksbill turtles exhibit temperature-dependent sex determination (Wibbels, 2003) suggesting that there may be a skewing of future hawksbill cohorts toward strong female bias. Loggerhead sea turtles are very sensitive to temperature as a determinant of sex while incubating. Ambient temperature increase by just 1º-2º C can potentially change hatchling sex ratios to all or nearly all female in tropical and subtropical areas (L. A. Hawkes, A. C. Broderick, M. H. Godfrey, & B. J. Godley, 2007). Over time, this can reduce genetic diversity, or even population viability, if males become a small proportion of populations (Hulin et al., 2009). Sea surface temperatures on loggerhead foraging grounds has also been linked to the timing of nesting, with higher temperatures leading to earlier nesting (Antonios D. Mazaris et al., 2009; Schofield et al., 2009). Green sea turtles emerging from nests at cooler temperatures likely absorb more yolk that is converted to body tissue than do hatchlings from warmer nests (Ischer, Ireland, & Booth, 2009). However, warmer temperatures may also decrease the energy needs of a developing embryo (K. A. Reid et al., 2009).

5.2.6 Pulsed Sound Generated by Seismic Surveys

High energy pulsed sound generated in the marine environment from seismic surveysand underwater detonations have the potential to increase stress levels, alter behavior, result in temporary or permanent hearing loss, and/or, in extreme cases, result in direct injury and even death to listed cetaceans depending on the proximity of the animal to the sound source (Christopher W. Clark & Ellison, 2004; Nowacek et al., 2007; NRC, 2003b, 2005; W. John Richardson, Charles R. Greene Jr., et al., 1995; Brandon L. Southall et al., 2007b; A. J. Wright et al., 2007).

Numerous surveys have been conducted in the northeast Pacific and northwest Atlantic using seismic airguns. Airguns are typically fired every 10-15 seconds with theoretical source levels of about 255 dB ± 3 dB which are detectable 50-75 km away in shallow water and over 100 kilometers away in survey areas deeper than 50 m (W. John Richardson, Charles R. Greene Jr., et al., 1995). As a general mitigation measure for surveys in U.S. waters or vessels under U.S. regulations, airguns are shutdown if marine mammals approach too closely (generally within the 180 dB isopleths for cetaceans), presumably avoiding the potential for temporary or permanent threshold shifts in cetaceans exposed to the airgun pulses. While onboard observers and passive acoustic monitoring help identify the presence of whales, the possibility exists that some non-vocalizing whales beneath the surface may be temporarily exposed to higher sound levels at an unspecified degree. In addition to possible physical trauma and stress, whales are known to respond behaviorally by actively avoiding the sound of the seismic survey vessel, thus causing some temporary habitat displacement upon exposure (Gallagher & Hall., 1993; J. C. George, 2010; D. Green & Ortiz-Crespo, 1982; W. John Richardson, Charles R. Greene Jr., et al., 1995; W. J. Richardson, McDonald, Greene, & Blackwell, 2004; W. John Richardson, Wells, & Würsig, 1985; W. John Richardson & Williams, 2003; W. J. Richardson & Williams, 2004; W. John Richardson, Würsig, & Greene Jr, 1990; Schick & Urban, 2000; Streever et al., 2008; Wartzok, Watkins, Wursig, & Malme., 1989).

Seismic survey activities occur throughout the world's oceans. Seismic surveys may be conducted to explore for oil and gas reserves as well as for scientific research on aspects of the earth's structure. For instance, during September-October 2008, Colombia University's Lamont-Doherty Earth Observatory (L-DEO) conducted a seismic survey in northeastern Gulf of Alaska (40-4,000 meters water depth) aboard the R/V Langseth using a 36-airgun array. It was estimated that 80 humpback whales were likely to be exposed to seismic sound in excess of 160 dB re: 1 μPa_{rms} presumably causing avoidance behavior and temporary habitat displacement to occur as a result. Subsequent passive acoustic monitoring effort resulted in 14 sightings of humpback whales (35 individuals), 2 sightings of unidentified whales (4 individuals), and 2 sightings of unidentified baleen whales (7 individuals). From July-September 2010, L-DEO conducted a seismic survey in the northwestern Pacific Ocean in deepwater using a 36-airgun array and estimated that up to 10 humpback whales and 16 fin whales were likely exposed to seismic sound in excess of 160 dB re: 1 μPa_{rms} (Holst & Beland, 2010). While exposure to these sound sources ended at the completion of the seismic surveys, we anticipate that some whales exposed to SURTASS LFA sonar would also be exposed to harassment from pulses generated by seismic surveys conducted in the action area.

5.2.7 Scientific Research

Scientific research permits issued by the NMFS currently authorize studies of listed species in the Pacific and Atlantic Oceans which occur primarily in the action area. Marine mammals have been the subject of field studies for decades. The primary objective of most of these studies has generally been monitoring populations or gathering data for behavioral and ecological studies.

Over time, NMFS has issued dozens of permits for various non-lethal forms of "take" of marine mammals in the proposed action area from a variety of activities, including aerial and vessel surveys, photo-identification, remote biopsy sampling, and attachment of scientific instruments.

For example, existing permits authorized activities in 2013 allow investigators to harass, pursue, shoot, and wound about 675 endangered North Pacific right whales each year for photo-identification and behavioral observation; harass, pursue, and shoot up to 60 of these right whales per year to place tags; harass, pursue, shoot, and wound 108 animals to take biopsy samples. Since the right whale population in the North Pacific has been estimated to consist of between 29 and 100 individuals (fewer than 30 individual whales have been identified since the 1950s), existing permits allow investigators to harass each of these endangered whales several times for different research purposes.

Existing permits (activities anticipated in 2013) authorize investigators to make close approaches of other endangered whales species for photographic identification, behavioral observations, passive acoustic recording, aerial photogrammetry, and underwater observation. Existing permits authorize approximately 20,000 close approaches of blue whales, 28,000 close approaches of fin whales, 62,000 close approaches of humpback whales, 1,045 close approaches of north Pacific right whales, 8,300 close approaches of sei whales, and 22,701 close approaches of sperm whales per year in the Pacific Ocean for these purposes. In addition, existing permits authorize close approaches to take biopsy samples of 2,800 blue whales, 5,000 fin whales, 7,000 humpback whales, 168 North Pacific right whales, 800 sei whales, and 3,700 sperm whales per year in the Pacific Ocean.

Pinniped takes consist of approach, capture, handling, restraint, measurements, anesthesia or drugging, temporary captivity, external tagging, translocation, mortality, swab sampling, blood and tissue sampling, lavage, hot branding, tooth extraction, and/or ultrasound (Table 15). Sea turtle researcher involves approach, capture, handling, restraint, PIT, flipper, satellite, or sonic tagging, lavage, mortality, ultrasound, blood or tissue sampling, captive experiments, laproscopy, imaging, and/or antibiotic injections (Table 15). Smalltooth sawfish may be captured via a variety of means, measured, tagged, tissue sampled, and/or ultrasounded. Research actions on sturgeon species include capture, handling, restraint, anesthesia, laproscopy, lavage, boroscopy, fin, operculum, or barbel clipping, PIT, floy, sonic, or satellite tagging, gonad sampling, prophylactic, and/or mortality. Rockfishes may undergo capture, handling, restraint, anesthesia, floy or sonic tagging, and mortality.

Table 15. Authorized scientific research takes of various listed marine species in the action area.

Species	2009-2013 lethal take	2009-2013 sub lethal take
Hawaiian monk seal	59	41,590
Green sea turtles-North Pacific	0	12,490
Hawksbill sea turtle-North Pacific	0	1,500

Leatherback sea turtle-North Pacific	0	2,957
Loggerhead sea turtle-North Pacific	0	1,886
Olive ridley sea turtle-North Pacific	0	7,686

5.2.8 Vessel Approaches – Commercial and Private Marine Mammal Watching

Although considered by many to be a non-consumptive use of marine mammals with economic, recreational, educational and scientific benefits, marine mammal watching is not without potential negative impacts. Whale watching has the potential to harass whales by altering feeding, breeding, and social behavior or even injure them if the vessel gets too close or strikes the whale. Another concern is that preferred habitats may be abandoned if disturbance levels are too high. In the Notice of Availability of Revised Whale Watch Guidelines for Vessel Operations in the Northeastern United States (64 FR 29270; June 1, 1999), NMFS noted that whale watch vessel operators seek out areas where whales concentrate, which has led to numbers of vessels congregating around groups of whales, increasing the potential for harassment, injury, or even the death of these animals. In addition to whale watching vessels, large cruise vessels also operate in waters off the coast of Alaska, and may pose a threat to humpback whales. Whale watching, particularly of humpback whales, is extensive in Hawaiian waters during winter. The interactions that individuals experience in these waters likely influence how they react to approaches by vessels in the future (Louis M. Herman, 1979).

Several studies have specifically examined the effects of whale watching on marine mammals, and investigators have observed a variety of short-term responses from animals, ranging from no apparent response to changes in vocalizations, duration of time spent at the surface, swimming speed, swimming angle or direction, respiration rate, dive time, feeding behavior, and social behavior (NMFS, 2006a). Responses appear to be dependent on factors such as vessel proximity, speed, and direction, as well as the number of vessels in the vicinity (W. W. L. Au & Green., 2000; Corkeron, 1995; C. Erbe, 2002; Magalhaes, Prieto, Silva, Goncalves, Afonso-Dias, & Santos., 2002; C. F. Richter, Dawson, & Slooten., 2003; Scheidat, Castro, Gonzalez, & Williams., 2004; Watkins, 1986; Williams, Bain, Ford, & Trites., 2002; Williams, Trites, & Bain., 2002). Foote et al. (2004) reported that southern resident killer whale call duration in the presence of whale watching boats increased by 10-15 percent between 1989-1992 and 2001-2003 and suggested this indicated compensation for a noisier environment. Disturbance by whale watch vessels has also been noted to cause newborn calves to separate briefly from their mothers' sides, which leads to greater energy expenditures by the calves (NMFS, 2006a). Although numerous short-term behavioral responses to whale watching vessels are documented, little information is available on whether long-term negative effects result from whale watching (NMFS, 2006a).

It is difficult to precisely quantify or estimate the magnitude of the risks posed to marine mammals in general from vessel approaches. Given that SURTASS LFA sonar will not occur in

areas with 22 km of land, few whale watching boats would be expected to co-occur with the SURTASS vessels.

5.2.9 Ongoing SURTASS LFA Sonar Training, Operations, and Testing

The Surveillance Towed Array Sensor System (SURTASS) is an element of the Integrated Undersea Surveillance System (IUSS), providing mobile detection, tracking, and reporting of submarine contacts at long range. SURTASS LFA sonar was developed and deployed in the early 1980s as the mobile, tactical arm of the IUSS, providing long range detection and cuing for tactical weapons platforms against both diesel and nuclear powered submarines. As the Navy conducted testing of the system in the mid-1990s, some public interest groups and scientists raised concerns that SURTASS/LFA may cause harm to marine mammals.

SURTASS LFA sonar noise makes up a very small part of the human-caused noise pollution in the ocean. There are two types of noise in the ocean, natural and anthropogenic (human-caused). Natural noise is caused by wind, waves, rain, earthquakes, and marine life. Human-caused noise is created mostly by shipping and in inshore waters by seismic and construction, and recreational boaters. Both ships and boats have sonar noise, in addition to vessel noise from their engines and props. Therefore, by definition, all noise sources, natural and human caused result in the total level of background noise in the oceanic region in which it takes place.

SURTASS LFA sonar however, is a coherent low frequency signal with a duty cycle of less than 20 percent, operating for a maximum of only 432 hours/year for each system or a total of 32 days/year. This compares to an approximate 21.9 million days/year for the world's shipping industry (presuming an 80 percent activity rate all the time). Thus, SURTASS LFA sonar noise would make up a very small part of the human-caused noise pollution in the ocean.

5.2.9.1 Previous MMPA Authorizations for SURTASS LFA Sonar Activities

The information below is a summary of annual and programmatic reports provided by the Navy as a requirement of the ESA and MMPA authorizations. Additional details may be found in the individual reports (Navy, 2003, 2004, 2005a, 2006a, 2007a, 2007b, 2008b, 2009b, 2010b, 2011a).

Past operation of SURTASS LFA sonar in the western and central North Pacific Ocean over the eleven-year period spanning 2002 through 2013 (ending at the August 2013 LOA/ITS reporting period) involved 156 completed missions conducted over 480 days during which LFA sonar was transmitted for a total of 930 hr (or about 39 of a possible 4,015 days). During those missions, only 10 marine mammals or sea turtles were visually observed, eight marine animals were detected passive-acoustically, and 149 marine mammals/animals were detected active-acoustically by the high frequency marine mammal monitoring (HF/M3) sonar system. These combined detections of marine animals led to 1603 suspensions/delays of LFA sonar transmissions, per the mitigation protocol for the operation of SURTASS LFA sonar.

The acoustic (passive and active [HF/M3]) and visual mitigation and monitoring measures have been employed aboard four SURTASS LFA sonar vessels/systems that have operated in the western North Pacific Ocean during the annual LOA and ITS reporting period from August 2012 through August 2013 and the most recent period commencing 15 August 2013. For the 2012 through 2013 annual reporting period, 47.3 hours (hr) of LFA sonar were transmitted during 12 missions over 24.4 mission days (Table 17). During this annual reporting period, in accordance with the mitigation monitoring protocol, LFA sonar was suspended 13 times due to three passive and 10 active acoustic detections of marine mammals. No ESA-listed species were observed nor were any dead or injured marine species observed during the 2012 to 2013 reporting period. The LOAs for SURTASS LFA permit up to 432 hr of LFA sonar transmissions per SURTASS LFA vessel annually for a combined total of 1,728 hr of LFA sonar transmissions in the western and central North Pacific Ocean. However, in the 2012 to 2013 LOA reporting period, all four SURTASS LFA vessels only transmitted a grand total of 47.3 hr of LFA sonar or 2.7% of the permitted sonar transmit time (**Table 17**).

During the first two quarters of the current LOA reporting period (15 August 2013 through 14 February 2014), two missions have been completed over 4 days, during which LFA sonar was transmitted for 9.75 hr with no visual, passive acoustic, or active acoustic detections of marine animals having been made, no shutdowns of the LFA sonar, and no reports of dead or injured marine animals (Table 3). Half way through the current LOAs reporting period, far less transmit time, only 9.75 hr for all vessels or 0.6% of the permitted total sonar transmissions, have been conducted thus far (**Table 18**).

Both the historical and the most recent results of the mitigation monitoring and effectiveness support the Navy's and NMFS' assertions that the Navy's three-part mitigation and monitoring protocols provide an effective means of avoiding risk of injury to protected marine species.

Table 16. Summary of SURTASS LFA sonar missions and mitigation measure detections from 2002 through 2nd Quarter 2014.

Year [1]	Vessel	Mission Summary			Visual Detections	Passive Acoustic Detections	Active Acoustic HF/M3 Detections	HF/M3 Unavailable[2]	Mitigation Protocol Suspensions/ Delays
		Number	Days	LFA Sonar Hours					
2002-2003	R/V *Cory Chouest*	7	34.2	82.2	0	0	3	0	3
2003-2004	R/V *Cory Chouest*	5	72.5	173.7	0	0	10	0	10
	USNS IMPECCABLE	5			0	0	6	2	8
2004-2005	R/V *Cory Chouest*	3	22.5	41.9	0	0	1	11	12
	USNS IMPECCABLE	2			0	0	1	0	1
2005-2006	R/V *Cory Chouest*	12	95.6	173.2	1	0	47	10	58
	USNS IMPECCABLE	6			2	0	3	0	5
2006-2007	R/V *Cory Chouest*	6	94.6	161.5	0	0	30	50	80
	USNS IMPECCABLE	12			1	0	0	0	1
2007-2008	R/V Cory Chouest	6	49.8	135.8	0	0	0	16	16
	USNS IMPECCABLE	8			1	0	19	7	27
2008-2009	USNS ABLE	3	23.7	32.5	1	0	1	3	5
	USNS IMPECCABLE	6			2	0	1	0	3
2009-2010	USNS ABLE	10	17.1	43.6	0	0	0	0	0
	USNS IMPECCABLE	21			1[3]	3[4]	3[4]	4	7
2010-2011	USNS ABLE	8	62.3	64.0	1[3]	1	1[3]	0	1
	USNS EFFECTIVE	1			2 [5]	0	2[6]	1	4

	USNS IMPECCABLE	7			0	0	2	0	2
2011-2012	USNS ABLE	6	28.5	66.3	0[7]	1	1	0	3[8]
	USNS EFFECTIVE	5			0	0	0	0	0
	USNS IMPECCABLE	3			0	0	0	0	0
	USNS VICTORIOUS	1			0	0	0	0	0
2012-2013	USNS ABLE	3	24.4	47.3	0	0	0	0	0
	USNS EFFECTIVE	4			0	3	9	12	16
	USNS IMPECCABLE	2			0	0	0	0	12
	USNS VICTORIOUS	3			0	0	1	1	1
2013-2014 Through 2nd Quarter	USNS ABLE	0	4	9.75	0	0	0	0	0
	USNS EFFECTIVE	2			0	0	0	0	0
	USNS IMPECCABLE	0			0	0	0	0	0
	USNS VICTORIOUS	0			0	0	0	0	0
	Totals	**157**	**549.2**	**1031.8**	**12**	**8**	**141**	**117**	**275**

[1] *August to August*
[2] *LFA sonar suspended during HF/M3 non-availability*
[3] *Contact made when LFA sonar not transmitting*
[4] *Marine mammal passive contacts verified by HF/M3 sonar system*
[5] *One contact confirmed with HF/M3 sonar, second was a sea turtle*
[6] *Only one contact confirmed visually*
[7] *1 visual sighting of a marine mammal after the mission ended and LFA sonar was non-operational*
[8] *1 shut down due to loss of passive acoustics system*

Table 17. Summary of SURTASS LFA Sonar Operations and Preventative Measures During the Annual LOAs and ITS Reporting Period from 15 August 2010 through 14 August 2013

ANNUAL SURTASS LFA SONAR OPERATIONS								
SURTASS LFA VESSEL	**MISSIONS**	**MISSION DURATION (DAYS)**	**LFA SONAR TRANSMISSIONS (HOURS)**	**VISUAL DETECTIONS**	**PASSIVE ACOUSTIC DETECTIONS**	**HF/M3-ACTIVE ACOUSTIC DETECTIONS**	**LFA SONAR SUSPENSIONS DUE TO DETECTIONS**	**TOTAL SUSPENSIONS OF LFA SONAR[1]**
USNS ABLE (T-AGOS 20)	3	2.5	5.4	0	0	0	0	0
USNS EFFECTIVE (T-AGOS 21)	4	12.5	22.5	0	3	9	12	16
USNS IMPECCABLE (T-AGOS 23)	2	2.5	5.2	0	0	0	0	12
USNS VICTORIOUS (T-AGOS 19)	3	6.9	14.2	0	0	1	1	1
Totals	**12**	**24.4**	**47.3**	**0**	**3**	**10**	**13**	**29**

[1]In addition to LFA sonar suspensions due to visual, passive acoustic, or active acoustic/HFM3 detections, suspensions of LFA sonar transmissions are also due to loss of the passive acoustic system, HFM3 system faults, HFM3 system artifacts, or impedance checks.

Table 18. Summary of SURTASS LFA Sonar Operations and Preventative Measures During the First Two Quarters (August through February) of the Annual LOAs and ITS Reporting Period from 15 August 2013 through 14 August 2014

ANNUAL SURTASS LFA SONAR OPERATIONS								
SURTASS LFA VESSEL	**MISSIONS**	**MISSION DURATION (DAYS)**	**LFA SONAR TRANSMISSIONS (HOURS)**	**VISUAL DETECTIONS**	**PASSIVE ACOUSTIC DETECTIONS**	**HF/M3-ACTIVE ACOUSTIC DETECTIONS**	**LFA SONAR SUSPENSIONS DUE TO DETECTIONS**	**TOTAL SUSPENSIONS OF LFA SONAR[1]**
USNS ABLE (T-AGOS 20)	0	0	0	0	0	0	0	0
USNS EFFECTIVE (T-AGOS 21)	2	4.0	9.75	0	0	0	0	0
USNS IMPECCABLE (T-AGOS 23)	0	0	0	0	0	0	0	0
USNS VICTORIOUS (T-AGOS 19)	0	0	0	0	0	0	0	0
Totals	**2**	**4**	**9.75**	**0**	**0**	**0**	**0**	**0**

[1]In addition to LFA sonar suspensions due to visual, passive acoustic, or active acoustic/HFM3 detections, suspensions of LFA sonar transmissions are also due to loss of the passive acoustic system, HFM3 system faults, HFM3 system artifacts, or impedance checks.

5.2.10 Ongoing Navy Pacific Fleet Training and Testing Activities

The Navy conducts military readiness activities, which can be categorized as either training or testing exercises, throughout the North Pacific Ocean. During training activities, existing and established weapon systems and tactics are used in realistic situations to simulate and prepare for combat. Testing activities are conducted for different purposes and include at-sea research, development, test, evaluation, and experimentation. The Navy performs testing activities to ensure that its military forces have the latest technologies and techniques available to them.

Training exercises in the Pacific can be classified in eight broad activity categories that include routine gunnery, missile, surface fire support, amphibious assault and landing, bombing, sinking, torpedo, tracking, torpedo, and mine exercises. In addition to routine training exercises, major Pacific training events are conducted that involve multiple training exercises and more ships, aircraft, and submarines than participate in one typical training exercise. Examples of major Pacific training events are the biennial Rim of the Pacific (RIMPAC) Exercise, Ship ASW (anti-submarine warfare) Readiness and Evaluation Measuring (SHAREM) Exercise, the Talisman Saber Exercise began on July 15 and ended on August 5, 2013, and Composite Training Unit Exercise (COMPTUEX). The most recent biennial RIMPAC Exercise was completed 3 August 2012 and involved the navies of 22 nations, more than 40 ships and submarines, as well as more than 200 aircraft that operated in and around the Hawaiian Islands.

Testing exercises are conducted to gather not only research data but also to gather data about new systems and technologies the Navy is interested in potentially acquiring and additionally provides a way in which these new systems and technologies can be demonstrated for the Navy. The Navy conducts more than fourteen categories of testing exercises in the North Pacific. Test activities may include testing and evaluation of underwater vehicles, ship and pier protection systems, and mine counter-measures; research and development of underwater acoustic communications; and evaluation of using undersea gliders to collect oceanographic, acoustic, and meteorology data.

The Office of Naval Research (ONR), including the Naval Research Laboratory, and the Space and Naval Warfare Systems Center Pacific (SPAWAR PAC) are two Navy organizations that conduct the majority of the Navy's research and develop technologies for future Navy use. SPAWAR PAC focuses its research and test efforts on developing and transitioning technologies associated the Navy operations such as command, control, communications, computers, intelligence, surveillance, and reconnaissance. ONR's purposes are to plan, foster, and encourage scientific research in recognition of its importance in maintaining future naval power and the preservation of national security. SPAWAR PAC has conducted research such as the Autonomous Oceanographic Research and Meteorology and Oceanography experiments that use ocean gliders and autonomous undersea vehicles (AUVs) to gather data on the physical and chemical properties of the oceans as well as on weather events in less costly ways than gathering the same information using ships. ONR has sponsored a North Pacific experiment titled the

Kauai Acoustic Communications Experiment in which oceanographic and environmental data were collected in coastal waters of Hawaii to better understand the coupling of oceanography, acoustics, and underwater communications.

5.3 The Impact of the Baseline on Listed Resources

Although listed resources are exposed to a wide variety of past and present state, Federal or private actions and other human activities in the action area; Federal projects in the action area that have already undergone formal or early section 7 consultation; and State or private actions that are contemporaneous with this consultation; the impact of those activities on the status, trend, or the demographic processes of threatened and endangered species remains largely unknown.

Historically, commercial whaling had occurred in the action area and had caused all of the large whales to decline to the point where the whales faced risks of extinction that were high enough to list them as endangered species. Commercial whaling has been radically reduced. However, all of the whale species have not recovered from those historic declines and scientists cannot determine if those initial declines continue to influence current populations of most large whale species. Species like North Pacific right whales have not begun to recover from the effects of commercial whaling on their populations and continue to face very high risks of extinction in the foreseeable future because of their small population sizes (on the order of 50 individuals) and low population growth rates. Relationships between potential stressors in the marine environments and the responses of these species that may keep their populations depressed are unknown.

Recent attention has focused on the emergence of a wide number of anthropogenic sound sources in the action area and their role as a putative pollutant in the marine environment. Relationships between specific sound sources, or anthropogenic sound generally, and the responses of marine mammals to those sources are still subject to extensive scientific research and public inquiry but no clear patterns have emerged. As a result, the consequences of these activities on threatened and endangered marine mammals remain uncertain.

The anthropogenic phenomena that represent potential risks to whales in the Action Area are — close approaches by whale-watching and research vessels, anthropogenic sound sources, pollution, and many fishery interactions. — These have the potential to cause injury and/or to affect the behavioral, physiological, or social ecology of whales in the region. The second line of evidence consists of reports that suggest that the response of whales to many of the human activities in the action area are probably short-lived, which suggests that the responses would not be expected to affect the fitness of individual whales. Most of these reports relate to humpback whales during their winter, breeding season; there are very few reports of the behavioral responses of other whale species to human activity in the action area. For example, annual reports from the North Gulf Oceanic Society and two other investigators reported that most

whales did not react to approaches by their vessels or only small numbers of whales reacted. That is, in their 1999 report on their research activities, non-governmental organizations reported observing signs that whales were "disturbed" in only 3 out of 51 encounters with whales and that the whales' behavioral responses consisted of breaching, slapping tail and pectoral fin, and diving away from research vessels.

Gauthier and Sears (1999) , Weinrich et al. (1992), Clapham and Mattila (1993), Clapham et al. (1993) concluded that close approaches for biopsy samples or tagging did not cause humpback whales to respond or caused them to exhibit "minimal" responses when approaches were "slow and careful." This caveat is important and is based on studies conducted by Clapham and Mattila (1993) of the reactions of humpback whales to biopsy sampling in breeding areas in the Caribbean Sea. These investigators concluded that the way a vessel approaches a group of whales had a major influence on the whale's response to the approach; particularly cow and calf pairs. Based on their experiments with different approach strategies, they concluded that experienced, trained personnel approaching humpback whales slowly would result in fewer whales exhibiting behavioral reactions to these approaches. At the same time, several lines of evidence suggest that these human activities might be greater consequences for individual whales (if not for whale populations). Several investigators reported behavioral responses to close approaches that suggest that individual whales might experience stress responses. Baker et al. (1983) described two responses of whales to vessels, including: (1) "horizontal avoidance" of vessels 2,000 to 4,000 m away characterized by faster swimming and fewer long dives; and (2) "vertical avoidance" of vessels from 0 to 2,000 m away during which whales swam more slowly, but spent more time submerged. Watkins et al. (1981) found that both fin and humpback whales appeared to react to vessel approach by increasing swim speed, exhibiting a startled reaction, and moving away from the vessel with strong fluke motions.

Bauer (1986) and Bauer and Herman (1986) studied the potential consequences of vessel disturbance on humpback whales wintering off Hawaii. They noted changes in respiration, diving, swimming speed, social exchanges, and other behavior correlated with the number, speed, direction, and proximity of vessels. Results were different depending on the social status of the whales being observed (single males when compared with cows and calves), but humpback whales generally tried to avoid vessels when the vessels were 0.5 to 1.0 kilometer from the whale. Smaller pods of whales and pods with calves seemed more responsive to approaching vessels.

Baker et al. (1983) and Baker and Herman (1987) summarized the response of humpback whales to vessels in their summering areas and reached conclusions similar to those reached by Bauer and Herman (1986): these stimuli are probably stressful to the humpback whales in the action area, but the consequences of this stress on the individual whales remains unknown. Studies of other baleen whales, specifically bowhead and gray whales document similar patterns of short-term, behavioral disturbance in response to a variety of actual and simulated vessel activity and

noise (Malme et al., 1983; W. John Richardson, Wells, et al., 1985). For example, studies of bowhead whales revealed that these whales oriented themselves in relation to a vessel when the engine was on, and exhibited significant avoidance responses when the vessel's engine was turned on even at distance of approximately 3,000 ft (900 m). Weinrich et al. (1992) associated "moderate" and "strong" behavioral responses with alarm reactions and stress responses, respectively.

Jahoda et al. (2003) studied the response of 25 fin whales in feeding areas in the Ligurian Sea to close approaches by inflatable vessels and to biopsy samples. They concluded that close vessel approaches caused these whales to stop feeding and swim away from the approaching vessel. The whales also tended to reduce the time they spent at surface and increase their blow rates, suggesting an increase in metabolic rates that might indicate a stress response to the approach. In their study, whales that had been disturbed while feeding remained disturbed for hours after the exposure ended. They recommended keeping vessels more than 200 meters from whales and having approaching vessels move a low speeds to reduce visible reactions in these whales.

Beale and Monaghan (2004) concluded that the significance of disturbance was a function of the distance of humans to the animals, the number of humans making the close approach, and the frequency of the approaches. These results would suggest that the cumulative effects of the various human activities in the action area would be greater than the effects of the individual activity. None of the existing studies examined the potential effects of numerous close approaches on whales or gathered information of levels of stress-related hormones in blood samples that are more definitive indicators of stress (or its absence) in animals.

There is mounting evidence that wild animals respond to human disturbance in the same way that they respond to predators (C.M. Beale & P. Monaghan, 2004; Frid, 2003; Frid & Dill, 2002; Gill & Sutherland, 2001; Romero, 2004). These responses manifest themselves as stress responses (in which an animal perceives human activity as a potential threat and undergoes physiological changes to prepare for a flight or fight response or more serious physiological changes with chronic exposure to stressors), interruptions of essential behavioral or physiological events, alteration of an animal's time budget, or some combinations of these responses (Frid & Dill, 2002; Romero, 2004; Robert M. Sapolsky, Romero, & Munck, 2000; Walker, Dee Boersma, & Wingfield, 2005). These responses have been associated with abandonment of sites (Sutherland & Crockford, 1993), reduced reproductive success (M. Giese, 1996; Müllner, Eduard Linsenmair, & Wikelski, 2004), and the death of individual animals (Daan, Deerenberg, & Dijkstra, 1996).

6 EFFECTS OF THE ACTION

"Effects of the action" means the direct and indirect effects of an action on the species or critical habitat that will be added to the environmental baseline, together with the effects of other activities that are interrelated or interdependent with that action (50 CFR § 402.02). Indirect effects are those that are caused by the proposed action later in time but are still reasonably certain to occur.

Here we identify the potential stressors that are likely to have direct and indirect effect on the physical, chemical, and biotic environment of the action area. Then we describe the mitigation proposed to reduce the likelihood of exposure to listed species from those stressors. Our analysis assumes that these stressors pose no risk to listed species if these stressors do not co-occur with those species in space or time. We recognize that the sonar could have indirect, adverse effects on listed species by disrupting marine food chains, a species' predators, or a species' competitors; however, we did not identify situations where this concern might apply to species under NMFS' jurisdiction. Next, we examine the likely responses of listed species given the exposure to the stressors. Last, we evaluate the risk to those species given the stressors, the exposure, and the potential responses to those stressors.

6.1 Stressors Associated with the Proposed Action

The primary stressors associated with the SURTASS LFA sonar operations are:

(1) The ships and ship traffic associated with the proposed sonar activities;

(2) The passive sonar system (SURTASS);

(3) The low-frequency active (LFA sonar) component of the sonar system; and

(4) The high-frequency (HF/M3) active component of the monitoring/mitigation system.

The potential effects of these stressors have been discussed in multiple documents including previous biological opinions on the U.S. Navy's SURTASS LFA sonar operations, biological assessments, and analysis conducted under the National Environmental Policy Act (NEPA) (Navy, 2001b, 2006b, 2007b, 2007c, 2011b, 2012a; NMFS, 2002, 2003, 2004c, 2005b, 2007a, 2007b, 2008, 2009c, 2010a). Below we provide a brief summary of the potential effects of the stressors listed above by species groups, cetaceans, pinnipeds, and sea turtles. We direct the reader to the documents referenced above for more detailed descriptions of the potential effects of these on listed species.

6.1.1 Vessel Operation - Noise

Potential adverse effects on listed species could occur the generation of engine and propeller noise. Sound emitted from large vessels, such as shipping and cruise ships, is the principal source

of low-frequency noise in the ocean today, and marine mammals are known to react to or be affected by that noise (Foote, Osborne, & Hoelzel, 2004; Leila T. Hatch & Wright, 2007; John Hildebrand, 2005; M. M. Holt, V. Veirs, & S. Veirs, 2008; Melcon et al., 2012; W. John Richardson, Greene Jr., Malme, & Thomson, 1995) (Kerosky et al. 2013). In short-term studies, researchers have noted changes in resting and surface behavior states of cetaceans to whale watching vessels (Acevedo, 1991; Aguilar Soto et al., 2006; Arcangeli & Crosti, 2009; W. W. L. Au & Green, 2000; Christiansen, Lusseau, Stensland, & Berggren, 2010; C. Erbe, 2002; Noren, Johnson, Rehder, & Larson, 2009; Stensland & Berggren, 2007; Stockin, Lusseau, Binedell, Wiseman, & Orams, 2008; Williams & Noren, 2009). Noren et al. (2009) conducted research in the San Juan Islands in 2005 and 2006 and their findings suggested that close approaches by vessels impacted the whales' behavior and that the whale-watching guideline minimum approach distance of 100 m may be insufficient in preventing behavioral responses.

Most studies of this type are opportunistic and have only examined the short-term response to vessel sound and vessel traffic ((Magalhaes, Prieto, Silva, Goncalves, Afonso-Dias, & Santos, 2002; Noren et al., 2009; W. John Richardson & Wursig, 1995; Watkins, 1981c). Long-term and cumulative implications of vessel sound on marine mammals remains largely unknown (National Marine Fisheries Service 2012a, b). Christopher W. Clark et al. (2009) provided a discussion on calculating the cumulative impacts of anthropogenic noise on baleen whales and estimated that in one Atlantic setting and with the noise from the passage of two vessels, the optimal communication space for North Atlantic right whale could be decreased by 84 percent (see also Hatch et al. 2013).

Bassett, Polagye, Holt, and Thomson (2012) recorded vessel traffic over a period of just under a year as large vessels passed within 20 km of a hydrophone site located at Admiralty Inlet in Puget Sound. During this period there were 1,363 unique Automatic Identification System transmitting vessels recorded.

6.1.1.1 Mysticetes

Fin whales may alter their swimming patterns by increasing speed and heading away from a vessel, as well as changing their breathing patterns in response to a vessel approach (Jahoda et al. 2003). Vessels that remained 328 ft. (100 m) or farther from fin and humpback whales were largely ignored in one study in an area where whale watching activities are common (Watkins 1981). Only when vessels approached more closely did the fin whales in this study alter their behavior by increasing time at the surface and exhibiting avoidance behaviors. Other studies have shown when vessels are near, some but not all fin whales change their vocalizations, surface time, swimming speed, swimming angle or direction, respiration rates, dive times, feeding behavior, and social interactions (Castellote, Clark, & Lammers, 2012) (Au and Green 2000; Richter et al. 2003; Williams et al. 2002).

Based on passive acoustic recordings and in the presence of sounds from passing vessels, Melcon et al. (2012) reported that blue whales had an increased likelihood of producing certain types of calls. Castellote et al. (2012) demonstrated that fin whales' songs had shortened duration and decreased bandwidth, center frequency, and peak frequency in the presence of high shipping noise levels such as those found in the Strait of Gibraltar. At present it is not known if these changes in vocal behavior corresponded to any other behaviors.

In the Watkins (1981) study, humpback whales did not exhibit any avoidance behavior but did react to vessel presence. In a study of regional vessel traffic, Baker et al. (1983) found that when vessels were in the area, the respiration patterns of the humpback whales changed. The whales also exhibited two forms of behavioral avoidance: horizontal avoidance (changing direction or speed) when vessels were between 1.24 and 2.48 mi. (2,000 and 4,000 m) away, and vertical avoidance (increased dive times and change in diving pattern) when vessels were within approximately 1.2 mi. (2,000 m; Baker and Herman 1983). Similar findings were documented for humpback whales when approached by whale watch vessels in Hawaii and having responses that including increased speed, changed direction to avoid, and staying submerged for longer periods of time (W. W. L. Au & Green, 2000).

Recently, Gende et al. (2011) reported on observations of humpback whale in inland waters of Southeast Alaska subjected to frequent cruise ship transits (i.e., in excess of 400 transits in a four-month season in 2009). The study was focused on determining if close encounter distance was a function of vessel speed. The reported observations, however, seem in conflict with other reports of avoidance at much greater distance so it may be that humpback whales in those waters are more tolerant of vessels (given their frequency) or are engaged in behaviors, such as feeding, that they are less willing to abandon. This example again highlights that context is critical for predicting and understanding behavioral reactions as concluded by Southall et al. (2007a, b) and Ellison, Southall, Clark, and Frankel (2012).

Sei whales have been observed ignoring the presence of vessels and passing close to the vessel. In the presence of approaching vessels, blue whales perform shallower dives accompanied by more frequent surfacing, but otherwise do not exhibit strong reactions (John Calambokidis, Barlow, Ford, Chandler, & Douglas, 2009). Minke whales in the Antarctic did not show any apparent response to a survey vessel moving at normal cruising speeds (about 12 knots) at a distance of 5.5 nm; however, when the vessel drifted or moved at very slow speeds (about 1 knot), many whales approached it (Stephen Leatherwood, Awbrey, & Thomas, 1982).

Although not expected to be in the Action Area, North Atlantic right whales tend not to respond to the sounds of oncoming vessels (Douglas P. Nowacek, Mark P. Johnson, & Peter L. Tyack, 2004). North Atlantic right whales continue to use habitats in high vessel traffic areas (Douglas P. Nowacek et al., 2004). Studies show that North Atlantic right whales demonstrate little if any reaction to sounds of vessels approaching or the presence of the vessels themselves (Terhune and

Verboom 1999, (Douglas P. Nowacek et al., 2004). Although this may minimize potential disturbance from passing ships, it does increase the whales' vulnerability to potential ship strike.

Using historical records, Watkins (1986) showed that the reactions of four species of mysticetes to vessel traffic and whale watching activities in Cape Cod had changed over the 25-year period examined (1957–1982). Reactions of minke whales changed from initially more positive reactions, such as coming towards the boat or research equipment to investigate, to more 'uninterested' reactions towards the end of the study. Finback [fin] whales, the most numerous species in the area, showed a trend from initially more negative reactions, such as swimming away from the boat with limited surfacing, to more uninterested (ignoring) reactions allowing boats to approach within 98.4 ft. (30 m). Right whales showed little change over the study period, with a roughly equal number of reactions judged to be negative and uninterested; no right whales were noted as having positive reactions to vessels. Humpback whales showed a trend from negative to positive reactions with vessels during the study period. The author concluded that the whales had habituated to the human activities over time (Watkins 1986).

Mysticetes have been shown to both increase and decrease calling behavior in the presence of vessel noise. An increase in feeding call rates and repetition by humpback whales in Alaskan waters was associated with vessel noise (Doyle et al., 2008); Melcón et al. (2012) also recently documented that blue whales increased the proportion of time spent producing certain types of calls when vessels were present. Conversely, decreases in singing activity by humpback whales have been noted near Brazil due to boat traffic (Sousa-Lima & Clark, 2008). The Central North Pacific stock of humpback whales is the focus of whale-watching activities in both its feeding grounds (Alaska) and breeding grounds (Hawaii). Regulations addressing minimum approach distances and vessel operating procedures are in place in Hawaii, however, there is still concern that whales may abandon preferred habitats if the disturbance is too high (B. M. Allen & Angliss, 2010).

6.1.1.2 Odontocetes

Sperm whales generally react only to vessels approaching within several hundred meters; however, some individuals may display avoidance behavior, such as quick diving (Magalhaes, Prieto, Silva, Goncalves, Afonso-Dias, & Santos, 2002; Wursig, Lynn, Jefferson, & Mullin, 1998). One study showed that after diving, sperm whales showed a reduced timeframe from when they emitting the first click than before vessel interaction (Richter et al. 2006). The smaller whale-watching and research vessels generate more noise in higher frequency bands and are more likely to approach odontocetes directly, and to spend more time near the individual whale. Reactions to Navy vessels are not well documented, but smaller whale-watching and research boats have been shown to cause these species to alter their breathing intervals and echolocation patterns.

Wursig et al. (1998) reported most *Kogia* species and beaked whales react negatively to vessels by quick diving and other avoidance maneuvers. Cox et al. (2006) noted very little information is available on the behavioral impacts of vessels or vessel noise on beaked whales. A single observation of vocal disruption of a foraging dive by a tagged Cuvier's beaked whale documented when a large noisy vessel was opportunistically present, suggests that vessel noise may disturb foraging beaked whales (Aguilar Soto et al. 2006). Tyack et al. (2011) noted the result of a controlled exposure to pseudorandom noise suggests that beaked whales would respond to vessel noise and at similar received levels to those noted previously and for mid-frequency sonar.

Most delphinids react neutrally to vessels, although both avoidance and attraction behavior is known (Hewitt, 1985; Wursig et al., 1998). Avoidance reactions include a decrease in resting behavior or change in travel direction (Bejder et al. 2006a). Incidence of attraction includes harbor porpoises approaching a vessel and common, rough-toothed, and bottlenose dolphins bow riding and jumping in the wake of a vessel (Kenneth S. Norris & Prescott, 1961; S. H. Shane, Wells, Wursig, & Wursig, 1986) (Ritter, 2002; Wursig et al., 1998). A study of vessel reactions by dolphin communities in the eastern tropical Pacific found that populations that were often the target of tuna purse-seine fisheries (spotted, spinner and common dolphins) show evasive behavior when approached; however populations that live closer to shore (within 100 nm; coastal spotted and bottlenose dolphins) that are not set on by purse-seine fisheries tend to be attracted to vessels (Archer, Mesnick, & Allen, 2010; Archer, Redfern, Gerrodette, Chivers, & Perrin, 2010).

Killer whales, the largest of the delphinids, are targeted by numerous small whale-watching vessels in the Pacific Northwest and, from 1998 to 2012 during the viewing season, have had an annual monthly average of nearly 20 vessels of various types within 0.5 mile of their location from between the hours of 9 a.m. and 6 p.m. (Eisenhardt 2012). For the 2012 season, it was reported that 1,590 vessel incidents were possible violations of the federal vessel approach regulations or MMPA and ESA laws as well (Eisenhardt 2012). Research suggests that whale-watching guideline distances may be insufficient to prevent behavioral disturbances due to vessel noise (Noren et al., 2009). In 2012, there were 79 U.S. and Canadian commercial whale watch vessels in the Haro Strait region (Eisenhardt 2012).These vessels have measured source levels that ranged from 145 to 169 dB re 1 µPa at 1 m and have the sound they produce underwater has the potential to result in behavioral disturbance, interfere with communication, and affect the killer whales' hearing (Erbe 2002). Killer whales foraged significantly less and traveled significantly more when boats were within 328 ft. (100 m) of the whales (Kruse, 1991; Trites & Bain, 2000; Williams & Noren, 2009; Williams, Trites, & Bain, 2002);Lusseau et al. 2009). These short-term feeding activity disruptions may have important long-term population-level effects (Lusseau et al. 2009; (Noren et al., 2009). The reaction of the killer whales to whale-watching vessels may be in response to the vessel pursuing them, rather than to the noise of the vessel itself, or to the number of vessels in their proximity.

Similar behavioral changes (increases in traveling and other stress-related behaviors) have been documented in Indo-Pacific bottlenose dolphins in Zanzibar (Christiansen et al., 2010; Englund & Berggren, 2002; Stensland & Berggren, 2007). Short-term displacement of dolphins due to tourist boat presence has been documented (Carrera, Favaro, & Souto, 2008), while longer term or repetitive/sustained displacement for some dolphin groups due to chronic vessel noise has been noted (Haviland-Howell et al., 2007; Miksis-Olds, Donaghay, Miller, Tyack, & Reynolds, 2007). Most studies of the behavioral reactions to vessel traffic of bottlenose dolphins have documented at least short-term changes in behavior, activities, or vocalization patterns when vessels are near, although the distinction between vessel noise and vessel movement has not been made clear (Acevedo, 1991; Arcangeli & Crosti, 2009; Berrow & Holmes, 1999; Gregory & Rowden, 2001; Janik & Thompson, 1996; David Lusseau, 2004; Mattson, Thomas, & Aubin, 2005; Scarpaci, Bigger, Corkeron, & Nugegoda, 2000).

Both finless porpoises (Li et al. 2008) and harbor porpoises (Polacheck & Thorpe, 1990) routinely avoid and swim away from large motorized vessels. The vaquita, which is closely related to the harbor porpoise in the Action Area, appears to avoid large vessels at about 2,995 ft. (913 m) Jaramillo-Legorreta, 1999 #67695}. The assumption is that the harbor porpoise would respond similarly to large Navy vessels.

Odontocetes have been shown to make short-term changes to vocal parameters such as intensity (M. Holt, V. Veirs, & S. Veirs, 2008) as an immediate response to vessel noise, as well as increase the pitch, frequency modulation, and length of whistling (May-Collado & Wartzok, 2008). Likewise, modification of multiple vocalization parameters has been shown in belugas residing in an area known for high levels of commercial traffic. These animals decreased their call rate, increased certain types of calls, and shifted upward in frequency content in the presence of small vessel noise (Lesage, Barrette, Kingsley, & Sjare, 1999). Another study detected a measurable increase in the amplitude of their vocalizations when ships were present (Scheifele et al., 2005a). Killer whales off the northwestern coast of the United States have been observed to increase the duration of primary calls once a threshold in observed vessel density (e.g., whale watching) was reached, which has been suggested as a response to increased masking noise produced by the vessels (Foote, Osborne, & Hoelzel, 2004). On the other hand, long-term modifications to vocalizations may be indicative of a learned response to chronic noise, or of a genetic or physiological shift in the populations. For example, the source level of killer whale vocalizations has been shown to increase with higher background noise levels associated with vessel traffic (the Lombard effect) (Hotchkin & Parks, 2013). In addition, calls with a high-frequency component have higher source levels than other calls, which may be related to behavioral state, or may reflect a sustained increase in background noise levels (Holt et al. 2008).

6.1.1.3 Pinnipeds

Little is known about pinniped reactions to underwater non-impulsive sounds (Southall et al. 2007a,) including vessel noise. In a review of reports on reactions of pinnipeds to small craft and

ships, Richardson et al. (1995) note that information is on pinniped reactions is limited and most reports are based on anecdotal observations. Specific case reports in Richardson et al. (1995) vary based on factors such as routine anthropogenic activity, distance from the vessel, engine type, wind direction, and ongoing subsistence hunting. As with reactions to sound reviewed by Southall et al. (2007a) pinniped responses to vessels are affected by the context of the situation and by the animal's experience. In summary, pinniped's reactions to vessels are variable and reports include a wide entire spectrum of possibilities from avoidance and alert to cases where animals in the water are attracted and cases on land where there is lack of significant reaction suggesting "habituation" or "tolerance" of vessels (Richardson et al. 1995).

A study of reactions of harbor seals hauled out on ice to cruise ship approaches in Disenchantment Bay, Alaska revealed that animals are more likely to flush and enter the water when cruise ships approach within 1,640 ft. (500 m) and four times more likely when the cruise ship approaches within 328 ft. (100 m) (Jansen et al. 2010). Navy vessels would generally not operate in vicinity of nearshore natural areas that are pinniped haul-out or rookery locations.

6.1.1.4 Sea Turtles

Based on sea turtle sensory biology, Hazel et al. (2007) suggested that green turtles rely more on visual cues than auditory cues when reacting to approaching water vessels. This suggests that sea turtles might not respond to vessel operations based on noise alone.

Based on the information available, endangered and threatened sea turtles may have a brief startle response, but are most likely to ignore Navy vessels entirely and continue behaving as if the vessels and any risks associated with those vessels did not exist.

6.1.1.5 SURTASS LFA Sonar Vessel Noise Conclusions

The SURTASS LFA sonar vessels will operate at speeds of approximately 3 kt (3.5 mph) during SURTASS LFA sonar operations and about 10 kt (11.6 mph) during transit. Other than transit entering or leaving ports, the vessels would generally not be operating close to land because of the geographical restrictions on SURTASS LFA sonar operation that prevent the transmission of the sound field produced by the LFA sonar above 180 dB SPL within 22 km (12 nmi) or in the 22 offshore biologically important areas (OBIAs) designated for low frequency marine mammals in biologically important seasons. Due to the low speed of the SURTASS LFA sonar vessel the generation of engine and propeller noise would be expected to be minimal and discountable.

6.1.2 Vessel Operation - Discharges

Operation of the SURTASS LFA sonar vessels will result in discharges incidental to normal operations of a vessel. The International Convention for the Prevention of Pollution from Ships (MARPOL73/78) prohibits certain discharges of oil, noxious liquid substances, sewage, garbage, and air pollution from vessels. The Convention is implemented by the Act to Prevent Pollution from Ships (APPS; 33 U.S.C. §§ 1905-1915 et seq.), which establishes requirements for the operation of U.S. Navy vessels. The SURTASS LFA sonar vessels will operate in compliance

with these requirements. The sonar system itself will not result in the discharge of any pollutants regulated under the Act to Prevent Pollution from Ships.

6.1.3 Vessel Operation – Ship Strike

Potential adverse effects on listed species could occur through ship strikes. Of the three species categories, cetaceans would have the largest risk of ship strikes. During SURTASS LFA sonar operations, the combination of ship speeds and the three elements of the U.S. Navy's monitoring and mitigation program would be expected to virtually eliminate the risk of ship strikes for cetaceans. During the ten years of MMPA authorizations for SURTASS LFA sonar vessel operations there have not been any ship strikes (Navy, 2011b). Therefore we determine the risk of ship strikes or disturbance from ships for cetaceans to be discountable.

Pinnipeds and sea turtles are not at risk of ship strike because of: (1) the slow speed of the SURTASS LFA sonar vessels (3 kt; 3.5 mph); (2) they would be rare in the offshore areas where the vessels would be maneuvering; (3) there is little evidence that the ESA-listed species of pinnipeds and turtles are vulnerable to ship strikes. Therefore, we determine the risk of ship strikes or disturbance from ships for pinnipeds and sea turtles to be discountable. Therefore, endangered or threatened marine mammals and sea turtles are not likely to be struck by the USNS ABLE, USNS EFFECTIVE, USNS IMPECCABLE, or USNS VICTORIOUS while these ships are underway and this potential stressor is not discussed further in this opinion.

6.1.4 Passive System Component - SURTASS

The passive, or listening, part of the SURTASS LFA sonar system is the Surveillance Towed Array Sensor System or "SURTASS." This system detects returning echoes from submerged objects, such as threat submarines, through the use of hydrophones. Part of the SURTASS transforms mechanical energy (received acoustic sound wave) to an electrical signal that can be analyzed by the processing system of the sonar. The SURTASS hydrophones are mounted on a receive array (horizontal line array [HLA]) that is towed astern of the vessel. The SURTASS LFA sonar vessel tows the hydrophone array at a minimum speed of 3 knots (3.5 mph) through the water to maintain the proper towed array geometry for maximum sonar system performance. The return signals, which are usually below background or ambient noise level, are then processed and evaluated to identify and classify potential underwater threats.

Because this sonar system does not transmit energy into the marine environment, we conclude that it is not likely to adversely affect listed species or critical habitat in the action area for this consultation. Therefore, this potential stressor is not discussed further in this opinion.

6.1.5 Active System Component - LFA sonar

The Low Frequency Active (LFA sonar) sonar system consists of up to 18 low-frequency acoustic-transmitting projectors that are suspended from a cable beneath a ship. The source level of an individual projector in the LFA sonar array is about 215 dB re: 1 μPa_{rms} at 1 meter (m), and the sound field of the array can never have a sound pressure level higher than that of an

individual projector. The typical LFA sonar signal is not a constant tone, but is a transmission of various signal types that vary in frequency and duration (including continuous wave and frequency-modulated signals). The Navy refers to a complete sequence of sound transmissions as a "ping" which can range from between 6 and 100 seconds, with no more than 10 seconds at any single frequency. The time between pings will typically range from 6 to 15 min. The Navy can control the average duty cycle (the ratio of sound "on" time to total time) for the system but the duty cycle cannot exceed 20 percent; based on operations since 2003, the duty cycle has averaged about 7.5 percent.

The LFA sonar mitigation zone covers a volume ensonified to a received level at or above180 dB re: 1 μPa_{rms} by the SURTASS LFA sonar transmit array. Based on spherical spreading, this zone will vary between the nominal ranges of 0.75 to 1.0 km (0.40 to 0.54 nmi) from the source array ranging over a depth of approximately 87 to 157 m (285 to 515 ft). The center of the array is at an approximate depth of 122 m (400 ft).

We assume that the propagation of signals transmitted from LFA sonar systems would be affected by surface ducts, sound channels, convergence zones, and bottom interactions. For more complete discussion of sound propagation in marine environments, readers should refer to Richardson et al. (1995), Appendix B of the Navy's EIS on SURTASS LFA sonar (Navy, 2001c), and the Effects of the Action section of NMFS' 30 May 30 2002 (NMFS, 2002) and 12 August 2007 (NMFS, 2007a) final rules, and biological opinions on SURTASS LFA sonar.

The potential effects to marine mammals and sea turtles from LFA sonar include non-auditory injury, auditory injury, behavioral changes, and masking.

6.1.5.1 Non-Auditory Injury

This includes the potential for resonance of the lungs/organs, tissue damage, and mortality from direct acoustic impacts on tissue, indirect acoustic impact on tissue surrounding a structure, and acoustically mediated bubble growth within tissues from supersaturated dissolved nitrogen gas.

Nowacek et al. (2007) and Southall et al. (2007b) reviewed potential areas for non-auditory injury to marine mammals from active sonar transmissions. These include direct acoustic impact on tissue, indirect acoustic impact on tissue surrounding a structure, and acoustically mediated bubble growth within tissues from supersaturated dissolved nitrogen gas.

Physical effects, such as direct acoustic trauma or acoustically enhanced bubble growth, require relatively intense received energy that would only occur at short distances from high-powered sonar sources (Nowacek et al., 2007; Zimmer & Tyack, 2007). The best available scientific information shows that, while resonance can occur in marine animals, this resonance does not necessarily cause injury, and any such injury is not expected to occur below a received level of 180 dB sound pressure level (SPL). Damage to the lungs and large sinus cavities of cetaceans from air space resonance is not regarded as a likely significant non-auditory injury because

resonance frequencies of marine mammal lungs are below that of the LFA sonar signal (Finneran, 2003). Further, biological tissues are heavily damped and tissue displacement at resonance is predicted to be exceedingly small. In addition, lung tissue damage is generally uncommon in acoustic-related strandings (Brandon L. Southall et al., 2007b).

There are no data on the potential for anthropogenic sound to cause injury in sea turtles. Although not directly related to SURTASS LFA sonar effects, a review of effects of explosives on turtles was done by Viada et al. (2008b). For explosive structure removals in the Gulf of Mexico, NMFS specified that the area within 915 m (3,000 ft) of the platform must be clear of sea turtles. Therefore, using a value of 180-dB sound pressure level injury threshold for sea turtles (within approximately 1,000 m (3,281 ft) of the LFA sonar array) is conservative.

Despite the increase in research and literature, there remains scientific disagreement and/or lack of scientific data regarding the evidence for gas bubble formation as a causal mechanism between certain types of acoustic exposures and stranding events in marine mammals. These issues include: (1) received acoustic exposure conditions; (2) pathological interpretation; (3) acoustic exposure conditions required to directly induce physiological trauma; (4) behavioral reactions caused by sound exposure such as atypical dive patterns; and (5) the extent of postmortem artifacts (Brandon L. Southall et al., 2007b).

6.1.5.2 Auditory Injury – Temporary and Permanent Hearing Loss

The most familiar effect of exposure to high intensity sound is hearing loss, meaning an increase in the hearing threshold. The meaning of the term “hearing loss” does not equate to “deafness.” This phenomenon is called a noise-induced threshold shift, or simply a threshold shift (Miller 1994). If high-intensity sound over stimulates tissues in the ear, causing a threshold shift, the impacted area of the ear (associated with and limited by the sound’s frequency band) no longer provides the same auditory impulses to the brain as before the exposure (Darlene R. Ketten, 2012). The distinction between PTS and TTS is based on whether there is a complete recovery of a threshold shift following a sound exposure. If the threshold shift eventually returns to zero (the threshold returns to the pre-exposure value), the threshold shift is a TTS.

For TTS, full recovery of the hearing loss (to the pre-exposure threshold) has been determined from studies of marine mammals, and this recovery occurs within minutes to hours for the small amounts of TTS that have been experimentally induced (Finneran, Carder, Schlundt, & Ridgway, 2005; Finneran & Schlundt, 2010; P. E. Nachtigall, A. Y. Supin, J. L. Pawloski, & W. W. L. Au, 2004). The recovery time is related to the exposure duration, sound exposure level, and the magnitude of the threshold shift, with larger threshold shifts and longer exposure durations requiring longer recovery times (Finneran et al., 2005; Finneran & Schlundt, 2010; T. Aran Mooney, Nachtigall, Breese, Vlachos, & Au, 2009; T. A. Mooney, Nachtigall, & Vlachos, 2009). In some cases, threshold shifts as large as 50 dB (loss in sensitivity) have been temporary, although recovery sometimes required as much as 30 days (Darlene R. Ketten, 2012). If the

threshold shift does not return to zero but leaves some finite amount of threshold shift, then that remaining threshold shift is a PTS. Figure 8 shows one hypothetical threshold shift that completely recovers, a TTS, and one that does not completely recover, leaving some PTS. The actual amount of threshold shift depends on the amplitude, duration, frequency, temporal pattern of the sound exposure, and on the susceptibility of the individual animal.

Both auditory trauma and auditory fatigue may result in hearing loss. Many are familiar with hearing protection devices (i.e., ear plugs) required in many occupational settings where pervasive noise could otherwise cause auditory fatigue and possibly result in hearing loss. The mechanisms responsible for auditory fatigue differ from auditory trauma and would primarily consist of metabolic fatigue and exhaustion of the hair cells and cochlear tissues. Note that the term "auditory fatigue" is often used to mean "temporary threshold shift"; however, the Navy uses a more general meaning is used to differentiate fatigue mechanisms (e.g., metabolic exhaustion and distortion of tissues) from trauma mechanisms (e.g., physical destruction of cochlear tissues occurring at the time of exposure). The actual amount of threshold shift depends on the amplitude, duration, frequency, and temporal pattern of the sound exposure.

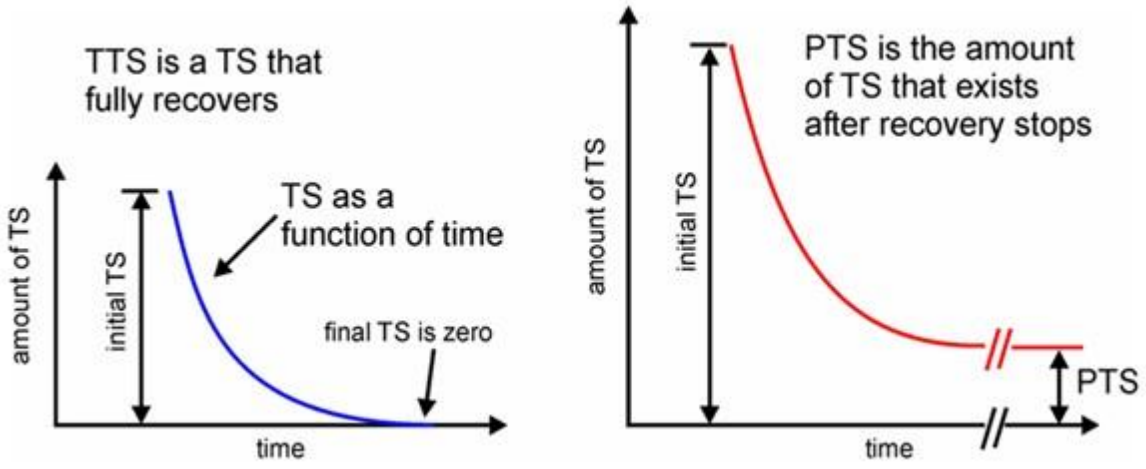

Figure 8. Two Hypothetical Threshold Shifts, Temporary and Permanent

Hearing loss, or auditory fatigue, in marine mammals has been studied by a number of investigators. (Finneran et al., 2005; Finneran & Schlundt, 2010; Finneran, Schlundt, Branstetter, & Dear, 2007a; Finneran, Schlundt, Dear, Carder, & Ridgway, 2000a, 2002; Lucke, Siebert, Lepper, & Blanchet, 2009; T. Aran Mooney et al., 2009; T. A. Mooney et al., 2009; Nachtigall, Pawloski, & Au, 2003b; C. E. Schlundt, J. J. Finneran, D. A. Carder, & S. H. Ridgway, 2000). The studies of marine mammal auditory fatigue were all designed to determine relationships between TTS and exposure parameters such as level, duration, and frequency.

In these studies, hearing thresholds were measured in trained marine mammals before and after exposure to intense sounds. The difference between the pre-exposure and post-exposure thresholds indicated the amount of TTS. Species studied include the bottlenose dolphin (total of 9 individuals), beluga (2), harbor porpoise (1), finless porpoise (2), California sea lion (3), harbor seal (1), and Northern elephant seal (1). Some of the more important data obtained from these studies are onset-TTS levels— exposure levels sufficient to cause a just-measurable amount of TTS, often defined as 6 dB of TTS (C. E. Schlundt et al., 2000). These criteria for onset-TTS are very conservative, and it is not clear that this level of threshold shift would have a functional effect on the hearing of a marine mammal in the ocean.

The primary findings of the marine mammal TTS studies are:

- The growth and recovery of TTS shift are analogous to those in terrestrial mammals. This means that, as in terrestrial mammals, threshold shifts primarily depend on the amplitude, duration, frequency content, and temporal pattern of the sound exposure.

- The amount of TTS increases with exposure sound pressure level and the exposure duration.

- For continuous sounds, exposures of equal energy lead to approximately equal effects (Ward 1997). For intermittent sounds, less hearing loss occurs than from a continuous exposure with the same energy (some recovery will occur during the quiet `period between exposures) (Kryter et al. 1965; Ward 1997).

- Sound exposure level is correlated with the amount of TTS and is a good predictor for onset-TTS from single, continuous exposures with similar durations. This agrees with human TTS data presented by Ward et al. (1958; 1959a, b). However, for longer duration sounds—beyond 16–32 seconds, the relationship between TTS and sound exposure level breaks down and duration becomes a more important contributor to TTS (Finneran & Schlundt, 2010).

- The maximum TTS after tonal exposures occurs one-half to one octave above the exposure frequency (Finneran et al., 2007a; C. E. Schlundt et al., 2000). TTS from tonal exposures can thus extend over a large (greater than one octave) frequency range.

- For bottlenose dolphins, sounds with frequencies above 10 kHz are more hazardous than those at lower frequencies (i.e., lower sound exposure levels required to affect hearing) (Finneran & Schlundt, 2010).

- The amount of observed TTS tends to decrease with increasing time following the exposure; however, the relationship is not monotonic. The amount of time required for complete recovery of hearing depends on the magnitude of the initial shift; for relatively

small shifts recovery may be complete in a few minutes, while large shifts (e.g., 40 dB) require several days for recovery.

- TTS can accumulate across multiple intermittent exposures, but the resulting TTS will be less than the TTS from a single, continuous exposure with the same sound exposure level. This means that predictions based on total, cumulative sound exposure level will overestimate the amount of TTS from intermittent exposures.

Although there have been no marine mammal studies designed to measure PTS, the potential for PTS in marine mammals can be estimated based on known similarities between the inner ears of marine and terrestrial mammals. Experiments with marine mammals have revealed their similarities with terrestrial mammals with respect to features such as TTS, age-related hearing loss (called Presbycusis), ototoxic drug-induced hearing loss, masking, and frequency selectivity. Therefore, in the absence of marine mammal PTS data, onset-PTS shift exposure levels may be estimated by assuming some upper limit of TTS that equates the onset of PTS, then using TTS growth relationships from marine and terrestrial mammals to determine the exposure levels capable of producing this amount of TTS.

Hearing loss resulting from auditory fatigue could effectively reduce the distance over which animals can communicate, detect biologically relevant sounds such as predators, and echolocate (for odontocetes). The costs to marine mammals with TTS, or even some degree of PTS have not been studied; however, it is likely that a relationship between the duration, magnitude, and frequency range of hearing loss could have consequences to biologically important activities (e.g., intraspecific communication, foraging, and predator detection) that affect survivability and reproduction.

For humans, acoustic damage-risk criteria rely on numeric thresholds based on “weighted” noise levels. Weighted noise levels are calculated by applying a frequency-dependent filter, or “weighting function” to the measured sound pressure before calculation of the overall sound pressure level (SPL). The weighting functions are designed to emphasize frequencies where sensitivity to sound is high and to de-emphasize frequencies where sensitivity is low. This technique allows for a single, weighted damage-risk criterion, regardless of the sound frequency. Weighting functions for humans are derived from equal loudness contours – graphs representing the SPLs that led to a sensation of equal loudness magnitude in the listener as a function of sound frequency (Suzuki and Takshima 2004). Equal loudness contours are derived from loudness experiments where the listener is asked to judge the relative loudness of two tones with different frequencies. Prior to (Finneran & Schlundt, 2011) there were no direct measurements of subjective loudness in non-human animals from which to develop equal loudness contours. Finneran and Schlundt (2011) trained a bottlenose dolphin to perform a loudness comparison test, where the listener indicated which of two sequential tones was louder. This study demonstrated that a non-human animal could be conditioned for subjective loudness testing and

therefore, it was possible to directly measure loudness levels in some species. Additional data is required to more accurately predict the relationship below 2.5 kHz. The weighting function derived here is substantially different than the "M-weighting function" proposed for mid-frequency cetaceans in Southall et al. (2007), which is nearly flat over the range of ~1-30 kHz and thus does not mirror the change in equal loudness contours observed over that frequency range. Nor does the M-weighting function capture the difference in TTS onset and growth reported for a single bottlenose dolphin tested at 3 and 20 kHz in Finneran and Schlundt (2010).

6.1.5.3 Behavioral Changes

Specific to U.S. Navy systems using low frequency sound, studies were undertaken in 1997–98 pursuant to the Navy's Low Frequency Sound Scientific Research Program. These studies found only short-term responses to low frequency sound by mysticetes (fin, blue, and humpback) including changes in vocal activity and avoidance of the source vessel (Christopher W. Clark & Fristrup, 2001a; D. A. Croll, C. W. Clark, J. Calambokidis, W. T. Ellison, & B. R. Tershy, 2001; Fristrup, Hatch, & Clark, 2003; P. J. O. Miller, Biassoni, Samuels, & Tyack, 2000b; Nowacek et al., 2007). Recent work by (Risch, Corkeron, Ellison, & Parijs, 2012) found that humpback whale vocalizations ("song") were reduced concurrent with pulses from the low frequency Ocean Acoustic Waveguide Remote Sensing source located approximately 200 km away. Baleen whales exposed to moderate low- frequency signals demonstrated no variation in foraging activity (Croll et al. 2001). However, five out of six North Atlantic right whales exposed to an acoustic alarm interrupted their foraging dives, although the alarm signal was long in duration, lasting several minutes, and purposely designed to elicit a reaction from the animals as a prospective means to protect them from ship strikes (Douglas P. Nowacek et al., 2004). Although the animal's received sound pressure level was similar in the latter two studies (133–150 dB re1 µPa), the frequency, duration, and temporal pattern of signal presentation were different. Additionally, the right whales did not respond to playbacks of either right whale social sounds or vessel noise, highlighting the importance of the sound characteristics, species differences, and individual sensitivity in producing a behavioral reaction.

Low-frequency signals of the Acoustic Thermometry of Ocean Climate sound source were not found to affect dive times of humpback whales in Hawaiian waters (A. S. Frankel & Clark, 2000) or to overtly affect elephant seal dives off California (Costa et al., 2003). However, they did produce subtle effects that varied in direction and degree among the individual seals, again illustrating the equivocal nature of behavioral effects and consequent difficulty in defining and predicting them.

Blue whales exposed to mid-frequency sonar in the Southern California Bight were less likely to produce low frequency calls usually associated with feeding behavior (Melcon et al., 2012). It is not known whether the lower rates of calling actually indicated a reduction in feeding behavior or social contact since the study used data from remotely deployed, passive acoustic monitoring buoys. In contrast, blue whales increased their likelihood of calling when ship noise was present,

and decreased their likelihood of calling in the presence of explosive noise, although this result was not statistically significant (Melcon et al., 2012). Additionally, the likelihood of an animal calling decreased with the increased received level of mid-frequency sonar, beginning at a sound pressure level of approximately 110–120 dB re 1 µPa (Melcon et al., 2012). Preliminary results from the 2010–2011 field season of an ongoing behavioral response study in Southern California waters indicated that in some cases and at low received levels, tagged blue whales responded to mid-frequency sonar but that those responses were mild and there was a quick return to their baseline activity (B. Southall et al., 2011) Blue whales responded to a mid-frequency sound source, with a source level between 160 and 210 dB re 1 µPa at 1 m and a received sound level up to 160 dB re 1 µPa, by exhibiting generalized avoidance responses and changes to dive behavior during CEEs (Goldbogen et al., 2013). However, reactions were not consistent across individuals based on received sound levels alone, and likely were the result of a complex interaction between sound exposure factors such as proximity to sound source and sound type (mid-frequency sonar simulation vs. pseudo-random noise), environmental conditions, and behavioral state. Surface feeding whales did not show a change in behavior during CEEs, but deep feeding and non-feeding whales showed temporary reactions that quickly abated after sound exposure. Whales were sometimes less than a mile from the sound source during CEEs.

6.1.5.4 Masking

Auditory masking occurs when a sound, or noise in general, limits the perception of another sound. As with hearing loss, auditory masking can effectively limit the distance over which a marine mammal can communicate, detect biologically relevant sounds, and echolocate (odontocetes). Unlike auditory fatigue, which always results in a localized stress response, behavioral changes resulting from auditory masking may not be coupled with a stress response. Another important distinction between masking and hearing loss is that masking only occurs in the presence of the sound stimulus, whereas hearing loss can persist after the stimulus is gone.

Critical ratios have been determined for pinnipeds (Brandon L. Southall, Schusterman, & Kastak, 2000; B. L. Southall, Schusterman, & Kastak, 2003) and bottlenose dolphins (C. Scott Johnson, 1967) and detections of signals under varying masking conditions have been determined for active echolocation and passive listening tasks in odontocetes (Whitlow W. L. Au & Pawloski, 1989; Erbe, 2000; C. S. Johnson, 1971). These studies provide baseline information from which the probability of masking can be estimated.

Clark et al. (2009) developed a methodology for estimating masking effects on communication signals for low frequency cetaceans, including calculating the cumulative impact of multiple noise sources. For example, their technique calculates that in Stellwagen Bank National Marine Sanctuary, when two commercial vessels pass through a North Atlantic right whale's optimal communication space (estimated as a sphere of water with a diameter of 20 km), that space is decreased by 84 percent. This methodology relies on empirical data on source levels of calls (which is unknown for many species), and requires many assumptions about ancient ambient

noise conditions and simplifications of animal behavior, but it is an important step in determining the impact of anthropogenic noise on animal communication. Subsequent research for the same species and location estimated that an average of 63–67 percent of North Atlantic right whale's communication space has been reduced by an increase in ambient noise levels, and that noise associated with transiting vessels is a major contributor to the increase in ambient noise (Leila T. Hatch, Clark, Parijs, Frankel, & Ponirakis, 2012).

Vocal changes in response to anthropogenic noise can occur across the repertoire of sound production modes used by marine mammals, such as whistling, echolocation click production, calling, and singing. Changes to vocal behavior and call structure may result from a need to compensate for an increase in background noise. In cetaceans, vocalization changes have been reported from exposure to anthropogenic noise sources such as sonar, vessel noise, and seismic surveying.

In the presence of low frequency active sonar, humpback whales have been observed to increase the length of their 'songs' (Fristrup et al., 2003; P. J. O. Miller et al., 2000b), possibly due to the overlap in frequencies between the whale song and the low frequency active sonar. North Atlantic right whales have been observed to shift the frequency content of their calls upward while reducing the rate of calling in areas of increased anthropogenic noise (S. E. Parks, Clark, & Tyack, 2007) as well as increasing the amplitude (intensity) of their calls (Susan E. Parks, 2009). In contrast, both sperm and pilot whales potentially ceased sound production during the Heard Island feasibility test (Bowles, Smultea, Wursig, Demaster, & Palka, 1994), although it cannot be absolutely determined whether the inability to acoustically detect the animals was due to the cessation of sound production or the displacement of animals from the area.

Differential vocal responding in marine mammals has been documented in the presence of seismic survey noise. An overall decrease in vocalization during active surveying has been noted in large marine mammal groups (Potter et al., 2007), while blue whale feeding/social calls increased when seismic exploration was underway (Di Lorio & Clark, 2010), indicative of a potentially compensatory response to the increased noise level. Melcón et al. (2012) recently documented that blue whales decreased the proportion of time spent producing certain types of calls when simulated mid-frequency sonar was present. Castellote et al. (2012) found that vocalizing fin whales in the Mediterranean left the area where a seismic survey was being conducted and that their displacement persisted beyond the completion of the survey. At present it is not known if these changes in vocal behavior corresponded to changes in foraging or any other behaviors. Controlled exposure experiments (CEEs) in 2007 and 2008 in the Bahamas recorded responses of false killer whales, short-finned pilot whales, and melon-headed whales to simulated MFA sonar (Stacy L. Deruiter et al., 2013). The responses to exposures between species were variable. After hearing each MFA signal, false killer whales were found to "increase their whistle production rate and made more-MFA-like whistles" (Stacy L. Deruiter et al., 2013). In contrast, melon-headed whales had "minor transient silencing" after each MFA

signal, while pilot whales had no apparent response. Consistent with the findings of other previous research (see, for example, Southall et al. 2007), DeRuiter et al. (2013) found the responses were variable by species and with the context of the sound exposure.

Evidence suggests that at least some marine mammals have the ability to acoustically identify potential predators. For example, harbor seals that reside in the coastal waters off British Columbia are frequently targeted by certain groups of killer whales, but not others. The seals discriminate between the calls of threatening and non-threatening killer whales (Deecke, Slater, & Ford, 2002), a capability that should increase survivorship while reducing the energy required for attending to and responding to all killer whale calls. The occurrence of masking or hearing impairment provides a means by which marine mammals may be prevented from responding to the acoustic cues produced by their predators. Whether or not this is a possibility depends on the duration of the masking/hearing impairment and the likelihood of encountering a predator during the time that predator cues are impeded.

6.1.5.5 Stranding

Stranding is a potential risk for cetaceans. Stranding occurs when marine mammals passively (unintentionally) or purposefully come ashore either alive, but debilitated or disoriented, or dead. Although some species of marine mammals, such as pinnipeds, routinely come ashore during all or part of their life history, stranded marine mammals are differentiated by their helplessness ashore and inability to cope with or survive their stranded situation (i.e., they are outside their natural habitat and survival envelope) (J.R. Geraci & V.J. Lounsbury, 2005). In the U.S., the MMPA defines a stranding as: a) a marine mammal is dead and is (i) on a beach or shore of the U.S.; or (ii) in waters under the jurisdiction of the U.S. (including any navigable waters); or b) a marine mammal is alive and is (i) on a beach or shore of the U.S. and is unable to return to the water; (ii) on a beach or shore of the U.S. and, although able to return to the water, is in need of apparent medical attention; or (iii) in the waters under the jurisdiction of the U.S. (including any navigable waters) but is unable to return to its natural habitat under its own power or without assistance (16 U.S. Code §1421h).

Strandings of multiple marine mammals, or mass strandings, however, occur only rarely. A mass stranding of marine mammals is the stranding of two or more unrelated cetaceans (i.e., not a mother-calf pair) of the same species coming ashore at the same time and place (J.R. Geraci & V.J. Lounsbury, 2005). Mass strandings typically involve pelagic odontocete marine mammal species that occur infrequently in coastal waters and are usually typified by highly developed social bonds. Marine mammal strandings and mortality events are natural events that have been recorded historically from as early as 350 B.C. (Aristotle, ca. 350 B.C.), and such events continue to occur throughout the world's oceans.

Over the last four decades, marine mammal stranding networks have become established, and the reporting of marine mammal stranding and mortality events has become better documented and

publicized. This has led to increased public awareness and concern, especially regarding the potential for anthropogenic causes of stranding and mortality events. Underwater noise, particularly sounds generated by military sonar or geophysical and geologic seismic exploration, has increasingly been implicated as the plausible cause for marine mammal mortality and stranding events. However, despite extensive and lengthy investigations and continuing scientific research, definitive causes or links are rarely determined for the vast majority of marine mammal mass strandings and unusual mortality events (UMEs). It is generally more feasible to exclude causes of strandings or UMEs than to resolve the specific causative factors leading to these events. For instance, although no definitive cause could be identified for the mass stranding and death of 26 common dolphins in the Cornwall region of the United Kingdom during 2008, more than 10 factors were excluded or were considered highly unlikely to have caused the stranding (P.D. Jepson & Deaville, 2009).

As a result of scientific investigations and research over the last decade, especially on beaked whales, the scientific understanding has increased regarding the association between behavioral reactions to natural as well as anthropogenic sources and strandings or deaths of marine mammals. Scientists now understand that for some species, particularly deep-diving marine mammals, behavioral reactions may begin a cascade of physiologic effects, such as gas and fat embolisms, that may result in injury, death, and strandings of marine mammals (Cox et al., 2006; Fernández et al., 2005; Zimmer & Tyack, 2007).

Globally from 2006 through early 2010, at least 27 mass strandings of 11 marine mammal species occurred. These 27 mass stranding and mortality events were researched and analyzed to substantiate if any occurred within or near SURTASS LFA sonar mission areas, or if any were potentially associated with the transmission of underwater sound from military sonar. Any mass strandings involving beaked whales were also examined, as strandings of this species group have been shown to have a significant correlation with MFA naval sonar activities in some geographic regions (in the Mediterranean and Caribbean Seas but not off the coasts of Japan or Southern California) (Filadelfo et al., 2009). Additionally, marine mammal stranding records from Japan were analyzed for spatial or temporal correlations to LFA sonar operations.

The use of SURTASS LFA sonar was not associated with any of the reported 27 mass stranding events or UMEs that occurred globally between 2006 and early 2010. There is no evidence that LFA sonar transmissions resulted in any difference in the stranding rates of marine mammals in Japanese coastal waters adjacent to LFA sonar mission areas. As has been reported previously (Navy, 2001b, 2007c, 2012a) and has been further documented here, the employment of LFA sonar is not expected to result in any sonar-induced strandings of marine mammals. Given the large number of natural factors that can result in marine mammal mortality, the high occurrence of marine mammal strandings, and the many years of LFA sonar operations without any reported associated stranding events, the likelihood of LFA sonar transmissions causing marine mammals to strand is negligible.

6.1.6 High Frequency (HF/M3) Sonar

As a mitigation measure to reduce the exposure of marine mammals and sea turtles to SURTASS LFA sonar, the Navy will operate the High Frequency Marine Mammal Monitoring (HF/M3) sonar continuously during the course of active sonar transmissions. The HF/M3 sonar, which is a Navy-developed, enhanced HF commercial sonar, designed to detect, locate, and track marine mammals and, to some extent, sea turtles, that may pass close enough to the SURTASS LFA sonar's transmitter array to enter the LFA sonar mitigation and buffer zones. Analysis and testing of the HF/M3 sonar operating capabilities indicate that the system substantially increases the probability of detecting marine mammals within the LFA sonar mitigation and buffer zones. The HF/M3 sonar provides an excellent monitoring capability (particularly for medium to large marine mammals) beyond the LFA sonar mitigation and buffer zones, out to 2 to 2.5 km (1.08 to 1.35 nm).

The source level required for the HF/M3 sonar to effectively detect marine mammals (and possibly sea turtles) out to the 180-dB LFA sonar mitigation and buffer zones under the most adverse oceanographic conditions (low echo return and high ambient noise) is on the order of 220 dB re: 1 μPa_{rms} at 1 m. The Navy designed the HF/M3 sonar to be as benign as possible within the marine environment to minimize potential effects to marine mammals and sea turtles. These features include:

1. The HF/M3 sonar source frequency is >30 kHz, which pushes its frequency band well away from the best hearing bandwidth of mysticetes, pinnipeds, and sea turtles, but within the best hearing bandwidth of odontocetes;
2. A duty cycle that is variable, but below 10 percent;
3. A maximum HF/M3 sonar pulse with a duration of 40 milliseconds (msec). Ridgway et al.(1997) and Schlundt et al. (2000) reported that measured TTS in bottlenose dolphins for a 20 kHz, 1-second pulse occurred at response levels of 193-196 dB. For a 30 kHz, 40-msec pulse, the estimated range from the HF/M3 sonar of 193 dB response level would be 22 m (72 ft);
4. A transmission loss that is very high because of the high frequency of the sound source.

In addition, as supplementary safety measures, HF/M3 sonar source level would be ramped up over a five-minute period to alert a marine mammal that was close to the sonar and provide it time to move away from the sound source.

The application of these operational procedures reduces potential impacts of the HF/M3 sonar on marine mammals and sea turtles to negligible levels. Therefore the HF/M3 sonar, as a potential stressor, will not be considered further in this opinion.

6.2 Mitigation to Minimize the Likelihood of Exposure

The Navy proposes to use a three-part monitoring program to avoid potentially exposing marine mammals to LFA sonar transmissions at high decibel levels. The monitoring program includes visual, passive acoustic, and active acoustic monitoring of a 180 dB LFA mitigation zone and an additional 1 km buffer zone.

The effectiveness of visual monitoring is limited to daylight hours, and its effectiveness declines during poor weather conditions. In line transect surveys, the range of visual sighting effectiveness (distance from the ship's track, called effective strip width) varies with an animal's size, group size, reliability of conspicuous behaviors (blows), pattern of surfacing behavior, and positions of the observers (which includes the observer's height above the water surface). For most large baleen whales, effective strip width can be about 3 km (1.6 nmi) up through Beaufort 6 (Buckland & Borchers, 1993). For harbor porpoises the effective strip width is about 250 m (273 yd), because they are much smaller and less demonstrative on the surface than the baleen whales (Palka, 1996). The percentage of animals that will pass unseen is difficult to determine, but for minke whales, Schweder et al. (1992) estimated that visual survey crews did not detect about half of the animals in a strip width. Palka (1996) and Barlow (1988) estimated that visual survey teams did not detect about 25 percent of the harbor porpoises in a strip width. Visual detections of marine mammals during SURTASS LFA sonar mission from 2002 to 2012 totaled 12 sightings.

The effectiveness of passive acoustic detection is considered to be higher than visual monitoring. Thomas et al. (1986) and Clark and Fristrup (1997) concluded that the effective strip width and detection rates for passive acoustic monitoring is greater than that for visual, but the percentage of animals that will be undetected by the methods is unknown. Frequency coverage for this mitigation method using the SURTASS passive array is between 0 and 500 Hz, so vocalizing animals are more likely to be detected than animals that do not vocalize. This would increase the detection rate of gray, humpback, fin, blue, and minke whales, and some of the beaked whale and dolphin species. Passive acoustic detections of marine mammals during SURTASS LFA sonar missions from 2002 to 2012 totaled five animals.

The HF/M3 sonar is the final measure the Navy proposes to use to detect animals within 1 to 2 kilometers of the projectors. Testing of the HF/M3 sonar demonstrated a probability of single-ping detection above 95 percent within the LFA sonar mitigation zone for many marine mammals (Ellison & Stein, 1999). Under normal operating conditions, marine mammals will receive multiple pings, increasing the probability of detection to near 100 percent (Navy, 2001a, 2001b, 2001c). If any of these monitoring methods detects animals within the LFA sonar mitigation and buffer zones, the LFA sonar projectors would be shut down until the animal(s) move out of these zones. The HF/M3 monitoring detected 131 marine mammals during SURTASS LFA sonar missions from 2002-2012 (153 marine mammals detected from 2002 through May 2013), and these detections resulted in the delay or suspension of SURTASS LFA

sonar transmissions. Combined with the visual monitoring and passive acoustic monitoring protocols and the delay/suspension protocols for LFA sonar transmissions, this should minimize the risk of marine mammals being exposed to sound pressure levels in excess of 180 dB re: μPa_{rms} at 1 m. In fact, the Navy's tests have shown that the HF/M3 system is nearly 100 percent effective using multiple pings in detecting marine mammals of any size.

The HF/M3 system also increases the likelihood of detecting marine turtles. Because the HF/M3 sonar is positioned at the top of the LFA vertical array, sea turtles would first have to swim through the 1-km HF/M3 sonar detection zone (*i.e.*, the very same 1-km buffer zone for marine mammals) where detection is highly likely, before entering the LFA sonar mitigation zone.

HF/M3 sonar ramp-up. Benda-Beckmann et al. (2014) investigated the effectiveness of ramp-up procedures in reducing the area within which changes in hearing thresholds can occur. They modeled the level of sound killer whales (*Orcinus orca*) exposed to a generic sonar operation preceded by different ramp-up schemes. Results indicate that ramp-up procedures reduced the risk of killer whales receiving sounds of sufficient intensity to affect their hearing. The main factors that limited effectiveness of ramp up in a typical antisubmarine warfare scenario were high source level, rapid moving sonar source, and long silences between consecutive sonar transmissions.

6.3 Exposure

Here we estimate the potential exposure of listed species and critical habitat to the stressors we determined posed a risk to listed species. Because of the reasons provided in the two previous sections (Sections 6.1 and 6.2), this exposure analysis only addresses exposure to signals generated by SURTASS LFA sonar. Our analyses consider the contributions of the intensity of received levels, the duration of a potential exposure, and how frequently individuals of a listed species might be exposed at different received levels. We assumed that exposure to SURTASS LFA sonar, regardless of the sound pressure level, is a pre-requisite for listed species or critical habitat to be adversely affected by the sonar.[4]

This section describes the conditions under which listed species could be exposed to LFA sonar based on evaluations of the available information and also describes potential relationships between differing levels of exposure to LFA sonar and potential effects on listed species. Therefore, we assume that the potential biological risk associated with exposure to LFA sonar is a function of an animal's exposure to a sound that could adversely affect the animal's hearing, behavior, psychology, or physiology.

[4] We recognize that LFA sonar could have indirect, adverse effects on species or critical habitat by disrupting marine food chains, a species predator, or species competitors; however, we could not identify a situation where this concern would be applicable to species under NMFS' jurisdiction.

As discussed previously, the LFA sonar system consists of up to 18 low-frequency acoustic-transmitting projectors that are suspended from a cable beneath a ship. The source level of an individual projector in the LFA sonar array is approximately 215 dB, and the sound field of the array can never have a sound pressure level higher than that of an individual projector. The typical LFA sonar signal is not a constant tone, but is a transmission of various signal types that vary in frequency and duration (including continuous wave and frequency-modulated signals). The Navy refers to a complete sequence of sound transmissions as a "ping" which can range from between 6 and 100 seconds, with no more than 10 seconds at any single frequency. The time between pings will typically range from 6 to 15 minutes. The Navy has stated that the duty cycle (the ratio of sound "on" time to total time) would not be greater than 20 percent. Based on operations since 2003, the average duty cycle for the LFA sonar system has been 7.5 to 10 percent (Navy, 2012a).

6.3.1 Intensity

Sound transmissions are usually measured in terms of sound pressure levels, which are denoted as decibels and which have a reference pressure value of 1 μPa. The logarithmic nature of the dB scale means that each 10 dB increase is a ten-fold increase in power (e.g., 20 dB is a 100-fold increase, 30 dB is a 1,000-fold increase). Humans perceive a 10 dB increase in noise as a doubling of sound level, or a 10 dB decrease in noise as a halving of sound level.

Inside the LFA sonar mitigation zone during a ping, a marine mammal could be exposed to sound levels at or above 180 dB re: 1 μPa_{rms} at 1 m and could experience PTS or other injury. However, implementing the LFA sonar mitigation and buffer zone which are approximately 2 km (1.2 mi; 1.1 nm) around the vessel will ensure that no marine mammals are exposed to a sound pressure level greater than approximately 175 dB re: 1 μPa. This ensures that marine mammals or sea turtles are exposed to these energy levels. Given the size of the LFA sonar mitigation zone (extending to approximately 0.75 to 1.00 km (0.40 to 0.56 nmi) from the transmitter), the detection probabilities associated with the HF/M3 sonar detecting marine mammals approaches 100 percent. Due to the depth of the transmitters, a marine mammal would have a high probability of being detected within the LFA sonar mitigation and buffer zones and, as a result, a low probability of being exposed to sound levels greater than 180 dB re: 1 μPa_{rms} at 1 m.

For an animal to be exposed at received levels greater than 180 dB re: 1 μPa_{rms} at 1 m, the animal would have to occur in the same approximately 1-kilometer wide 180 dB re: 1 μPa_{rms} at 1 m sound field of the LFA sonar transmit array (centered at approximate depth of 122 m), would have to enter the LFA sonar mitigation zone without being detected, and would have to remain in the LFA sonar mitigation zone when the LFA sonar transmitter was operating. Based on the available information, we believe the probability of all of these events occurring, although possible, is extremely improbable.

Further, SURTASS LFA sonar will be operated to ensure that sonar sound fields do not exceed 180 dB re: 1 μPa_{rms} within 22 km (12 nmi) of any coastline, including offshore islands, or designated offshore areas that are biologically important for marine mammals outside the 22 km zone during seasons specified for a particular area. When in the vicinity of known recreational and commercial dive sites, SURTASS LFA sonar would be operated to ensure that the sound field at these sites would not exceed 145 dB re: 1 μPa_{rms} at 1 m, adding an additional level of protection for marine mammals and sea turtles located within designated dive sites.

Thus far, the combination of geographic operating constraints, operating protocols, monitoring/mitigation measures, and shut-down procedures appear to have prevented most threatened and endangered species of marine mammal and sea turtles from being exposed to SURTASS LFA sonar at received levels exceeding 180 dB. Further, these factors have prevented listed species from being exposed in areas that are critical to their ecology, critical to large portions of their populations, or both. The Navy proposes to continue using these measures in the prescribe manner. Therefore, based on the evidence available, we conclude that only a few marine animals would have the potential to be exposed to received levels of LFA sonar at or above 180 dB re: 1 μPa_{rms} at 1 m.

6.3.2 Exposure Duration

The Navy proposes to conduct active operations of SURTASS LFA sonar systems on up to four vessels for 240 days. The maximum time of active sonar transmissions per vessel would be 432 hours each year (based on a 7.5 percent duty cycle). The duration of a typical SURTASS LFA sonar ping would range from 6 to 100 seconds, with no more than 10 seconds at a single frequency; intervals between pings would range from 6 to 15 minutes. Pings would consist of various signal types that vary in frequency (between 100 and 500 Hz) and duration (including continuous wave and frequency-modulated signals). When the system is turned off, no additional energy would enter the ocean's environment.

The duration of an animal's exposure to SURTASS LFA sonar signals will depend on their proximity to the transmitter and their location in the water column. Nevertheless, because of the length of individual pings, individual animals could be exposed to SURTASS LFA sonar transmissions for periods ranging from 6 to 100 seconds at a time.

6.3.3 Exposure Frequency

Individual animals or groups of animals have a low statistical probability of being exposed to SURTASS LFA sonar signals on multiple, separate occasions. However, the number of times an animal could be exposed to sound pressure levels associated with SURTASS LFA sonar transmissions will depend on the deployment schedule of the SURTASS LFA sonar vessels.

There are no published data on marine mammals regarding responses to repeated exposure to low frequency sound. Evidence from studies of repeated exposure to other impulsive sounds suggest that the risk threshold is lowered by 5 dB per ten-fold increase in the number of pulses

per exposure if the number of pulses per exposure is less than 100 (W. John Richardson, Charles R. Greene Jr., et al., 1995, citing Kryter 1985).

The Navy represented the probability of risk, using stressor-response functions generated by mathematical simulation. These functions, which are represented as cumulative probability distributions or cumulative distribution functions, have values near zero at very low exposures, and values near one for very high exposures. From this distribution function, received levels of 150 dB re: 1 μPa_{rms} at 1 m generally had a 2.5 percent likelihood of producing a behavioral response that is important to the biology of marine mammals. Received levels of 165 dB **re:** 1 μPa_{rms} at 1 m generally had a 50 percent probability of producing such a behavioral response in marine mammals and received levels of 180 dB re: 1 μPa_{rms} at 1 m generally have a 95 percent probability of producing such a behavioral response, although this is a generalization for all marine mammals and did not necessarily apply to particular marine mammal species.

6.3.4 Mitigation Measures to Minimize the Likelihood of Exposure

As also discussed in the description of the action, the Navy proposes several measures to reduce or eliminate the likelihood of exposing marine mammals to LFA sonar transmissions at high decibel levels including a monitoring program (visual, passive sonar and the HF/M3 active sonar system), delay/suspension protocols for LFA sonar transmissions, and geographic constraints on the use of SURTASS LFA sonar (coastal mitigation, LFA sonar mitigation and buffer zones, OBIAs, and avoidance of known recreational and commercial dive sites).

6.3.4.1 Monitoring Program

The monitoring program includes visual, passive acoustic, and active acoustic monitoring of a 180 dB mitigation zone plus an additional 1 km buffer zone. The effectiveness of visual monitoring is limited to daylight hours, and its effectiveness declines during high sea states. The percentage of animals that will pass unseen is difficult to determine, but for minke whales, Schweder et al. (1992) estimated that visual survey crews did not detect about half of the animals in a strip width. Palka (1996) and Barlow (1988) estimated that visual survey teams did not detect about 25 percent of the harbor porpoises in a strip width.

Based on actual trials, the passive acoustic detection has been more effective than visual monitoring. Thomas et al. (1986) and Clark and Fristrup (1997) concluded that the effective strip width and detection rates for passive acoustic monitoring are greater than for visual monitoring, but the percentage of animals that will be undetected by the methods is unknown. This would increase the detection rate of gray, humpback, fin, blue, and unlisted minke whales, and some of the unlisted beaked whale and dolphin species.

6.3.4.2 High Frequency Marine Mammal Monitoring Sonar

The HF/M3 active sonar system appears to be the most effective measure the Navy uses to detect animals within 2 kilometers of the projectors. Recent testing of the HF/M3 active sonar system demonstrated a probability of single-ping detection above 95 percent within the LFA sonar

mitigation zone for many marine mammals (Ellison & Stein, 1999). Under normal operating conditions, marine mammals will receive multiple pings, increasing the probability of detection to near 100 percent (Navy, 2012a). If any of these monitoring methods detects animals within the LFA sonar mitigation or buffer zones, the projectors would be shut down until the animal(s) move out of the mitigation and buffer zones. Combined with the visual monitoring and passive acoustic monitoring protocols, this should minimize the risk of marine mammals being exposed to sound pressure levels at or above 180 dB re: 1 μPa_{rms} at 1 m.

The Navy operated the HF/M3 active sonar system continuously during the course of the missions conducted from 2002 through May 2013. During this period, there were 153 HF/M3 alerts that were identified as possible marine mammal or sea turtle detections that resulted in LFA sonar transmissions delays/suspensions. The Navy followed the appropriate protocols and LFA sonar transmissions were suspended or delayed due to mitigation protocols. As a result, marine mammals that had been detected were not exposed to sonar transmissions at received levels above 180 dB.

HF/M3 sonar ramp-up. Benda-Beckmann et al. (2014) investigated the effectiveness of ramp-up procedures in reducing the area within which changes in hearing thresholds can occur. They modeled the level of sound killer whales (*Orcinus orca*) exposed to a generic sonar operation preceded by different ramp-up schemes. Results indicate that ramp-up procedures reduced the risk of killer whales receiving sounds of sufficient intensity to affect their hearing. The main factors that limited effectiveness of ramp up in a typical antisubmarine warfare scenario were high source level, rapid moving sonar source, and long silences between consecutive sonar transmissions.

6.3.4.3 LFA Sonar Mitigation and Buffer Zones

Inside the LFA sonar mitigation and buffer zones during a ping, listed species could be exposed to sound levels at or above 180 dB and could experience threshold shifts (loss in hearing sensitivity) or more serious auditory injury. However, the LFA sonar mitigation and buffer zones were established and designed to prevent marine mammal or sea turtles from being exposed to energy levels high enough to produce these outcomes. Several aspects of the LFA sonar mitigation and buffer zones reduce a listed species' chances of being exposed to LFA sonar at sound pressure levels at or above 180 dB; (a) the mitigation zone extends approximately 0.75 to 1.00 km (0.40 to 0.56 nmi) from the transmitters and is supplemented by a 1-km buffer zone, (b) the detection probabilities associated with the HF/M3 sonar (based on a single ping) exceed 95 percent for small dolphins at about 750 m (0.4 nmi), whale calves at 1,000 m (0.56 nmi) and large whales at more than 1,500 m (0.81 nmi); (c) during actual operations marine mammal or larger sea turtles will receive approximately 5 pings from the HF/M3 sonar with the probablility of detection t approaching 100 percent; (d) the depth of the transmitters would reduce the risk of exposing animals located near the ocean's surface or in the mixed layer; and (e) marine mammals and larger sea turtles have a high probability of being detected within the LFA sonar

mitigation and buffer zones and, as a result, a low probability of being exposed to sound pressure levels greater than 180 dBre 1 μPa_{rms} at 1 m.

For an animal to be exposed to LFA sonar transmissions at sound pressure levels greater than 180 dB, the animal would have to enter the LFA sonar mitigation and buffer zones without being detected by marine mammal observers or the HF/M3 sonar system and would have to remain in the LFA sonar mitigation and buffer zones when LFA sonar transmitters were operating. The monitoring results of the deployments of the SURTASS LFA sonar system from 2002 through May of 2013 support this conclusion. The active acoustic monitoring (HF/M3 sonar) resulted in a total of 153 possible marine mammal/sea turtle contacts that resulted in suspensions of operations in accordance with the terms and conditions of the existing permit. No contacts were reported within the 180-dB LFA sonar mitigation zone during transmissions, which suggests that no marine mammals were subjected to LFA sonar exposures that might have resulted in injuries or that a very small percentage of animals when undetected, which is consistent with our analysis. Based on this monitoring information, the probability of all of these events occurring, although possible, is extremely low.

6.3.4.4 Geographic Restrictions

The SURTASS LFA sonar system would be operated in a manner that would not cause sonar sound fields to exceed 180 dB re: 1 μPa_{rms} within the 22 km (12 nmi) coastal standoff range. For any annual Letter of Authorization, NMFS' regulations establish a coastal standoff range of 22 km (12 nmi) of any coastline, including offshore islands, and designated OBIAs for marine mammals outside the 22 km coastal standoff range during seasons specified for a particular area (see Table 1). In addition the LFA sonar fields will not exceed 180 dB re: 1 μPa_{rms} at 1 m within a buffer zone 1-km seaward of any OBIA boundary. When in the vicinity of known recreational and commercial dive sites, SURTASS LFA sonar will be operated to ensure that the sound field at these sites would not exceed 145 dB.

These operational constraints would further reduce the likelihood of exposing threatened and endangered species of whales to SURTASS LFA sonar at sound pressure levels exceeding 180 dB re: 1 μPa_{rms} at 1 m in areas that are critical to their ecology, critical to large portions of their populations, or both.

6.3.5 Exposure Estimates

For the recent FSEIS/SOEIS (Navy, 2012a) and third MMPA rulemaking, the Navy modeled 19 worldwide ocean areas where SURTASS LFA sonar operations could occur. Since the Navy could not model every part of the ocean waters in which SURTASS LFA sonar operations could potentially occur, 19 sites relevant to U.S. national security interests were selected as nominal examples. The Navy proposes to conduct 20 missions using SURTASS LFA sonar in the following areas (Table 19):

- Up to 16 missions in the northwestern Pacific Ocean which includes the following operational areas: east of Japan; the North Philippine Sea; the west Philippine Sea; offshore Guam; the Sea of Japan; the East China Sea; the South China Sea; and offshore northwest Pacific Ocean (offshore Japan) (25° to 40° N and 10° to 25° N).

- Up to 4 missions in the north-central Pacific Ocean, which includes the northern and southern Hawaii mission areas located within the Hawaii Range Complex.

As under the 2012 MMPA Final Rule, all missions would be conducted under the Letters of Authorization to ensure that no more than 12 percent of any marine mammal stock, including those of ESA-listed species, would be taken by Level B harassment annually over the course of the five-year regulation, regardless of the number of LFA sonar vessels operating.

Table 19. Number of proposed SURTASS LFA sonar missions by area proposed to occur 15 August 2014 through 14 August 2015.

Site #	Modeled Site	Number of Missions
1	East of Japan	1
2	North Philippine Sea	3
3	West Philippine Sea	3
4	Offshore Guam (Mariana Islands Range Complex)	3
	Sea of Japan	2
6	East China Sea	1
7	South China Sea	1
8	Offshore - NW Pacific 25° to 40°N (Offshore Japan)	1
9	Offshore NW Pacific 10° to 25°N	1
10	Hawaii North (Hawaii Range Complex)	2
11	Hawaii South (Hawaii Range Complex)	2

A single mission scenario was modeled in these areas to estimate potential impacts to marine mammals. Nominal operational assumptions for the model were a mission lasting 7 days with an LFA sonar duty cycle of 7.5 percent, which is equal to 12.6 hours of LFA sonar transmission per mission (Table 20).

The Navy defined the basement value for the risk model, also called the “B parameter”, below which the risk is so low that calculations are impractical as 119-dB. This level is taken as the estimate of received level below which the risk of significant change in a biologically important behavior approaches zero for the SURTASS LFA sonar risk assessment. This level is the value at which avoidance reactions have been noted in bowhead, beluga, and gray whales (which are mitigated by geographic restrictions on SURTASS LFA sonar operations). We recognize that for actual risk of changes in behavior to be zero, the signal-to-noise ratio at the animal must also be zero. However, the present convention of ending the risk calculation at 119 dB single ping equivalent (SPE) has a negligible impact on subsequent calculations, because the risk function does not attain appreciable values until received levels exceed 130 dB single ping equivalent. Hence, the values estimated in the analysis (Table 21) are the proportion of each species stock that would be expected to be exposed to SURTASS LFA sonar levels between 119 and 180 dB.

Table 20. Critical Parameters Used in the Navy's SURTASS LFA Sonar Model

LFA sonar Acoustic Source Parameters	Value
Vertical line array elements	**18**
Nominal source level for single element	215 dB re: 1 µPascal (Pa) @ 1 m (rms) (SPL)
Nominal effective source level for array	227.5 dB re: 1 µPa @ 1 m (rms) (SPL)
Nominal frequency	300 Hz
Signal bandwidth	~30 Hz
Nominal duty cycle	7.5 percent
Array depth	87 to 157 m (285 to 515 ft)
Array center depth	122 m (400 ft)
Acoustic Integration Model (AIM) Input	
Species/stock name/stock abundance/density	animals per km²
Coordinates of operating area	Latitude/Longitude
Season	Summer/fall/winter/spring
Mission length	7 days (168 hrs)
Active transmission time (based on 7.5 percent duty cycle)	12.6 hours per 7-day mission
Vessel speed	7.4 km/hr (4 knots)
Ship movement	triangular path, 8 hours per leg
Animat/Species Specific AIM Input	
Surface time (min/max)	Heading variable (angle/time)
Surface/dive angle	Swim speed (min/max)
Dive depth (min/max)	Swim speed distribution (normal, gamma)
Dive duration (min/max)	
LFA sonar Acoustic Propagation Modeling	
Standard Parabolic Equation Model for transmission loss versus depth and range from source	
Gridded Global Relief Data from the NOAA National Geophysical Data Center (NGDC) for bathymetry	
Generalized Digital Environmental Model (GDEM) (OAML, 2000) for sound velocity profiles for each location and season	
Bechmann-Spezzichino formula modified by Leibiger (1978) to calculate surface losses using a wind speed of 15 knots	
Consolidated bottom loss upgrade (CBLUG) database (OAML, 2000) for bottom loss	

Table 21. Percentage estimated of each ESA-listed marine mammal stock, based on a single 7-day mission in each potential operating area, that may be affected by the operation of SURTASS LFA sonar (with mitigation measures applied).

Species	Marine Mammal Stock	Est. Number in Stock	Percent Stock Affected <180 dB
East of Japan			
Blue whale	Central North	9,250	N/A
Fin whale	Western North	9,250	0.05
Sei whale	North Pacific	8,600	0.07
North Pacific right whale	Western North	922	N/A
Sperm whale	North Pacific	102,112	0.02
North Philippine Sea			
Blue whale	Central North	9,250	0.01
Fin whale	Western North	9,250	0.18
Humpback whale	Western North	1,107	6.89
North Pacific right whale	Western North	922	0.06
Sperm whale	North Pacific	102,112	0.09
West Philippine Sea			
Blue whale	Central North	9,250	0.01
Fin whale	Western North	9,250	0.09
Humpback whale	Western North	1,107	1.47
Sperm whale	North Pacific	102,112	0.10
Offshore Guam			
Blue whale	Central North	9,250	0.01
Fin whale	Western North	9,250	0.01
Sei whale	North Pacific	8,600	0.20
Humpback whale	Central North	10,103	0.18
Sperm whale	North Pacific	102,112	0.09
Sea of Japan			
Fin whale	Western North	9,250	0.77
North Pacific right whale	Western North	922	0.05
Western Pacific gray whale	Western North	121	0.10
Sperm whale	North Pacific	102,112	0.12
East China Sea			
Fin whale	East China Sea	500	1.48
North Pacific right whale	Western North	922	N/A
Western Pacific gray whale	Western North	121	N/A
Sperm whale	North Pacific	102,112	0.03
South China Sea			
Fin whale	Western North	9,250	0.04
North Pacific right whale	Western North	922	0.04
Western Pacific gray whale	Western North	121	0.31
Sperm whale	North Pacific	102,112	0.01
Offshore North Pacific (25° to 40°N) (Offshore Japan)			
Blue whale	Central North	9,250	N/A
Fin whale	Western North	9,250	0.05

Species	Marine Mammal Stock	Est. Number in Stock	Percent Stock Affected <180 dB
Sei whale	North Pacific	8,600	0.07
Sperm whale	North Pacific	102,112	0.05
Hawaiian monk seal	Hawaiian	1,212	0.04
Offshore North Pacific (10° to 25°N) (Offshore Japan)			
Blue whale	Central North	9,250	0.01
Fin whale	Western North	9,250	0.01
Sei whale	North Pacific	8,600	0.05
Sperm whale	North Pacific	102,112	0.04
Hawaii North			
Blue whale	Central North Pacific	9,250	0.14
Fin whale	Hawaiian	174	3.59
Sei whale	Hawaiian	77	0.11
Humpback whale	Central North Pacific	10,103	0.09
Sperm whale	Hawaiian	6,919	3.23
Main Hawaiian Island Insular False Killer Whale	Main Hawaiian Island Insular	151	0.22
Hawaiian monk seal	Hawaiian	1,212	0.80
Hawaii South			
Blue whale	Central North Pacific	9,250	0.08
Fin whale	Hawaiian	174	2.25
Sei	Hawaiian	77	0.11
Humpback whale	Central North Pacific	10,103	0.69
Sperm whale	Hawaiian	6,919	0.62
Main Hawaiian Island Insular False Killer Whale	Main Hawaiian Island Insular	151	1.06
Hawaiian monk seal	Hawaiian	1,212	0.28

An entry of N/A in the Percent Stock Affected <180 dB indicates that no exposures of individuals of those stocks are expected to occur.

6.3.5.1 Blue Whale

Blue whales will be exposed to SURTASS LFA sonar in five of the areas that that Navy will be operating under the proposed Letters of Authorization (Table 22). Blue whales may be exposed to SURTASS LFA sonar during a proposed 14 missions in six mission areas in which the Navy proposes to operate during August 2014 to August 2015. Based on the effects analysis, a total of 30 blue whales in the Central North Pacific (CNP) stock or 0.26% of the stock, which numbers an estimated 9,250 individuals, may be exposed to SURTASS LFA sonar during the annual reporting period. The highest number of blue whale potentially exposed to LFA sonar may occur is in the Hawaii-North mission area, where during two proposed missions 14 blue whales or 0.14% of the CNP stock could be exposed to RLs of 120 to 180 dB SPE (with mitigation applied).

Table 22. Estimated blue whale exposures to SURTASS LFA sonar between 120dB and 180dB

Site #	Operating Area	Number of Missions	Marine Mammal Stock	Number in Stock	Estimated Exposures without Mitigation	Estimated Exposures with Mitigation
2	North Philippine Sea	3	Central North Pacific	9,250	60	2
3	West Philippine Sea	3	Central North Pacific	9,250	49	2
4	Offshore Guam	3	Central North Pacific	9,250	48	2
9	Offshore North Pacific (10° to 25°N) (Offshore Japan)	1	Central North Pacific	9,250	20	1
10	Hawaii North (25°N, 158°W)	2	Central North Pacific	9,250	549	14
11	Hawaii South (19.5°N 158.5°W)	2	Central North Pacific	9,250	860	9
	Totals	**14**			**1,586**	**30**

6.3.5.2 Fin Whale

Based on the results of the Navy's modeling, the highest estimated number of fin whale exposures to SURTASS LFA sonar at the 120 to 180 dB SPE level would be 73 whales in the Sea of Japan mission area where the Navy proposes to conduct two missions. In total, three stocks of fin whales, the Western North Pacific (WNP), the East China Sea (ECS), and Hawaiian, would be exposed to LFA sonar in the western and central North Pacific mission areas during 20 possible missions; in total during these 20 missions, 137 fin whales in three stocks would potentially be exposed to SURTASS LFA sonar at the 120 to 180 dB SPE level. The highest percentage and number of animals of the three stocks affected is predicted for the WNP at 1.11% or 117 animals, respectively, during a total for 15 proposed missions in eight mission areas.

Table 23. Estimated fin whale exposures to SURTASS LFA sonar between 120dB and 180dB.

Site #	Operating Area	Number of Missions	Marine Mammal Stock	Number in Stock	Estimated Exposures without Mitigation	Estimated Exposures with Mitigation
1	East of Japan	1	Western North Pacific	9,250	541	5
2	North Philippine Sea	3	Western North Pacific	9,250	1,175	18
3	West Philippine Sea	3	Western North Pacific	9,250	499	8
4	Offshore Guam	3	Western North Pacific	9,250	48	2
5	Sea of Japan	2	Western North Pacific	9,250	4,804	73
6	East China Sea	1	East China Sea	500	196	8
7	South China Sea	1	Western North Pacific	9,250	650	4

8	Offshore North Pacific (25° to 40°N) (Offshore Japan)	1	Western North Pacific	9,250	645	6
9	Offshore North Pacific (10° to 25°N) (Offshore Japan)	1	Western North Pacific	9,250	20	1
10	Hawaii North (25°N, 158°W)	2	Hawaii	174	279	7
11	Hawaii South (19.5°N 158.5°W)	2	Hawaii	174	423	5
	Totals	**20**			**9,279**	**137**

6.3.5.3 Western Pacific Gray Whale

During three of four proposed SURTASS LFA sonar missions in two mission areas of the Western North Pacific Ocean, three Western North Pacific gray whales or 0.41% of one stock are estimated to be exposed to LFA sonar transmissions at 120 to 180 dB SPE. Gray whales in the WNP stock, which numbers an estimated 121 whales, may not occur in all seasons during which missions are proposed.

Table 24. Estimated Western Pacific gray whale exposures to SURTASS LFA sonar between 120dB and 180dB.

Site #	Operating Area	Number of Missions	Marine Mammal Stock	Number in Stock	Estimated Exposures without Mitigation	Estimated Exposures with Mitigation
5	Sea of Japan	2	Western North Pacific	121	52	2
6	East China Sea	1	Western North Pacific	121	0	0
7	South China Sea	1	Western North Pacific	121	23	1
	Totals	**4**			**75**	**3**

6.3.5.4 Humpback Whale

Over the course of 13 proposed missions, 120 humpback whales in two stocks, the Western North Pacific (WNP) and Central North Pacific (CNP), may be exposed to SURTASS LFA sonar transmissions in five mission areas, based on the results of the Navy's modeling. The most affected of the humpback stocks is predicted to be the WNP stock, which is composed of 1,107 individuals, and would have 98 whales or 8.54% of the stock affected at the 120 to 180 dB SPE exposure level during nine missions in three mission areas. Twenty-two humpback whales or 0.2% of the CNP stock of 10,103 individuals would potentially be exposed to SURTASS LFA sonar transmissions during four missions in two mission areas.

Table 25. Estimated humpback whale exposures to SURTASS LFA sonar between 120dB and 180dB.

Site #	Operating Area	Number of Missions	Marine Mammal Stock	Number in Stock	Estimated Exposures without Mitigation	Estimated Exposures with Mitigation
2	North Philippine Sea	3	Western North Pacific	1,107	5,176	78
3	West Philippine Sea	3	Western North Pacific	1,107	4,246	18
4	Offshore Guam	3	Central North Pacific	10,103	4,081	2
10	Hawaii North (25°N, 158°W)	2	Central North Pacific	10,103	2,441	10
11	Hawaii South (19.5°N 158.5°W)	2	Central North Pacific	10,103	3,644	12
	Totals	**10**			**19,588**	**120**

6.3.5.5 North Pacific Right Whale

One stock, the WNP stock, of North Pacific right whales may be exposed to SURTASS LFA sonar transmissions during six of eight proposed missions in five of the western North Pacific mission areas; North Pacific right whales are not expected to occur in all mission areas during the seasons in which the missions were proposed. Of the 922 estimated whales in the WNP stock, a total of four animals or 0.15% of the right whale stock possibly would be exposed to SURTASS LFA sonar transmissions at the 120 to 180 dB SPE exposure level.

Table 26. Estimated North Pacific right whale exposures to SURTASS LFA sonar between 120dB and 180dB.

Site #	Operating Area	Number of Missions	Marine Mammal Stock	Number in Stock	Estimated Exposures without Mitigation	Estimated Exposures with Mitigation
1	East of Japan	1	Western North Pacific	922	0	0
2	North Philippine Sea	3	Western North Pacific	922	59	2
5	Sea of Japan	2	Western North Pacific	922	24	1
6	East China Sea	1	Western North Pacific	922	0	0
7	South China Sea	1	Western North Pacific	922	22	1
	Totals	**8**			**105**	**4**

6.3.5.6 Sei Whale

Two stocks, the Hawaiian and NP, of sei whales may be exposed to SURTASS LFA sonar transmissions at the 120 to 180 dB SPE level during 10 proposed missions in the central and western North Pacific Ocean. Based on the Navy's effects analysis, the highest number of animals and percentage of a sei whale stock affected are 36 animals and 0.39%, respectively, of the NP stock of 8,600 individuals during six annual proposed missions in four mission areas. Three of the 77 whales in the Hawaiian stock of sei whales, or 0.80% may be exposed to LFA transmissions over four missions in the two central North Pacific mission areas.

Table 27. Estimted sei whale exposures to SURTASS LFA sonar between 120dB and 180dB

Site #	Operating Area	Number of Missions	Marine Mammal Stock	Number in Stock	Estimated Exposures without Mitigation	Esimated Exposures with Mitigation
1	East of Japan	1	North Pacific	8,600	1,533	7
4	Offshore Guam	3	North Pacific	8,600	1,406	18
8	Offshore North Pacific (25° to 40°N) (Offshore Japan)	1	North Pacific	8,600	1,869	6
9	Offshore North Pacific (10° to 25°N) (Offshore Japan)	1	North Pacific	8,600	256	5
10	Hawaii North (25°N, 158°W)	2	Hawaiian	77	27	1
11	Hawaii South (19.5°N 158.5°W)	2	Hawaiian	77	42	2
	Totals	**10**			**5,133**	**39**

6.3.5.7 Sperm Whale

Based on the results of the Navy's modeling, the highest total number of sperm whales in any stock potentially exposed to SURTASS LFA sonar at RLs of 120 to 180 dB SPE would be 574 whales or 0.55% of the NP stock. The NP stock of sperm whales, consisting of an estimated 102,112 individuals, may be exposed to LFA sonar during 16 missions in all nine of the western North Pacific mission areas. Only one other stock of sperm whales, the Hawaiian, potentially will be exposed to LFA sonar. In total, 872 sperm whale exposures in all 11 SURTASS LFA mission areas are estimated. Sperm whales in the Hawaiian stock, which is estimated to include 6,919 whales, may be exposed to LFA sonar during four proposed missions in the two central North Pacific mission areas, resulting in 298 sperm whale exposures, which may affect 4.29% of the Hawaiian stock.

Table 28. Estimated sperm whale exposures to SURTASS LFA sonar between 120dB and 180dB

Site #	Operating Area	Number of Missions	Marine Mammal Stock	Number in Stock	Estimated Exposures without Mitigation	Esimated Exposures with Mitigation
1	East of Japan	1	North Pacific	102,112	3,517	24
2	North Philippine Sea	3	North Pacific	102,112	10,650	90
3	West Philippine Sea	3	North Pacific	102,112	8,787	108
4	Offshore Guam	3	North Pacific	102,112	8,793	98
5	Sea of Japan	2	North Pacific	102,112	5,818	120
6	East China Sea	1	North Pacific	102,112	927	30
7	South China Sea	1	North Pacific	102,112	2,405	13
8	Offshore North Pacific (25° to 40°N) (Offshore Japan)	1	North Pacific	102,112	14,348	49
9	Offshore North Pacific (10-25°N	1	North Pacific	102,112	4,365	42
10	Hawaii North (25°N, 158°W)	2	Hawaii	6,919	15,558	224
11	Hawaii South (19.5°N 158.5°W)	2	Hawaii	6,919	11,497	74
	Totals	**20**			**86,665**	**872**

6.3.5.8 Main Hawaiian Insular DPS of False Killer Whale

Only one stock of false killer whales in the North Pacific Ocean, the Main Hawaiian Islands Insular DPS, is listed under the ESA. An estimated four false killer whales or 0.84% of the Main Hawaiian Islands Insular stock of 151 individuals may be exposed during four missions to SURTASS LFA sonar at the RL of 120 to 180 dB SPE, with mitigation applied, in two central North Pacific mission areas in which the Navy proposes to operate in August 2014 through August 2015.

Table 29. Estimated Main Hawaiian Insular false killer whale exposures to SURTASS LFA sonar between 120dB and 180dB.

Site #	Operating Area	Number of Missions	Marine Mammal Stock	Number in Stock	Estimated Exposures without Mitigation	Estimated Exposures with Mitigation
10	Hawaii North (25°N, 158°W)	2	Hawaii Insular	151	6,666	2
11	Hawaii South (19.5°N 158.5°W)	2	Hawaii Insular	151	5,316	2

	Totals	4			11,982	4

6.3.5.9 Hawaiian Monk Seal

The Hawaiian stock, composed of an estimated 1,212 Hawaiian monk seals, may be exposed to SURTASS LFA sonar transmissions at RLs of 120 to 180 dB SPE during five proposed missions. Based on the results of the Navy's modeling, the total number of Hawaiian monk seal's exposed to SURTASS LFA sonar transmissions during the annual reporting period is estimated at 16, for 1.12% of the Hawaiian monk seal stock potentially affected in three proposed mission central North Pacific mission areas. The highest number of exposures, 10, is estimated for the Hawaii-North mission area.

Table 30. Estimated Hawaiian monk seal exposures to SURTASS LFA sonar between 120dB and 180dB

Site #	Operating Area	Number of Missions	Marine Mammal Stock	Number in Stock	Estimated Exposures without Mitigation	Estimated Exposures with Mitigation
8	Offshore North Pacific (25° to 40°N) (Offshore Japan)	1	Hawaii	1,161	67	1
10	Hawaii North (25°N, 158°W)	2	Hawaii	1,161	543	10
11	Hawaii South (19.5°N 158.5°W)	2	Hawaii	1,161	467	5
	Totals	**5**			**1,077**	**16**

6.3.5.10 Sea Turtles

The Navy did not simulate potential exposure of sea turtles to SURTASS LFA sonar transmissions. Due to the seasonal and life stage changes in habitat occupation, sea turtle populations are particularly difficult to census. Abundance estimates are based on the most current information available regarding counts of the most accessible member of the population, nesting females, which does not account for the abundance of male sea turtles. The lack of detailed population data and scarcity of density data for sea turtles do not allow for density estimates to be derived for populations in the North Pacific Ocean.

Juvenile and adult stages of sea turtles could potentially be exposed to SURTASS LFA sonar transmissions when residing in pelagic environments. We assume that the monitoring protocols associated with SURTASS LFA sonar would be more effective with larger sea turtles, such as adult leatherback and loggerhead turtles, than with smaller species such as olive ridley, hawksbill, or green turtles or smaller individual leatherback or loggerhead turtles. Monitoring measures may not detect some individual or smaller sea turtles, which would increase their risk of exposure to sound pressure levels associated with SURTASS LFA sonar within the mitigation

zone (that is, 180 dB) if they encountered SURTASS LFA sonar vessels during sonar transmission.

Due to the small size of the LFA mitigation zone relative to the enormous area and volume of the ocean and the water column usage by sea turtles, and the three-part monitoring system, sea turtles are unlikely to be exposed to SURTASS LFA sonar transmission levels ≥180 dB re 1 μPa (rms), the threshold at which they are considered to be injured. For instance, in 2011, a sea turtle was observed by visual monitors aboard the USNS EFFECTIVE at a distance of about 15 m (49 ft) from the vessel over eight minutes after LFA sonar transmissions ceased. Due to the proximity to the vessel, an assessment was made to determine the potential for the sea turtle to be within the LFA mitigation zone (180 dB sound field) and determine whether the HF/M3 sonar should have detected the turtle. The assessment concluded that due to the position of the HF/M3 sonar system above the LFA VLA, the sea turtle would have had to swim from the surface through the HF/M3 sonar detection zone to enter into the 180-dB LFA mitigation zone, making the potential for an acoustic detection of the turtle highly likely. The lack of detection by the HF/M3 sonar would indicate that the sea turtle remained at or near the surface and did not dive into the 180 dB LFA mitigation zone during transmissions. Thus, it is improbable that the sea turtle received SPL from SURTASS LFA sonar at or above 180 dB re 1 μPa (rms).

Although the potential of an interaction between SURTASS LFA sonar and individuals of any sea turtle species is statistically small (the Navy's analyses concluded that the possible number of times a leatherback sea turtle could be in the vicinity of a SURTASS LFA sonar vessel would be less than three out of 18,000 animals per year per vessel and with the monitoring protocols, the Navy concluded that this number would approach zero); this scenario assumes that the sea turtles and SURTASS LFA sonar vessels would be randomly distributed in the ocean (in such vast ocean areas, two randomly-distributed objects would have a low probability of co-occurring). The potential exposure could increase depending on the deployment of the SURTASS LFA sonar vessels. Although the decibel levels that could cause potential tissue damage in sea turtles remains unknown, sea turtles are unlikely to be more sensitive to sound-induced tissue damage than marine mammals (D. A. Croll et al., 1999).

6.4 Response Analysis

This section addresses the potential responses of ESA-listed species to SURTASS LFA sonar. For species that are exposed to SURTASS LFA sonar transmission, this response analysis examines the relationship between received levels and ecological effects; the plausibility of adverse effects occurring as a result of exposure; and linkages between measurable ecological effects and vital rates or biologically-important behavior in populations of listed species, which are the endpoints for this assessment. In this analysis we consider potential for non-auditory injury, auditory injury, temporary threshold shift, behavioral changes, masking, and stranding as a result of exposure to SURTASS LFA sonar.

To assess the plausibility of adverse effects resulting from exposure to SURTASS LFA sonar, we evaluated various lines of evidence from published and unpublished sources on the effects of SURTASS LFA sonar on threatened and endangered species, other studies of low frequency sound, and reports on the effects of other sonars. We evaluate the same lines of evidence to establish linkages between the potential effects of SURTASS LFA sonar on biologically important behavior of listed species exposed to the sonar. We also consider information provided in annual and summary reports on previous SURTASS LFA sonar activities.

6.4.1 Cetaceans

To establish relationships between the LFA sonar as a stressor and its effects on threatened and endangered species, our assessment relied on the Navy's acoustic modeling and risk continuum analyses (Navy, 2001b, 2001c, 2011b, 2012a), which estimate the risk of injury and changes to biologically important behavior that might be caused by SURTASS LFA sonar and the Navy's analyses correlating the risk of biologically significant behavior to received levels (single ping equivalents in decibels) using probability distribution functions. The results of the Navy's analyses are presented as continuous functions that are analogous to the stressor-response curves: low received levels would not be expected to elicit a negative response in a species; at the other end of these curves, high received levels would be expected to elicit much more serious, negative responses.

6.4.1.1 Potential Non-Auditory Injury

Nowacek et al. (2007) and Southall et al. (2007b) reviewed potential areas for non-auditory injury to marine mammals from active sonar transmissions. These include direct acoustic impact on tissue, indirect acoustic impact on tissue surrounding a structure, and acoustically mediated bubble growth within tissues from supersaturated dissolved nitrogen gas.

6.4.1.1.1 Direct Acoustic Impacts

Physical effects, such as direct acoustic trauma or acoustically enhanced bubble growth, require relatively intense received energy that would only occur at short distances from high-powered sonar sources (Nowacek et al., 2007; Zimmer & Tyack, 2007). The best available scientific information shows that, while resonance can occur in marine animals, this resonance does not necessarily cause injury, and any such injury is not expected to occur below a received level of 180 dB sound pressure level. Damage to the lungs and large sinus cavities of cetaceans from air space resonance is not regarded as a likely significant non-auditory injury because resonance frequencies of marine mammal lungs are below that of the LFA sonar signal (Finneran, 2003). Further, biological tissues are heavily damped and tissue displacement at resonance is predicted to be exceedingly small. In addition, lung tissue damage is generally uncommon in acoustic-related strandings (Brandon L. Southall et al., 2007b).

6.4.1.1.2 Gas Bubble Formation

Presently, there are discussions among researchers regarding the potential for marine mammals to suffer from a form of decompression sickness caused by in vivo nitrogen gas-bubble growth.

Jepson et al. (2003; 2005) and Fernandez et al. (2005) reported results of necropsies of stranded beaked whales, some of which coincided with naval sonar exercises, which they interpreted as consistent with a decompression-like syndrome (Nowacek et al., 2007).

Scientists have documented bone lesions (osteonecrosis), which may be a chronic result of nitrogen bubbles, in the rib and chevron bone articulations, nasal bones, and deltoid crests of sperm whale specimens from the Atlantic and Pacific Oceans dating from the late 1800s to 2003, (M. J. Moore & Early, 2004). This suggests that nonlethal pathologies related to gas bubbles may occur during the normal life span of, at least, the deep-diving sperm whale. Houser (2008) assessed the potential for nitrogen bubble formation in a trained dolphin. Based on repetitive dives to depths of 10, 30, 50, 70, and 100 m (32.8, 98.4, 164, 230, and 328 ft), ultrasound inspections were completed on the portal and innominate veins (i.e., the left and right brachiocephalic veins). Blood samples were also taken over a 20-minute (min) period at the end of each of the 50, 70, and 100 m (164, 230, and 328 ft) dives for the assessment of nitrogen partial pressure. There were no vascular bubbles found in any post-dive ultrasound. Nitrogen partial pressures from blood samples were not significantly elevated from those of the dolphin at rest (20 min post dive). Results suggest that repetitive, prolonged dives up to 100 m (328 ft) accumulate insufficient nitrogen to generate asymptomatic intravascular bubbles in bottlenose dolphins.

Zimmer and Tyack (2007) modeled nitrogen tension and bubble growth in beaked whales during normal diving behavior and for several hypothetical dive profiles to assess the risk of nitrogen bubble formation. These authors concluded that macroscopic bubbles are unlikely to pose a risk of decompression-like syndrome from a simple interruption of a normal deep foraging dive, even when accompanied by an unrealistic ascent rate. Zimmer and Tyack (2007) concluded, contrary to the findings of Jepson et al. (2003), that the interruption and rapid ascent from a regular deep foraging dive is unlikely to pose a risk of decompression-like syndromes; they suggested that gas bubble lesions in stranded beaked whales reported by Jepson et al. (2003; 2005) and Fernandez et al. (2005) might be caused by repetitive dives of short to medium surfacing duration without exceeding the depth of alveolar collapse. Also, Zimmer and Tyack (2007) found that the longer the dive time compared to surfacing time, the greater the risk; the authors suggested the hypothesis that beaked whales have an avoidance response to killer whales and great white sharks, which are their primary near-surface predators, resulting in their swimming at depths of approximately 25 m (82 ft) without exceeding alveolar collapse. This hypothesis requires more behavioral and physiological research.

Baird et al. (2008) investigated the variation in diving behavior from time-depth recorders on six Blainville's and two Cuvier's beaked whales. Both species demonstrated ascent rates from dives deeper than 800 m (2,625 ft) that were significantly slower than decent rates, both during the day and at night, suggesting some physiological purpose for the slower ascents. The whales also spent more time in dives to mid-water depths (100 to 600 m [328 to 1,969 ft]) during the day. At

night, the whales spent more time in shallow (<100 m) dives. This diel variation in behavior suggests that beaked whales may spend less time in surface waters during the day to avoid visually oriented predators, including sharks and killer whales.

Fahlman et al. (2009) modeled the effects of lung compression and collapse (pulmonary shunt) on the uptake and removal of oxygen, carbon dioxide, and nitrogen in blood and tissue, and on end-dive nitrogen concentrations for breath-holding marine mammals (e.g., elephant seals, Weddell seals, and beaked whales). Fahlman et al. (2009) suggested that repeated dives might result in tissue and blood levels of nitrogen sufficient to cause symptomatic bubble formation.

Based on the current knowledge of gas exchange and physiology of marine mammals, Hooker et al. (2009) developed a mathematical model to predict blood and tissue levels of nitrogen gas for three species of beaked whales: northern bottlenose, Cuvier's, and Blainville's beaked whales. Hooker et al. suggested that deep-diving marine mammals live with and manage high levels of nitrogen gas in their tissues and blood. Due to differences in dive behavior, predicted nitrogen levels were higher in Cuvier's beaked whales than in northern bottlenose whales and Blainville's beaked whales. Hooker et al. (2009) state that while the prevalence of Cuvier's beaked whale strandings after naval sonar exercises could be explained by a higher abundance of the species in the area, their results suggest that species differences in behavior and/or physiology may also play a role.

Moore et al. (2009) performed gross histologic and radiographic observations related to the presence of gas bubbles in the tissues and blood of seals and dolphins drowned in gillnets, set at a depth of approximately 80 m (263 ft). The majority (15 of 23) of the seals and dolphins had extensive bubble formation in multiple tissues and blood. In addition, computer tomography (CT), which was performed on four randomly-selected marine mammals, identified gas bubbles in various tissues. Due to the good condition of the carcasses, absence of bacteria and autolytic (self-digestion) changes, the study concluded that peri- or post-mortem phase change of supersaturated blood and tissues was the most likely cause of the bubbles. Overall, Moore et al. (2009) found a high prevalence of vascular and interstitial bubbles in seals and dolphins drowned in gillnets set at a depth of approximately 80 m (263 ft). In contrast, a very low prevalence of bubble lesions was found for beach-stranded marine mammals in this study (one of 41) and in a study by Jepson et al. (2005) (10 of 2,376). The results of the Moore et al. (2009) analyses support the modeling of simulated dive profiles by Zimmer and Tyack (2007), which suggest an increase in risk of bubble formation caused by repetitive dives with short to medium surface durations, without exceeding the depth of alveolar collapse, which is estimated to be about 80 m (263 ft) for dolphins.

Hooker et al. (2012) reviewed evidence for gas-bubble incidence in marine mammal tissues and suggest that diving mammals vary their physiological responses according to multiple stressors. The authors state that while there is strong evidence that marine mamamls have some control

(voluntary or reflexive) over the intensity of the dive response and the possible formation of gas bubbles as well as evidence that in some situations marine mammals routinely exceed their presumed limits(e.g. exceeding their aerobic capacity and possibly tolerating a build-up of lactic acid to optimize foraging efficiency), it might then be possible that certain preconditions of high saturation levels, combined with behavioral or physiological responses to a perceived threat, exacerbate saturation levels and lead to the appearance of bubbles. So far, observations of potential decompression injury have been coincident with anthropogenic triggers, but little is known about marine mammal perceptions and responses to natural versus anthropogenic threats, and the authors could not rule out that the same response could be triggered by rare natural events.

Despite the increase in research and literature, there remains scientific disagreement and/or lack of scientific data regarding the evidence for gas bubble formation as a causal mechanism between certain types of acoustic exposures and stranding events. These issues include: 1) received acoustic exposure conditions; 2) pathological interpretation; 3) acoustic exposure conditions required to directly induce physiological trauma; 4) behavioral reactions caused by sound exposure such as atypical dive patterns; and 5) the extent of postmortem artifacts (Brandon L. Southall et al., 2007a).

The hypotheses for gas bubble formation related to beaked whale strandings is that beaked whales potentially have strong avoidance responses to MFA sonars because they sound similar to their main predator, the killer whale (R. W. Baird, Webster, et al., 2008; Cox et al., 2006; Sascha K. Hooker et al., 2009; Brandon L. Southall et al., 2007b; Zimmer & Tyack, 2007). Because SURTASS LFA sonar transmissions are lower in frequency (<500 Hz) and dissimilar in characteristics from those of marine mammal predators, the above scientific studies do not provide additional evidence that SURTASS LFA sonar has caused behavioral reactions, specifically avoidance responses, in beaked whales. Thus, SURTASS LFA sonar transmissions are not expected to cause gas bubble formation or beaked whale strandings.

Kvadsheim et al. (2012) investigate the formation of nitrogen gas bubbles in blood and tissue with an increased risk of decompression sickness following the exposure to sonar transmissions of sperm whales, killer whales, long-finned pilot whales, Blainville's beaked whale and Cuvier's beaked whales. They concluded that the possibility that a combination of behavioral and physiological responses to sonar have the potential to alter the blood and tissue end-dive nitrogen tension to levels which could cause decompression sickness and formation of *in vivo* bubbles, but the actually observed behavioral responses of cetaceans to sonar in their study, did not imply any significantly increased risk of decompression sickness as the result of exposure to sonar transmissions.

6.4.1.2 Potential Auditory Injury- Permanent Threshold Shift

Permanent Threshold Shift (PTS) is defined as the deterioration of hearing due to prolonged or repeated exposure to sounds that accelerate the normal process of gradual hearing loss (K. Kryter, 1985) and the permanent hearing damage from brief exposure to extremely high sound levels (W. John Richardson, Charles R. Greene Jr., et al., 1995). PTS results in a permanent elevation in hearing threshold—an unrecoverable reduction in hearing sensitivity (Brandon L. Southall et al., 2007b). Therefore, PTS is considered an injury.

Since the boundary line between TTS and PTS is neither clear, definitive, nor predictable for marine mammals, NMFS adopted the standard that 20 dB of threshold shift defines the onset of PTS (i.e., a shift of 20 dB in hearing threshold) (NMFS, 2002). NMFS used this same standard in the second Final Rule (72 FR 46846). Southall et al. (2007b) proposed injury criteria for individual LF/MF/HF marine mammals exposed to non-pulsed sound types, which included discrete acoustic exposures from SURTASS LFA sonar. The proposed injury criteria for cetaceans and pinnipeds in water are sound exposure levels of 215 dB received level and 203 dB received level, respectively. An 18-dB adjustment must be made for the longer LFA sonar signal (nominally 60 seconds) resulting in injury criteria for SURTASS LFA sonar for LF/MF/HF cetaceans of a sound exposure level of 197 dB received level and for pinnipeds in water an sound exposure level of 185 dB received level. The SURTASS LFA sonar injury criterion for all marine mammals was an sound pressure level of 180 dB received level (Navy, 2001b, 2007c, 2012a), which is noticeably lower and, therefore, more conservative, than the injury criteria proposed by Southall et al. (2007b).

The center of LFA sonar arrays would operate at a depth of 122 m (400 feet), which would generally place the array and the main sound transmission beam produced by the arrays below the mixed-layer depth of an ocean. The transmitted signal would move in an extended wave through the ocean, initially moving away from the surface, then rising at some distance from the source, before sinking again (in some instances affected by ducting, sound channels, convergence zones, and bottom interactions). At the source, an LFA sonar signal is approximately 215 dB re: 1 μPa_{rms} at 1 m for a single projector. This signal would attenuate with time and distance from the source. Received levels from the array attenuate to 180 dB re: 1 μPa_{rms} at 1 m (sound pressure level) nominally from 750 to 1,000 m (2,460 to 3,281 ft).

The SURTASS LFA sonar has the greatest potential for injuring threatened or endangered species if an animal is close to one or more of the projectors during a ping. If that occurred, the animal could be exposed to sound pressure levels between 215 and 180 dB re: 1 μPa_{rms} at 1 m (sound pressure level), which could injure listed species by causing permanent threshold shifts. To minimize the likelihood of this occurring, the Navy proposes to use HF/M3 sonar, visual monitoring, and passive acoustic monitoring, to detect animals within 1 to 2 kilometers of the projectors; if any of these monitoring methods detects animals within this zone, the projectors would be shut down until the animal(s) moved out of the zone.

Testing of the HF/M3 sonar has demonstrated a probability of single-ping detection above 95 percent within the LFA sonar mitigation and buffer zones for many marine mammals (Ellison & Stein, 1999). Under normal operating conditions, marine mammals will receive multiple pings, increasing the probability of detection within the LFA mitigation zone to near 100 percent (Navy 2001b). Combined with the visual monitoring and passive acoustic monitoring protocols, the risk of marine mammals being exposed to sound pressure levels in excess of 180 dB re: 1 μPa_{rms} at 1 m approaches zero.

Therefore, unless an animal occurred within the 180-dB re: 1 μPa_{rms} at 1 m LFA sonar mitigation and buffer zones during a ping, it is not likely to experience physical injury, including damage to tissues. Because of the mitigation measures, a marine mammal's or sea turtle's likelihood of being exposed to these sound pressure levels is very low.

6.4.1.3 Potential Temporary Threshold Shift

In addition to the possibility of causing permanent injury to hearing, sound may cause TTS, a temporary and reversible loss of hearing that may last for minutes to days. The following physiological mechanisms may result in TTS:

1. Reduced sensitivity of the sensory hair cells in the inner ear as a result of their being over-stimulated;
2. Modification of the chemical environment within sensory cells;
3. Displacement of certain inner ear membranes;
4. Increased blood flow; and
5. Post-stimulation reduction in both efferent (impulses traveling from the central nervous system to the peripheral sensory tissue) and sensory output (K. D. Kryter, 1994; Brandon L. Southall et al., 2007b).

In the 2002 and 2007 SURTASS LFA sonar Final Rules (67 FR 46712 and 72 FR 46846), NMFS stated that TTS is not an injury. The duration of TTS depends on a variety of factors including intensity and duration of the stimulus. Southall et al. (Brandon L. Southall et al., 2007b) considered that the temporary elevation of a hearing threshold by 6 dB was a sufficient definition for TTS onset. For cetaceans, most of the published TTS data are limited to bottlenose dolphins and belugas (Finneran & Houser., 2005; Finneran, Schlundt, Branstetter, & Dear, 2007b; Finneran, Schlundt, Carder, & Ridgway, 2002; Finneran, Schlundt, Dear, Carder, & Ridgway, 2000b; Nachtigall, Pawloski, & Au, 2003a; P. E. Nachtigall, A. Y. Supin, J. Pawloski, & W. W. L. Au, 2004; C. R. Schlundt et al., 2000).

A study of TTS in harbor porpoises used a seismic airgun as a stimulus (Lucke et al., 2009). Airguns produce an impulsive signal and have a broad frequency range but also have substantial energy in the low frequency region. A small airgun was used in proximity to the animals

(between 14 to 150 m), a context that is likely to enhance behavioral responsiveness. The harbor porpoises showed a behavioral response at a received level of 174 dB re: 1 µPa (peak-to-peak), which is equivalent to an sound exposure level of 145 dB re: 1 µPa2-sec (Lucke et al., 2009). Harbor porpoise hearing was tested at a frequency of 4 kHz and TTS was detected at a received level of 199.7 dB re: 1 µPa (peak-to-peak), which is equivalent to an sound exposure level of 164.3 dB re: 1 µPa2-sec (Lucke et al., 2009). These are the lowest received sound levels that produce TTS yet reported. These data are intriguing and clearly indicate a need for additional research. Unfortunately, only one individual was tested in this study. The applicability of these results to SURTASS LFA sonar is uncertain, given the large differences in source characteristics between airguns and LFA sonar. Furthermore, LFA sonar typically operates in water deeper and further offshore than most harbor porpoise habitats. Indeed, harbor porpoises are found in only one of the SURTASS LFA sonar OPAREAs analyzed, for which zero exposures at levels >180 dB sound pressure level were found. Nevertheless, this study indicates that further study of TTS in porpoises is warranted. Ideally, additional harbor porpoise individuals as well as additional high-frequency hearing species would be tested. If this type of results are confirmed for harbor porpoise or found in other HF hearing species, then the analyses for those species would merit revision.

In a study on the effects of noise level and duration of TTS in a bottlenose dolphin, Mooney et al. (2009) exposed a bottlenose dolphin to octave-band noise (4 to 8 kHz) of varying durations (2 to 30 minutes) and sound pressure level received levels (130 to 178 dB re: 1 µPa). The results of the Mooney et al. study indicated that shorter-duration sound exposures often require greater sound energy to induce TTS than longer-duration exposures and also supported the trend that longer-duration exposures often induce greater amounts of TTS, which concurrently require longer recovery times.

In a controlled exposure experiment, Mooney et al. (2009) demonstrated that MFA sonar could induce temporary hearing loss in a bottlenose dolphin (*Tursiops truncatus*). Temporary hearing loss was induced by repeated exposure to a sound exposure level of 214 dB re: 1 µPa2-sec. Subtle behavioral alterations were also associated with the sonar exposures. At least with one odontocete species (common bottlenose dolphin), sonar can induce both TTS and mild behavioral effects; but exposures must be prolonged with high exposure levels to generate these effects. The received level used in the Mooney et al. (2009) experiment was an sound pressure level of 203 dB, which equates to the received level approximately 40 m (131 ft) from an MFA sonar operated at an sound pressure level of 235 dB (source level). Mooney et al. (2009) concluded that in order to receive an sound exposure level of near 214 dB, an animal would have to remain in proximity of the moving sonar, which is transmitting for 0.5 sec every 24 sec over an approximately 2 to 2.5 min period, an unlikely situation.

Sound exposure levels necessary for TTS onset for pinnipeds in water have been measured for harbor seals, California sea lions, and northern elephant seals. As reported by Southall et al.

(2007), Kastak et al. (2005) presented comparative analysis of underwater TTS for pinnipeds. This indicated that in harbor seals, a TTS of ~6 dB occurred with a 25-min exposure to 2.5 kHz octave-band noise of 152 dB sound pressure level (183 dB sound exposure level); a California sea lion showed TTS-onset under the same conditions at 174 dB sound pressure level (206 dB sound exposure level); and a northern elephant seal under the same conditions experienced TTS-onset at 172 dB sound pressure level (204 dB sound exposure level). Finneran et al. (2003) exposed two California sea lions to single underwater pulses from an arc-gap transducer and found no measurable TTS following exposures of up to 183 dB sound pressure level (215 dB sound exposure level).

Animals suffering from TTS over longer periods of time, such as hours to days, may be considered to have a change in a biologically significant behavior, as they may be prevented from detecting sounds that are biologically relevant, including communication sounds, sounds of prey, or sounds of predators. As noted by Mooney et al. (2009), shorter duration sound exposures can require greater sound energy to induce TTS than longer duration exposures, and longer duration exposures can induce greater amounts of TTS. In assessing the potential for LFA sonar transmissions to cause TTS, the much shorter length of the LFA sonar signal (1 min) versus the above studies (2 to 30 min) must be considered.

This recent scientific information supports the findings that the likelihood that SURTASS LFA sonar, with a sound pressure level of 180 dB received level, may cause TTS in marine mammals is negligible. Further, mitigation measures, such as mitigation zones and shutdown protocols are employed where there is the potential for a marine mammal to incur TTS and prevent any animal from incurring PTS.

6.4.1.4 Potential Behavioral Responses

The primary potential deleterious effect from SURTASS LFA sonar is change in a biologically significant behavior. The National Research Council (NRC, 2005) discussed biologically significant behaviors and possible effects and stated that an action or activity becomes biologically significant to an individual animal when it affects the ability of the animal to grow, survive, and reproduce. These are the effects on individuals that can have population-level consequences and affect the viability of the species (NRC, 2005). For military readiness activities, such as the use of SURTASS LFA sonar, Level B “harassment” under the MMPA is defined as any act that disturbs or is likely to disturb a marine mammal by causing disruption of natural behavioral patterns to a point where the patterns are abandoned or significantly altered. Behaviors include migration, surfacing, nursing, breeding, feeding, and sheltering.

The Low Frequency Sound Scientific Research Program (LFS SRP) in 1997 to 1998 provided important results on, and insights into, the types of responses of baleen whales to LFA sonar signals and how those responses scaled relative to received level and context. The results of the LFS SRP confirmed that some portion of the total number of whales exposed to LFA sonar

responded behaviorally by changing their vocal activity, moving away from the source vessel, or both; but the responses were short-lived (Christopher W. Clark & Fristrup, 2001b).

In the LFS SRP LFA sonar playback experiment (Phase II), migrating gray whales avoided exposure to LFA sonar signals (source levels of 170 and 178 dB sound pressure level) when the source was placed in the center of their migration corridor. Responses were similar for the 170-dB source level LFA sonar stimuli and for the 170-dB source level 1/3rd-octave, band-limited noise with timing and frequency band similar to the LFA sonar stimulus. However, during the LFA sonar playback experiments, in all cases, whales resumed their normal activities within tens of minutes after the initial exposure to the LFA sonar signal (Christopher W. Clark & Fristrup, 2001b). Essentially, the whales made minor course changes to go around the source. When the source was relocated within the outer portion of the migration corridor (twice the distance offshore), and the source level was increased to reproduce the same sound field for the central corridor playback condition, the gray whales showed little to no response to the LFA sonar source. This result stresses the importance of context in interpreting the animals' behavioral responses to underwater sounds and demonstrates that received level is not necessarily a good predictor of behavioral impact.

The LFS SRP also conducted field tests to examine the effects of LFA sonar transmissions on foraging fin and blue whales off San Nicolas Island, California (Phase I). Overall, whale encounter rates and dive behavior appeared to be more strongly linked to changes in prey abundance associated with oceanographic parameters rather than LFA sonar sound transmissions (Donald A. Croll et al., 2001).

In the final phase of the LFS SRP (Phase III), the effect of LFA sonar on humpback whales during the winter mating season was investigated. Both Miller et al. (2000a) and Fristrup et al. (2003) published results from tests conducted with male humpback singers off the Big Island, Hawaii during which they evaluated variation in song length as a function of exposure to LFA sonar sounds. Fristrup et al. (2003) used a larger data set to describe song length variability and to explain song length variation in relation to LFA sonar broadcasts. In spite of methodological and sample size differences, the results of the two analyses were generally in agreement, and both studies indicated that humpback whales might lengthen their songs in response to LF broadcasts.

The Fristrup et al. (2003) results also provided a detailed picture of short-term response as compared to behavioral variation observed in the absence of the stimuli. These responses were relatively brief in duration, with all observed effects occurring within 2 hrs of the last LFA sonar source transmission. It should be noted that these effects were not obvious to the acoustic observers on the scene, but were revealed by careful, complex post-test statistical analyses (Fristrup et al., 2003). Aside from the delayed responses, other measures failed to indicate cumulative effects from LFA sonar broadcasts, with song-length response being dependent

solely on the most recent LFA sonar transmission, and not the immediate transmission history. The modeled seasonal factors (changes in density of whales sighted near shore) and diurnal factors (changes in surface social activities) did not show trends that could be plausibly explained by cumulative exposure. Increases in song length from early morning to afternoon were the same on days with and without LFA sonar transmissions, and the fraction of variation in song length that could be attributed to LFA sonar broadcast was small (<10 percent). Fristrup et al. (2003) found high levels of natural variability in humpback song length and interpreted the whales' responses to LFA sonar broadcasts to indicate that exposure to LFA sonar would not impose a risk of dramatic changes in humpback whale singing behavior that would have demographic consequences.

Southall et al. (2007b) reviewed the relatively extensive behavioral observations of low frequency cetaceans exposed to non-pulse sources. While there are clearly major areas of uncertainty, Southall et al. concluded that the literature indicated that there were no (or very limited) responses to received levels of 90 dB to 120 dB sound pressure level with an increasing probability of avoidance and other behavioral effects in the 120 to 160 dB sound pressure level (received level) range.

6.4.1.4.1 Potential Behavioral Avoidance

There are few empirical studies of avoidance responses of free-living cetaceans to mid-frequency sonars. Much more information is available on the avoidance responses of free-living cetaceans to other acoustic sources, like seismic airguns and low frequency sonar.

Richardson et al. (1995) noted that avoidance reactions are the most obvious manifestations of disturbance in marine mammals. Richardson et al. (1985) and Richardson (1997) used controlled playback experiments to study the response of bowhead whales in Arctic Alaska. In their studies, bowhead whales tended to avoid drill ship noise at estimated received levels of 110 to 115 dB and seismic sources at estimated received levels of 110 to 132 dB. Richardson et al. (1995) concluded that some marine mammals would tolerate continuous sound at received levels above 120 dB re: 1 μPa for a few hours. These authors concluded that most marine mammals would avoid exposures to received levels of continuous underwater noise greater than 140 dB when source frequencies were in the animal's most sensitive hearing range.

Several authors noted that migrating whales are likely to avoid stationary sound sources by deflecting their course slightly as they approached a source (LGL and Greenridge 1987 in W. John Richardson, Charles R. Greene Jr., et al., 1995). A study examined responses of gray whales migrating along the California coast to various sound sources located in their migration corridor (Malme et al., 1983, 1984). Gray whales showed statistically significant responses to four different underwater playbacks of continuous sound at received levels of approximately 120 dB. The sources of the playbacks were typical of a drillship, semisubmersible, drilling platform, and production platform.

This study was replicated in Phase II of the Scientific Research Program using SURTASS LFA sonar stimuli. However, the Phase II research demonstrated that it may be invalid to apply the inshore (2 km from shore) response model (when 50 percent of the whales avoided SURTASS LFA sonar stimuli at received levels of 141 +3 dB) to sources that are offshore (4 km from shore) of migrating whales, and that whales did not avoid offshore sources at received levels of 140 dB.. This implies that the inshore avoidance model, in which 50 percent of the whales avoid exposure to levels of 141 +3 dB, may not be valid for whales in proximity to an offshore source (Buck & Tyack, 2000).

Taken together, the data generated during the three phases of the LFS Scientific Research Program did not support the initial hypothesis that most baleen whales exposed to received levels near 140 dB would exhibit behavioral changes and avoid the area. These experiments, which exposed baleen whales to received levels ranging from 120 to about 155 dB, detected only minor, short-term behavioral responses whose potential effects on behavior. These conclusions are supported by other studies of whale responses to low frequency, anthropogenic sounds. For example, bowhead (*Balaena mysticetus*) and gray whales (*Eschrichtius robustus*), showed clear patterns of short-term, behavioral disturbance in response to a variety of actual and simulated vessel activity and noise (Malme et al., 1983; W. John Richardson, Davids, Evans, & Norton, 1985). Richardson et al. (W. J. Richardson, Finley, Miller, & Davis, 1995) noted that whales have been seen within a few kilometers of operating seismic vessels, although they added that any discomfort the seismic sound pulses may have caused remains unknown. Humpback whales showed similar patterns on their summering grounds (G. Bauer & Herman, 1986) and on their wintering grounds (G. B. Bauer, 1986) in response to vessel noise. Richardson et al. (W. J. Richardson et al., 1995) argued that intermittent pulses with peak levels between 160 to 180 dB are less likely to cause discomfort than continuous sounds at the same sound pressure levels.

In the Caribbean, sperm whales avoided exposure to mid-frequency submarine sonar pulses, in the range 1000 Hz to 10,000 Hz (IWC, 2005). Blue and fin whales have occasionally been reported in areas ensonified by airgun pulses. Systematic data on their reactions to airguns are generally lacking. Sightings by observers on seismic vessels off the United Kingdom suggest that, at times of good sightability, the number of blue, fin, sei, and humpback whales seen when airguns are shooting are similar to the numbers seen when the airguns are not shooting (C. J. Stone, 1997, 1998, 2000, 2001). However, fin and sei whale sighting rates were higher when airguns were shooting, which may be due to a tendency to remain at or near the surface at times of airgun operation (C. J. Stone, 2003). The analysis of the combined data from all years indicated that baleen whales stayed farther from airguns during periods of shooting (C. J. Stone, 2003). Baleen whales also altered course more often during periods of shooting and more were headed away from the vessel at these times, indicating some level of localized avoidance of seismic activity (C. J. Stone, 2003).

Sperm whales reacted to military sonar, apparently from a submarine, by dispersing from social aggregations, moving away from the sound source, remaining relatively silent and becoming difficult to approach (Watkins, Morre, & Tyack, 1985). Captive bottlenose dolphins and a white whale exhibited changes in behavior when exposed to 1 sec pulsed sounds at frequencies similar to those emitted by multi-beam sonar that is used by geophysical surveys (Ridgway et al., 1997; C. R. Schlundt et al., 2000), and to shorter broadband pulsed signals (Finneran, Schlundt, Carder, et al., 2000; Finneran, Schlundt, Carder, et al., 2002).

Behavioral changes typically involved what appeared to be deliberate attempts to avoid a sound exposure or to avoid the location of the exposure site during subsequent tests (Finneran, Schlundt, Carder, et al., 2000; C. R. Schlundt et al., 2000). Dolphins exposed to 1-sec intense tones exhibited short-term changes in behavior above received sound levels of 178 to 193 dB re: 1 μPa (rms) and belugas did so at received levels of 180 to 196 dB and above. Received levels necessary to elicit such reactions to shorter pulses were higher (Finneran, Schlundt, Carder, et al., 2000; Finneran, Schlundt, Carder, et al., 2002). Test animals sometimes vocalized after exposure to pulsed, mid-frequency sound from a watergun (Finneran, Schlundt, Dear, et al., 2002). In some instances, animals exhibited aggressive behavior toward the test apparatus (Finneran, Schlundt, Carder, et al., 2000; Ridgway et al., 1997).

Nowacek et al. (2004) conducted controlled exposure experiments on North Atlantic right whales using ship noise, social sounds of conspecifics, and an alerting stimulus (frequency modulated tonal signals between 500 Hz and 4.5 kHz). Animals were tagged with acoustic sensors (D-tags) that simultaneously measured movement in three dimensions. Whales reacted strongly to alert signals at received levels of 133-148 dB sound pressure level, mildly to conspecific signals, and not at all to ship sounds or actual vessels. The alert stimulus caused whales to immediately cease foraging behavior and swim rapidly to the surface.

6.4.1.4.2 Potential Disturbance Responses

There is evidence that wild animals respond to human disturbance in the same way that they respond to predators (Colin M. Beale & Pat Monaghan, 2004; Frid, 2003; Frid & Dill, 2002; Gill & Sutherland, 2001; Romero, 2004). These responses manifest themselves as stress responses (in which an animal perceives human activity as a potential threat and undergoes physiological changes to prepare for a flight or fight response or more serious physiological changes with chronic exposure to stressors), interruptions of essential behavioral or physiological events, alteration of an animal's time budget, or some combinations of these responses (Frid & Dill, 2002; Romero, 2004; Robert M. Sapolsky et al., 2000; Walker et al., 2005). These responses have been associated with abandonment of sites (Sutherland & Crockford, 1993), reduced reproductive success (M. Giese, 1996; Müllner et al., 2004), and the death of individual animals (Daan et al., 1996).

Brownell (2004) reported observations of the effects of behavioral disturbance on the endangered western gray whale population off the northeast coast of Sakhalin Island associated with seismic activities in that region. In 1997, various behavioral disturbances concomitant with seismic activities were observed including changes in swimming speed and orientation, respiration rates, and distribution offshore. Cumulative impacts of these short-term disturbances are not known. In 2001, seismic activities were conducted in the known feeding area of these whales. It was observed that whales left the feeding ground during these activities and moved to areas farther south. They only returned to the feeding ground after the seismic activities ceased days later. The potential impacts on these whales, especially mother-calf pairs and "skinny whales", of being displaced to the south outside the normal feeding area are not known but are cause for concern. As reported previously, whales observed to be much skinnier than normal were first observed in 1999 and continue to be observed in the population but in smaller numbers. Any disruption of feeding can be expected to impact the ability of these animals to store sufficient food reserves prior to migration.

6.4.1.4.3 Potential for No Response

One study of blue whales reported that when pulses from air guns were produced off Oregon, blue whales continued vocalizing at the same rate as before the pulses, suggesting that at least their vocalization behavior was undisturbed by the sound (Mark A. McDonald, Hildebrand, Webb, Dorman, & Fox, 1993).

Watkins (1986) reviewed data on the reactions of fin, humpback, right and minke whales that were exposed to continuous, broadband low-frequency shipping and industrial noise in Cape Cod Bay is informative. Watkins (1986) concluded that underwater sound was the primary cause of a reaction in these species of whales and that whales responded to acoustic stimuli in their range of hearing. Watkins also noted that whales showed the strongest reactions to sounds in the 15 Hz to 28 kHz range, although negative reactions (avoidance, interruptions in vocalizations, etc.) were generally associated with sounds that were either unexpected, too loud, suddenly louder or different, or perceived as being associated with a potential threat (such as an approaching ship on a collision course). In particular, whales seemed to react negatively when they were within 100 m of the source or when received levels increased suddenly in excess of 12 dB relative to ambient sounds. At other times, the whales ignored the source of the signal and all four species habituated to these sounds.

Nevertheless, Watkins concluded that whales ignored most sounds in the background of ambient noise, including the sounds from distant human activities even though these sounds may have had considerable energies at frequencies well within the whale's range of hearing. Further, Watkins (1986) noted that fin whales were initially the most sensitive of the four species of whales, followed by humpback whales; right whales were the least likely to be disturbed and generally did not react to low-amplitude engine noise. By the end of his period of study, Watkins

(1986) concluded that fin and humpback whales have generally habituated to the continuous, broad-band, noise of Cape Cod Bay while right whales did not appear to change their response.

This study covered a long enough period (the data covered 25 years of time) to provide some insight into possible long-term effects of low-frequency noise on whales, particularly since the four whale species would be exposed to continuous, low-frequency noise from shipping and other industrial sources. Given that whales in Cape Cod Bay reacted negatively to these continuous sources of anthropogenic sounds only under specific circumstances and, over time, habituated to these sounds (rather than abandon the area), it seems unlikely that an additional, intermittent signal lasting from 6 to 100 seconds that is designed to mimic background, low frequency sound would have a greater negative effect on at least, these species of whales (fin, humpback, right, and minke whales). The studies associated with the LFS Scientific Research Program suggest the same conclusions may also apply to blue, gray, and fin whales.

A recent study of a new low-frequency active sonar system that was being developed for use by the British Navy monitored the behavioral responses of marine mammals to the low-frequency active sonar (Aicken et al., 2005). During these trials, fin whales, sperm whales, Sowerby's beaked whales, long-finned pilot whales, Atlantic white-sided dolphins, and common bottlenose dolphins were observed and recorded vocalizing. These studies found no evidence of behavioral responses that could be attributed to the low-frequency active sonar during these trials (some of the responses that were observed may have been to the vessels used for the monitoring).

6.4.1.4.4 Allostasis

Classic stress responses begin when an animal's central nervous system perceives a potential threat to its homeostasis. That perception triggers stress responses regardless of whether a stimulus actually threatens the animal; the mere perception of a threat is sufficient to trigger a stress response (Moberg, 2000; R. M. Sapolsky, 2006; Seyle, 1950). Once an animal's central nervous system perceives a threat, it mounts a biological response or defense that consists of a combination of the four general biological defense responses: behavioral responses, autonomic nervous system responses, neuroendocrine responses, or immune response.

In the case of many stressors, the first and most economical (in terms of biotic costs) response is behavioral avoidance of the potential stressor or avoidance of continued exposure to a stressor. An animal's second line of defense to stressors involves the autonomic nervous system and the classical "fight or flight" response which includes the cardiovascular system, the gastrointestinal system, the exocrine glands, and the adrenal medulla to produce changes in heart rate, blood pressure, and gastrointestinal activity that humans commonly associate with stress. These responses have a relatively short duration and may or may not have significant long-term effect on an animal's welfare.

An animal's third line of defense to a stressor involves its neuroendocrine systems, usually hormones associated with the hypothalmus-pituitary-adrenal system (most commonly known as the HPA axis in mammals or the hypothalamus-pituitary-interrenal axis in fish and some reptiles). Unlike stress responses associated with the autonomic nervous system, virtually all neuroendocrine functions that are affected by stress – including immune competence, reproduction, metabolism, and behavior – are regulated by pituitary hormones. In the majority of stress studies, the TTS axis has been the primary neuroendocrine axis monitored. Stress-induced changes in the secretion of pituitary hormones have been implicated in failed reproduction (Moberg, 1987; Rivier, 1985) and altered metabolism (Elsasser, Klasing, Filipov, & Thompson, 2000), immune competence (Blecha, 2000) and behavior. Increases in the circulation of gluco-corticosteroids (cortisol, corticosterone, and aldosterone in marine mammals) have been equated with stress for many years.

The primary distinction between stress (which is adaptive and does not normally place an animal at risk) and distress is the biotic cost of the response. During stress an animal uses glycogen stores that can be quickly replenished once the stress is alleviated. In such circumstances, the cost of the stress response does not pose a risk to the animal's welfare.

However, when an animal has insufficient biotic reserves to satisfy the biotic cost of a stress response, then resources must be shifted away from other biotic functions. When sufficient reserves are diverted from these functions, the functions are impaired. For example, when stress shifts metabolism away from growth, young animals no longer thrive and growth is stunted. When energy is shifted from supporting reproduction, reproductive success is diminished.

In these cases, animals have entered a pre-pathological state and are experiencing "distress" (sensu Seyle, 1950) or "allostatic loading" (sensu McEwen & Wingfield, 2003). This period of distress will last until the animal replenishes its biotic reserves sufficient to restore normal function.

Relationships between these physiological mechanisms, animal behavior, and the costs of stress responses have also been documented fairly well through controlled experiment; because this physiology exists in every vertebrate that has been studied, it is not surprising that stress responses and their costs have been documented in both laboratory and free-living animals (for examples see, (Holberton, Helmuth, & Wingfield, 1996; Hood, Boersma, & Wingfield, 1998; Jessop, Tucker, Limpus, & Whittier, 2003; Lankford, Adams, Miller, & Cech, 2005).

No information has been collected on the physiological responses of marine mammals upon exposure to anthropogenic sounds, studies of other marine animals and terrestrial animals would lead us to expect some marine mammals to experience physiological stress responses and, perhaps, physiological responses that would be classified as "distress" upon exposure to mid-frequency and low-frequency sounds. However, when bowhead and gray whales were exposed to

various types of man-made noise, their surfacings become unusually brief with fewer blows per surfacing (Malme et al., 1989; W. John Richardson, Wells, et al., 1985; W. John Richardson et al., 1990; W. John Richardson, Würsig, & Greene Jr., 1986). These authors concluded that changes in surfacing, respiration, and diving behavior may be indicators of stress, although their consequences on the population ecology of the animals that are affected remain unknown.

Stress responses in response to sound exposures have been studied in other animal species, including humans. Jansen (1998) reported on the relationship between acoustic exposures and physiological responses that are indicative of stress responses in humans (for example, elevated respiration and increased heart rates). Jones (1998) reported on reductions in human performance when faced with acute, repetitive exposures to acoustic disturbance. Trimper et al. (1998) reported on the physiological stress responses of osprey to low-level aircraft noise while Krausman et al. (2004) reported on the auditory and physiology stress responses of endangered Sonoran pronghorn to military overflights. Smith et al. (2004; 2004) identified noise-induced physiological stress responses in hearing-specialist fish that accompanied short- (TTS) and long-term (PTS) hearing losses. Welch and Welch (1970) reported physiological and behavioral stress responses that accompanied damage to the inner ears of fish and several mammals.

Hearing is one of the primary senses cetaceans use to gather information about their environment and to communicate with conspecifics. Although empirical information on the relationship between sensory impairment (TTS, PTS, and acoustic masking) on cetaceans remains limited, it seems reasonable to assume that reducing an animal's ability to gather information about its environment and to communicate with other members of its species would be stressful for animals that use hearing as their primary sensory mechanism. Therefore, we assume that acoustic exposures sufficient to trigger onset PTS or TTS would be accompanied by physiological stress responses because terrestrial animals exhibit those responses under similar conditions (NRC, 2003b). More importantly, marine mammals might experience stress responses at received levels lower than those necessary to trigger onset TTS. Based on empirical studies of the time required to recover from stress responses (Moberg, 2000), we also assume that stress responses are likely to persist beyond the time interval required for animals to recover from TTS and might result in pathological and pre-pathological states that would be as significant as behavioral responses to TTS.

6.4.1.5 Responses of Baleen Whales

Like Richardson et al. (1995), we assume that baleen whales are very sensitive to low-frequency sounds. As a result, masking effects could be significant for many of these whales because they vocalize at low frequencies and are thought to have hearing that is sensitive at the SURTASS LFA sonar frequencies. This is especially true for those animals that use the same frequency bands as SURTASS LFA sonar. For example, Dahlheim et al. (1984) concluded that gray whales in the San Ignacio Lagoon, Baja, California shifted the frequencies of their vocalizations away from the predominant ambient noise producers in the lagoon to overcome masking effects.

In contrast, Biassoni et al. (2001) concluded that the intermittent sounds produced by LFA sonar were unlikely to mask humpback whale songs, but the similarities of its sounds to those produced by the whales could cause some humpback whales to perceive LFA sonar as a competing male. Based on their studies, they concluded that humpback whales could adapt to the presence of LFA sonar and concluded that singing whales could compensate for interference from sound sources like LFA sonar.

Acoustic masking from low frequency ocean noise is increasingly being considered as a threat, especially to low frequency hearing specialists such as baleen whales (C. W. Clark et al., 2009). Most underwater low frequency anthropogenic noise is generated by commercial shipping, which has contributed to the increase in oceanic background noise over the past 150 years (S. E. Parks et al., 2007). Shipping noise is primarily in the 20 to 200 Hz frequency band and is increasing yearly (D. Ross, 2005). Andrew et al. (2002) demonstrated an increase in oceanic ambient noise of 10 dB sound pressure level since 1963 in the 20 to 80 Hz frequency band as sampled on the continental slope off Point Sur, California, and they ascribed this increase to increased commercial shipping. McDonald et al. (2006) compared data sets from 1964 to 1966 and 2003 to 2004 for continuous measurements west of San Nicolas Island, California, and found an increase in ambient noise levels of 10 to 12 dB sound pressure level in the 30 to 50 Hz band. This increase in LF background noise is likely having a widespread impact on marine mammal low frequency hearing specialists by reducing their access to acoustic information essential for con-specific communication and other biologically important activities, such as navigation and prey/predator detection. Clark et al. (2009) considered this long-term, large-scale increase in low frequency background noise a chronic impact that results in a reduction in communication space, and the loss of acoustic habitat.

Parks et al. (2007) provided evidence of behavioral changes in the acoustic behaviors of the endangered North Atlantic right whale, and the South Atlantic right whale, and suggested that these were correlated to increased underwater noise levels. The study indicated that right whales might shift the frequency band of their calls to compensate for increased in-band background noise. The significance of their result is the indication of potential species-wide behavioral change in response to gradual, chronic increases in underwater ambient noise. DiLorio and Clark (2010) showed that blue whale calling rates vary in association with seismic sparker survey activity, with whales calling more on days with survey than on days without surveys. They suggested that the whales called more during seismic survey periods as a way to compensate for the elevated noise conditions.

Changes in behavior are not limited to low frequency species. Holt et al. (2009) measured killer whale call source levels and background noise levels in the 1 to 40 kHz band. The whales increased their call source levels by 1 dB for every 1 dB increase in background noise level. A similar rate of increase in vocalization activity was reported for St. Lawrence River belugas in response to passing vessels (Scheifele et al., 2005b).

6.4.1.6 SURTASS LFA Sonar Potential for Masking

Masking effects from SURTASS LFA sonar signals will be limited for a number of reasons. First, the bandwidth of any LFA sonar transmitted signal is limited (30 Hz), and the instantaneous bandwidth at any given time of the signal is small, on the order of ~10 Hz. Therefore, within the frequency range in which masking is possible, the effect will be limited because animals that use this frequency range typically use signals with greater bandwidths. Thus, only a portion of frequency band for the animal's signal is likely to be masked by the LFA sonar transmissions. Furthermore, when LFA sonar is in operation, the LFA sonar source is active only 7.5 to 10 percent of the time (based on historical LFA sonar operational parameters), which means that for 90 to 92.5 percent of the time there is no risk that an animal's signal will be masked by LFA sonar. Therefore, within the area in which energetic masking is possible, any effect of LFA sonar transmissions will be minimal because of the limited bandwidth and intermittent nature of the signal, and the fact that animals that use this frequency region typically produce signals with greater bandwidth that are repeated for many hours.

Hildebrand (2005) provided a comparison of anthropogenic underwater sound sources by their annual energy output. On an annual basis, four LFA sonar systems were estimated to have a total energy output of 6.8 x 1011 Joules/yr. Seismic airgun arrays and mid-frequency military sonars were two orders of magnitude greater, with an estimated annual output of 3.9 and 2.6 x 1013 Joules/year, respectively. Super tankers were greater at 3.7 x 1012 Joules/year. Hildebrand (2005) concluded that anthropogenic sources most likely to contribute to increased underwater noise in order of importance are: commercial shipping, offshore oil and gas exploration and drilling, and naval and other uses of sonar. The use of LFA sonar is not scheduled to increase beyond the originally analyzed four systems during the next five-year regulation period under the MMPA. The percentage of the total anthropogenic acoustic energy budget added by each LFA sonar source is estimated to be 0.21 percent per system (or less), when other man-made sources are considered (John. Hildebrand, 2005). When combined with the naturally occurring and other man-made sources of noise in the oceans, the intermittent LFA sonar signals barely contribute a measurable portion of the total acoustic energy.

6.4.1.7 Risk to Baleen Whales

Although the number of studies is limited, the available evidence suggests that at received levels below 180 dB, exposure to LFA sonar transmissions are not likely to result in injury, masking, stranding, resonance effects, or other behavioral effects in baleen whales. The best scientific and commercial data available suggest that LFA sonar transmissions could elicit short-term behavioral responses in whales, particularly the baleen whales. However, those responses are not known to have long-term, adverse consequences for the biology or ecology of the individual whales exposed to the LFA sonar signal. For example, the information available on bowhead whales, which have very sensitive hearing and are extremely sensitive to noise, suggests that bowhead whales will alter their migratory pathways to avoid industrial sound sources and may

reduce their calling rates (W. John Richardson, Charles R. Greene Jr., et al., 1995), although these reactions varied by season and ambient sound levels.

More importantly, the best scientific and commercial data available suggest that exposing individual baleen whales to SURTASS LFA sonar may cause those whales to react behaviorally, but those behavioral reactions are not likely to adversely affect the ability of the whales to forage, detect predators, select a mate, or reproduce successfully. We also would not expect these responses to be symptomatic of chronic stress that might depress a whale's immune responses and increase their susceptibility to disease. At received levels between 120 and 180 dB re: 1μPa, the information available would not lead us to expect baleen whales to have responses that significantly alter behavior for extended periods. As discussed previously, based on the performance of the SURTASS LFA sonar system and the mitigative measures the Navy uses with the sonar system over the past 12 years (2002-2012), we do not expect any baleen whales to be exposed to received levels equal to or greater than 180 dB.

Numerous studies of the ecology of populations have demonstrated the relationship between a population's reproduction (which includes fecundity schedules, age at maturity, and reproductive lifespan), numbers (which includes age- or stage-specific abundance and survival rates), or distribution (which includes the number of populations and sub-populations, immigration rates, and emigration rates), and a population's risk of extinction, summarized in Steans (1992), (see also Caswell, 1980; Caswell & Fujiwara, 2004; Gotelli, 2001; McEwen & Wingfield, 2003; Stark, Banks, & Vargas, 2004). Available information leads us to conclude that exposing baleen whales to LFA sonar transmissions might elicit short-term responses in individual whales but is not likely to adversely affect the fitness (longevity and reproductive success) of those individual whales to a degree that would reduce their reproduction, numbers, or distribution and, based on the evidence available, does not appear likely to predispose them to experience fitness consequences when exposed to other stressors in their environment.

Nevertheless, because of the many unknowns about the relationships between an animal's behavioral responses and their population ecology, the U.S. Navy and NMFS have interpreted any data or other information conservatively when dealing with the SURTASS LFA sonar program. As a result, when an animal's responses to LFA sonar transmissions appear to be part of the animal's normal behavioral repertoire (such as a tail flick or head turn), a low-level response to a novel stimulus (such as an alert response or a startle response), or a response with unknown consequence, the U.S. Navy and NMFS has chosen to treat these responses as symptomatic of potential disruptions one or more behavioral patterns that are essential to an individual animal's life history or to the animal's contribution to a population (that is, harassment).

6.4.1.8 Responses of Toothed Whales

The echolocation calls of toothed whales are subject to masking by high frequency sound. Human data indicate low frequency sound can mask high frequency sounds (i.e., upward masking). Studies on captive odontocetes by Au et al. (1993; 1985; 1974) indicate that some species may use various processes to reduce masking effects (e.g., adjustments in echolocation call intensity or frequency as a function of background noise conditions). Since echolocation calls occur at much higher frequencies than SURTASS LFA sonar, the extent of upward masking (i.e., low frequencies masking high frequencies) would be limited. There is also evidence that the directional hearing abilities of odontocetes are useful in reducing masking at the high frequencies used for echolocation, but not at the low-moderate frequencies used for communication (Zaitseva, Morozov, & Akopian, 1980).

Although low frequency hearing has not been studied in many odontocete species, those species that have been tested (beluga, killer whale, false killer whale, Risso's dolphin, and bottlenose dolphin) exhibit poor audiometric and behavioral sensitivity to low frequency sound. For sounds dominated by low frequency components, the maximum radius of audibility for most odontocete species may often be noise-limited when sensitivity is good, and sensitivity-limited when sensitivity is poor. At a maximum 20 percent duty cycle, it is anticipated that any masking of odontocetes would be temporary (i.e., at least 80 percent of the time an animal would be able to perceive incoming signals through low frequency sounds). The possibility of effective masking would only occur for environmental sounds that happen during the ping transmission (maximum 100 seconds) and are at, or at least close to, the frequencies in the 30-Hz-wide bandwidth signal, during the 10 seconds the SURTASS LFA sonar was transmitting in that bandwidth. As a result, the available evidence does not lead us to expect masking to directly reduce the reproduction, numbers, or distribution of threatened or endangered odontocetes or elicit behavioral responses that would reduce the reproduction, numbers, or distribution of these species.

6.4.1.9 Risk to Toothed Whales

If exposed to LFA sonar transmissions, the evidence available suggests that sperm whales and MHI Insular false killer whales, like other toothed whales, are not very sensitive to low-frequency sounds. Despite the limited number of studies, the available evidence suggests that the risk of injury, masking, stranding, resonance effects, or behavioral effects in sperm whales is very low. The best scientific and commercial data available suggests that exposing sperm whales or false killer whales to LFA sonar transmissions is likely to elicit short-term effects on these whales that are not known to have long-term, adverse consequences for the biology or ecology of the individual whales exposed to the LFA sonar signal. Therefore SURTASS LFA sonar might elicit short-term responses in individual sperm and Hawaii Insular false killer whales.

Nevertheless, because of the many unknowns about the relationships between an animal's behavioral responses and their population ecology, the U.S. Navy and NMFS have interpreted any data or other information conservatively when dealing with the SURTASS LFA sonar

program. As a result, when an animal's responses to LFA sonar transmissions appear to be part of the animal's normal behavioral repertoire (such as a tail flick or head turn), a low-level response to a novel stimulus (such as an alert response or a startle response), or a response with unknown consequence, the U.S. Navy and NMFS has chosen to treat these responses as symptomatic of potential disruptions one or more behavioral patterns that are essential to an individual animal's life history or to the animal's contribution to a population (that is, harassment).

6.4.1.10 Responses of Pinnipeds

Hearing capabilities and sound production is highly developed in all pinniped species studied to date. It is assumed that pinnipeds rely heavily on sound and hearing for breeding activities and social interactions (Schusterman, 1978; Van Parijs & Kovacs, 2002). Sensitivity to sounds at frequencies above 1 kHz has been well documented. However, there have been few studies on their sensitivity to low frequency sounds. Kastak and Schusterman (1998) suggest that the pinniped ear may respond to acoustic pressure rather than particle motion when in the water. Sound intensity level and the measurement of the rate of energy flow in the sound field was used to describe amphibious thresholds in an experiment studying low frequency hearing in two California sea lions, a harbor seal, and an elephant seal. Results suggest that California sea lions are relatively insensitive to most low frequency sound in the water, as sea lions have a higher hearing threshold (116 to 119 dB received level) at frequencies of 100 Hz. Harbor seals are approximately 20 dB more sensitive to signals at 100 Hz compared to California sea lions and thus are more likely to hear low frequency anthropogenic noise. Elephant seals are the most sensitive to low frequency sound underwater with a hearing threshold of around 90 dB received level at 100 Hz. Elephant seals also are deep divers, which may expose them to higher sound levels in the deep sound channel. Kastak and Schusterman (1996, 1998) also suggest that elephant seals may not habituate well to certain types of sound (in contrast to sea lions and harbor seals), but in fact may become more sensitive to disturbing noises and environmental features associated with the noises.

In a 2002 study, the California sea lion was most sensitive between approximately 2.5 and 10 kHz (David Kastak & Schusterman, 2002). Other otariid species (eared seals) with documented vocalizations are the South American sea lions and northern fur seals (Fernandez-Juricic, Campagna, Enriquez, & Ortiz, 1999; Insley, 2000). Otariid hearing abilities are thought to be intermediate between Hawaiian monk seal and other phocids (true seals), with a cutoff in hearing sensitivity at the high frequency end between 36 and 40 kHz. Underwater low frequency sensitivity is between approximately 100 Hz and 1 kHz. The underwater hearing of fur seals is most sensitive with detection thresholds of approximately 60 dB received level at frequencies between 4 and 28 kHz (W. John Richardson, Charles R. Greene Jr., et al., 1995).

Other sound experiments have shown some pinniped sensitivity to low frequency sound. Ringed, harbor, and harp seal audiograms show that they can hear frequencies as low as 1 kHz, with the

harp seal responding to stimuli as low as 760 Hz. Hearing thresholds of ringed, harbor and harp seals are relatively flat from 1 to 50 kHz with thresholds between 65 and 85 dB received level (Mohl, 1968) (Terhune & Ronald, 1972, 1976; Terhune & Ronald., 1985).

6.4.1.11 Risk to Pinnipeds

If exposed to LFA sonar transmissions, the evidence available suggests that Hawaiian monk seals and Steller sea lions may detect LFA sonar transmission. However, that exposure is not likely to kill or injure these pinnipeds or cause them to experience temporary or permanent shifts in hearing sensitivity. As a result, the best scientific and commercial data available suggests that exposing Hawaiian monk seals, Guadalupe fur seals, or Steller sea lions to LFA sonar transmissions is not likely to elicit short-term responses or responses that are known to have long-term, adverse consequences for the biology or ecology of these pinnipeds. Therefore exposing these species to LFA sonar is not likely to adversely affect the fitness (longevity and reproductive success) of individual members of these species and, based on the evidence available, does not appear likely to predispose them to experience fitness consequences when exposed to other stressors in their environment.

6.4.1.12 Potential Acoustic Masking

The obscuring of sounds of interest by interfering sounds, generally at similar frequencies is referred to as masking (W. John Richardson, Charles R. Greene Jr., et al., 1995). In humans, masking has been measured as an increase in detection threshold of the sound of interest in the presence of a masking sound (compared to the detection threshold when there is no masker). Two types of masking have been described: energetic masking and informational masking (Kidd et al., 2007; Pollack, 1975; Watson, 2005). The definitions of energetic and informational masking and their physiological mechanisms, however, continue to be debated. Energetic masking is thought to result from an interfering sound(s) within the same critical band(s) as the signal of interest. It is usually ascribed to peripheral acoustic processing; i.e., the ear itself. A definition for informational masking has been even less forthcoming, and as a default position, informational masking has often been taken to mean masking that is greater than would be predicted by energetic masking alone (Kidd et al., 2007). Informational masking is associated with uncertainty of the signal of interest (Watson, 2005) and is generally assumed to occur as a result of central neural processing that includes analytic (e.g., auditory stream segregation and discrimination) and attentive components (e.g., distraction) (Kidd et al., 2007). As a general statement, the more similar the characteristics (i.e., frequency band, duration) of a masking sound are to the sound of interest, the greater its potential for masking.

Acoustic masking from low frequency ocean noise is increasingly being considered as a threat, especially to low frequency hearing specialists such as baleen whales (C. W. Clark et al., 2009). Most underwater low frequency anthropogenic noise is generated by commercial shipping, which has contributed to the increase in oceanic background noise over the past 150 years (S. E. Parks et al., 2007). Shipping noise is primarily in the 20 to 200 Hz frequency band and is

increasing yearly (D. Ross, 2005). Andrew et al. (2002) demonstrated an increase in oceanic ambient noise of 10 dB sound pressure level since 1963 in the 20 to 80 Hz frequency band as sampled on the continental slope off Point Sur, California, and they ascribed this increase to increased commercial shipping. McDonald et al. (2006) compared data sets from 1964 to 1966 and 2003 to 2004 for continuous measurements west of San Nicolas Island, California, and found an increase in ambient noise levels of 10 to 12 dB sound pressure level in the 30 to 50 Hz band. This increase in LF background noise is likely having a widespread impact on marine mammal low frequency hearing specialists by reducing their access to acoustic information essential for conspecific communication and other biologically important activities, such as navigation and prey/predator detection. Clark et al. (2009) considered this long-term, large-scale increase in low frequency background noise a chronic impact that results in a reduction in communication space, and the loss of acoustic habitat.

6.4.1.13 Marine Mammal Behavioral Responses to Masking Sounds

Parks et al. (2007) provided evidence of behavioral changes in the acoustic behaviors of the endangered North Atlantic right whale, and the South Atlantic right whale, and suggested that these were correlated to increased underwater noise levels. The study indicated that right whales might shift the frequency band of their calls to compensate for increased in-band background noise. The significance of their result is the indication of potential species-wide behavioral change in response to gradual, chronic increases in underwater ambient noise. DiLorio and Clark (2010) showed that blue whale calling rates vary in association with seismic sparker survey activity, with whales calling more on days with survey than on days without surveys. They suggested that the whales called more during seismic survey periods as a way to compensate for the elevated noise conditions.

Changes in behavior are not limited to low frequency species. Holt et al. (2009) measured killer whale call source levels and background noise levels in the 1 to 40 kHz band. The whales increased their call source levels by 1 dB sound pressure level for every 1 dB sound pressure level increase in background noise level. A similar rate of increase in vocalization activity was reported for St. Lawrence River belugas in response to passing vessels (Scheifele et al., 2005b).

6.4.1.14 SURTASS LFA Sonar Potential for Masking

Masking effects from SURTASS LFA sonar signals will be limited for a number of reasons. First, the bandwidth of any LFA sonar transmitted signal is limited (30 Hz), and the instantaneous bandwidth at any given time of the signal is small, on the order of ~10 Hz. Therefore, within the frequency range in which masking is possible, the effect will be limited because animals that use this frequency range typically use signals with greater bandwidths. Thus, only a portion of frequency band for the animal's signal is likely to be masked by the LFA sonar transmissions. Furthermore, when LFA sonar is in operation, the LFA sonar source is active only 7.5 to 10 percent of the time (based on historical LFA sonar operational parameters), which means that for 90 to 92.5 percent of the time there is no risk that an animal's signal will be

masked by LFA sonar. Therefore, within the area in which energetic masking is possible, any effect of LFA sonar transmissions will be minimal because of the limited bandwidth and intermittent nature of the signal, and the fact that animals that use this frequency region typically produce signals with greater bandwidth that are repeated for many hours.

Hildebrand (2005) provided a comparison of anthropogenic underwater sound sources by their annual energy output. On an annual basis, four LFA sonar systems were estimated to have a total energy output of 6.8 x 1011 Joules/yr. Seismic airgun arrays and mid-frequency military sonars were two orders of magnitude greater, with an estimated annual output of 3.9 and 2.6 x 1013 Joules/year, respectively. Super tankers were greater at 3.7 x 1012 Joules/year. Hildebrand (2005) concluded that anthropogenic sources most likely to contribute to increased underwater noise in order of importance are: commercial shipping, offshore oil and gas exploration and drilling, and naval and other uses of sonar. The use of LFA sonar is not scheduled to increase beyond the originally analyzed four systems during the remaining period of the five-year regulations under the MMPA or in the reasonably foreseeable future. The percentage of the total anthropogenic acoustic energy budget added by each LFA sonar source is estimated to be 0.21 percent per system (or less), when other man-made sources are considered (John. Hildebrand, 2005). When combined with the naturally occurring and other man-made sources of noise in the oceans, the intermittent LFA sonar signals barely contribute a measurable portion of the total acoustic energy.

The recent research reviewed above provides no substantial changes to the knowledge or understanding for the potential of SURTASS LFA sonar to cause acoustic masking in marine mammals. Any masking in marine mammals due to narrowband, intermittent (low duty cycle) LFA sonar signal transmissions are expected to be minimal and unlikely.

6.4.1.15 Potential Stranding

Several mass strandings of cetaceans have been associated with military operations. Although the majority of these events did not involve endangered or threatened species and did not implicate SURTASS LFA sonar, we discuss them in this opinion in response to concerns raised by members of the public that SURTASS LFA sonar could be another cause of marine mammal stranding events. Naval maneuvers have been identified as the potential cause in cetacean strandings in at least two reports (Frantzis, 1998; M. P. Simmonds & Lopez-Jurado, 1991. These reports associated offshore naval operations with a mass stranding of Cuvier's beaked whales (Ziphius cavirostris) in the eastern Mediterranean Sea in 1996 {Frantzis, 1998 #2693) and mass strandings of Gervais' beaked whales (*Mesoplodon europaeus*), de Blainville's dense-beaked whales (*M. densirostris*), and Cuvier's beaked whales off the coast of the Canary Islands in the late 1980s (M. P. Simmonds & Lopez-Jurado, 1991).

Public concern about potential causal relationships between SURTASS LFA sonar and marine mammal stranding resurfaced after a beaked whale stranding in the Bahamas in 2000. Fox et al.

(2001) ruled out exposure to natural sounds as a possible cause of the stranding, which pointed to an anthropogenic source. In 2001, the Joint Interim Report, Bahamas Marine Mammal Stranding Event of 14-16 March 2000 (D. L. Evans & England, 2001) concluded that "tactical mid-range frequency sonars onboard U.S. Navy ships that were in use during the sonar exercise in question were the most plausible source of this acoustic or impulse trauma." The report also went on to conclude, "the cause of this stranding event was the confluence of Navy tactical mid-range frequency sonar and the contributory factors acting together." The factors that were assumed to contribute to the stranding event included "a complex acoustic environment that included the presence of a strong surface duct, unusual underwater bathymetry, intensive use of multiple sonar over an extended period of time, a constricted channel with limited access, and the presence of beaked whales that appear to be sensitive to the frequencies produced by these sonars." This report also stated that, "SURTASS LFA sonar, and other Navy sonar, had no involvement in this event."

6.4.1.16 Strandings Possibly Involving Military Sonar or Beaked Whales

Of the 27 mass stranding events that occurred globally from 2006 through early 2010, only two were possibly linked to military sonar transmissions with just one of those events involving beaked whales.

6.4.1.16.1 Spain (2006)

On January 26 through 27, 2006, four Cuvier's beaked whales were reported stranded along the southeast coast of Spain near Almeria in the western Mediterranean Sea. Of the four stranded beaked whales, two live-stranded while the remaining two whales were dead when discovered. All the whales ultimately died. Necropsies were performed on all four of the whales. Although the pathologists that conducted the necropsies concluded that anthropogenic acoustic activities were the likely cause of the whales stranding, no pathological results supporting this conclusion were ever presented, and no further documentation has been published.

A North Atlantic Treaty Organization (NATO) surface ship group (seven ships including one U.S. ship under NATO command) conducted active sonar training against a Spanish submarine target from January 25 through 26, 2006 in the Cartagena Exercise Area, which is located within 93 km (50 nmi) of the stranding sites. Although no definitive pathological or causal linkage between the naval exercises and the mass stranding has been documented, it appears likely that a confluence of factors such as: 1) the water depths in which the naval exercises occurred (1,000 m [3,281 ft] with steeply grading slope); 2) the multiple ships equipped with MFA sonar operating in proximity within the same area for a long duration (~20 hrs); and 3) the topography of the area in which deep water is surrounded by land masses that may have caused sound to be directed toward a channel or embayment, cutting off the whales' egress, may have contributed to the strandings of the Cuvier's beaked whales. As presented in Dolman et al. (2010), Fernandez (2006) concluded that the Almeria strandings were similar to previous atypical mass strandings

of beaked whales that were spatially and temporally associated with military naval sonar exercises, such as in the Bahamas (2000) and the Canary Islands (2002).

6.4.1.16.2 Cornwall, United Kingdom (2008)

On June 9, 2008, 26 common dolphins died after mass stranding in a small tidal tributary, Porth Creek, of the Fal Estuary in Cornwall, southwestern England. An even larger number of common dolphins were refloated and herded back into deeper water. In the days preceding the mass stranding, a large group(s) of dolphins was observed very close to shore. All of the dead stranded dolphins were necropsied; and detailed pathological, histological, and other diagnostic testing was conducted, as was an investigation of the area, environmental conditions, and interviews with witnesses and responders.

An international naval exercise was conducted in the South Coast Exercise Area, located off the south coast of Cornwall, Devon, and Dorset, from 1 through 9 June, 2008 with peak activity on 4 to 5 June. The naval exercise involved up to 20 Royal Navy (United Kingdom) surface and submarine vessels as well as 11 international ships (P.D. Jepson & Deaville, 2009). The joint exercise involved the use of several acoustic sources, including MFA (2 to 8 kHz) sonar, standard echosounders, acoustic modems, sonobuoys, high-frequency (100 kHz) side-scan sonar; the firing of inert and live ammunition and at least one SEAWOLF missile; and helicopter and fixed-wing aircraft flights. No helicopter or fixed-wing flights occurred over the area of the mass stranding. The MFA sonars were employed at least 45 to 50 km (24 to 27 nmi) from the stranding location. Approximately 60 hours lapsed between the end of MFA sonar transmissions and the mass stranding event.

The results of the investigation of this mass stranding event were reported by Jepson and Deaville (2009); the pathological and other analysis results were presented with no finding of significant infectious disease, contaminants, biotoxins, or acute physical injury in the dead dolphins. The ears of all the dolphins were normal with no damaged tissue. Jepson and Deaville (2009) concluded that the following potential causes for the stranding could be excluded or were considered highly unlikely to have caused the mass stranding: infectious disease, fat or gas embolisms (decompression sickness), boat strike, fisheries bycatch, predation, feeding unusually close to shore, ingestion of biotoxins or harmful contaminants, abnormal weather conditions, and high-intensity underwater acoustic sound from airguns or earthquakes. While no definitive cause could be identified for the mass stranding event, the investigation did conclude that an adverse behavioral reaction to some specific trigger or stimuli within a group of healthy dolphins resulted in the mass stranding and death of the 26 common dolphins (P.D. Jepson & Deaville, 2009). The investigation also noted that the dolphin's unusual proximity to shore prior to the mass stranding, or a combination of factors including errors in navigation and other natural or anthropogenic factors, could have led to an increased risk of stranding. While the investigators did acknowledge that the use of the MFA sonar could have led to the dolphins being closer to shore than normal,

they considered it highly unlikely that the MFA sonar directly triggered the mass stranding event (P.D. Jepson & Deaville, 2009).

6.4.1.17 Risk of Stranding

The use of SURTASS LFA sonar was not associated with any of the reported 27 mass stranding events or unusual mortality events that occurred globally between 2006 and early 2010. There is no evidence that LFA sonar transmissions resulted in any difference in the stranding rates of marine mammals in Japanese coastal waters adjacent to LFA sonar mission areas. As has been reported previously (Navy, 2001a, 2001b, 2007c, 2012a), the employment of LFA sonar is not expected to result in any sonar-induced strandings of marine mammals. Given the large number of natural factors that can result in marine mammal mortality, the high occurrence of marine mammal strandings, and the many years of LFA sonar operations without any reported associated stranding events, the likelihood of LFA sonar transmissions causing marine mammals to strand is negligible.

6.4.2 Sea Turtles

There are very few studies of the potential effects of underwater sound on sea turtles and most of these examined the effects of sounds of much longer duration or of different types (e.g., seismic airgun) than the SURTASS LFA sonar signals. The analysis of the potential effects on sea turtle species is summarized based on the following SURTASS LFA sonar operational parameters:

- Small number of SURTASS LFA sonar systems to be deployed;
- Geographic restrictions imposed on system employment;
- Narrow bandwidth of the SURTASS LFA sonar active signal (approximately 30 Hz);
- Slowly moving ship, coupled with low system duty cycle, would mean that a sea turtle would spend less time in the LFA sonar mitigation and buffer zones (180-dB sound pressure level sound field); therefore, with a ship speed of less than 5 kt, the potential for animals being in the sonar transmit beam during the estimated 7.5 to 10 percent of the time the sonar is actually transmitting is very low; and
- Small size of the LFA sonar mitigation zone (180-dB sound pressure level sound field) relative to open ocean areas.

Due to the lack of more definitive data on sea turtle stock distributions in the open ocean, it is not feasible to estimate the percentage of a stock that could be located in a SURTASS LFA sonar operations area at a potentially vulnerable depth, during an LFA sonar sound transmission. Data on sea turtle sound production and hearing are very limited.

6.4.2.1 Non-Auditory Injury

There are no data on the potential for anthropogenic sound to cause injury in sea turtles. Although not directly related to SURTASS LFA sonar effects, a review of effects of explosives on turtles was done by Viada et al. (2008b). For explosive structure removals in the Gulf of Mexico, NMFS specified that the area within 3,000 ft (915 m) of the platform must be clear of

sea turtles. Therefore, using a value of 180-dB sound pressure level injury threshold for sea turtles (within approximately 1,000 m [3,281 ft] of the LFA sonar array) is conservative. The probability of a sea turtle being within the 180-dB mitigation zone is considered negligible because of the mitigation measures employed during sonar operations and the five SURTASS LFA sonar operational parameters listed above.

6.4.2.2 Auditory Injury - Permanent Hearing Loss

Very little is known what may cause a sea turtle to incur permanent loss of hearing. However, data support the premise that using a value of 180-dB injury threshold for sea turtles is conservative. A sea turtle would have to be within the LFA sonar mitigation zone (≥180 dB re: 1 μPa_{rms} received level) when the sonar was transmitting to be at risk of injury, including permanent loss of hearing (i.e., PTS). Despite the lack of scientific data on the potential effects of low frequency sound on sea turtle hearing and on PTS in sea turtles caused by low frequency sound, the potential for SURTASS LFA sonar to cause PTS in sea turtles can be considered negligible.

6.4.2.3 Temporary Hearing Loss

As with PTS, there are no published scientific data on temporary loss of hearing in sea turtles caused by low frequency sound. Further, the five SURTASS LFA sonar operational parameters listed above support the conclusion that the potential for SURTASS LFA sonar to cause TTS in sea turtles can be considered to be negligible.

6.4.2.4 Behavioral Changes

Sea turtles can travel many kilometers per day in the open ocean, as shown in tagging studies (Luschi, Hays, & Papi, 2003; Papi, Liew, Luschi, & Chan, 1995); and the use of magnetic positional information for long-range navigation has been demonstrated in several diverse animals, including sea turtles (Benhamou et al., 2011; Lohmann, Putman, & Lohmann, 2012). Sea turtles make extensive migrations and movements either for foraging opportunities or to breed. Their migration tracks may extend to thousands of kilometers (M. S. Allen, 2007; Arendt et al., 2012; Luschi et al., 2006; Shillinger et al., 2008).

Changes in movement patterns or other behaviors due to exposure to a high intensity sound source that causes prolonged displacement of animals from the site of their normal activities could be considered a deleterious effect. Displacement can occur in two dimensions: vertical and horizontal. For example, a sea turtle could move to the surface, where anthropogenic low frequency sound would be weaker, possibly exposing it to a higher degree of predation. As for horizontal displacement, this is probably of greatest importance for non-pelagic sea turtle species (green, olive ridley, hawksbill), for which displacement from preferred benthic habitats could be construed as more serious.

Behavioral responses to human activity have been investigated for only a few species of sea turtles: green and loggerhead (McCauley et al., 2000; J. O'Hara & Wilcox, 1990); and olive

ridley, leatherbacks, loggerhead, and 160 unidentified turtles (hard-shell species) (C. W. Weir, 2007). The work by O'Hara and Wilcox (1990) and McCauley et al. (2000) reported behavioral changes of sea turtles in response to seismic airguns. O'Hara and Wilcox (1990) reported avoidance behaviors by loggerheads in response to airguns with sound levels (received level) of 175 to 176 dB re: 1 µPa (peak-to-peak). McCauley et al. (2000) reported noticeable increases in swimming behavior for both green and loggerhead turtles at received levels of 166 dB re: 1 µPa (peak-to-peak). At 175 dB re: 1 µPa (peak-to-peak) received level, both green and loggerhead sea turtles displayed increasingly erratic behavior (McCauley et al., 2000).

Based on the hearing data, it is possible that if a sea turtle happened to be in proximity of a SURTASS LFA sonar operations area, it will hear the low frequency transmissions. Given that the majority of sea turtles encountered would probably be transiting in the open ocean from one site to another, the possibility of significant displacement would be unlikely.

6.4.2.5 Masking

One critical question to ask is whether there are sufficient anthropogenic sounds in the normal environment of sea turtles to suggest that hearing might be masked. While no masking studies on marine turtles have been conducted, an indirect study looked at the potential for masking by examining sounds in an area known to be inhabited by turtles. These underwater sound recordings were made in one of the major coastal foraging areas for juvenile sea turtles (mostly loggerhead, Kemp's ridley and green sea turtles) in the Peconic Bay Estuary system in Long Island, NY (Samuel, Morreale, Clark, Greene, & Richmond, 2005). The recording season of the underwater environment coincided with the sea turtle activity season in an inshore area where there is considerable boating and recreational activity, especially during the July to September timeframe. During this time period, received levels at the data collection hydrophone system in the 200 to 700 Hz band ranged from 83 dB re: 1 μPa_{rms} (night) up to 113 dB re: 1 μPa_{rms} (weekend day). Therefore, during much of the season when sea turtles are actively foraging in New York waters, they are undoubtedly exposed to these levels of noise, most of which is anthropogenic. However, there were no data collected on any behavioral changes in the sea turtles as a consequence of anthropogenic noise or otherwise during this study, so it cannot be stated whether this level of ambient sound would have any physiological and/or behavioral effects on the sea turtles.

Masking effects may occur for sea turtle species that have critical hearing bandwidths at the same frequencies as the SURTASS LFA sonar. However, masking would probably be temporary. The geographical restrictions imposed on all SURTASS LFA sonar operations would limit the potential for masking of sea turtles in the vicinity of their nesting sites. In summary, masking effects are not expected to be significant because of the nominal 7.5 percent duty

cycle[10], the maximum 100-sec signal duration, the fact that the ship is always moving, the limited 30 Hz sonar bandwidth, and the signals not remaining at a single frequency for more than ten seconds.

6.4.3 Responses of Sea Turtles

Data on sound production and hearing in sea turtles is very limited. There is little known about the mechanism of sound detection by turtles, including the pathway by which sound gets to the inner ear and the structure and function of the inner ear of sea turtles (Moein Bartol & Musick, 2003). The limited information available suggests that the auditory capabilities of sea turtles are centered in the low-frequency range (<1 kHz) (Soraya Moein Bartol, Musick, & Lenhardt, 1999a; M. L. Lenhardt, Bellmund, Byles, Harkins, & Musick, 1983; M.L. Lenhardt, Moein, Musick, & Barnard, 1994; J. O'Hara & Wilcox, 1990; Ridgway et al., 1969).

The Navy's Supplemental EIS referred to studies conducted by Streeter and colleagues on a female green sea turtle that had been trained to respond to acoustic signals (Navy, 2007c). Those authors established that this sea turtle had a hearing range of at least 100 to 500 Hz (the maximum frequency that could be used in the study, as opposed to what may be a wider hearing range) with hearing thresholds of 120-130 dB received level. We could not be certain whether and to what degree the results of this study might apply to other sea turtles, particularly sea turtles in the wild. First, only one individual was studied and that individual was an older individual that had lived its entire life in captivity. Second, the study was conducted at the New England Aquarium, which is a relatively noisy environment. As a result, the thresholds reported may have been masked by the background noise and the "absolute thresholds" (the lowest detectable signal within a noisy environment) may be several dB lower than the reported results.

McCauley et al. (2000) studied the response of green and loggerhead sea turtles to air-gun arrays at 2 km and at 1 km with received levels of 166 dB re: 1 μPa and 175 db re: 1 μPa, respectively. They reported that the sea turtles responded consistently above received levels of about 166 dB re: 1 μPa: they increased their swimming activity compared to periods during which the airgun was not operating. Above 175 dB re: 1 μPa their behavior became erratic and might have indicated an agitated state.

O'Hara and Wilcox (1990) exposed loggerhead sea turtles to low-frequency sound from two types of seismic airguns with source levels approximating 256 and 262 dB re: 1μPa (the paper does not state whether this was peak, peak-to-peak, or root mean square). With airguns firing at a rate of four times per minute and source levels of 256 dB, the distribution of the loggerhead sea turtles was not significantly different than during control trials. When they increased source levels to about 262 dB, the sea turtles avoided the seismic airgun when it fired at four or eight

[10] Average duty cycle (ratio of sound "on" time to total time) of the SURTASS LFA sonar active transmission mode is less than 20 percent. The typical duty cycle, based on historical LFA sonar operational parameters since 2003 is nominally 7.5 to 10 percent. During the remaining 80 to 92.5 percent of the time, LFA sonar transmitters would be off, thus adding no sound to the water.

times per minute. Although the sound source in this experiment is not directly comparable to the low-frequency active sonar associated with the proposed operations of the SURTASS LFA sonar system, the results of this study illustrate the relative insensitivity of sea turtles (at least loggerhead sea turtles) to low-frequency sound.

6.4.4 Risk to Sea Turtles

Based on the limited evidence available, sea turtles exposed to received levels of active low-frequency sonar may hear the sound. At levels below 175 dB re: 1 μPa, they may not react at all or they may increase their swimming speed. Any changes in their swimming behavior would not result in a measurable risk to individual sea turtles.

6.4.4.1 Effects of the HF/M3 Sonar

The source level required for the HF/M3 sonar to effectively detect marine mammals (and possibly sea turtles) out to the 180-dB LFA sonar mitigation and buffer zones under the most adverse oceanographic conditions (low echo return and high ambient noise) is on the order of 220 dB. The Navy designed the HF/M3 sonar to be as benign as possible within the marine environment in order to minimize potential effects to marine mammals and sea turtles. These features include:

(1) The HF/M3 sonar source frequency is >30 kHz, which pushes its frequency band well away from the best hearing bandwidth of mysticetes, pinnipeds, and sea turtles, but within the best hearing bandwidth of odontocetes;

(2) A duty cycle that is variable, but usually below 10 percent;

(3) A maximum HF/M3 sonar pulse with a duration of 40 milliseconds (msec). As noted above, at received levels below 175 dB re: 1 μPa (rms) sea turtles would exhibit little reaction. For a 30 kHz, 40-msec pulse, the estimated range from the HF/M3 sonar of 175 dB response level would be less than 60 m (192 ft); and

(4) A transmission loss that is very high because of the high frequency of the sound source.

In addition, as supplementary safety measures, the following operational procedures would be applied to operation of the HF/M3 sonar:

(1) The HF/M3 sonar source level would be ramped up over a five-minute period to alert a marine animal that was close to the sonar and provide it time to move away from the sound source; and

(2) The HF/M3 sonar source level would not be increased once a marine mammal was detected. Once the marine animal is no longer detected by the HF/M3 sonar, passive, or visual monitoring, the HF/M3 sonar ramp-up may resume.

The application of these operational procedures reduces potential impacts of the HF/M3 sonar on marine mammals and sea turtles to negligible levels.

6.4.5 Interaction of SURTASS LFA Sonar and Other Sound Sources

Several investigators and organizations have expressed concern about the "cumulative impact" (in the NEPA sense of the term) of marine sounds on the ocean environment and its organisms. Any man-made sound that is strong enough to be audible (detectable above natural background noise) will increase total background levels and could interfere with an animal's ability to detect sound signals if the signal is weak relative to total noise levels. Concern about the cumulative impact of man-made sounds focuses on impacts from individual actions that are insignificant or minor when considered in isolation, but combine to produce effects that are greater than any individual action (either because the effects are synergistic - effects that occur when two or more phenomena interact - multiplicative, or additive). In this opinion, our assessment has focused on the effect of adding mid-frequency sonar and low-frequency active sonar to underwater ambient noise levels during the same series of exercises.

Richardson et al. (1995) provided extensive information and arguments about the potential cumulative effects of man-made noise on marine mammals. Those effects included masking, physiological effects and stress, habituation, and sensitization. Those concerns were echoed by Clark and Fristrup (2001b), The National Research Council (NRC, 2003b), the National Resources Defense Council (Jasny, Reynolds, Horowitz, & Wetzler, 2005), and others. Although all of these responses have been measured in terrestrial animals reacting to airborne, man-made noises, those studies are counterbalanced by studies of other terrestrial mammals that did not exhibit these responses to similar acoustic stimuli.

Broadband, continuous low-frequency shipping noise is more likely to affect marine mammals than narrowband, low duty cycle SURTASS LFA sonar. Moreover, SURTASS LFA sonar bandwidth is limited (approximately 30 Hz), the average maximum pulse length is 60 seconds, signals do not remain at a single frequency for more than 10 seconds, and during an operation the system is off nominally 90 to 92.5 percent of the time. Most mysticetes vocalizations are in the low frequency band below 1 kHz. No direct auditory measurements have been made for any mysticete, but it is generally believed that their frequency band of best hearing is below 1,000 Hz, where their calls have the greatest energy (Christopher W Clark, 1990; Darlene R Ketten, 2000). However, with the nominal duty cycle of 7.5 to 10 percent, masking would be temporary. For these reasons, any masking effects from SURTASS LFA sonar are expected to be negligible and extremely unlikely.

Odontocetes have a broad acoustic range and hearing thresholds measure between 400 Hz and 100 kHz (Finneran, Schlundt, Dear, et al., 2002; W. John Richardson, Charles R. Greene Jr., et al., 1995). It is believed that odontocetes communicate above 1,000 Hz and echolocate above 20 kHz (Wursig & Richardson, 2009). While the upward spread of masking is known to exist, the

phenomenon has a limited range in frequency. Kidd (2007) showed that magnitude of the masking effect decreases as the difference between signal and masking frequency increase; i.e., the masking effect is lower at 3 times the frequency of the masker than at 2 times the frequency. Gorga et al. (2002) demonstrated that for a 1.2-kHz masking signal, the upward spread of masking was extinguished at frequencies of 6 kHz and higher. Therefore, while the phenomenon of upward spread of masking does exist, it is unlikely that LFA sonar would have any significant effect on the hearing of higher frequency animals. Gorga et al. (2002) also demonstrated that the upward spread of masking is a function of the received level of the masking signal. Therefore, a large increase in the masked bandwidth due to upward masking would only occur at high received levels of the LFA sonar signal.

Hildebrand (2005) provided a comparison of anthropogenic underwater sound sources by their annual energy output. On an annual basis, four LFA sonar systems were estimated to have a total energy output of 6.8 x 1011 Joules/yr. Seismic airgun arrays and mid-frequency military sonars were two orders of magnitude greater, with an estimated annual output of 3.9 and 2.6 x 1013 Joules/year, respectively. Super tankers were greater at 3.7 x 1012 Joules/year. Hildebrand (2005) concluded that anthropogenic sources most likely to contribute to increased underwater noise in order of importance are: commercial shipping, offshore oil and gas exploration and drilling, and naval and other uses of sonar. The use of LFA sonar is not scheduled to increase beyond the originally analyzed four systems during the next five-year regulation period under the MMPA. The percentage of the total anthropogenic acoustic energy budget added by each LFA sonar source is estimated to be 0.21 percent per system (or less), when other man-made sources are considered (John A. Hildebrand, 2005). When combined with the naturally occurring and other man-made sources of noise in the oceans, the intermittent LFA sonar signals barely contribute a measurable portion of the total acoustic energy.

In a recently released report entitled “Ad-Hoc Group on the Impact of Sonar on Cetaceans,” the International Council for the Exploration of the Sea (ICES, 2005) concluded that shipping accounts for more than 75 percent of all human sound in the sea, and sonar amounts to no more than 10 percent or so. It further stated that sonar (noise budget) will probably never exceed 10 percent, but that sonar deployment seems likely to increase in the future.

In some instances, one or more of the vessels equipped with SURTASS LFA sonar would participate in Navy training exercises that involve anti-submarine warfare and mid-frequency active sonar. During these exercises, sound fields produced by LFA sonar might co-occur with sound fields generated by mid-frequency active sonar or animals might be exposed to LFA sonar and mid-frequency active sonar in a short time interval. Potential interactions between LFA sonar and other sonars were analyzed on the Navy’s FSEIS/SOEIS (Navy, 2012a). Although the SURTASS LFA sonar and mid-frequency active (MFA) sonars (AN/SQS 53C and AN/SQS 56) are similar in the underlying transmission types, specifically frequency-modulated (FM) sweeps and continuous wave (CW) transmissions, LFA sonar and MFA sonars are dissimilar in other

respects. In addition to these multiple differences, the duty cycle, (i.e., the amount of time during sonar operations that the sonar is actually transmitting), is different for SURTASS LFA sonar as opposed to MFA sonar. During SURTASS LFA sonar operations, LFA sonar transmits approximately 10 percent of the time (1 minute out of 10). During MFA sonar operations, MFA sonar transmits approximately 1.7 percent of the time (1 second out of 60). This means that for any given period of time that both SURTASS LFA sonar and MFA sonars are operating concurrently, the LFA sonar 60-sec transmission will be overlapped by 1 sec of MFA transmission, or 1.7 percent of the 60-sec LFA sonar ping (1 sec/60 sec). During the 10-min LFA sonar transmission cycle, the most an animal could be simultaneously exposed from both transmissions is 1 sec for every 600 sec, or about 0.17 percent of the time that both sonars are operating.

The ocean volumes of Level A harassment received levels for each source are relatively small (1 km [0.54 nmi] radius or less). It is not reasonably foreseeable that SURTASS LFA sonar and MFA sonars would operate simultaneously within ranges less than 9.3 km (5 nmi). Thus, it is not reasonably foreseeable that the Level A harassment volumes of the two sonars could ever overlap during simultaneous transmissions.

The results of two separate analysis methodologies, parametric analysis and underwater acoustic model analysis, were consistent—concurrent MFA/SURTASS LFA sonar operations produce no Level B harassment risk greater than that obtained by simply adding the risks from the individual sources. Therefore, two separate analytic approaches have concluded that there is no potential increase in risk for Level B harassment from concurrent MFA/SURTASS LFA sonar operations. Thus, the potential for this occurring is small and should be considered very conservative. See FSEIS/SOEIS (Navy, 2012a) Subchapter 4.7.4 and Appendix E for additional detail.

Because of the potential for seismic surveys to interfere with the reception of passive signals and return echoes, SURTASS LFA sonar operations are not expected to be close enough to these activities to have any synergistic effects. Because of the differences between the LFA sonar coherent signal and seismic air gun impulsive “shots,” there is little chance of producing a “synergistic” sound field. Marine animals would perceive these two sources of underwater sound differently and any addition of received signals would be insignificant. This situation would present itself only rarely, as LFA sonar testing and training operations have not been, and are not expected to be conducted in proximity to any seismic survey activity.

If SURTASS LFA sonar operations were to occur concurrent with other military and commercial sonar systems, synergistic effects are not probable because of differences between these systems. In order for the sound fields to converge, the multiple sources would have to transmit exactly in phase (at the same time), requiring similar signal characteristics, such as time of transmissions, depth, frequency, bandwidth, vertical steering angle, waveform, wavetrain, pulse length, pulse repetition rate, and duty cycle. The potential for this occurring is small.

Therefore, because of major differences in signal characteristics between LFA sonar, MFA, and seismic air guns, there is negligible chance of producing a "synergistic" sound field. It is also unlikely that LFA sonar sources, if operated in proximity to each other would produce a sound field so complex that marine animals would not be able to escape.

6.5 Cumulative Effects

"Cumulative effects" are those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation (50 CFR § 402.02). Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the Act.

During this consultation, we searched for information on future State, tribal, local, or private actions that were reasonably certain to occur in the action area. Most of the action area includes federal military reserves or is outside of territorial waters of the United States of America, which would preclude the possibility of future state, tribal, or local action that would not require some form of federal funding or authorization. We conducted electronic searches of business journals, trade journals, and newspapers using *First Search*, Google, and other electronic search engines. Those searches produced no evidence of future private action in the action area that would not require federal authorization or funding and is reasonably certain to occur. As a result, we are not aware of any actions of this kind that are likely to occur in the action area during the foreseeable future.

The potential accumulative impacts of these "take" estimates include (1) impacts or effects that accumulate in the environment in the form of stressors or combinations of stressors and (2) impacts or effects that represent either the response of individuals, populations, or species to that accumulation of stressors in the environment or the accumulated responses of individuals, populations, and species to sequences of exposure to stressors.

In the sense of Item 1, which captures the normal usage of "cumulative impacts," we concluded that phenomena like sound do not accumulate, although phenomena like the acreage of habitat destroyed and concentrations of toxic chemicals, sediment, and other pollutants accumulate. In the sense of Item 2, we considered phenomena that accumulate in individuals and individually contribute or collectively determine the probable fitness of the individuals that comprise a population. These include, the passage of time and its corollary, the passage or loss of time (specifically, the loss of time to reproduce, to forage, and to migrate, etc.); reproductive success; longevity; energy debt, including allostatic loading; body burdens of toxic chemicals; the fitness costs of behavioral decisions (canonical costs); and injuries and tissue damage; overstimulation of sensory organs (which would include noise-induced losses of hearing sensitivity).

At the level of populations, phenomena that "accumulate" include reproductive success; the number or percent of individuals that emigrate from a population per unit time; the number or percent of individuals that immigrate into a population per unit time; mortality within a particular age or stage over generation time; and the reservoir of juveniles in a population that have a high probability of surviving to the age of reproduction (population momentum or its absence).

At the species level, we would accumulate those phenomena that would allow us to estimate the extinction risks facing a species. These include increases or decreases in the number of occurrences or populations; the extinction probability of particular occurrences; variance in the rates of population growth or decline; and demographic stochasticity.

Based on our consideration of these phenomena, we concluded that one of the four primary stressors — the probability of a ship strike — accumulated, in the sense that the probabilities of collisions associated with multiple transits are higher than the probabilities associated with a single transit. We factored those considerations into our estimation of the probability of a collision associated with multiple transits. We concluded that the passive sonar (SURTASS) was not a stressor because it does not add any energy to the environment. Otherwise, we concluded that two of the primary stressors (LFA sonar and HF/M3 sonar) associated with the U.S. Navy testing, training and operation of SURTASS LFA sonar do not accumulate in either of the two senses of we discussed earlier in this section. Specifically, we concluded that the effects of multiple exposures to active sonar were not likely to accumulate through altered energy budgets caused by avoidance behavior (reducing the amount of time available to forage), physiological stress responses (mobilizing glucocorticosteroids, which increases an animal's energy demand), or the canonical costs of changing behavioral states (small decrements in the current and expected reproductive success of individuals exposed to the stressors) because these costs primarily occur because of avoidance behavior and altered energy budgets. In particular, we concluded that listed species would be exposed on foraging areas or migration routes where trivial increases in feeding duration, effectiveness, or transit, would eliminate the costs of these phenomena on the individuals that might be exposed.

The number of individuals "taken" gets larger when we accumulative them through addition, but the effect of that "take" on the survival or reproductive success of the animals themselves would not accumulate in the same way. As a result, we did not expect that the instances of exposing whales to SURTASS LFA sonar or HF/M3 sonar each year or over five years would result in effects that would be greater than we would expect from a single exposure event. To the contrary, we did not expect the effects of the "take" to have any additive, interactive, or synergistic effect on the individual animals, the population(s) those individuals represent, or the species those population(s) comprise.

6.6 Potential for Long-Term or Additive Impacts

In order to address a recent Court ruling that we take a long-term view of the "action," we assumed for purposes of this Opinion that the Navy's SURTASS LFA training, testing, and operational activities and associated impacts will continue into the reasonably foreseeable future at the levels assessed in this Opinion. This assumption raises the question of whether the Navy's activities are reasonably certain to cause any aggregate or long-term impacts over time, beyond the effects of individual takes that could occur in a given year. Further information is provided below.

To address the likelihood of long-term additive or accumulative impacts, we first considered (1) impacts or effects that accumulate in the environment in the form of stressors or reservoirs of stressors and (2) impacts or effects that represent either the response of individuals, populations, or species to that accumulation of stressors in the environment or the accumulated responses of individuals, populations, and species to sequences of exposure to stressors.

In regards to Item 1, which captures the normal usage of "cumulative impacts," we concluded that phenomena like sound do not accumulate like other phenomena, such as acreage of habitat destroyed and concentrations of toxic chemicals, sediment, and other pollutants, tend to accumulate[11]. In regards to Item 2, we considered phenomena that accumulate in individuals and individually contribute or collectively determine the probable fitness of the individuals that comprise a population. These include: the passage of time and its corollary, the loss of time (specifically, the loss of time to reproduce, to forage, and to migrate, etc.); reproductive success; longevity; energy debt, including allostatic loading; body burdens of toxic chemicals; the fitness costs of behavioral decisions (canonical costs); injuries and tissue damage; and overstimulation of sensory organs (which would include noise-induced losses of hearing sensitivity).

Based on our consideration of these phenomena, we concluded that one of the three primary stressors — the probability of a ship strike — accumulated, in the sense that the probabilities of collisions associated with multiple transits are higher than the probabilities associated with a single transit. We factored those considerations into our estimation of the probability of a collision associated with multiple transits. Even with the consideration that the probability of ship strike accumulated over time, based on the relatively low density of animals and the low number of training exercises, in combination with the information that no ship strikes involving Naval vessels have occurred in the action area, we concluded that this stressor remains discountable.

Otherwise, we concluded that two of the three primary stressors associated with the U.S. Navy training, active sonar and underwater detonations, do not accumulate in either of the two ways discussed earlier in this section. Specifically, we concluded that the effects of multiple exposures to active sonar or underwater detonations were not likely to accumulate through altered energy

[11] These example phenomena are not a concern in this Opinion.

budgets caused by avoidance behavior (reducing the amount of time available to forage), physiological stress responses (mobilizing glucocorticosteroids, which increases an animal's energy demand), or the canonical costs of changing behavioral states (small decrements in the current and expected reproductive success of individuals exposed to the stressors). In particular, we concluded that the species would be exposed on foraging areas and would experience trivial increases in feeding duration, effectiveness, or both, that would not accumulate in a manner that is likely to result in avoidance behavior or altered energy budgets. In short, the vast majority of impacts expected from sonar exposure and underwater detonations are behavioral in nature, temporary and comparatively short in duration, relatively infrequent, and not of the type or severity that would be expected to be additive for the small portion of the stocks and species likely to be exposed either annually or over the remaining period of the five-year MMPA regulations or in the reasonably foreseeable future.

Under certain conditions, chronic exposure to acoustic sources or other stimuli that can cause individual stress or behavioral responses can also lead to additional long-term adverse impacts. For example, investigators concluded that gray whales and humpback whales abandoned some of their coastal habitat in California and Hawai'i, respectively, because of persistent underwater noise associated with extensive vessel traffic (Gard 1974, Reeves 1977, Salden 1988). Another study of terrestrial mammals suggests that while short-term stress responses are often beneficial, conditions of chronic or long-term stress can lead to adverse physiological effects (Romero, et al., 2007). However, the Navy's SURTASS LFA sonar activities involving low frequency active sonar are infrequent and short-term. Even though an animal's exposure to low frequency active sonar may be more than one time, the intermittent nature of the sonar signal, its low duty cycle, and the fact that both the vessel and animal are moving provide a very small chance that exposure to active sonar for individual animals and stocks would be repeated over extended periods of time. Consequently, the Navy's SURTASS LFA sonar activities do not create conditions of chronic, continuous underwater noise and are unlikely to lead to habitat abandonment or long-term hormonal or physiological stress responses in marine mammals.

As documented above, the vast majority of impacts from sonar exposure are expected to be behavioral in nature, temporary and comparatively short in duration, relatively infrequent, and not of the type or severity that would be expected to be additive for the small portion of the stocks and species likely to be exposed annually, into the reasonably foreseeable future. Thus, while the number of individuals "taken" by active sonar increases over time, the effect of each "take" on the survival or reproductive success of the animals themselves would not accumulate in the same way. As a result, for example, we do not expect that exposing one whale to low frequency active active sonar per year at a level that we would consider a take in the form of behavioral harassment, as predicted by the Navy's modeling described above, would result in effects over the long-term that would be greater than what we would expect from a single exposure event. To the contrary, we do not expect the effects of that "take" to have any additive,

interactive, or synergistic effect on the individual animals, the population(s) those individuals represent, or the species those population(s) comprise.

The preliminary findings from Melcón et al. (2012) and Goldbogen et al. 2013, discussed above, are also consistent with our determination that behavioral responses of mysticetes to active sonar and other active acoustic sources are unlikely to have any measurable adverse impact on the long-term fitness or reproductive success of individual animals or long-term adverse population-level effects. Although Goldbogen et al. 2013 speculates that "frequent exposures to mid-frequency anthropogenic sounds may pose significant risk to the recovery rates of endangered blue whale populations," the authors acknowlege that the actual responses of individual blue whales to simulated mid-frequency sonar documented in the study "typically involves temporary avoidance responses that appear to abate quickly after sound exposure." Moreover, the most significant response documented in the study (figure 1(b)) occurred not as a result of exposure to simulated mid-frequency sonar but as a result of exposure to pseudo-random noise. Therefore, the overall weight of scientific evidence indicates that substantive behavioral responses by mysticetes, if any, from exposure to low-frequency active sonar and other active acoustic sources evaluated in this Opinion are likely to be temporary and are unlikely to have any long-term adverse impact on individual animals or affected populations.

Also as discussed above, while the New et al. (2013) model provides a test case for future researching the potential for long-term impacts, this pilot study has very little of the critical data necessary to form any conclusions applicable to current management decisions

With respect to threatened and endangered marine mammals, our conclusion that the annual predicted behavioral takes resulting from exposure to active sonar and impulsive acoustic sources, continuing into the reasonably foreseeable future, are unlikely to result in accumulated adverse impacts is consistent with the negligible impact determination contained in the MMPA rulemaking, which is incorporated by reference. See 75 FR 69317-18; 74 FR 33828, 33884-92.

Our assessment that the continuation of the Navy's SURTASS LFA sonar training, testing and operational activities into the reasonably foreseeable future is unlikely to have any adverse additive or long-term impacts on the affected threatened or endangered species (assuming current levels of activity and no significant changes in the status of species or to the environmental baseline) is also consistent with the absence of any documented population-level or adverse aggregate impacts resulting from Navy activities to date, despite decades of Navy SURTASS LFA sonar use. Most of the SURTASS LFA sonar training, testing and operational activities are similar, if not identical, to activities that have been occurring in the same locations for decades.

The best assessment of long-term consequences from SURTASS LFA sonar training, testing, and operational activities will be to monitor the populations over time within the action area. A U.S. workshop on Marine Mammals and Sound (Fitch et al. 2011) indicated a critical need for

baseline biological data on marine mammal abundance, distribution, habitat, and behavior over sufficient time and space to evaluate impacts from human-generated activities on long-term population survival.

6.7 Integration and Synthesis

In this section, we will add the effects of the action to the environmental baseline and the cumulative effects to formulate the agency's biological opinion as to whether the proposed action is likely to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing its numbers, reproduction, or distribution. These assessments are made in full consideration of the status of the species and critical habitat.

6.7.1 Blue Whale

Blue whales will be exposed to SURTASS LFA sonar in five of the areas that that Navy will be operating under the proposed Letters of Authorization (Table 22). Based on the Navy's modeling, approximately 30 exposures of blue whales to SURTASS LFA sonar would occur. These individuals would be from the Central North Pacific stock of blue whales which is estimated at 9,250 blue whales. The highest number of exposures would occur in the Hawaii-North mission area (14). The 30 exposures would result in 0.26 percent of the stock being exposed to SURTASS LFA sonar transmissions during 14 missions.

Because of their migratory habit, blue whales are not likely to occur in water east of Japan or off Guam in the summer. The evidence available suggests that when exposed to SURTASS LFA sonar transmissions, blue whales, like other baleen whales, are not likely to be killed or experience injury, masking, stranding, resonance effects, or behavioral responses that might reduce the longevity or reproductive success of individuals that have been exposed. Instead, the best scientific and commercial data available suggests that SURTASS LFA sonar transmissions are likely to elicit short-term responses in blue whales that are known to have no long-term, adverse consequences for the biology or ecology of the individual whales exposed to an LFA sonar signal.

We conclude that the SURTASS LFA sonar training, testing and operations the U.S. Navy proposes to conduct in the action area on an annual basis consistent with the MMPA letters of authorization, cumulatively over five-year period of the MMPA rule (August 15, 2012 through August 14, 2017), and ongoing for the reasonably foreseeable future is not likely to have short-term, adverse effects on individual whales and is not likely to adversely affect the population dynamics of blue whales in ways that would reduce their reproduction, numbers, or distribution. As a result, these transmissions would not be expected to appreciably reduce the blue whales' likelihood of surviving and recovering in the wild.

6.7.2 Fin Whale

Based on the results of the Navy's modeling, the highest number of exposures (56) would be in the Sea of Japan where Navy proposes to conduct two SURTASS LFA sonar missions (

Table 23). Elsewhere in the Pacific Ocean, the exposures ranged from 1 to 18 fin whales in areas where one to three missions will be conducted. In total, 137 exposures of fin whales to SURTASS LFA sonar would occur during 16 missions. These exposures would be from three stocks of fin whales; the Western North Pacific stock - 107 exposures out of a total stock estimated to include 9,250 fin whales; the East China Sea stock - 8 exposures out of stock estimated at 500 individuals; and the Hawaii stock - 12 exposures out of the estimated 174 individuals. The 137 exposures would result in 1.11 percent of the Western North Pacific stock, 1.48 percent of the East China Sea stock, and 5.84 percent of the Hawaii stock being exposed to SURTASS LFA sonar transmissions during 20 missions.

The evidence available suggests that fin whales, like other baleen whales, exposed to SURTASS LFA sonar transmissions are not likely to be killed or experience injury, masking, stranding, resonance effects, or behavioral responses that might reduce the longevity or reproductive success of individuals that have been exposed. Instead, the best scientific and commercial data available suggests that SURTASS LFA sonar transmissions are likely to elicit short-term responses in fin whales that are known to have no long-term, adverse consequences for the biology or ecology of the individual whales exposed to an LFA sonar signal.

We conclude that the SURTASS LFA sonar training, testing and operations the U.S. Navy proposes to conduct in the action area on an annual basis consistent with the MMPA letters of authorization, cumulatively over five-year period of the MMPA rule (August 15, 2012 through August 14, 2017), and ongoing for the reasonably foreseeable future is not likely to have short-term, adverse effects on individual whales and is not likely to adversely affect the population dynamics of fin whales in ways that would reduce their reproduction, numbers, or distribution. As a result, these transmissions would not be expected to appreciably reduce the fin whales' likelihood of surviving and recovering in the wild.

6.7.3 Western Pacific Gray Whale

Based on Navy modeling, we expect three exposures of Western Pacific gray whales to SURTASS LFA sonar transmissions (Table 24). The western North Pacific stock of gray whales is estimated to include only 121 animals. The two exposures would result in 0.41 percent of the Western North Pacific stock being exposed to SURTASS LFA sonar transmissions during 4 missions.

The evidence available suggests that western Pacific gray whales, like other baleen whales, exposed to SURTASS LFA sonar transmissions are not likely to be killed or experience injury, masking, stranding, resonance effects, or behavioral responses that might reduce the longevity or reproductive success of individuals that have been exposed. Instead, the best scientific and commercial data available suggests that SURTASS LFA sonar transmissions are likely to elicit short-term responses in western Pacific gray whales that are known to have no long-term,

adverse consequences for the biology or ecology of the individual whales exposed to an LFA sonar signal.

We conclude that the SURTASS LFA sonar training, testing and operations the U.S. Navy proposes to conduct in the action area on an annual basis consistent with the MMPA letters of authorization, cumulatively over five-year period of the MMPA rule (August 15, 2012 through August 14, 2017), and ongoing for the reasonably foreseeable future is not likely to have short-term, adverse effects on individual whales and is not likely to adversely affect the population dynamics of western Pacific gray whales in ways that would reduce their reproduction, numbers, or distribution. As a result, these transmissions would not be expected to appreciably reduce the gray whales' likelihood of surviving and recovering in the wild.

6.7.4 Humpback Whale

Based on the results of the Navy's modeling, humpback whales will be exposed in five mission areas over the course of 13 SURTASS LFA sonar missions Over the course of 13 proposed missions, 120 humpback whales in two stocks, the WNP and CNP, may be exposed to SURTASS LFA sonar transmissions in five mission areas, based on the results of the Navy's modeling. The most affected of the humpback stocks is predicted to be the WNP stock, which is composed of 1,107 individuals, and would have 98 whales or 8.54% of the stock affected at the 120 to 180 dB SPE exposure level during nine missions in three mission areas. Twenty-two humpback whales or 0.2% of the CNP stock of 10,103 individuals would potentially be exposed to SURTASS LFA sonar transmissions during four missions in two mission area. Over the course of 13 proposed missions, 120 humpback whales in two stocks, the WNP and CNP, may be exposed to SURTASS LFA sonar transmissions in five mission areas, based on the results of the Navy's modeling. The most affected of the humpback stocks is predicted to be the WNP stock, which is composed of 1,107 individuals, and would have 98 whales or 8.54% of the stock affected at the 120 to 180 dB SPE exposure level during nine missions in three mission areas. Twenty-two humpback whales or 0.2% of the CNP stock of 10,103 individuals would potentially be exposed to SURTASS LFA sonar transmissions during four missions in two mission areas.

Over the course of 13 proposed missions, 120 humpback whales in two stocks, the WNP and CNP, may be exposed to SURTASS LFA sonar transmissions in five mission areas, based on the results of the Navy's modeling. The most affected of the humpback stocks is predicted to be the WNP stock, which is composed of 1,107 individuals, and would have 98 whales or 8.54% of the stock affected at the 120 to 180 dB SPE exposure level during nine missions in three mission areas. Twenty-two humpback whales or 0.2% of the CNP stock of 10,103 individuals would potentially be exposed to SURTASS LFA sonar transmissions during four missions in two mission areas. (Over the course of 13 proposed missions, 120 humpback whales in two stocks, the Western North Pacific (WNP) and Central North Pacific (CNP), may be exposed to SURTASS LFA sonar transmissions in five mission areas, based on the results of the Navy's modeling. The most affected of the humpback stocks is predicted to be the WNP stock, which is

composed of 1,107 individuals, and would have 98 whales or 8.54% of the stock affected at the 120 to 180 dB SPE exposure level during nine missions in three mission areas. Twenty-two humpback whales or 0.2% of the CNP stock of 10,103 individuals would potentially be exposed to SURTASS LFA sonar transmissions during four missions in two mission areas.

Table 25). Three missions in the North Philippine Sea would result in 78 humpback whales exposures while three missions in the West Philippine Sea would exposure 18 humpback whales to SURTASS LFA sonar transmissions. Three mission off Guam, two missions in the Hawaii-North area and two missions in the Hawaii-South area would result in approximately 2, 10 and 12 exposures of humpback whales, respectively to SURTASS LFA sonar transmissions. The 98 exposures would result in 8.54 percent of the Western North Pacific stock being exposed to SURTASS LFA sonar transmissions during 9 missions and 22 exposures resulting in 0.20 percent of the Central North Pacific stock being exposed to SURTASS LFA sonar transmissions during 4 missions.

The evidence available suggests that humpback whales, like other baleen whales, exposed to SURTASS LFA sonar transmissions are not likely to be killed or experience injury, masking, stranding, resonance effects, or behavioral responses that might reduce the longevity or reproductive success of individuals that have been exposed. Instead, the best scientific and commercial data available suggests that SURTASS LFA sonar transmissions are likely to elicit short-term responses in humpback whales that are known to have no long-term, adverse consequences for the biology or ecology of the individual whales exposed to an LFA sonar signal.

We conclude that the SURTASS LFA sonar training, testing and operations the U.S. Navy proposes to conduct in the action area on an annual basis consistent with the MMPA letters of authorization, cumulatively over five-year period of the MMPA rule (August 15, 2012 through August 14, 2017), and ongoing for the reasonably foreseeable future is not likely to have short-term, adverse effects on individual whales and is not likely to adversely affect the population dynamics of humpback whales in ways that would reduce their reproduction, numbers, or distribution. As a result, these transmissions would not be expected to appreciably reduce the humpback whales' likelihood of surviving and recovering in the wild.

6.7.5 North Pacific Right Whale

Based on Navy modeling, four exposures of North Pacific right whales to SURTASS LFA sonar transmissions would occur during six out of a total of eight missions (Table 13). The North Pacific stock of right whales is estimated at 922 animals. The four exposures would result in 0.15 percent of the North Pacific stock being exposed to SURTASS LFA sonar transmissions.

If exposed to SURTASS LFA sonar transmissions, the evidence available suggests that North Pacific right whales, like other baleen whales, are not likely to be killed or experience injury, masking, stranding, resonance effects, or behavioral responses that might reduce the longevity or reproductive success of individuals that have been exposed. Instead, the best scientific and commercial data available suggests that SURTASS LFA sonar transmissions are likely to elicit short-term responses in North Pacific right whales that are known to have no long-term, adverse consequences for the biology or ecology of the individual whales exposed to an LFA sonar signal.

We conclude that the SURTASS LFA sonar training, testing and operations the U.S. Navy proposes to conduct in the action area on an annual basis consistent with the MMPA letters of authorization, cumulatively over five-year period of the MMPA rule (August 15, 2012 through August 14, 2017), and ongoing for the reasonably foreseeable future is not likely to have short-term, adverse effects on individual whales and is not likely to adversely affect the population dynamics of right whales in ways that would reduce their reproduction, numbers, or distribution. As a result, these transmissions would not be expected to appreciably reduce the right whales' likelihood of surviving and recovering in the wild.

6.7.6 Sei Whale

Based on the results of the Navy's modeling, approximately 39 exposures of sei whales (36 exposures of the North Pacific stock and 3 exposures of the Hawaiian stock of sei whales) will occur during 10 SURTASS LFA sonar missions (Table 14). The 39 exposures would result in 0.39 percent of the North Pacific stock being exposed to SURTASS LFA sonar transmissions during six missions and 0.80 percent of the Hawaiian stock being being exposed to SURTASS LFA sonar transmissions during four missions.

If exposed to SURTASS LFA sonar transmissions, the evidence available suggests that sei whales, like other baleen whales, are not likely to be killed or experience injury, masking, stranding, resonance effects, or behavioral responses that might reduce the longevity or reproductive success of individuals that have been exposed. Instead, the best scientific and commercial data available suggests that SURTASS LFA sonar transmissions are likely to elicit short-term responses in sei whales that are known to have no long-term, adverse consequences for the biology or ecology of the individual whales exposed to an LFA sonar signal.

We conclude that the SURTASS LFA sonar training, testing and operations the U.S. Navy proposes to conduct in the action area on an annual basis consistent with the MMPA letters of authorization, cumulatively over five-year period of the MMPA rule (August 15, 2012 through August 14, 2017), and ongoing for the reasonably foreseeable future is not likely to have short-term, adverse effects on individual whales and is not likely to adversely affect the population dynamics of sei whales in ways that would reduce their reproduction, numbers, or distribution.

As a result, these transmissions would not be expected to appreciably reduce the sei whales' likelihood of surviving and recovering in the wild.

6.7.7 Sperm Whale

Based on the results of the Navy's modeling, approximately 872 exposures of sperm whales will occur during 20 missions Based on the results of the Navy's modeling, the highest total number of sperm whales in any stock potentially exposed to SURTASS LFA sonar at RLs of 120 to 180 dB SPE would be 574 whales or 0.55% of the NP stock. The NP stock of sperm whales, consisting of an estimated 102,112 individuals, may be exposed to LFA sonar during 16 missions in all nine of the western North Pacific mission areas. Only one other stock of sperm whales, the Hawaiian, potentially will be exposed to LFA sonar. In total, 872 sperm whale exposures in all 11 SURTASS LFA mission areas are estimated. Sperm whales in the Hawaiian stock, which is estimated to include 6,919 whales, may be exposed to LFA sonar during four proposed missions in the two central North Pacific mission areas, resulting in 298 sperm whale exposures, which may affect 4.29% of the Hawaiian stock. Based on the results of the Navy's modeling, the highest total number of sperm whales in any stock potentially exposed to SURTASS LFA sonar at RLs of 120 to 180 dB SPE would be 574 whales or 0.55% of the NP stock. The NP stock of sperm whales, consisting of an estimated 102,112 individuals, may be exposed to LFA sonar during 16 missions in all nine of the western North Pacific mission areas. Only one other stock of sperm whales, the Hawaiian, potentially will be exposed to LFA sonar. In total, 872 sperm whale exposures in all 11 SURTASS LFA mission areas are estimated. Sperm whales in the Hawaiian stock, which is estimated to include 6,919 whales, may be exposed to LFA sonar during four proposed missions in the two central North Pacific mission areas, resulting in 298 sperm whale exposures, which may affect 4.29% of the Hawaiian stock. Based on the results of the Navy's modeling, the highest total number of sperm whales in any stock potentially exposed to SURTASS LFA sonar at RLs of 120 to 180 dB SPE would be 574 whales or 0.55% of the NP stock. The NP stock of sperm whales, consisting of an estimated 102,112 individuals, may be exposed to LFA sonar during 16 missions in all nine of the western North Pacific mission areas. Only one other stock of sperm whales, the Hawaiian, potentially will be exposed to LFA sonar. In total, 872 sperm whale exposures in all 11 SURTASS LFA mission areas are estimated. Sperm whales in the Hawaiian stock, which is estimated to include 6,919 whales, may be exposed to LFA sonar during four proposed missions in the two central North Pacific mission areas, resulting in 298 sperm whale exposures, which may affect 4.29% of the Hawaiian stock. Based on the results of the Navy's modeling, the highest total number of sperm whales in any stock potentially exposed to SURTASS LFA sonar at RLs of 120 to 180 dB SPE would be 574 whales or 0.55% of the NP stock. The NP stock of sperm whales, consisting of an estimated 102,112 individuals, may be exposed to LFA sonar during 16 missions in all nine of the western North Pacific mission areas. Only one other stock of sperm whales, the Hawaiian, potentially will be exposed to LFA sonar. In total, 872 sperm whale exposures in all 11 SURTASS LFA mission areas are estimated. Sperm whales in the Hawaiian stock, which is estimated to include

6,919 whales, may be exposed to LFA sonar during four proposed missions in the two central North Pacific mission areas, resulting in 298 sperm whale exposures, which may affect 4.29% of the Hawaiian stock. (Based on the results of the Navy's modeling, the highest total number of sperm whales in any stock potentially exposed to SURTASS LFA sonar at RLs of 120 to 180 dB SPE would be 574 whales or 0.55% of the NP stock. The NP stock of sperm whales, consisting of an estimated 102,112 individuals, may be exposed to LFA sonar during 16 missions in all nine of the western North Pacific mission areas. Only one other stock of sperm whales, the Hawaiian, potentially will be exposed to LFA sonar. In total, 872 sperm whale exposures in all 11 SURTASS LFA mission areas are estimated. Sperm whales in the Hawaiian stock, which is estimated to include 6,919 whales, may be exposed to LFA sonar during four proposed missions in the two central North Pacific mission areas, resulting in 298 sperm whale exposures, which may affect 4.29% of the Hawaiian stock.

Table 28). The highest number of exposures, 224, would be during two missions in the Hawaii-North area. Missions in the Hawaii-South area would result in 74 exposures to SURTASS LFA sonar transmissions during two missions. These exposures would be animals from the Hawaii stock that is estimated at 6,919 sperm whales. Another 574 exposures of sperm whales to SURTASS LFA sonar transmissions would occur in other mission areas. These exposures would be animals from the North Pacific stock of sperm whales that is estimated at 102,112 individuals. Approximately 0.55 percent of the North Pacific stock and 4.29 percent of the Hawaii stock of sperm whales would be exposed to SURTASS LFA sonar transmissions during 16 and 4 missions, respectively.

The evidence available suggests that sperm whales, like other toothed whales, are not very sensitive to low-frequency sounds. Despite the limited number of studies, the available evidence suggests that the risk of injury, masking, stranding, resonance effects, or behavioral effects in sperm whales is very low. The best scientific and commercial data available suggests that SURTASS LFA sonar transmissions is likely to elicit short-term effects on sperm whales that are known to have no long-term, adverse consequences for the biology or ecology of the individual whales exposed to the LFA sonar signal.

We conclude that the SURTASS LFA sonar training, testing and operations the U.S. Navy proposes to conduct in the action area on an annual basis consistent with the MMPA letters of authorization, cumulatively over five-year period of the MMPA rule (August 15, 2012 through August 14, 2017), and ongoing for the reasonably foreseeable future is not likely to have short-term, adverse effects on individual whales and is not likely to adversely affect the population

dynamics of sperm whales in ways that would reduce their reproduction, numbers, or distribution. As a result, these transmissions would not be expected to appreciably reduce the sperm whales' likelihood of surviving and recovering in the wild.

6.7.8 Main Hawaiian Island Insular False Killer Whale

Based on the results of the Navy's modeling approximately 4 Main Hawaiian Island Insular false killer whales could be exposed to SURTASS LFA sonar transmissions during 4 missions (Table 29). These exposures would occur during the 2 missions each in the Hawaii-North and Hawaii-South mission areas. Abundance for Main Hawaiian Island Insular false killer whales is estimated at 151 individuals. Approximately 0.84 percent of the of the Main Hawaiian Island Insular stock of false killer whales would be exposed to SURTASS LFA sonar transmissions during four missions.

The evidence available suggests, however, that Hawaii Insular false killer whales, like other toothed whales, are not very sensitive to low-frequency sounds. Despite the limited number of studies, the available evidence suggests that the risk of injury, masking, stranding, resonance effects, or behavioral effects in these whales is very low. The best scientific and commercial data available suggests that SURTASS LFA sonar transmissions is likely to elicit short-term effects on Hawaii Insular false killer whales that are known to have no long-term, adverse consequences for the biology or ecology of the individual whales exposed to the LFA sonar signal.

We conclude that the SURTASS LFA sonar training, testing and operations the U.S. Navy proposes to conduct in the action area on an annual basis consistent with the MMPA letters of authorization, cumulatively over five-year period of the MMPA rule (August 15, 2012 through August 14, 2017), and ongoing for the reasonably foreseeable future is not likely to have short-term, adverse effects on individual whales and is not likely to adversely affect the population dynamics of Hawaii Insular false killer whales in ways that would reduce their reproduction, numbers, or distribution. As a result, these transmissions would not be expected to appreciably reduce the Hawaii Insular false killer whales' likelihood of surviving and recovering in the wild.

6.7.9 Hawaiian Monk Seal

Although Hawaiian monk seals generally reside in coastal waters near haulout areas, they forage in deep water and dive to at least 490 m (1,608 ft) (Randall R. Reeves, Stewart, & Leatherwood., 1992), which could expose them to low frequency sounds from SURTASS LFA sonar. The Navy's simulation modeling suggest that 16 Hawaiian monk seals would be exposed to SURTASS LFA sonar transmissions in areas north of Hawaii and south of Hawaii. Approximately 1.12 percent of the Hawaiian monk seals would be exposed to SURTASS LFA sonar transmissions during 5 missions (Table 30).

Hawaiian monk seals have their most sensitive hearing at 12 to 28 kHz; their high frequency sensitivity drops off sharply above 30 kHz (Jeanette A. Thomas, Moore, Withrow, & Stoermer, 1990b). Below 8 kHz, Hawaiian monk seals have less sensitive hearing than other pinnipeds.

Based on the monk seal's limited sensitivity to low frequency sound (D. A. Croll et al., 1999; W. John Richardson, Charles R. Greene Jr., et al., 1995), we conclude that the SURTASS LFA sonar training, testing and operations the U.S. Navy proposes to conduct in the action area on an annual basis consistent with the MMPA letters of authorization, cumulatively over five-year period of the MMPA rule (August 15, 2012 through August 14, 2017), and ongoing for the reasonably foreseeable future is not likely to have short-term, adverse effects on individual whales and is not likely to adversely affect the population dynamics of Hawaiian monk seal and use of SURTASS LFA sonar would not be expected to appreciably reduce this monk seal's likelihood of surviving and recovering in the wild.

6.7.10 Sea Turtles

The Navy did not simulate potential exposure of sea turtles to SURTASS LFA sonar transmissions. Because of their ecology, only the juvenile and adult stages of sea turtles could be potentially exposed to SURTASS LFA sonar transmissions. We assume that the monitoring protocols associated with SURTASS LFA sonar would be more effective with larger sea turtles, like adult leatherback and loggerhead turtles, than with species like olive ridley, smaller leatherback, hawksbill, green, and some loggerhead turtles; the monitoring protocols may not detect some individual members of these species, which would increase their risk of exposure to sound pressure levels associated with SURTASS LFA sonar within the mitigation zone (that is, 180 dB) if they encountered SURTASS LFA sonar vessels during a ping.

Although the probability of an interaction between SURTASS LFA sonar and individuals of any of these sea turtles is statistically small (the Navy's analyses concluded that the possible number of times a leatherback sea turtle could be in the vicinity of a SURTASS LFA sonar vessel would be less than three out of 18,000 animals per year per vessel; with the monitoring protocols, the Navy concluded that this number would approach zero), the probability could increase depending on the deployment of the SURTASS LFA sonar vessels.

Nevertheless, sea turtles have a small probability of being exposed to SURTASS LFA sonar transmissions. Sea turtles are able to detect low-frequency sounds and will be able to detect SURTASS LFA transmissions. Information on their behavioral response to these decibel levels is limited. However, green sea turtles were observed to avoid passing through a sound barrier created by an array of air guns with a broadband spectrum of 20 to 1,000 Hz; received levels were 141 to 150 dB (J. O'Hara & Wilcox, 1990). The probability that a sea turtle would be within an ensonified area that would elicit a similar or other behavioral response is low because most of the turtles make shallow dives of about 300 m (984 ft; dive observed for the olive ridley sea turtle). As for the leatherback sea turtles, which can dive to depths of 1,000 m (3,280 ft), the opportunity for a behavioral response is also considered to be low because 95 percent of their dives are less than 200 m (656 ft) deep, which would minimize their exposure to the SURTASS LFA sonar 180-dB sound field. Based on the hearing data, it is possible that if a sea turtle

happened to be in proximity of a SURTASS LFA sonar operations area, it will hear the LF transmissions.

Given that the majority of sea turtles encountered would probably be transiting in the open ocean from one site to another, the possibility of significant displacement would be unlikely, we conclude that the SURTASS LFA sonar training, testing and operations the U.S. Navy proposes to conduct in the action area on an annual basis consistent with the MMPA letters of authorization, cumulatively over five-year period of the MMPA rule (August 15, 2012 through August 14, 2017), and ongoing for the reasonably foreseeable future would not reduce the reproduction, numbers, or distribution of sea turtles. As a result, these sonar transmissions would not be expected to appreciably reduce these turtles likelihood of surviving and recovering in the wild.

7 CONCLUSION

After reviewing the current status of the listed species, the environmental baseline within the action area, the effects of the proposed action, any effects of interrelated and interdependent actions, and cumulative effects, it is NMFS' biological opinion that the proposed action is not likely to jeopardize the continued existence of endangered blue, fin, western North Pacific gray, humpback, North Pacific right, sei, Main Hawaiian Insular false killer whales or sperm whales; Hawaiian monk seals; green, hawksbill, loggerhead, olive ridley, or leatherback sea turtles; or to destroy or adversely modify critical habitat designated for those species.

While the number of individuals "taken" gets larger over time, the effect of each "take" on the survival or reproductive success of the animals themselves would not accumulate in the same way. As a result, for example, we do not expect that instances of exposing whales to SURTASS LFA sonar sonar in a single year, or instances of exposing them to LFA sonar over the remaining period of the MMPA rule or into the reasonably foreseeable future, would result in effects that would be greater than what we would expect from a single exposure event. To the contrary, we did not expect the effects of that "take" to have any additive, interactive, or synergistic effect on the individual animals, the population(s) those individuals represent, or the species those population(s) comprise. With respect to threatened and endangered marine mammals, our conclusion that the aggregate number of exposures over the duration of the MMPA regulations or and into the reasonably foreseeable future is unlikely to result in accumulated adverse impacts is also supported by the negligible impact determination and response to comments contained in the MMPA rulemaking.

Our regulations require us to consider, using the best available scientific data, effects of the action that are "likely" and "reasonably certain" to occur rather than effects that are speculative or uncertain. See 50 C.F.R. § 402.02 (defining to "jeopardize the continued existence of" and "effects of the action"). For the reasons set forth above, and taking into consideration the best available scientific evidence documented throughout this Opinion, we conclude that the

continuation of SURTASS LFA sonar training, testing and operational activities into the reasonable foreseeable future, at the levels described in the current five-year MMPA rule (and assuming no change in the status of species or the environmental baseline), are unlikely to lead to any adverse, long-term additive or cumulative impacts on individuals or affected populations, and that such long-term impacts are not reasonably certain to occur based on the information that is currently available. Furthermore, our analysis and conclusions in this Opinion are based on modeled estimates of exposures and take assuming that the Navy conducts the maximum number of authorized training activities for the maximum number of authorized hours. Therefore, our assumption that the Navy's activities will continue into the reasonably foreseeable future does not alter our conclusion that the Navy's activites are unlikely to jeopardize the continued existence of any ESA-listed species or destroy or adversely modify critical habitat that has been designated for such species.

Therefore, it is NMFS' opinion that SURTASS LFA sonar training, testing, and operations are likely to adversely affect but are not likely to jeopardize the continued existence of these threatened and endangered species under NMFS' jurisdiction and are not likely to result in the destruction or adverse modification of critical habitat that has been designated for endangered or threatened species in the action area annually, over the remaining period of the five-year MMPA, rule or in the reasonably foreseeable future, assuming that the type, amount and extent of training, testing, and operations do not exceed levels assessed in this opinion and/or the status of the species affected by these actions does not change significantly from that assessed in this Opinion.

This opinion also concludes that the NMFS' issuance of the letters of authorization pursuant to the MMPA five-year rule for the Navy to take marine mammals for a period beginning in August 15, 2014 and ending in Augist 14, 2015, incidental to the SURTASS LFA sonar training, testing, and operations are likely to adversely affect but are not likely to jeopardize the continued existence of these threatened and endangered species under NMFS' jurisdiction and are not likely to result in the destruction or adverse modification of critical habitat that has been designated for endangered or threatened species in the action area.

8 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and Federal regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by regulation to include significant habitat modification or degradation that results in death or injury to listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. For this consultation, we interpret "harass" to mean an intentional or negligent

action that has the potential to injure an animal or disrupt its normal behaviors to a point where such behaviors are abandoned or significantly altered.[12] Section 7(b)(4) and section 7(o)(2) provide that taking that is incidental to an otherwise lawful agency action is not considered to be prohibited taking under the ESA if that action is performed in compliance with the terms and conditions of this Incidental Take Statement.

ESA section 7(b)(4) states that take of ESA-listed marine mammals must be authorized under MMPA section 101(a)(5) before the Secretary can issue an incidental take statement for listed marine mammals. NMFS' implementing regulations for section 101 (a)(5)(A) specify that a letter of authorization is required to conduct activities pursuant to any regulations for a specific activity that will "take" marine mammals. NMFS has authorized the incidental take of marine mammals under letters of authorization pursuant to 50 CFR § 218 Subpart X.

8.1 Amount and Extent of Take

The analysis contained in this opinion concluded that individual blue whales, fin whales, western Pacific gray whales, humpback right whales, sei whales, sperm whales, Main Hawaiian Islands (MHI) Insular false killer whales, and Hawaiian monk seals as well as listed sea turtles will be exposed to SURTASS LFA sonar transmissions in the North Pacific Ocean. Any animals that would be exposed to LFA sonar transmissions would occur in portions of the North Pacific Ocean:

- East of Japan; the North Philippine Sea; the west Philippine Sea; offshore Guam; the Sea of Japan; the East China Sea; the South China Sea; and offshore Japan (25° to 40° N and 10° to 25° N).
- The north-central Pacific Ocean, which includes the Hawaii-North and Hawaii-South mission areas within the Hawaii Range Complex.

Any threatened or endangered species that are exposed to LFA sonar transmissions may elicit behavioral responses that might be considered “harassment.” NMFS does not expect any threatened or endangered species to be injured or killed as a result of exposure to LFA sonar transmissions.

The section 7 regulations require NMFS to estimate the number of individuals that may be taken by proposed actions or the extent of land or marine area that may be affected by an action, if we cannot assign numerical limits for animals that could be incidentally taken during the course of an action (51 Federal Register, 19926, 19953, 3 June 1986).

[12] NMFS has not adopted a regulatory definition of harassment under the ESA. The World English Dictionary defines harass as “to trouble, torment, or confuse by continual persistent attacks, questions, etc.” The U.S. Fish and Wildlife Service defines “harass” in its regulations as “an intentional or negligent act or omission which creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering (50 CFR 17.3). The interpretation we adopt in this consultation is consistent with our understanding of the dictionary definition of harass and is consistent with the Service’s interpretation of the term.

The amount of take resulting from LFA sonar transmissions was difficult to estimate because we have little empirical information on (a) the actual number of listed species that are likely to occur in the action area, (b) the actual number of individuals of those species that are likely to be exposed to LFA sonar transmissions, (c) the circumstances associated with any exposure, and (d) the range of responses we would expect different individuals of the different species to exhibit upon exposure.

Because this information was not available, this biological opinion relied on the Navy's computer simulations to estimate the "number" of certain marine mammals that might be harassed during the employment of SURTASS LFA sonar; the results of these simulations appear in Table 21, Table 22, Table 23, Table 24, Table 25, Table 26, Table 27, Table 28, Table 29, and Table 30. Because these estimates were produced by computer simulations, they should not be treated literally; instead, they should be treated as an index of the order of magnitude of potential exposure rather than the actual number of animals that would be exposed.

For the purposes of this biological opinion and incidental take statement, we assumed that any non-zero value in Table 31 indicates that an individual whale has a probability of being exposed to received levels that might be expected to result in behavioral responses characteristic of an animal that has been harassed. That assumption results in the estimates shown in Table 31 below.

Table 31. The number of individuals that are likely to be "taken" as a result of their exposure to U.S. Navy Surveillance Towed Array Sensor System Low Frequency Active Sonar by mission area number.

Species	Estimated Annual Take by Species & Mission Area											Total Annual Take Estimates	
	1	2	3	4	5	6	7	8	9	10	11	Behavioral Harrassment*	Harm (Injury, PTS)
Number of Missions	*1*	*3*	*3*	*3*	*2*	*1*	*1*	*1*	*1*	*2*	*2*		
Blue Whale	-	2	2	2	-	-	-	-	1	14	9	**30**	**0**
Fin Whale	5	18	8	2	73	8	4	6	1	7	5	**137**	**0**
Western Pacific Gray Whale	-	-	-	-	2	-	1	-	-	-	-	**3**	**0**
Humpback Whale	-	78	18	2	-	-	-	-	-	10	12	**120**	**0**
North Pacific Right Whale	-	2	-	-	1	-	1	-	-	-	-	**4**	**0**
Sei Whale	7	-	-	18	-	-	-	6	5	1	2	**39**	**0**
Sperm Whale	24	90	108	98	120	30	13	49	42	224	74	**872**	**0**
MHI Insular False Killer Whale	-	-	-	-	-	-	-	-	-	2	2	**4**	**0**
Hawaiian Monk Seal	-	-	-	-	-	-	-	1	-	10	5	**16**	**0**
Sea Turtles	**	**	**	**	**	**	**	**	**	**	**	**	**0**

***Behavioral Harrassment. We do not expect any instances of Temporary Threshold Shift (TTS) in sea turtles due to mitigation.**
****Unspecified Number. While the potential for behavioral harrassment of sea turtles exists, it is very difficult to estimate the number and species composition of turtles that could be "taken." Take will be exceeded if activity levels as proposed are exceeded or if the monitoring program detects any turtle species that have exposed to received levels greater than 180 dB, or if during or after LFA sonar operations it is determined that an animal was exposed to sound levels of 180 dB or higher, or if a vessel strike occurs.**

We did not conduct computer simulations for sea turtles because the data necessary to develop computer models were not available. Due to the seasonal and life stage changes in habitat occupation, sea turtle populations are particularly difficult to census. Abundance estimates are based on the most current information available regarding counts of the most accessible member of the population, nesting females, which does not account for the abundance of male sea turtles. The lack of detailed population data and scarcity of density data for sea turtles do not allow for density estimates to be derived for populations in the North Pacific Ocean; therefore, we could not assign numerical limits for take estimates. Rather than specifying an amount of take for sea turtles, this incidental take statement specifies an extent of take as follows:

Adult and sub-adult sea turtles may be taken, in the form of harassment, in areas outside the LFA mitigation zone and the additional buffer zone required by the Letters of Authorization. (Because they tend to remain at or within a few meters of the ocean surface, we do not expect hatchling or juvenile sea turtles to be exposed to LFA sonar transmissions.) Because of their size and the density of their shells, we assume that the Navy's monitoring programs, particularly the HF/M3 sonar system, will detect these larger turtles if they are in the mitigation zone.

Take of these species will have been exceeded if the monitoring program detects any individuals of these species that have been harmed, injured, or killed as result of exposure to LFA sonar transmissions (from which NMFS might infer that they had been exposed to received levels greater than 180 dB), or if during or after LFA sonar operations it is determined that an animal was exposed to sound levels of 180 dB or higher (i.e., it was not detected until after it was inside the 180 dB isopleth), or if a vessel strike occurs. Temporary threshold shift or PTS would not be expected to occur in sea turtles due to the mitigation.

8.1.1 Activity Levels Indicator of Take for Sea Turtles

Detection of behavioral responses of juvenile or adult sea turtles in coastal waters or at-sea during Navy SURTASS LFA training, testing, or operational activities would be extremely difficult. Most forms of behavioral responses would not be detected. Also, monitoring techniques to calculate actual take of including detection and collection of individuals and assessment of injuries or death is not feasible for sea turtles at the scale of SURTASS LFA sonar activities. Therefore, we must rely on predicted take associated with levels of activities and any opportunistic observations of potential behavioral responses or injured or dead or juvenile or adult sea turtles during activities as measurements of take and a trigger for reinitiation of consultation. In the absence of observations of unanticipated levels of behavioral responses, injury or mortality, exceedance of an activity level will require the Navy to reinitiate consultation. Exceedances at the activity level or in other planned training events must be reported to NMFS prior to carrying out or immediately following, if reporting would interrupt Navy training activities.

8.2 Effect of the Take

Based on analysis provided in this biological opinion, we conclude that small numbers of the endangered Hawaiian monk seal, blue whale, western Pacific population of gray whale, fin whale, humpback whale, North Pacific right whales, sei whale, sperm whale, MHI Insular false killer whales and threatened and endangered sea turtles have will be exposed to SURTASS LFA sonar transmissions during the 20 missions that the U.S. Navy proposes to conduct between 15 August and 14 August 2015.

The assessment contained in this opinion also considers the probable responses of those species to exposure by LFA sonar and concludes that, based on the best scientific and commercial data available, individuals of these species might respond to that exposure by exhibiting behavioral responses that might constitute harassment. We do not expect any threatened or endangered species to be injured or killed as a direct or indirect result of exposure to LFA sonar transmissions.

After considering the number of marine mammals that might be exposed to LFA sonar and the geographic area in which sea turtles might be exposed; received levels of LFA sonar associated with that exposure; and the probable responses of individuals of these different species to an LFA sonar exposure, we have concluded that exposing these species to LFA sonar during the missions the Navy proposes to conduct between 15 August 2014 and 14 August 2015, or over the remaining period of the MMPA rule, or in the reasonably forseeable future, is not likely to reduce the fitness of individuals of these species and, therefore, is not likely to reduce the viability of the populations those individuals represent, and, as a result, is not likely to jeopardize the continued existence of these threatened or endangered species.

8.2.1 Reasonable and Prudent Measures

"Reasonable and prudent measures" are nondiscretionary measures to minimize the amount or extent of incidental take (50 CFR § 402.02). NMFS believes the reasonable and prudent measures described below are necessary and appropriate to minimize the impacts of incidental take on threatened and endangered species:

(1) The authorization shall be valid only for the unintentional taking of the species of marine mammals identified in 50 CFR § 218.230(b) and condition 5 of the Authorization governing the taking of these animals incidental to the activity specified below and shall be valid only for takings consistent with the terms and conditions set out in 50 CFR § 218 Subpart X and the terms of NMFS' Letters of Authorization.

(2) NMFS' Permits and Conservation Division shall require the U.S. Navy to implement a program to mitigate the potential effects of LFA sonar transmissions on threatened or endangered species as specified in the final regulations for the Taking of Marine Mammals Incidental to Operation of the Surveillance Towed Array Sensor System Low Frequency Active Sonar (50 CFR § 218 Subpart X).

(3) NMFS' Permits and Conservation Division shall require the U.S. Navy to implement a program to monitor potential interactions between LFA sonar transmissions and threatened or endangered species.

8.3 Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA the Permits and Conservation Division and the U.S. Navy must comply with the following terms and conditions, which implement the Reasonable and Prudent Measures described above and outlines the mitigation, monitoring and reporting measures required by the section 7 regulations (50 CFR 402.14(i)). These terms and conditions are non-discretionary. If the Permits and Conservation Division and the U.S. Navy fail to ensure compliance with these terms and conditions and their implementing reasonable and prudent measures, the protective coverage of section 7(o)(2) may lapse.

(1) The authorization shall be valid only for the activities associated with the operation of the Surveillance Towed Array Sensor System Low Frequency Active Sonar onboard the USNS IMPECCABLE (T-AGOS 23), USNS ABLE (T-AGOS 20), USNS EFFECTIVE (T-AGOS 21) and USNS VICTORIOUS (T-AGOS 19). The signals transmitted by the SURTASS LFA sonar source must be between 100 and 500 Hertz (Hz) with a source level for each projector no more than 215 dB (re: 1 micro Pascal (μPa) at 1 meter (m)) and a maximum duty cycle of 20 percent.

(2) The U.S. Navy shall be required to (a) establish shut-down criteria for the SURTASS LFA sonar whenever a marine mammal or sea turtle is detected within the 1 km (0.54 nmi) buffer zone beyond the SURTASS LFA sonar mitigation zone (180 dB sound field), (b) not broadcast the SURTASS LFA sonar signal at a frequency greater than 500 Hz, and (c) plan its missions to ensure no greater than 12 percent of any marine mammal stock is incidentally harassed during the effective period of the letters of authorization.

(3) If a marine mammal or sea turtle is detected within the area subjected to a sound pressure level of 180 dB or greater (mitigation zone) or within the 1 km (0.5 nmi) buffer zone extending beyond the 180 dB mitigation zone, SURTASS LFA sonar transmissions shall be immediately delayed or suspended. Transmissions shall not resume earlier than 15 minutes after:

a. All marine mammals or sea turles have left the area of the LFA mitigation and buffer zone; and

b. There is no further detection of any marine mammal or sea turtle within the LFA mitigation and buffer zones as determined by the visual and/or passive or active acoustic monitoring.

(4) The High Frequency Marine Mammal Monitoring (HF/M3) sonar source described in 50 CFR § 218.235 shall be ramped-up to operating levels over a period of no less

than 5 minutes. The HF/M3 source level shall not be increased if a marine mammal or sea turle is detected during ramp-up. The HF/M3 ramp-up may continue once marine mammals or sea turtles are no longer detected. The HF/M3 sonar shall be ramped-up:

a. At least 30 minutes prior to any SURTASS LFA sonar transmissions;

b. Prior to any SURTASS LFA sonar calibrations or testing that are not part of regular SURTASS LFA sonar transmissions described in 50 CFR § 218.230; and

c. Any time after the HF/M3 source has been powered down for more than 2 minutes.

(5) The SURTASS LFA sonar shall not be operated such that the SURTASS LFA sonar sound field exceeds 180 dB (re 1 microPascal$_{rms}$):

a. At a distance of 22 kilometers (km) (12 nautical miles (nmi)) from any coastline, including offshore islands;

b. At a distance of 1 km (0.5 nmi) seaward of the outer perimeter of any designated OBIA (i.e., the Hawaiian Islands Humpback Whale National Marine Sanctuary—Penguin Bank OBIA (OBIA #16) located in the north-central Pacific Ocean) during the biologically important season for that particular area in accordance with 50 CFR § 218.230.

(6) The U.S. Navy shall deliver an annual report no later than 45 days after the expiration of any Letter of Authorization issued for the operation of SURTASS LFA sonar. This report shall include numbers and locations of threatened and endangered species sightings, and all information required by the Letter of Authorization, including the results, if any, of coordination with coastal marine mammal stranding networks. The annual reports shall be submitted to the following NMFS offices: (1) Chief, Permits and Conservation Division, 1315 East-West Highway, Silver Spring, Maryland; and (2) Chief, Endangered Species Act Interagency Cooperation Division, 1315 East-West Highway, Silver Spring, Maryland.

(7) The Navy shall collect specific data on any apparent avoidance reactions of threatened or endangered species in response to exposure to LFA sonar transmissions, including the distance from the LFA sonar transmission, conditions of the exposure (location coordinates, depth of the species, time of day, ocean conditions, the animal's behavior before and after the exposure, and estimates of the received levels that elicited the response). These data must be reported in the annual reports described in condition 6 (above).

(8) If the Navy's monitoring programs identify any threatened or endangered species that demonstrate acute effects in response to exposure to LFA sonar transmissions, such as injury or death, the Navy shall immediately initiate the source shut-down protocol for the sonar system.

(9) The U.S. Navy shall carry out all mitigation, monitoring and reporting requirements contained in the Letters of Authorization issued under section 101(a)(5)(A) of the Marine Mammal Protection Act.

(10) Systematically observe SURTASS LFA sonar operations for injured or disabled marine mammals and monitor the principal marine mammal stranding networks and other media to correlate analysis of any whale strandings that could potentially be associated with SURTASS LFA sonar operations.

These reasonable and prudent measures, with their implementing terms and conditions, are designed to minimize the impact of incidental take that might otherwise result from the proposed action. If, during the course of the action, the level of incidental take specified in this Incidental Take Statement is exceeded, NMFS' Permits and Conservation Division and the U.S. Navy must immediately reinitiate consultation and review the reasonable and prudent measures provided. NMFS' Permits and Conservation Division and U.S. Navy must immediately provide an explanation, in writing, of the causes of any take and discuss possible modifications to the reasonable and prudent measures with NMFS Endangered Species Act Interagency Cooperation Division.

9 CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. Specifically, conservation recommendations are suggestions regarding discretionary measures to minimize or avoid adverse effects of a proposed action on listed species or critical habitat or regarding the development of information (50 CFR § 402.02).

1. The HF/M3 source should be ramped-up to operating levels over a period of no less than 5 minutes:

    a. No later than 30 minutes before the first SURTASS LFA sonar transmission;

    b. Prior to any SURTASS LFA sonar calibrations or testing that are not part of regular SURTASS LFA sonar transmissions; and

    c. Anytime after the HF/M3 source has been powered down for a period of time greater than two minutes.

2. SURTASS LFA sonar would be operated such that the sound field does not exceed 180 dB (re 1 μPa_{rms}):

   a. At a distance of 22 kilometers (12 nautical miles) from any coastline, including offshore islands (coastal standoff range);

   b. Within in 1-km of the seaward boundary of the 22 designated OBIAs in accordance with 50 CFR § 218.234(f), which include the following National Marine Sanctuaries and critical habitats: (1) Monterey Bay, (2) Gulf of the Farallones, (3) Cordell Bank, (4) Stellwagen Bank, (5) Hawaiian Islands Humpback Whale (Penguin Bank), (6) North Atlantic Right Whale, and (7) North Pacific Right Whale;

   c. Within 37.4 km (23 nmi) of the coastline during the months of December, January, March, April, and May of each year in the Olympic Coast National Marine Sanctuary.

3. SURTASS LFA sonar would be operated such that the sound field does not exceed 145 dB re: 1 μPa_{rms} within in known dive sites to include the following National Marine sanctuaries: (1) Florida Keys, (2) Gray's Reef, (3) Flower Garden Banks, (4) Monitor, and (5) Channel Islands.

NMFS Permits and Conservation Division has indicated that these actions are being taken and are effective for mitigation. NMFS asks the U.S. Navy to notify us of any changes to the implementation of these conservation recommendations.

10 REINITIATION OF CONSULTATION

This concludes formal consultation on the U.S. Navy's proposed use of Surveillance Towed Array Sensor System Low Frequency Active Sonar and NMFS' proposed issuance of four Letters of Authorization that would allow the Navy to "take" marine mammals incidental to its employment of the Surveillance Towed Array Sensor System Low Frequency Active Sonar System from 15 August 2014 through 14 August 2015, pursuant to the provisions of section 7 of the Endangered Species Act and Marine Mammal Protection Act. As provided in 50 CFR § 402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded; (2) new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this opinion; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action.

11 INFORMATION QUALITY ACT

Section 515 of the Treasury and General Government Appropriations Act for Fiscal Year 2001 (Public Law 106-554, AKA the Data Quality Act or Information Quality Act) directed the Office of Management and Budget (OMB) to issue government-wide guidelines that "provide policy and procedural guidance to federal agencies for ensuring and maximizing the quality, objectivity, utility, and integrity of information (including statistical information) disseminated by federal agencies." OMB complied by issuing guidelines which direct each federal agency to 1) issue its own guidelines; 2) establish administrative mechanisms allowing affected persons to seek and obtain correction of information that does not comply with the OMB 515 Guidelines or the agency guidelines; and 3) report periodically to OMB on the number and nature of complaints received by the agency and how the complaints were handled. The OMB Guidelines can be found at: http://www.whitehouse.gov/omb/fedreg/reproducible2.pdf

The Department of Commerce Guidelines can be found at:
http://ocio.os.doc.gov/ITPolicyandPrograms/Information_Quality/index.htm

The NOAA Section 515 Information Quality Guidelines, created with input and reviews from each of the components of NOAA Fisheries, went into effect on October 1, 2002. The NOAA Information Quality Guidelines are posted on the NOAA Office of the Chief Information Officer Webpage. http://www.cio.noaa.gov/Policy_Programs/info_quality.html

12 REFERENCES

Abbott, R., & Bing-Sawyer, E. (2002). Assessment of pile driving impacts on the Sacramento blackfish (*Orthodon microlepidotus*): Draft report prepared for California Department of Transportation District 4.

Abernathy, K. (1999). *Foraging ecology of Hawaiian monk seals at French Frigate Shoals, Hawaii.* (M.S. Thesis), University of Minnesota, St. Paul, MN.

Aburto, A., Rountry, D. J., & Danzer, J. L. (1997). Behavioral responses of blue whales to active signals (pp. 95). San Diego, CA: Naval Command, Control and Ocean Surveillance Center, RDT&E division.

Acevedo-Whitehouse, K., & Duffus., A. L. J. (2009). Effects of environmental change on wildlife health. *Philosophical Transactions of the Royal Society of London B Biological Sciences, 364*(1534), 3429-3438.

Acevedo, A. (1991). Interactions between boats and bottlenose dolphins, *Tursiops truncatus*, in the entrance to Ensenada de la Paz, Mexico. *Aquatic Mammals, 17*(3), 120-124.

Achinstein, P. (2001). *The Book of Evidence*. New York, New York: Oxford University Press.

Addison, D. S. (1997). Sea turtle nesting on Cay Sal, Bahamas, recorded June 2-4, 1996. *Bahamas Journal of Science, 5*, 34-35.

Addison, D. S., & Morford, B. (1996). Sea turtle nesting activity on the Cay Sal Bank, Bahamas. *Bahamas Journal of Science, 3*, 31-36.

Agler, B. A., Beard, J. A., Bowman, R. S., Corbett, H. D., Frohock, S. E., Hawvermale, M. P., . . . Seipt, I. E. (1990). Fin whale (*Balaenoptera physalus*) photographic identification: Methodology and preliminary results from the western North Atlantic. *Report of the International Whaling Commission Special Issue, 12*, 349-356.

Aguayo, L. A. (1974). Baleen whales off continental Chile. In W. E. Schevill (Ed.), *The Whale Problem: A Status Report* (pp. 209-217). Cambridge, Massachusetts: Harvard University Press.

Aguilar, A. (1983). Organochlorine pollution in sperm whales, *Physeter macrocephalus*, from the temperate waters of the eastern North Atlantic. *Marine Pollution Bulletin, 14*(9), 349-352.

Aguilar, A. (2002). Fin whale, Balaenoptera physalus: Encyclopedia of Marine Mammals. San Diego: Academic Press.

Aguilar, A., & Borrell, A. (1988). Age- and sex-related changes in organochlorine compound levels in fin whales (*Balaenoptera physalus*) from the Eastern North Atlantic. *Marine Environmental Research, 25*(1988?), 195-211.

Aguilar, A., & Borrell, A. (1994). Assessment of organochlorine pollutants in cetaceans by means of skin and hypodermic biopsies *Non-destructive Biomarkers in Vertebrates. C. Fossi and C Leonzio (eds.). Lewis Publishers, CRC Press, Boca Ration, FL. p.245-267.*

Aguilar, A., Forcada, J., Arderiu, A., Gazo, M., & Silvani, L. (1999). Status and conservation threats of small cetaceans in Spanish waters. *European Research on Cetaceans, 13*, 75.

Aguilar, A., & Lockyer, C. H. (1987). Growth, physical maturity, and mortality of fin whales (*Balaenoptera physalus*) inhabiting the temperate waters of the northeast Atlantic. *Canadian Journal of Zoology, 65*, 253-264.

Aguilar, R., Mas, J., & Pastor, X. (1995). *Impact of Spanish swordfish longline fisheries on the loggerhead sea turtle Caretta caretta population in the western Mediterranean*. Paper

presented at the Proceedings of the Twelfth Annual Workshop on Sea Turtle Biology and Conservation, Jekyll Island, Georgia.

Aguilar Soto, N., Johnson, M., Madsen, P. T., Tyack, P. L., Bocconcelli, A., & Borsani, J. F. (2006). Does intense ship noise disrupt foraging in deep-diving Cuvier's beaked whales (*Ziphius cavirostris*)? *Marine Mammal Science, 22*(3), 690-699.

Aguirre, A. A., Spraker, T. R., Balazs, G. H., & Zimmerman, B. (1998). Spirochidiasis and fibropapillomatosis in green turtles of the Hawaiian Islands. *journal of wildlife diseases, 34*, 91-98.

Aicken, W., Clements, E., Harland, E., Healy, S., Smith, G., Ward, P., . . . Pierpont, C. (2005). STUFT2 Trial: Environmental protection data analysis report *Unpublished report prepared by QuinetiQ Limted for the United Kingdon Ministry of Defense*. Hampshire, United Kingdom.

Al-Bahry, S. N., Mahmoud, I. Y., Al-Amri, I. S., Ba-Omar, T. A., Melgheit, K. O., & Al-Kindi, A. Y. (2009). Ultrastructural features and elemental distribution in eggshell during pre and post hatching periods in the green turtle, *Chelonia mydas* at Ras Al-Hadd, Oman. *Tissue and Cell, 41*(3), 214-221. doi: 10.1016/j.tice.2008.11.002

Alava, J. J., Keller, J. M., Kucklick, J. R., Wyneken, J., Crowder, L., & Scott, G. I. (2006). Loggerhead sea turtle (*Caretta caretta*) egg yolk concentrations of persistent organic pollutants and lipid increase during the last stage of embryonic development. *Science of the Total Environment, 367*(1), 170-181.

Allen, B. M., & Angliss, R. P. (2010). Alaska marine mammal stock assessments, 2009 (pp. 287): National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center.

Allen, B. M., & Angliss, R. P. (2011). Alaska marine mammal stock assessments, 2010 (pp. 301): NMFS.

Allen, K. R. (1980). *Conservation and Management of Whales*. Seattle, Washington: University of Washington Press.

Allen, M. S. (2007). Three millennia of human and sea turtle interactions in Remote Oceania. *Coral Reefs, 26*(4), 959-970. doi: 10.1007/s00338-007-0234-x

Anan, Y., Kunito, T., Watanabe, I., Sakai, H., & Tanabe, S. (2001). Trace element accumulation in hawksbill turtles (Eretmochelys imbricata) and green turtles (Chelonia mydas) from Yaeyama Islands, Japan. *Environmental Toxicology and Chemistry, 20*(12), 2802-2814.

Anderson, J. J. (2000). A vitality-based model relating stressors and environmental properties to organism survival. *Ecological Monographs, 70*(3), 445-470.

Andrade, A. L. V., Pinedo, M. C., & Barrreto., A. S. (2001). Gastrointestinal parasites and prey items from a mass stranding of false killer whales, Pseudorca crassidens, in Rio Grande do Sul, Southern Brazil. *Revista Brasileira de Biologia, 61*(1), 55-61.

Andrew, R. K., Howe, B. M., Mercer, J. A., & Dzieciuch, M. A. (2002). Ocean ambient sound: comparing the 1960s with the 1990s for a receiver off the California coast. *Acoustics Research Letters Online, 3*(2), 65-70.

Andrews, R. C. (1914). Monographs of the Pacific Cetacea. I: The California gray whale (Rhachianectes glaucus Cope). (Eschrichtius robustus). *Memoirs of the American Museum of Natural History New Series, 1*(5), 227-287,+228pls&txt.

Andrews, R. C. (1916). The sei whale (Balaenoptera borealis Lesson). *Memoirs of the American Museum of Natural History, New Series, 1*(6), 291-388.

Angliss, R. P., & Allen, B. M. (2009). Alaska marine mammal stock assessments, 2008.

Angliss, R. P., & Lodge, K. L. (2004). Alaska marine mammal stock assessments, 2003 *NOAA Technical Memorandum* (pp. 230): U.S. Department of Commerce.

Angliss, R. P., & Outlaw, R. B. (2005). Alaska marine mammal stock assessments, 2005 *NOAA Technical Memorandum* (pp. 250): U.S. Department of Commerce.

Angliss, R. P., & Outlaw, R. B. (2008). Alaska marine mammal stock assessments, 2007 *NOAA Technical Memorandum* (pp. 252): Department of Commerce.

Aprill, M. L. (1994). *Visitation and predation of the olive ridley sea turtle, Lepidochelys olivacea, at nest sites in Ostional, Costa Rica*. Paper presented at the Proceedings of the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation.

Aprill, M. L. (1994). Visitation and predation of the olive ridley sea turtle, *Lepidochelys olivacea*, at nest sites in Ostional, Costa Rica. *Pp.3-6 In: K.A. Bjorndal, A.B. Bolten, D.A. Johnson, and P.J. Eliazer (Compilers), Proceedings of the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation. Department of Commerce, NOAA Technical Memorandum NMFS-SEFSC-351 323p.*

Arauz, R. (2001). *Impact of high seas longline fishery operations on sea turtle populations in the Exclusive Economic Zone (EEZ) of Costa Rica - a second look*. Paper presented at the 21st Annual Symposium on Sea Turtle Biology and Conservation, Philadelphia, PA.

Arcangeli, A., & Crosti, R. (2009). The short-term impact of dolphin-watching on the behaviour of bottlenose dolphins (*Tursiops truncatus*) in western Australia. *Journal of Marine Animals and their Ecology, 2*(1), 3-9.

Archer, F. I., Mesnick, S. L., & Allen, A. C. (2010). Variation and predictors of vessel response behavior in a tropical dolphin community (pp. 59): National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Archer, F. I., Redfern, J. V., Gerrodette, T., Chivers, S. J., & Perrin, W. F. (2010). Estimation of relative exposure of dolphins to fishery activity. *Marine Ecology Progress Series, 410*, 245-255.

Arenas, P., & Hall, M. (1991a). The association of sea turtles and other pelagic fauna with floating objects in teh eastern tropical Pacific Ocean. *Pp.7-10 In: M. Salmon and J. Wyneken (Compilers), Proceedings of the Eleventh Annual Symposium on Sea Turtle Biology and Conservation. Department of Commerce, NOAA Technical Memorandum NMFS-SEFSC-302. 195p.*

Arenas, P., & Hall, M. (1991b). *The association of sea turtles, and other pelagic fauna with floating objects in the eastern tropical Pacific Ocean.* Paper presented at the The Eleventh Annual Workshop on Sea Turtle Biology and Conservation.

Arendt, M. D., Segars, A. L., Byrd, J. I., Boynton, J., Schwenter, J. A., Whitaker, J. D., & Parker, L. (2012). Migration, distribution, and diving behavior of adult male loggerhead sea turtles (Caretta caretta) following dispersal from a major breeding aggregation in the Western North Atlantic. *Marine Biology, 159*(1), 113-125. doi: 10.1007/s00227-011-1826-0

Arnbom, T., Papastavrou, V., Weilgart, L. S., & Whitehead, H. (1987). Sperm whales react to an attack by killer whales. *Journal of Mammalogy, 68*(2), 450-453.

Arndt, D. S., Baringer, M. O., Johnson, M. R., Alexander, L. V., Diamond, H. J., Fogt, R. L., . . . Willett, K. M. (2010). State of the climate in 2009. *Bulletin of the American Meteorological Society, 91*(7), S1-S224.

Atkinson, A., Siegel, V., Pakhomov, E., & Rothery, P. (2004). Long-term decline in krill stock and increase in salps within the Southern Ocean. *Nature, 432*, 100-103.

Au, W. W. L. (1993). *The Sonar of Dolphins*. New York: Springer-Verlag.

Au, W. W. L., Carder, D. A., Penner, R. H., & Scronce., B. L. (1985). Demonstration of adaptation in beluga whale echolocation signals. (Delphinapterus leucas). *Journal of the Acoustical Society of America, 77*(2), 726-730.

Au, W. W. L., Floyd, R. W., Penner, R. H., & Murchison., A. E. (1974). Measurement of echolocation signals of the Atlantic bottlenose dolphin, Tursiops truncatus Montagu in open waters. *Journal of the Acoustical Society of America, 56*(4), 1280-1290.

Au, W. W. L., & Green, M. (2000). Acoustic interaction of humpback whales and whale-watching boats. *Marine Environmental Research, 49*(5), 469-481.

Au, W. W. L., & Green., M. (2000). Acoustic interaction of humpback whales and whale-watching boats. *Marine Environmental Research, 49*(5), 469-481.

Au, W. W. L., Pack, A. A., Lammers, M. O., Herman, L. M., Deakos, M. H., & Andrews, K. (2006). Acoustic properties of humpback whale songs. *Journal of Acoustical Society of America, 120*(August 2006), 1103-1110.

Au, W. W. L., & Pawloski, J. L. (1989). Detection of noise with rippled spectra by the Atlantic bottlenose dolphin. *Journal of the Acoustical Society of America, 86*(2), 591-596.

Au, W. W. L., Popper, A. N., & Fay, R. R. (2000). *Hearing by whales and dolphins*. New York: Springer-Verlag.

Baier, C. T., & Napp, J. M. (2003). Climate-induced variability in Calanus marshallae populations. *Journal of Plankton Research, 25*, 771-782.

Bailey, A. M. (1952). The Hawaiian monk seal. *Museum Pictorial, Denver Museum of Natural History, 7*(1-32).

Bailey, K. M., Piatt, J. F., Royer, T. C., Macklin, S. A., Reed, R. K., Shima, M., . . . Wooster, W. S. (1995). ENSO events in the northern Gulf of Alaska, and effects on selected marine fisheries. *CalCOFI Reports, 36*, 78-96.

Baird, R. W. (2009). *A review of false killer whales in Hawaiian waters: biology, status, and risk factors*: Cascadia Research Collective.

Baird, R. W. (2009). A review of false killer whales in Hawaiian waters: Biology, status, and risk factors.: U.S. Marine Mammal Commission.

Baird, R. W., & Gorgone, A. M. (2005). False killer whale dorsal fin disfigurements as a possible indicator of long-line fishery interactions in Hawaiian waters. *Pacific Science, 59*(4), 593 - 601.

Baird, R. W., & Gorgone, A. M. (2005). False Killer Whale Dorsal Fin Disfigurements as a Possible Indicator of Long-Line Fishery Interactions in Hawaiian Waters 1. *Pacific Science, 59*(4), 593-601.

Baird, R. W., Gorgone, A. M., McSweeney, D. J., Webster, D. L., Salden, D. R., Deakos, M. H., . . . Mahaffy, S. D. (2008). False killer whales (*Pseudorca crassidens*) around the main Hawaiian Islands: Long-term site fidelity, inter-island movements, and association patterns. *Marine Mammal Science, 24*(3), 591-612. doi: DOI 10.1111/j.1748-7692.2008.00200.x

Baird, R. W., Hanson, M., Schorr, G. S., Webster, D. L., McSweeney, D. J., Gorgone, A. M., . . . Andrews, R. D. (2012). Range and primary habitats of Hawaiian insular false killer whales: informing determination of critical habitat. *Endangered Species Research, 18*(1), 47-61.

Baird, R. W., Hanson, M. B., Schorr, G. S., Webster, D. L., McSweeney, D. J., Gorgone, A. M., . . . Andrews, R. D. (2012). Range and primary habitats of Hawaiian insular false killer

whales: informing determination of critical habitat. *Endangered Species Research, 18*(1), 47-61. doi: 10.3354/esr00435

Baird, R. W., Schorr, G. S., Webster, D. L., McSweeney, D. J., Hanson, M., & Andrews, R. D. (2010). Movements and habitat use of satellite-tagged false killer whales around the main Hawaiian Islands: DTIC Document.

Baird, R. W., Schorr, G. S., Webster, D. L., McSweeney, D. J., Hanson, M. B., & Andrews, R. D. (2011). Movements and Spatial Use of False Killer Whales in Hawaii: Satellite Tagging Studies in 2009: DTIC Document.

Baird, R. W., Schorr, G. S., Webster, D. L., Mcsweeney, D. J., Hanson, M. B., & Andrews., R. D. (2010). Movements and habitat use of satellite-tagged false killer whales around the main Hawaiian Islands. *Endangered Species Research, 10*(1), 107-121.

Baird, R. W., Webster, D. L., Schorr, G. S., McSweeney, D. J., & Barlow, J. (2008). Diel variation in beaked whale diving behavior. *Marine Mammal Science, 24*(3), 630-642. doi: 10.1111/j.1748-7692.2008.00211.x

Baker, C. S. (1985). *The population structure and social organization of humpback whales (Megaptera novaeangliae) in the central and eastern North Pacific.* (Ph.D. dissertation), University of Hawaii, Honolulu.

Baker, C. S., Dalebout, M. L., Lento, G. M., & Funahashi, N. (2002). Gray whale products sold in commercial markets along the Pacific coast of Japan. *Marine Mammal Science, 18*(1), 6.

Baker, C. S., & Herman, L. M. (1987). Alternative population estimates of humpback whales (*Megaptera novaeangliae*) in Hawaiian waters. *Canadian Journal of Zoology, 65*(11), 2818-2821.

Baker, C. S., Herman, L. M., Bays, B. G., & Bauer, G. B. (1983). The impact of vessel traffic on the behavior of humpback whales in southeast Alaska: 1982 season (pp. 86): National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center, National Marine Mammal Laboratory.

Baker, C. S., & Herman., L. M. (1987). Alternative population estimates of humpback whales (Megaptera novaeangliae) in Hawaiian waters. *Canadian Journal of Zoology, 65*(11), 2818-2821.

Baker, J. D. (2008). Variation in the relationship between offspring size and survival provides insight into causes of mortality in Hawaiian monk seals. *Endang Species Res, 5*, 55-64. doi: 10.3354/esr00122

Baker, J. D., Harting, A. L., Wurth, T. A., & Johanos, T. C. (2011). Dramatic shifts in Hawaiian monk seal distribution predicted from divergent regional trends. *Marine Mammal Science, 27*(1), 78-93.

Baker, J. D., Polovina, J. J., & Howell, E. A. (2007). Effect of variable oceanic productivity on the survival of an upper trophic predator, the Hawaiian monk seal Monachus schauinslandi. *Marine Ecology Progress Series, 346*, 277-283.

Baker, J. D., & Thompson, P. M. (2007). Temporal and spatial variation in age-specific survival rates of a long-lived mammal, the Hawaiian monk seal. *Proceedings of the Royal Society B: Biological Sciences, 274*(1608), 407-415.

Balazs, G. H., Puleloa, W., Medeiros, E., Murakawa, S. K. K., & Ellis, D. M. (1998). *Growth rates and incidence of fibropapillomatosis in Hawaiian green turtles utilizing coastal foraging pastures at Palaau, Molokai*. Paper presented at the Seventeenth Annual Sea Turtle Symposium.

Baldwin, R., Hughes, G. R., & Prince, R. I. T. (2003). Loggerhead turtles in the Indian Ocean. In A. B. Bolten & B. E. Witherington (Eds.), *Loggerhead Sea Turtles* (pp. 218-232). Washington, D. C.: Smithsonian Institution Press.

Baldwin, R. M. (1992). Nesting turtles on Masirah Island: Management issues, options, and research requirements: Ministry of Regional Municipalities and Environment, Oman.

Banner, A. (1967). Evidence of sensitivity to acoustic displacements in the lemon shark, *Negaprion brevirostris* (Poey). In P. H. Cahn (Ed.), *Lateral Line Detectors* (pp. 265-273). Bloomington, Indiana: Indiana University Press.

Barbieri, E. (2009). Concentration of heavy metals in tissues of green turtles (Chelonia mydas) sampled in the Cananeia estuary, Brazil. *Brazilian Journal of Oceanography, 57*(3), 243-248.

Barlow, J. (1988). Harbor porpoise, *Phocoena phocoena*, abundance estimation for California, Oregon, and Washington: I. Ship Surveys. *Fishery Bulletin, 86*(3), 417-432.

Barlow, J. (1995). Abundance of cetaceans in California waters. Part I: Ship surveys in summer and fall of 1991. *Fishery Bulletin, 93*(1), 1-14.

Barlow, J. (1997). Preliminary estimates of cetacean abundance off California, Oregon, and Washington based on a 1996 ship survey and comparisons of passing and closing modes (pp. 25). La Jolla, California: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Barlow, J. (2003). Preliminary estimates of the abundance of cetaceans along the U.S. West Coast: 1991-2001 *Administrative Report* (pp. 31): National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center

Barlow, J. (2006). Cetacean abundance in Hawaiian waters estimated from a summer/fall survey in 2002. *Marine Mammal Science, 22*(2), 446-464.

Barlow, J., Forney, K. A., Hill, P. S., Robert L. Brownell, J., Caretta, J. A., Demaster, D. P., . . . Reeves, R. R. (1997a). U.S. Pacific Marine Mammal Stock Assessments: 1996 (pp. 224): Southwest Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration.

Barlow, J., Forney, K. A., Hill, P. S., Robert L. Brownell, J., Caretta, J. A., Demaster, D. P., . . . Reeves, R. R. (1997b). U.S. Pacific Marine Mammal Stock Assessments: 1996 (pp. 224): Southwest Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration.

Bartol, S. M., & Ketten, D. R. (2006). Turtle and tuna hearing. *Pp.98-103 In: Swimmer, Y. and R. Brill (Eds), Sea Turtle and Pelagic Fish Sensory Biology: Developing Techniques to Reduce Sea Turtle Bycatch in Longline Fisheries. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-PIFSC-7*.

Bartol, S. M., Musick, J. A., & Lenhardt, M. (1999a). Auditory Evoked Potentials of the Loggerhead Sea Turtle (Caretta caretta). *Copeia, 3*, 836-840.

Bartol, S. M., Musick, J. A., & Lenhardt, M. (1999b). Evoked Potentials of the Loggerhead Sea Turtle (*Caretta caretta*). *Copeia, 3*, 836-840.

Barton, B. T., & Roth, J. D. (2008). Implications of intraguild predation for sea turtle nest protection. *Biological Conservation, 181*(8), 2139-2145.

Bass, A. L., Epperly, S. P., Braun, J., Owens, D. W., & Patterson, R. M. (1998). Natal origin and sex ratios of foraging sea turtles in the Pamlico-Albemarle Estuarine Complex *NOAA Technical Memorandum* Miami, Florida: U.S. Department of Commerce, National

Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

Bassett, C., Polagye, B., Holt, M., & Thomson, J. (2012). A vessel noise budget for Admiralty Inlet, Puget Sound, Washington (USA). *Journal of the Acoustical Society of America, 132*(6), 3706-3719.

Bauer, G., & Herman, L. M. (1986). Effects of vessel traffic on the behavior of humpback whales in Hawaii (pp. 151). Honolulu, Hawaii: National Marine Fisheries Service.

Bauer, G. B. (1986). *The behavior of humpback whales in Hawaii and modifications of behavior induced by human interventions. (Megaptera novaeangliae).* (Ph.D. dissertation), University of Hawaii. 314p.

Baumgartner, M. F., Cole, T. V. N., Clapham, P. J., & Mate, B. R. (2003). North Atlantic right whale habitat in the lower Bay of Fundy and on the SW Scotian Shelf during 1999-2001. *Marine Ecology Progress Series, 264*, 137-154.

Baumgartner, M. F., & Mate, B. R. (2003). Summertime foraging ecology of North Atlantic right whales. *Marine Ecology Progress in Series, 264*, 123–135.

Baylis, H. A. (1928). Parasites of whales. *Natural History Magazine, 1*(2), 55-57.

Beale, C. M., & Monaghan, P. (2004). Behavioural responses to human disturbance: A matter of choice? *Animal Behaviour, 68*(5), 1065-1069.

Beale, C. M., & Monaghan, P. (2004). Human disturbance: people as predation-free predators? *Journal of Applied Ecology, 41*, 335-343.

Beamish, R., Noakes, D., McFarlane, G., Klyashtorin, L., Ivanov, V., & Kurashov, V. (1999). The regime concept and natural trends in the production of Pacific salmon. *Canadian Journal of Fisheries and Aquatic Sciences, 56*(3), 516-526.

Beavers, S. C., & Cassano, E. R. (1996). Movements and dive behavior of a male sea turtle (Lepidochelys olivacea) in the eastern tropical Pacific. *Journal of Herpetology, 30*(1), 97-104.

Bellido, J. J., Báez, J. C., Castillo, J. J., Pinto, F., Martín, J. J., Mons, J. L., & Real, R. (2010). Loggerhead strandings and captures along the southern Spanish Coast: Body size–based differences in natural versus anthropogenic injury. *Chelonian Conservation and Biology, 9*(2), 276-282. doi: 10.2744/ccb-0802.1

Benda-Beckmann, A. M., Wensveen, P. J., Kvadsheim, P. H., Lam, F. P. A., Miller, P. J. O., Tyack, P. L., & Ainslie, M. A. (2014). Modeling effectiveness of gradual increases in source level to mitigate effects of sonar on marine mammals. *Conservation Biology, 28*(1), 119-128.

Benhamou, S., Sudre, J., Bourjea, J., Ciccione, S., De Santis, A., & Luschi, P. (2011). The Role of Geomagnetic Cues in Green Turtle Open Sea Navigation. *PLoS ONE, 6*(10). doi: e26672 10.1371/journal.pone.0026672

Benson, A. J., & Trites, A. W. (2002). Ecological effects of regime shifts in the Bering Sea and eastern North Pacific Ocean. *Fish and Fisheries, 3*(2), 95-113.

Benson, S. R., Croll, D. A., Marinovic, B. B., Chavez, F. P., & Harvey, J. T. (2002). Changes in the cetacean assemblage of a coastal upwelling ecosystem during El Niño 1997-98 and La Niña 1999. *Progress in Oceanography, 54*, 279-291.

Benson, S. R., Forney, K. A., Harvey, J. T., Carretta, J. V., & Dutton, P. H. (2007). Abundance, distribution, and habitat of leatherback turtles (*Dermochelys coriacea*) off California, 1990-2003. *Fishery Bulletin, 105*, 337-347.

Berchok, C. L., Bradley, D. L., & Gabrielson, T. B. (2006). St. Lawrence blue whale vocalizations revisited: Characterization of calls detected from 1998 to 2001. *Journal of the Acoustical Society of America, 120*(4), 2340-2354.

Berkson, H. (1967). Physiological adjustments to deep diving in the Pacific green turtle (*Chelonia mydas agassizii*). *Comp Biochem Physiol, 21*(3), 507-524.

Berman-Kowalewski, M., Gulland, F. M. D., Wilkin, S., Calambokidis, J., Mate, B., Cordaro, J., . . . Dover, S. (2010). Association Between Blue Whale (Balaenoptera musculus) Mortality and Ship Strikes Along the California Coast. *Aquatic Mammals, 36*(1), 59-66. doi: 10.1578/am.36.1.2010.59

Berrow, S. D., & Holmes, B. (1999). Tour boats and dolphins: Quantifying the activities of dolphin watching vessels in the Shannon Estuary. *European Research on Cetaceans, 13*, 77-80.

Bérubé, M., Aguilar, A., Dendanto, D., Larsen, F., Sciara, G. N. d., Sears, R., . . . Palsbøll, P. J. (1998). Population genetic structure of North Atlantic, Mediterranean and Sea of Cortez fin whales, *Balaenoptera physalus (*Linnaeus 1758): analysis of mitochondrial and nuclear loci. *Molecular Ecology, 7*, 585-599.

Bérubé, M., R., J. U., Dizon, r. E., Brownell, R. L., & Palsboll, P. J. (2002). Genetic identification of a small and highly isolated population of fin whales (*Balaenoptera physalus*) in the Sea of Cortez, México. *Conservation Genetics, 3*(2), 183-190.

Berzin, A. A., & Rovnin, A. A. (1966a). The distribution and migrations of whales in the northeastern part of the Pacific, Chuckchee and Bering Seas *Soviet Research on Marine Mammals in the Far East. K. I. Panin (ed.). p.103-136*.

Berzin, A. A., & Rovnin, A. A. (1966b). Distribution, and migration of whales in the northeastern part of the Pacific Ocean, Bering, and Chukchi seas. *Izv. TINRO, 58*, 179-207.

Berzin, A. A., & Vladimirov, V. L. (1989). Current distribution and abundance of cetaceans in the Sea of Okhotsk [Pacific Ocean]. *Biologiya Morya-Vladivostok, 0*(2), 15-23.

Berzin, A. A., & Vladimirov, V. L. (1990). Recent distribution and abundance of cetaceans in the Sea of Okhotsk. *Soviet Journal of Marine Biology, 15*(2), 84-90.

Berzin, A. A., & Yablokov, A. V. (1978). Abundance and population structure of important exploited cetacean species of the world ocean. *Zoologichesky Zhurnal*(12), 1771-1785.

Best, P. B. (1987). Estimates of the landed catch of right (and other whalebone) whales in the American fishery, 1805-1909. *Fishery Bulletin, 85*(3), 403-418.

Best, P. B., Canham, P. A. S., & Macleod, N. (1984). Patterns of reproduction in sperm whales, *Physeter macrocephalus*. *Report of the International Whaling Commission Special Issue, 6*, 51-79. Reproduction in Whales, Dolphins and Porpoises. Proceedings of the Conference Cetacean Reproduction Estimating Parameters For stock Assessment and Management.

Best, P. B., & Lockyer, C. H. (2002). Reproduction, growth and migrations of sei whales *Balaenoptera borealis* off the west coast of South Africa in the 1960s. *South African Journal of Marine Science, 24*, 111-133.

Biassoni, N., Miller, P. J. O., & Tyack., P. L. (2001). *Humpback whales, Megaptera novaeangliae, alter their song to compensate for man-made noise.* Paper presented at the Fourteenth Biennial Conference on the Biology of Marine Mammals, 28 November-3 December Vancouver Canada. p.24.

Bjorndal, K. A. (1997). Foraging ecology and nutrition of sea turtles *The Biology of Sea Turtles* (pp. 199–231). Boca Raton, Florida: CRC Press.

Bjorndal, K. A., & Bolten, A. B. (2000). Proceedings on a workshop on accessing abundance and trends for in-water sea turtle populations: NOAA.

Blecha, F. (2000). Immune system response to stress. In G. P. Moberg & J. A. Mench (Eds.), *The Biology of Animal Stress* (pp. 111-122): CABI Publishing.

Blokhin, S. A. (1985). Investigations of gray whales taken off Chukotka in 1983. *Reports of the International Whaling Commission, 35*, 371-374.

Boggs, C. H. (1992). Depth, capture time, and hooked longevity of longline-caught pelagic fish: timing bites of fish with chips. *Fishery Bulletin, 90*, 642-658.

Bolten, A. B., K.A. Bjorndal, & Martins, H. R. (1994). Life history model for the loggerhead sea turtle (*Caretta caretta*) populations in the Atlantic: Potential impacts of a longline fishery. . *Pp.48-55 In: Balazs, G.J. and S.G. Pooley (eds), Research Plan to Assess Marine Turtle Hooking Mortality: Results of an Expert Workshop Held in Honolulu, Hawaii, November 16-18, 1993. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SEFSC-201, pp.48-55.*

Bolten, A. B., Wetherall, J. A., Balazs, G. H., & Pooley, S. J. (1996). Status of marine turtles in the Pacific Ocean relevant to incidental take in the Hawaii-based pelagic longline fishery *NOAA Technical Memorandum*

Borrell, A. (1993). PCB and DDTs in blubber of cetaceans from the northeastern North Atlantic. *Marine Pollution Bulletin, 26*(3), 146.

Borrell, A., & Aguilar, A. (1987). Variations in DDE percentage correlated with total DDT burden in the blubber of fin and sei whales. *Marine Pollution Bulletin, 18*(2), 70-74.

Bouchard, S., Moran, K., Tiwari, M., Wood, D., Bolten, A., Eliazar, P., & Bjorndal, K. (1998). Effects of exposed pilings on sea turtle nesting activity at Melbourne Beach, Florida. *Journal of Coastal Research, 14*(4), 1343-1347.

Bourgeois, S., Gilot-Fromont, E., Viallefont, A., Boussamba, F., & Deem, S. L. (2009). Influence of artificial lights, logs and erosion on leatherback sea turtle hatchling orientation at Pongara National Park, Gabon. *Biological Conservation, 142*(1), 85-93. doi: 10.1016/j.biocon.2008.09.028

Bowen, B. W., Abreu-Grobois, F. A., Balazs, G. H., Kamezaki, N., Limpus, C. J., & Ferl., R. J. (1995). TransPacific migrations of the loggerhead turtle (Caretta caretta) demonstrated with mitochondrial DNA markers. *Proceedings of the National Academy of Sciences USA, 92*, 3731-3734.

Bowen, B. W., Bass, A. L., Chow, S.-M., Bostrom, M., Bjorndal, K. A., Bolten, A. B., . . . Dutton, P. H. (2004). Natal homing in juvenile loggerhead turtles (*Caretta caretta*). *Molecular Ecology, 13*, 3797–3808.

Bowen, B. W., Bass, A. L., Soares, L., & Toonen, R. J. (2005). Conservation implications of complex population structure lessons from the loggerhead turtle (Caretta caretta). *Molecular Ecology, 14*, 2389-2402.

Bowen, S. L. (1974). Probable extinction of the Korean stock of the gray whale (*Eschrichtius robustus*). *Journal of Mammalogy, 55*(1), 208-209.

Bowles, A. E., Smultea, M., Wursig, B., Demaster, D. P., & Palka, D. (1994). Relative abundance and behavior of marine mammals exposed to transmissions from the Heard Island feasibility test. *Journal of the Acoustical Society of America, 96*(4), 2469-2484.

Boyd, I. L. (2002). Antarctic marine mammals. *Encyclopedia of Marine Mammals. W. F. Perrin, B. Wursig AND J. G. M. Thewissen (eds.). p.30-36. Academic Press, San Diego, CA. 1414pgs.*

Boyle, M. C., FitzSimmons, N. N., Limpus, C. J., Kelez, S., Velez-Zuazo, X., & Waycott, M. (2009). Evidence for transoceanic migrations by loggerhead sea turtles in the southern Pacific Ocean. *Proceedings of the Royal Society B-Biological Sciences, 276*(1664), 1993-1999. doi: 10.1098/rspb.2008.1931

Bradford, A. L., Forney, K. A., Oleson, E. M., & Barlow, J. (2012). Line-transect Abundance Estimates of False Killer Whales (*Pseudorca crassidens*) in the Pelagic Region of the Hawaiian Exclusive Economic Zone and in the Insular Waters of the Northwestern Hawaiian Islands. Honolulu, HI: Pacific Islands Fisheries Science Center, National Marine Fisheries Service, NOAA.

Braham, H. W. (1991). Endangered whales: A status update. A report on the 5-year status of stocks review under the 1978 amendments to the U.S. Endangered Species Act. Seattle, Washington: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center, National Marine Mammal Laboratory.

Braham, H. W., & Rice, D. W. (1984). The right whale, *Balaena glacialis*. *Marine Fisheries Review, 46*(4), 38-44.

Brashares, J. S. (2003). Ecological, behavioral, and life-history correlates of mammal extinctions in West Africa. *Conservation Biology, 17*, 733-743.

Brautigam, A., & Eckert, K. L. (2006). *Turning the Tide: Exploitation, Trade and Management of Marine Turtles in the Lesser Antilles, Central America, Columbia and Venezuela*. Cambridge, UK: TRAFFIC International.

Breiwick, J. M., Reeves, R. R., & Mitchell., E. D. (1993). *Pre-exploitation abundance of right whales off the eastern United States.* Paper presented at the Tenth Biennial Conference on the Biology of Marine Mammals, Galveston, Texas.

Brock, K. A., Reece, J. S., & Ehrhart, L. M. (2009). The effects of artificial beach nourishment on marine turtles: Differences between loggerhead and green turtles. *Restoration Ecology, 17*(2), 297-307. doi: 10.1111/j.1526-100X.2007.00337.x

Brodeur, R. D., Frost, B. W., Hare, S. R., Francis, R. C., & Jr., W. J. I. (1996). Interannual variations in zooplankton biomass in the Gulf of Alaska, and covariation with California Current zooplankton biomass. *CalCOFI Reports, 37*, 80-99.

Brodeur, R. D., Frost, B. W., Hare, S. R., Francis, R. C., & W.J. Ingraham, J. (1996). Interannual variations in zooplankton biomass in the Gulf of Alaska, and covariation with California Current zooplankton biomass. *CalCOFI Reports, 37*, 80-99.

Brodeur, R. D., & Ware, D. M. (1992). Long-term variability in zooplankton biomass in the subarctic Pacific Ocean. *Fisheries Oceanography, 1*(1), 32-38.

Brown, C. H., & Brown, W. M. (1982). Status of sea turtles in the southeastern Pacific: emphasis on Peru. In K. A. Bjorndal (Ed.), *Biology and conservation of sea turtles*. Washington, D.C.: Smithsonian Institution Press.

Brown, C. J., Fulton, E. A., Hobday, A. J., Matear, R. J., Possingham, H. P., Bulman, C., . . . Richardson, A. J. (2009). Effects of climate-driven primary production change on marine food webs: implications for fisheries and conservation. *Global Change Biology, 16*(4), 1194-1212. doi: 10.1111/j.1365-2486.2009.02046.x

Brown, S. G. (1976). Sightings and numbers of blue and humpback whales seen on the Icelandic whaling grounds 1969-74. *Reports of the International Whaling Commission, 26*, 297-299.

Brownell, J., R. L. (2007). Entrapment of western gray whales in Japanese fishing gear: Population threats.: Unpublished paper to the IWC Scientific Committee. 9 pp. Anchorage, AK, May (SC/59/BRG38).

Brownell, J., Robert L., Clapham, P. J., Miyashita, T., & Kasuya., T. (2001). Conservation status of North Pacific right whales. *Journal of Cetacean Research and Management Special Issue, 2*, 269-286.

Brownell Jr., R. L., Clapham, P. J., Miyashita, T., & Kasuya, T. (2001). Conservation status of North Pacific right whales. *Journal of Cetacean Research and Management*(Special Issue 2), 269-286.

Brownell, R. L. (2004). Oil development threats to western gray whales off Sakhalin Island.: Unpublished paper to the IWC Scientific Committee. 10 pp. Sorrento, Italy, July (SC/56/BRG39).

Brownell, R. L., Jr., & Chun, C.-i. (1977). Probable Existence of the Korean Stock of the Gray Whale (Eschrichtius robustus). *Journal of Mammalogy, 58*(2), 237-239.

Brownell, R. L., Jr., Weller, D. W., Wursig, B., Blokhin, S. A., & Burdin., A. M. (1999). The Asian or western gray whale population: Past exploitation, current status, and new threats.: Unpublished paper to the IWC Scientific Committee. 13 pp. Grenada, West Indies, May (SC/51/AS26).

Brownell, R. L., & Weller., D. W. (2002). Prolonged calving intervals in western gray whales: Nutritional stress and pregnancy.: Unpublished paper to the IWC Scientific Committee. Shimonoseki, Japan, April (SC/54/BRG12).

Brownell, R. L. J., Clapham, P. J., Miyashita, T., & Kasuya, T. (2001). Conservation status of north Pacific right whales. *Journal of Cetacean Research and Management (Special issue), 2*, 269-286.

Bryant, P. J., Lafferty, C. M., & Lafferty., S. K. (1984). Reoccupation of Laguna Guerrero Negro, Baja California, Mexico, by gray whales. (*Eschrichtius robustus*). In M. L. Jones, S. L. Swartz & S. Leatherwood (Eds.), *The Gray Whale, Eschrichtius robustus*. New York: Academic Press.

Buck, J. R., & Tyack, P. L. (2000). Response of gray whales to low-frequency sounds. *The Journal of the Acoustical Society of America, 107*, 2774.

Buckland, S. T., & Borchers, D. L. (1993). The design and analysis of sightings surveys for assessing cetacean abundance. *European Research on Cetaceans, 7*, 104-108.

Busnel, R.-G., & Dziedzic, A. (1968). Caracteristiques physiques des signaux acoustiques de Pseudorca crassidens Owen (Cetace Odontocete). *Mammalia, 32*(1), 1-5.

Calambokidis, J. (1997). The humpbacks of Costa Rica. Humpback whales and the California-Costa Rica connection. *Whale-Journal of the Oceanic Society, 1*(1), 4-7,10.

Calambokidis, J., Barlow, J., Ford, J. K. B., Chandler, T. E., & Douglas, A. B. (2009). Insights into the population structure of blue whales in the eastern North Pacific from recent sightings and photographic identification. *Marine Mammal Science, 25*(4), 816-832.

Calambokidis, J., Barlow, J., Ford, J. K. B., Chandler, T. E., & Douglas., A. B. (2009). Insights into the population structure of blue whales in the eastern North Pacific from recent sightings and photographic identification. *Marine Mammal Science, 25*(4), 816-832.

Calambokidis, J., & Chandler., T. (2000). Marine mammal observations and mitigation associated with USGS seismic surveys in the Southern California Bight in 2000.: Cascadia Research Collective report. Prepared for the U.S. Geological Survey. 13pp.

Calambokidis, J., Falcone, E., Douglas, A., Schlender, L., & Huggins, J. (2010). Photographic identification of humpback and blue whales off the U.S. West Coast: results and updated abundance estimates from 2008 field season. Final Report for Contract AB133F08SE2786 from Southwest Fisheries Science Center. 18pp.

Calambokidis, J., Falcone, E. A., Quinn, T. J., Burdin, A. M., Clapham, P. J., Ford, J. K. B., . . . Maloney, N. (2008). SPLASH: Structure of Populations, Levels of Abundance and Status of Humpback Whales in the North Pacific *Final Report* (pp. 57). Seattle, Washington: U.S. Dept of commerce, Western Administrative Center.

Calambokidis, J., Oleson, E., McDonald, M., Burgess, B., Francis, J., Marshall, G., . . . Hildebrand, J. (2003). *Feeding and vocal behavior of blue whales determined through simultaneous visual-acoustic monitoring and deployment of suction-cap attached tags*. Paper presented at the Abstracts of the 15th Biennial Conference on the Biology of Marine Mammals, Greensboro, North Carolina.

Calambokidis, J., Steiger, G. H., Cubbage, J. C., Balcomb, K. C., Ewald, C., Kruse, S., . . . Sears, R. (1990). Sightings and movements of blue whales off central California 1986-88 from photo-identification of individuals. *Report of the International Whaling Commission*(Special Issue 12), 343-348.

Calambokidis, J., Steiger, G. H., Ellifrit, D. K., Troutman, B. L., & Bowlby, C. E. (2004). Distribution and abundance of humpback whales (*Megaptera novaeangliae*) and other marine mammals off the northern Washington coast. *Fishery Bulletin, 102*(4), 563-580.

Calambokidis, J., Steiger, G. H., Evenson, J. R., Flynn, K. R., Balcomb, K. C., Claridge, D. E., . . . Green, G. A. (1996). Interchange and isolation of humpback whales off California and other North Pacific feeding grounds. *Marine Mammal Science, 12*(2), 215-226.

Calambokidis, J., Steiger, G. H., Straley, J. M., Herman, L. M., Cerchio, S., Salden, D. R., . . . Quinn, T. J. (2001). Movements and population structure of humpback whales in the North Pacific. *Marine Mammal Science, 17*(4), 769-794.

Calambokidis, J., Steiger, G. H., Straley, J. M., Quinn, T., Herman, L. M., Cerchio, S., . . . Rasmussen, K. (1997a). Abundance and population structure of humpback whales in the North Pacific basin *Final Contract Report* (pp. 72). La Jolla, CA: Southwest Fisheries Science Center.

Calambokidis, J., Steiger, G. H., Straley, J. M., Quinn, T., Herman, L. M., Cerchio, S., . . . Rasmussen, K. (1997b). Abundance and population structure of humpback whales in the North Pacific basin *Final Contract Report* (pp. 72). La Jolla, CA: Southwest Fisheries Science Center.

CalTrans. (2004). Fisheries and hydroacoustic monitoring program compliance report *Prepared by Strategic Environmental Consulting, Inc. and Illingworth & Rodkin, Inc. San Francisco for the Oakland Bay Bridge East Span Seismic Safety Project*. California Department of Transportation.

Canning-Clode, J. o., Fowler, A. E., Byers, J. E., Carlton, J. T., & Ruiz, G. M. (2011). 'Caribbean creep' chills out: Climate change and marine invasive species. *PLoS ONE, 6*(12), e29657. doi: 10.1371/journal.pone.0029657.g001

Carder, D. A., & Ridgway., S. H. (1990). Auditory brainstem response in a neomatal sperm whale, Physeter spp. *Journal of the Acoustical Society of America, 88*(Suppl.1), S4.

(2Ab1). the 120th Meeting of the Acoustical Society of American, San Diego, Ca 126-130 November.

Cardillo, M. (2003). Biological determinants of extinction risk: Why are smaller species less vulnerable? *Animal Conservation, 6*, 63-69.

Cardillo, M., Mace, G. M., Jones, K. E., & Bielby, J. (2005). Multiple causes of high extinction risk in large mammal species. *Science, 309*, 1239-1241.

Carey, F. G., & Robinson, B. H. (1981). Daily patterns in the activities of swordfish, *Xiphias gladius*, observed by acoustic telemetry. *Fishery Bulletin, 79*, 277-292.

Carr, A., Carr, M. H., & Meylan, A. B. (1978). The ecology and migration of sea turtles, 7. the west Caribbean turtle colony. *Bulletin of the American Museum of Natural History, New York, 162*(1), 1-46.

Carr, A. F. (1986). RIPS, FADS, and little loggerheads. *BioScience, 36*(2), 92-100.

Carrera, M. L., Favaro, E. G. P., & Souto, A. (2008). The response of marine tucuxis (*Sotalia fluviatilis*) towards tourist boats involves avoidance behaviour and a reduction in foraging. *Animal Welfare, 17*(2), 117-123.

Carretta, J. V., Barlow, J., Forney, K. A., Muto, M. M., & Baker, J. (2001). U.S. Pacific Marine Mammal Stock Assessments: 2001: U.S. Department of Commerce.

Carretta, J. V., Chivers, S. J., & Danil, K. (2005). Preliminary estimates of marine mammal bycatch, mortality, and biological sampling of cetaceans in California gillnet fisheries for 2004. (pp. 18): National Marine Fisheries Service.

Carretta, J. V., & Chivers., S. J. (2004). Preliminary estimates of marine mammal mortality and biological sampling of cetaceans in California gillnet fisheries for 2003.: Unpublished paper to the IWC Scientific Committee. 20 pp. Sorrento, Italy, July (SC/56/SM1).

Carretta, J. V., Forney, K. A., Lowry, M. S., Barlow, J., Baker, J., Hanson, B., & Muto, M. M. (2007). U.S. Pacific marine mammal stock assessments: 2007 *U.S. Department of Commerce*.

Carretta, J. V., Forney, K. A., Lowry, M. S., Barlow, J., Baker, J., Hanson, B., & Muto, M. M. (2009). U.S. Pacific Marine Mammal Stock Assessments: 2008 *NOAA Technical Memorandum NMFS-SWFSC-434. 334p.*: U.S. Department of Commerce.

Carretta, J. V., Forney, K. A., Lowry, M. S., Barlow, J., Baker, J., Johnston, D., . . . Carswell, L. (2009). U.S. Pacific Marine Mammal Stock Assessments: 2008 (pp. 341): U.S. Department of Commerce, NOAA.

Carretta, J. V., Forney, K. A., Lowry, M. S., Barlow, J., Baker, J., Johnston, D., . . . Carswell, L. (2008). U.S. Pacific Marine Mammal Stock Assessments: 2008: NOAA Technical Memorandum NMFS-SWFSC-434. 340p.

Carretta, J. V., Forney, K. A., Muto, M. M., Barlow, J., Baker, J., & Lowry, M. (2005). U.S. Pacific marine mammal stock assessments: 2004 (pp. 322): National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Center.

Carretta, J. V., Forney, K. A., Oleson, E., Martien, K., Muto, M. M., Lowry, M. S., . . . Hill, M. C. (2011). U.S. Pacific Marine Mammal Stock Assessments: 2011: NMFS.

Carretta, J. V., Lynn, M. S., & LeDuc, C. A. (1994). Right whale (Eubalaena glacialis) sighting off San Clemente Island, California. *Marine Mammal Science, 10*(1), 101-105.

Carretta, J. V., Oleson, E., Weller, D. W., Lang, A. R., Forney, K. A., Baker, J., . . . Hill, M. C. (2013). U.S. Pacific Marine Mammal Stock Assessments: 2012: NMFS.

Carruthers, E. H., Schneider, D. C., & Neilson, J. D. (2009). Estimating the odds of survival and identifying mitigation opportunities for common bycatch in pelagic longline fisheries. *Biological Conservation, 142*(11), 2620-2630. doi: 10.1016/j.biocon.2009.06.010

Casale, P., D'Astore, P. P., & Argano, R. (2009). Age at size and growth rates of early juvenile loggerhead sea turtles (*Caretta caretta*) in the Mediterranean based on length frequency analysis. *Herpetological Journal, 19*(1), 29-33.

Casale, P., Mazaris, A. D., Freggi, D., Vallini, C., & Argano, R. (2009). Growth rates and age at adult size of loggerhead sea turtles (*Caretta caretta*) in the Mediterranean Sea, estimated through capture-mark-recapture records. *Scientia Marina, 73*(3), 589-595. doi: 10.3989/scimar.2009.73n3589

Castellote, M., Clark, C. W., & Lammers, M. O. (2012). Acoustic and behavioural changes by fin whales (*Balaenoptera physalus*) in response to shipping and airgun noise. *Biological Conservation, 147*(1), 115-122.

Castroviejo, J., Juste, J., Delval, J. P., Castelo, R., & Gil, R. (1994). Diversity and Status of Sea-Turtle Species in the Gulf of Guinea Islands. *Biodiversity and Conservation, 3*(9), 828-836.

Caswell, H. (1980). On the equivalence of maximizing reproductive value and maximizing fitness. *Ecology, 6*, 19-24.

Caswell, H., & Fujiwara, M. (2004). Beyond survival estimation: Mark-recapture, matrix population models, and population dynamics. *Animal Biodiversity and Conservation, 27*(1), 471-488.

Cato, D. H., & McCauley., R. D. (2001a). Ocean ambient noise from anthropogenic and natural sources in the context of marine mammal acoustics. *Journal of the Acoustical Society of America, 110*(5 Pt. 2), 2751. 2142nd Meeting of the Acoustical Society of America.

Cato, D. H., & McCauley., R. D. (2001b). Ocean ambient noise from anthropogenic and natural sources in the context of marine mammal acoustics. *Journal of the Acoustical Society of America, 110*(5 Pt. 2), 2751. 2142nd Meeting of the Acoustical Society of America.

Caurant, F., Bustamante, P., Bordes, M., & Miramand, P. (1999). Bioaccumulation of cadmium, copper and zinc in some tissues of three species of marine turtles stranded along the French Atlantic coasts. *Marine Pollution Bulletin, 38*(12), 1085-1091.

Caut, S., Guirlet, E., & Girondot, M. (2009). Effect of tidal overwash on the embryonic development of leatherback turtles in French Guiana. *Marine Environmental Research, 69*(4), 254-261.

Celik, A., Kaska, Y., Bag, H., Aureggi, M., Semiz, G., Kartal, A. A., & Elci, L. (2006). Heavy metal monitoring around the nesting environment of green sea turtles in Turkey. *Water Air and Soil Pollution, 169*(1-4), 67-79.

Cerchio, S., Jacobsen, J. K., Cholewiak, D. M., & Falcone, E. A. (2005). *Reproduction of female humpback whales off the Revillagigedo Archipelago during a severe El Niño event*. Paper presented at the Sixteenth Biennial Conference on the Biology of Marine Mammals, San Diego, California.

Cerchio, S., Strindberg, S., Collins, T., Bennett, C., & Rosenbaum, H. (2014). Seismic surveys negatively affect humpback whale singing activity off northern Angola. *PloS one, 9*(3), e86464.

CETAP. (1982). A characterization of marine mammals and turtles in the mid- and north-Atlantic areas of the U.S. Outer Continental Shelf (pp. 538). Washington, D. C.: Cetacean and Turtle Assessment Program, Bureau of Land Management.

Chaloupka, M., Bjorndal, K. A., Balazs, G. H., Bolten, A. B., Ehrhart, L. M., Limpus, C. J., . . . Yamaguchi, M. (2008). Encouraging outlook for recovery of a once severely exploited marine megaherbivore. *Global Ecology and Biogeography, 17*(2), 297-304. doi: 10.1111/j.1466-8238.2007.00367.x

Chaloupka, M. Y., Kamezaki, N., & Limpus, C. (2008). Is climate change affecting the population dynamics of the endangered Pacific loggerhead sea turtle? *Journal of Experimental Marine Biology and Ecology, 356*(1-2), 136-143.

Chan, E. H., & Liew, H. C. (1996). Decline of the leatherback population in Terengganu, Malaysia, 1956-1995. *Chelonian Conservation and Biology, 2*(2), 192-203.

Chan, S. K. F., Cheng, I. J., Zhou, T., Wang, H. J., Gu, H. X., & Song, X. J. (2007). A comprehensive overview of the population and conservation status of sea turtles in China. *Chelonian Conservation and Biology, 6*(2), 185-198.

Chapman, N. R., & Price., A. (2011). Low frequency deep ocean ambient noise trend in the Northeast Pacific Ocean. *Journal of the Acoustical Society of America, 129*(5), EL161-EL165.

Charif, R. A., Mellinger, D. K., Dunsmore, K. J., Fristrup, K. M., & Clark, C. W. (2002). Estimated source levels of fin whale (*Balaenoptera physalus*) vocalizations: Adjustments for surface interference. *Marine Mammal Science, 18*(1), 81-98.

Cherfas, J. (1989). *The Hunting of the Whale*. New York, New York: Viking Penguin Inc.

Cheung, W. W. L., Lam, V. W. Y., Sarmiento, J. L., Kearney, K., Watson, R., Zeller, D., & Pauly, D. (2010). Large-scale redistribution of maximum fisheries catch potential in the global ocean under climate change. *Global Change Biology, 16*, 24-35.

Chevalier, J., Desbois, X., & Girondot, M. (1999). The Reason of Decline of Leatherback Turtles (Dermochelys coriacea) in French Guiana: a Hypothesis. In C. Miaud & R. Guyétant (Eds.), *Current Studies in Herpetology SEH, Le Bourget du Lac* (pp. 79-87).

Chevalier, J., & Girondot, M. (1998). Nesting dynamics of marine turtles in French Guiana during the 1997 nesting season. *Bulletin de la Societe Herpetologique de France, 85-86*, 5-19.

Childers, A. R., Whitledge, T. E., & Stockwell, D. A. (2005). Seasonal and interannual variability in the distribution of nutrients and chlorophyll a across the Gulf of Alaska shelf: 1998-2000. *Deep-Sea Research II, 52*, 193-216.

Chou, C. C., Chen, Y. N., & Li., C. S. (2004). Congener-specific polychlorinated biphenyls in cetaceans from Taiwan waters. *Archives of Environmental Contamination and Toxicology, 47*(4), 551-560.

Christal, J., & Whitehead, H. (1997). Aggregations of mature male sperm whales on the Galápagos Islands breeding ground. *Marine Mammal Science, 13*(1), 59-69.

Christal, J., Whitehead, H., & Lettevall, E. (1998). Sperm whale social units: Variation and change. *Canadian Journal of Zoology, 76*(8), 1431-1440.

Christiansen, F., Lusseau, D., Stensland, E., & Berggren, P. (2010). Effects of tourist boats on the behaviour of Indo-Pacific bottlenose dolphins off the south coast of Zanzibar. *Endangered Species Research, 11*(1), 91-99.

Clapham, P., Shelden, K., & Wade, P. (2005). Review of information relating to possible Critical Habitat for Eastern North Pacific right whales. Review of information relating to possible Critical Habitat for Eastern North Pacific right whales. Seattle, Washington: NMML, NMFS.

Clapham, P. J. (1994). Maturational changes in patterns of association in male and female humpback whales, Megaptera novaeangliae. *Journal of Zoology, 234*(2), 265-274.

Clapham, P. J. (1996). The social and reproductive biology of humpback whales: an ecological perspective. *Mammal Review, 26*, 27-49.

Clapham, P. J., & Mattila, D. K. (1993). Reactions of humpback whales to skin biopsy sampling on a West-Indies breeding ground. *Marine Mammal Science, 9*(4), 382-391.

Clapham, P. J., Mikhalev, Y. A., Franklin, W., Paton, D., Baker, C. S., Ivashchenko, Y. V., & Brownell Jr, R. L. (2009). Catches of Humpback Whales, Megaptera novaeangliae, by the Soviet Union and Other Nations in the Southern Ocean, 1947–1973. *United States Department of Commerce*, 86 pp.

Clapham, P. J., Palsboll, P. J., & Mattila, D. K. (1993). High-energy behaviors in humpback whales as a source of sloughed skin for molecular analysis. *Marine Mammal Science, 9*(2), 213-220.

Clapham, P. J., Young, S. B., & Brownell Jr., R. L. (1999). Baleen whales: conservation issues and the status of the most endangered populations. *Mammal Review, 29*(1), 35-60.

Clark, C. W. (1990). Acoustic behavior of mysticete whales. In J. A. Thomas & R. A. Kastelein (Eds.), *Sensory Abilities of Cetaceans: Laboratory and Field Evidence* (pp. 571-583). New York: Plenum Press.

Clark, C. W. (1995). Annex M: matters arising out of the discussion of blue whales: Annex M1. Application of the U. S. Navy underwater hydrophone arrays for scientific research on whales. *Reports of the International Whaling Commission, 45*, 210-212.

Clark, C. W., Borsani, J. F., & Notarbartolo-di-Sciara, G. (2002). Vocal activity of fin whales, *Balaenoptera physalus*, in the Ligurian Sea. *Marine Mammal Science, 18*(1), 286-295.

Clark, C. W., & Ellison, W. T. (2004). Potential use of low-frequency sounds by baleen whales for probing the environment: Evidence from models and empirical measurements. In J. A. Thomas, C. F. Moss & M. Vater (Eds.), *Echolocation in Bats and Dolphins* (pp. 564-582): University of Chicago Press.

Clark, C. W., Ellison, W. T., Southall, B. L., Hatch, L., Parijs, S. V., Frankel, A., & Ponirakis, D. (2009). Acoustic masking in marine ecosystems as a function of anthropogenic sound sources (pp. 19). Madeira, Portugal: International Whaling Commission Scientific Committee.

Clark, C. W., Ellison, W. T., Southall, B. L., Hatch, L., Van Parijs, S. M., Frankel, A., & Ponirakis, D. (2009). Acoustic masking in marine ecosystems: intuitions, analysis, and implication. *Marine Ecology-Progress Series, 395*, 201-222. doi: 10.3354/meps08402

Clark, C. W., & Fristrup, K. M. (1997). Whales '95: A combined visual and acoustic survey of blue and fin whales off southern California. (*Balaenoptera musculus, Balaenoptera physalus*). *Report of the International Whaling Commission, 47*, 583-600.-Sc/548/Np518).

Clark, C. W., & Fristrup, K. M. (2001a). Baleen whale responses to low-frequency human-made underwater sounds. *Journal of the Acoustical Society of America, 110*(5 Part 2), 2751.

Clark, C. W., & Fristrup, K. M. (2001b). Baleen whale responses to low-frequency human-made underwater sounds. *Journal of the Acoustical Society of America, 110*(5 part 2), 2751.

Clark, C. W., & Gagnon, G. J. (2004). Low-frequency vocal behaviors of baleen whales in the North Atlantic: Insights from Integrated Undersea Surveillance System detections, locations, and tracking from 1992 to 1996. *Journal of Underwater Acoustics (USN), 52*(3), 48.

Clark, C. W., Tyack, P. L., & Ellison., W. T. (1999). Acoustic responses of baleen whales to low-frequency, man-made sounds. *Journal of the Acoustical Society of America, 106*(4 Part 2), 2279-2280.

Clarke, C. W., & Charif, R. A. (1998). Acoustic monitoring of large whales to the west of Britain and Ireland using bottom mounted hydrophone arrays, October 1996-September 1997 (Vol. JNCC Report Number 281).

Clarke, M. R. (1976). Observations on sperm whale diving. *Journal of the Marine Biological Association of the United Kingdom, 56*(3), 809-810.

Clarke, R. (1980). Catches of sperm whales and whalebone whales in the southeast Pacific between 1908 and 1975. *Report of the International Whaling Commission, 30*, 285-288.-Sc/231/Doc 226).

Cliffton, K. D., Cornejo, O., & Felger, R. S. (1982). Sea turtles of the Pacific coast of Mexico. In K. A. Bjorndal (Ed.), *Biology and conservation of sea turtles* (pp. 199-209). Washington, D.C.: Smithsonian Institution Press.

Clifton, K., Cornejo, D. O., & Felger, R. (1982). Sea turtles of the Pacific coast of Mexico *Biology and Conservation of Sea Turtles* (pp. 199-209). Washington, D. C.: Smithsonian Institution Press.

Cole, T. V. N., Hartley, D. L., & Merrick, R. L. (2005). Mortality and serious injury determinations for large whales stocks along the eastern seaboard of the United States, 1999-2003 (Vol. NOAA Northeast Fisheries Science Center 05-08): NOAA, NMFS, NEFSC.

Collett, R. (1909). A few notes on the whale *Balaena glacialis* and its capture in recent years in the North Atlantic by Norwegian whalers. *Proceedings of the General Meetings for Scientific Business of the Zoological Society of London, 7*, 91-98.

Conant, T. A., Dutton, P. H., Eguchi, T., Epperly, S. P., Fahy, C. C., Godfrey, M. H., . . . Witherington, B. E. (2009). Loggerhead sea turtle (Caretta caretta) 2009 status review under the U.S. Endangered Species Act. (Vol. Report of the Loggerhead Biological Review Team to the National Marine Fisheries Service, pp. 222).

Considine, D. M. (1995). Van Nostrand's scientific encyclopedia. Eight Edition. In D. M. Considine (Ed.). Reinhold, NY: Van Nostrand

Cook, S. L., & Forrest, T. G. (2005). Sounds produced by nesting leatherback sea turtles (*Dermochelys coriacea*). *Herpetological Review, 36*(4), 387-390.

Corkeron, P. J. (1995). Humpback whales (*Megaptera novaeangliae*) in Hervey Bay, Queensland: Behaviour and responses to whale-watching vessels. *Canadian Journal of Zoology, 73*(7), 1290-1299.

Cornelius, S. E. (1986). The sea turtles of Santa Rosa National Park. San José, Costa Rica: Fundacion Parques Nacionales.

Cornelius, S. E., Arauz, R., Fretey, J., Godfrey, M. H., M., R. M.-., & Shanker, K. (2007). Effect of land-based harvest of Lepidochelys. In P. T. Plotkin (Ed.), *Biology and conservation of Ridley sea turtles* (pp. 231-251). Baltimore, Maryland: Johns Hopkins University Press.

Cornelius, S. E., & Robinson, D. C. (1986). Post-nesting movements of female olive ridley turtles tagged in Costa Rica. *Vida Silvestre Neotropical, 1*(11), 12-23.

Corsolini, S., Aurigi, A., & Focardi, S. (2000). Presence of polychlorobiphenyls (PCBs), and coplanar congeners in the tissues of the Mediterranean loggerhead turtle Caretta caretta. *Marine Pollution Bulletin, 40*(11), 952-960.

Corsolini, S., Aurigi, S., & Focardi, S. (2000). Presence of polychlorobiphenyls (PCBs) and coplanar congeners in the tissues of the Mediterranean loggerhead turtle *Caretta caretta*. *Marine Pollution Bulletin, 40*(11), 952-960.

Corwith, H. L., & Wheeler, P. A. (2002). El Niño related variations in nutrient and chlorophyll distributions off Oregon. *Progress in Oceanography, 54*, 361-380.

Costa, D. P., Crocker, D. E., Gedamke, J., Webb, P. M., Houser, D. S., Blackwell, S. B., . . . Boeuf, B. J. L. (2003). The effect of a low-frequency sound source (acoustic thermometry of the ocean climate) on the diving behavior of juvenile northern elephant seals, *Mirounga angustirostris*. *Journal of the Acoustical Society of America, 113*(2), 1155-1165.

Cox, T. M., Ragen, T. J., Read, A. J., Vos, E., Baird, R. W., Balcomb, K., . . . Benner, L. (2006). Understanding the impacts of anthropogenic sound on beaked whales. *Journal of Cetacean Research and Management, 7*(3), 177-187.

Craig, M. P., & Ragen, T. J. (1999). Body size, survival, and decline of juvenile Hawaiian monk seals, *Monachus schauinslandi*. *Marine Mammal Science, 15*(3), 786-809.

Craig, M. P., & Ragen, T. J. (1999). Body size, survival, and decline of juvenile Hawaiian monk seals, *Monachus schauinslandi*. *Marine Mammal Science, 15*(3), 24.

Cranford, T. W. (1992). *Functional morphology of the odontocete forehead: Implications for sound generation.* (Ph.D. dissertation), University of California, Santa Cruz CA. 276pp.

Crognale, M. A., Eckert, S. A., Levenson, D. H., & Harms, C. A. (2008). Leatherback sea turtle *Dermochelys coriacea* visual capacities and potential reduction of bycatch by pelagic longline fisheries. *Endangered Species Research, 5*, 249-256.

Croll, D., Benson, S., Marinovic, B., Demer, D., Hewitt, R., & Chavez., F. (1999). *From wind to whales: Foraging ecology of rorquals in the California Current.* Paper presented at the Thirteen Biennial Conference on the Biology of Marine Mammals, 28 November - 3 December Wailea Maui HI. p.41.

Croll, D. A., Acevedo-Gutierrez, A., Tershy, B. R., & Urban-Ramirez, J. (2001). The diving behavior of blue and fin whales: is dive duration shorter than expected based on oxygen stores? *Comparative Biochemistry and Physiology a-Molecular & Integrative Physiology, 129*(4), 797-809.

Croll, D. A., Clark, C. W., Acevedo, A., Tershy, B., Flores, S., Gedamke, J., & Urban, J. (2002). Only male fin whales sing loud songs. *Nature, 417*, 809.

Croll, D. A., Clark, C. W., Calambokidis, J., Ellison, W. T., & Tershy, B. R. (2001). Effect of anthropogenic low-frequency noise on the foraging ecology of *Balaenoptera* whales. *Animal Conservation, 4*(1), 13-27.

Croll, D. A., Clark, C. W., Calambokidis, J., Ellison, W. T., & Tershy, B. R. (2001). Effect of anthropogenic low-frequency noise on the foraging ecology of *Balaenoptera* whales. *Animal Conservation, 4*(1), 15.

Croll, D. A., Tershy, B. R., Acevedo, A., & Levin, P. (1999). Marine vertebrates and low frequency sound *Technical report for LFA EIS* (pp. 437): Marine Mammal and Seabird Ecology Group, Institute of Marine Sciences, University of California Santa Cruz.

Crouse, O. T., Crowder, L. B., & Caswell, H. (1987). A site based population model for loggerhead sea turtles and implications for conservation. *Ecology, 68*(5), 1412-1423.

Cummings, W. C., & Fish, J. F. (1971). A synopsis of marine animal underwater sounds in eight geographic areas: Special report prepared for NUC Code 14. 97pp. 28 May.

Curran, M. A. J., Ommen, T. D. V., Morgan, V. I., Phillips, K. L., & Palmer, A. S. (2003). Ice core evidence for Antarctic sea ice decline since the 1950s. *Science, 302*(5648), 1203-1206.

Curtis, K. B., Howe, B. M., & Mercer, J. A. (1999). Low-frequency ambient sounds in the North Pacific long time series observations. *The Journal of the Acoustical Society of America, 106*, 3189-3200.

D'Spain, G., & Wartzok, D. (2004). Ocean noise and marine mammals: A tutorial lecture. *Journal of the Acoustical Society of America, 116*(4), 2499.

Daan, S., Deerenberg, C., & Dijkstra, C. (1996). Increased daily work precipitates natural death in the kestrel. *The Journal of Animal Ecology, 65*(5), 6.

Dahlheim, M. E. (1987). *Bio-acoustics of the gray whale (Eschrichtius robustus).* (Ph.D. dissertation), University of British Columbia, Canada. 315pp.

Dahlheim, M. E., Fisher, H. D., & Schempp., J. D. (1984). Sound production by the gray whale and ambient noise levels in Laguna San Ignacio, Baja California Sur, Mexico. (Eschrichtius robustus) *The Gray Whale, Eschrichtius robustus. M. L. Jones, S. L. Swartz and S. Leatherwood (eds.). p.511-542. Academic Press, New York.*

Dahlheim, M. E., & Ljungblad, D. K. (1990). Preliminary hearing study on gray whales (Eschrichtius robustus) in the field. In J. A. Thomas & R. A. Kastelein (Eds.), *Sensory Abilities of Cetaceans: Laboratory and Field Evidence* (pp. 335-346). New York: Plenum Press.

Dähne, M., Gilles, A., Lucke, K., Peschko, V., Adler, S., Krügel, K., . . . Siebert, U. (2013). Effects of pile-driving on harbour porpoises (*Phocoena phocoena*) at the first offshore wind farm in Germany. *Environmental Research Letters, 8*(2), 025002. doi: 10.1088/1748-9326/8/2/025002

Danil, K., & Chivers, S. J. (2005). *Habitat-based spatial and temporal variability of life history characteristics of female common dolphins (Delphinus delphis) in the eastern tropical Pacific*. Paper presented at the Sixteenth Biennial Conference on the Biology of Marine Mammals, San Diego, California.

Danilewicz, D., Tavares, M., Moreno, I. B., Ott, P. H., & Trigo., C. C. (2009). Evidence of feeding by the humpback whale (Megaptera novaeangliae) in mid-latitude waters of the western South Atlantic. *JMBA2 - Biodiversity Records-Published Online, 3Pgs.*

Darling, J. D., & Cerchio, S. (1993). Movement of a Humpback Whale (Megaptera-Novaeangliae) between Japan and Hawaii. *Marine Mammal Science, 9*(1), 84-89.

Darling, J. D., & Morowitz, H. (1986). Census of Hawaiian humpback whales (Megaptera novaeangliae) by individual identification. *Canadian Journal of Zoology, 64*(1), 105-111.

Das, S. (2008). Hundreds of sea turtles die along Odisha coast. *Kalinga Times*.

Davenport, J., Wrench, J., McEvoy, J., & Carnacho-Ibar, V. (1990). Metal and PCB concentrations in the "Harlech" leatherback. *Marine Turtle Newsletter, 48*, 1-6.

Davis, R. W., Evans, W. E., & Würsig, B. (2000). Cetaceans, Sea Turtles and Seabirds in the Northern Gulf of Mexico: Distribution, Abundance and Habitat Associations. Volume II: Technical Report (pp. 346). Galveston: Texas A&M.

Davis, R. W., Jaquet, N., Gendron, D., Markaida, U., Bazzino, G., & Gilly, W. (2007). Diving behavior of sperm whales in relation to behavior of a major prey species, the jumbo squid, in the Gulf of California, Mexico. *Marine Ecology Progress Series, 333*, 291-302.

Deecke, V. B., Slater, P. J. B., & Ford, J. K. B. (2002). Selective habituation shapes acoustic predator recognition in harbour seals. *Nature, 417*(6912), 171-173.

Deem, S. L., Norton, T. M., Mitchell, M., Segars, A., Alleman, A. R., Cray, C., . . . Karesh, W. B. (2009). COMPARISON OF BLOOD VALUES IN FORAGING, NESTING, AND STRANDED LOGGERHEAD TURTLES (CARETTA CARETTA) ALONG THE COAST OF GEORGIA, USA. *Journal of Wildlife Diseases, 45*(1), 41-56.

Deruiter, S. L., Boyd, I. L., Claridge, D. E., Clark, C. W., Gagnon, C., Southall, B. L., & Tyack, P. L. (2013). Delphinid whistle production and call matching during playback of simulated military sonar. *Marine Mammal Science, 29*(2), E46-E59.

DeRuiter, S. L., & Larbi Doukara, K. (2012). Loggerhead turtles dive in response to airgun sound exposure. *Endangered Species Research, 16*(1), 55-63. doi: 10.3354/esr00396

Di Lorio, L., & Clark, C. W. (2010). Exposure to seismic survey alters blue whale acoustic communication. *Biology Letters, 6*(1), 51-54.

Dill, H. R., & Bryan, W. A. (1912). Report on an expedition to Laysan Island in 1911. *U.S. Department of Agriculture Biological Survey Bulletin, 42*, 1-30.

Dodd, C. K. (1988). Synopsis of the biological data on the loggerhead sea turtle: Caretta caretta (Linnaeus 1758). *Fish and Wildlife Service Biological Report, 88*(14), 110.

Dodd, C. K. J. (1988). Synopsis of the biological data on the loggerhead sea turtle *Caretta caretta* (Linnaeus 1758). *USFWS Biological Report, 88*(14), 110 pp.

Dodd Jr., C. K. (1988a). Synopsis of the biological data on the loggerhead sea turtle *Caretta caretta* (Linnaeus 1758) *Biological Report* (pp. 110): U.S. Fish and Wildlife Service.

Dodd Jr., C. K. (1988b). Synopsis of the biological data on the loggerhead sea turtle, Caretta caretta (Linnaeus 1758) *U.S. Fish and Wildlife Service Biological Report* (Vol. 88).

Dodd, K. T., Mundie, T. G., Lagutchik, M. S., & Morris, J. R. (1997). Cardiopulmonary effects of high-impulse noise exposure. *Journal of Trauma-Injury Infection and Critical Care, 43*(4), 656-666. doi: 10.1097/00005373-199710000-00016

Dohl, T. P. (1983). *Return of the humpback whale (Megaptera novaeangliae) to central California.* Paper presented at the Fifth Biennial Conference on the Biology of Marine Mammals, 27 November-1 December New England Aquarium Boston MA. p.23-24.

Doksaeter, L., Handegard, N. O., Godo, O. R., Kvadsheim, P. H., & Nordlund, N. (2012). Behavior of captive herring exposed to naval sonar transmissions (1.0-1.6 kHz) throughout a yearly cycle. *Journal of the Acoustical Society of America, 131*(2), 1632-1642. doi: 10.1121/1.3675944

Dolman, S. J., Pinn, E., Reid, R. J., Barley, J. P., Deaville, R., Jepson, P. D., . . . Simmonds., M. P. (2010). A note on the unprecedented strandings of 56 deep-diving whales along the UK and Irish coast. *Marine Biodiversity Records, 3*, E16.

Dolphin, W. F. (1987). Ventilation and dive patterns of humpback whales, Megaptera novaeangliae, on their Alaskan feeding grounds. *Canadian Journal of Zoology, 65*(1), 83-90.

Donovan, G. P. (1984). Blue whales off Peru, December 1982, with special reference to pygmy blue whales. (Balaenoptera musculus). *Report of the International Whaling Commission, 34*, 473-476.-Sc/435/Ps427).

Donovan, G. P. (1991). A review of IWC stock boundaries (pp. 39-68): Report of the International Whaling Commission (Special Issue 13).

Doroshenko, N. V. (2000). Soviet whaling for blue, gray, bowhead, and right whales in the North Pacific Ocean, 1961-1979 *Soviet Whaling Data (1949-1979)* (pp. 96-103). Moscow: Center for Russian Environmental Policy Marine Mammal Council.

Douglas, A. B., Calambokidis, J., Raverty, S., Jeffries, S. J., Lambourn, D. M., & Norman, S. A. (2008). Incidence of ship strikes of large whales in Washington State. *Journal of the Marine Biological Association of the United Kingdom, 88*(6), 1121-1132. doi: 10.1017/s0025315408000295

Dow Piniak, W. E., Mann, D. A., Eckert, S. A., & Harms, C. A. (2012). Amphibious hearing in sea turtles *The Effects of Noise on Aquatic Life* (pp. 83-87): Springer.

Doyle, L. R., McCowan, B., Hanser, S. F., Chyba, C., Bucci, T., & Blue, J. E. (2008). Applicability of information theory to the quantification of responses to anthropogenic noise by southeast Alaskan humpback whales. *Entropy, 10*(2), 33-46.

Dufault, S., Whitehead, H., & Dillon, M. (1999). An examination of the current knowledge on the stock structure of sperm whales (*Physeter macrocephalus*) worldwide. *Journal of Cetacean Research and Management, 1*, 1-10.

Dunlop, R. A., Cato, D. H., & Noad, M. J. (2008). Non-song acoustic communication in migrating humpback whales (Megaptera novaeangliae). *Marine Mammal Science, 24*(3), 613-629. doi: 10.1111/j.1748-7692.2008.00208.x

Dutton, P. (2003). *Molecular ecology of Chelonia mydas in the Eastern Pacific Ocean*. Paper presented at the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation.

Dutton, P. H. (2003). *Molecular ecology of the eastern Pacific green turtle.* Paper presented at the 22nd Annual Symposium on Sea Turtle Biology and Conservation.

Eckert, K. L. (1993). The biology and population status of marine turtles in the North Pacific Ocean. Honolulu, Hawaii: NOAA, NMFS, SWFSC.

Eckert, K. L. (1993). The biology and population status of marine turtles in the North Pacific Ocean *Final Report* (pp. 166). Honolulu, HI: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Eckert, K. L., & Eckert, S. A. (1988). Pre-reproductive movements of leatherback sea turtles (*Dermochelys coriacea*) nesting in the Caribbean. *Copeia*(2), 400-406.

Eckert, S. A. (1998). *Perspectives on the use of satellite telemetry and electronic technologies for the study of marine turtles, with reference to the first year long tracking of leatherback sea turtles*. Paper presented at the Proceedings of the 17th Annual Symposium on Sea Turtle Biology and Conservation.

Eckert, S. A. (1999). Habitats and migratory pathways of the Pacific leatherback sea turtle *Final Report* (Vol. 99-290, pp. 7 pp.): National Marine Fisheries Service, Office of Protected Resources.

Eckert, S. A., Eckert, K. L., & Richardson, T. H. (1989). Proceedings of the Ninth Annual Workshop on Sea Turtle Conservation and Biology *NOAA Technical Memorandum* (pp. 306): U.S. Department of Commerce.

Eckert, S. A., & Lien, J. (1999). Recommendations for Eliminating Incidental Capture and Mortality of Leatherback Turtles, Dermochelys coriacea, by Commercial Fisheries in Trinidad and Tobago: A Report to the Wider Caribbean Sea Turtle Conservation Network (WIDECAST). *Hubbs Sea World Research Institute Technical Report No. 2000-310*, 7 pp.

Eckert, S. A., & Sarti, L. (1997). Distant fisheries implicated in the loss of the world's largest leatherback nesting population. *Marine Turtle Newsletter, 78*, 2-7.

Edds-Walton, P. L. (1997). Acoustic communication signals of mysticete whales. *Bioacoustics-the International Journal of Animal Sound and Its Recording, 8*, 47-60.

Edds, P. L. (1988). Characteristics of finback Balaenoptera physalus vocalizations in the St. Lawrence estuary. *Bioacoustics, 1*, 131-149.

Edds, P. L., & Macfarlane, J. A. F. (1987). Occurrence and general behavior of balaenopterid cetaceans summering in the St. Lawrence Estuary, Canada. *Canadian Journal of Zoology, 65*(6), 1363-1376.

Edwards, M., Johns, D. G., Licandro, P., John, A. W. G., & Stevens, D. P. (2007). Ecological Status Report: results from the CPR survey 2005/2006. Plymouth, UK.

Eguchi, T., Gerrodette, T., Pitman, R. L., Seminoff, J. A., & Dutton, P. H. (in preperation). At-sea density and abundance estimates of the olive ridley turtle (Lepidochelys olivacea) in the Eastern Tropical Pacific.

Ehrhart, L. M., Bagley, D. A., & Redfoot, W. E. (2003). Loggerhead turtles in the Atlantic Ocean: Geographic distribution, abundance, and population status. In A. B. Bolten & B. E. Witherington (Eds.), *Loggerhead Sea Turtles* (pp. 157-174). Washington D.C.: Smithsonian Books.

Ehrhart, L. M., Redfoot, W. E., & Bagley, D. A. (2007). Marine turtles of the central region of the Indian River Lagoon System, Florida. *Florida Scientist, 70*(4), 415-434.

Eldredge, L. G. (1991). Annotated checklist of the marine mammals of Micronesia. *Micronesica, 24*(2), 217-230.

Eldredge, L. G. (2003). The marine reptiles and mammals of Guam. *Micronesica, 35*(36), 653-660.

Elfes, C. T., Vanblaricom, G. R., Boyd, D., Calambokidis, J., Clapham, P. J., Pearce, R. W., . . . Krahn., M. M. (2010). Geographic variation of persistent organic pollutant levels in humpback whale (Megaptera novaeangliae) feeding areas of the North Pacific and North Atlantic. *Environmental Toxicology and Chemistry, 29*(4), 824-834.

Ellison, W. T., Southall, B. L., Clark, C. W., & Frankel, A. S. (2012). A new context-based approach to assess marine mammal behavioral responses to anthropogenic sounds. *Conservation Biology, 26*(1), 21-28.

Ellison, W. T., & Stein, P. J. (1999). SURTASS LFA High Frequency Marine Mammal Monitoring (HF/M3) Sonar: System Description and Test & Evaluation.

Elsasser, T. H., Klasing, K. C., Filipov, N., & Thompson, F. (2000). The metabolic consequences of stress: targets for stress and priorities of nutrien use. In G. P. Moberg & J. A. Mench (Eds.), *The Biology of Animal Stress: Basic Principles and Implications for Animal Welfare*. New York, New York: CABI Publishing.

Elvin, S. S., & Hogart, C. T. (2008). Right whales and vessels in Canadian waters. *Marine Policy, 32*(3), 379-386.

Endo, T., Hisamichi, Y., Kimura, O., Haraguchi, K., Lavery, S., Dalebout, M. L., . . . Baker., C. S. (2010). Stable isotope ratios of carbon and nitrogen and mercury concentrations in 13 toothed whale species taken from the western Pacific Ocean off Japan. *Environmental Science and Technology, 44*(7), 2675-2681.

Englund, A., & Berggren, P. (2002). The impact of tourism on Indo-Pacific bottlenose dolphins (*Tursiops aduncus*) in Menai Bay, Zanzibar. Shimonoseki, Japan: International Whaling Commission Scientific Committee.

Erbe, C. (2000). Detection of whale calls in noise: Performance comparison between a beluga whale, human listeners, and a neural network. *Journal of the Acoustical Society of America, 108*(1), 297-303.

Erbe, C. (2002). Hearing abilities of baleen whales.: Defence R&D Canada – Atlantic report CR 2002-065. Contract Number: W7707-01-0828. 40pp.

Erbe, C. (2002). Underwater noise of whale-watching boats and potential effects on killer whales (*Orcinus orca*), based on an acoustic impact model. *Marine Mammal Science, 18*(2), 394-418.

Erhart, L. M., D.A. Bagley, & Redfoot, W. E. (2003). Loggerhead Turtles in the Atlantic Ocean : Geographic Distribution, Abundance, and Population Status. *Pp.157-174 In: Bolten, A.B. and B.E. Witherington (eds), Loggerhead Sea Turtles. Smithsonian Books, Washington D.C.*

Euroturtle. (2009). Hawksbill sea turtle, *Eretmochelys imbricata*. from http://www.euroturtle.org/hawksbill_habitat.htm

Evans, D. L., & England, G. R. (2001). Joint interim report Bahamas marine mammal stranding event 15 - 16 March 2000. Washington, D.C. and Silver Spring, Maryland: U.S. Department of the Navy and U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Evans, K., Hindell, M., & Hince, G. (2004). Concentrations of organochlorines in sperm whales (Physeter macrocephalus) from Southern Australian waters. *Marine Pollution Bulletin, 48*, 486-503.

Evans, P. G. H. (1987). *The natural history of whales & dolphins*: Helm.

Fahlman, A., Hooker, S. K., Szowka, A., Bostrom, B. L., & Jones., D. R. (2009). Estimating the effect of lung collapse and pulmonary shunt on gas exchange during breath-hold diving: The Scholander and Kooyman legacy. *Respiratory Physiology & Neurobiology, 165*(1), 28-39.

Farr, R. A., & Kern, J. C. (2004). Green sturgeon population characteristics in Oregon *Annual Progress Report: Fish Research Project Oregon* (pp. 32). Clackamas, Oregon: Oregon Department of Fish and Wildlife.

Feldkamp, S. D., DeLong, R. L., & Antonelis, G. A. (1991). Effects of El Niño 1983 on the foraging patterns of California sea lions (Zalophus californianus) near San Miguel Island, California. In F. Trillmich & K. A. Ono (Eds.), *Pinnipeds and El Niño: Responses to environmental stress* (pp. 146-155). Berlin, Germany: Springer-Verlag.

Fernandez-Juricic, E., Campagna, C., Enriquez, V., & Ortiz, C. L. (1999). Vocal communication and individual variation in breeding South American sea lions. *Behaviour, 136*(4), 495-517.

Fernández, A., Edwards, J. F., Rodríguez, F., Espinosa de los Monteros, A., Herráez, P., Castro, P., . . . Arbelo, M. (2005). "Gas and Fat Embolic Syndrome" Involving a Mass Stranding of Beaked Whales (Family *Ziphiidae*) Exposed to Anthropogenic Sonar Signals. *Veterinary Pathology, 42*, 446-457.

Fernandez, A., Martín, P. C. V., Gallardo, T., & Arbel, M. (2006). *New beaked whale mass stranding in Canary Islands associated with naval military exercises (Majestic Eagle 2004).* Paper presented at the Abstracts of the International Policy Workshop on Sound and Marine Mammals, London, England.

FFWCC. (2007a). Florida statewide nesting beach survey data–2005 season: Florida Fish and Wildlife Conservation Commission.

FFWCC. (2007b). Long-term monitoring program reveals a continuing loggerhead decline, increases in green turtle and leatherback nesting: Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute.

Fiedler, P. C., Reilly, S. B., Hewitt, R. P., Demer, D., Philbrick, V. A., Smith, S., . . . Mate, B. R. (1998). Blue whale habitat and prey in the California Channel Islands. *Deep-Sea Research Part Ii-Topical Studies in Oceanography, 45*(8-9), 1781-1801.

Filadelfo, R., Mintz, J., Michlovich, E., D'Amico, A., Tyack, P. L., & Ketten, D. R. (2009). Correlating military sonar use with beaked whale mass strandings: What do the historical data show? *Aquatic Mammals, 35*(4), 435-444.

Findlay, K. P., & Best, P. B. (1995). Summer incidence of humpback whales on the west coast of South Africa. (*Megaptera novaeangliae*). *South African Journal of Marine Science, 15*, 279-282.

Finkbeiner, E. M., Wallace, B. P., Moore, J. E., Lewison, R. L., Crowder, L. B., & Read, A. J. (2011a). Cumulative estimates of sea turtle bycatch and mortality in USA fisheries between 1990 and 2007. *Biological Conservation*. doi: 10.1016/j.biocon.2011.07.033

Finkbeiner, E. M., Wallace, B. P., Moore, J. E., Lewison, R. L., Crowder, L. B., & Read, A. J. (2011b). Cumulative estimates of sea turtle bycatch and mortality in USA fisheries between 1990 and 2007. *Biological Conservation, 144*(11), 2719-2727. doi: 10.1016/j.biocon.2011.07.033

Finneran, J. J. (2003). Whole-lung resonance in a bottlenose dolphin (Tursiops truncatus) and white whale (Delphinapterus leucas). *The Journal of the Acoustical Society of America, 114*(1), 7.

Finneran, J. J., Carder, D. A., Schlundt, C. E., & Ridgway, S. H. (2005). Temporary threshold shift in bottlenose dolphins (*Tursiops truncatus*) exposed to mid-frequency tones. *Journal of the Acoustical Society of America, 118*(4), 2696-2705.

Finneran, J. J., Dear, R., Carder, D. A., & Ridgway, S. H. (2003). Auditory and Behavioral Responses of California Sea Lions (*Zalophus californianus*) to Single Underwater Impulses From an Arc-Gap Transducer. *J. Acoust. Soc. Am., 114*(3), 1667-1677.

Finneran, J. J., & Houser., D. S. (2005). *Large sample study of dolphin hearing using auditory evoked potentials.* Paper presented at the Sixteenth Biennial Conference on the Biology of Marine Mammals, San Diego, CA.

Finneran, J. J., & Schlundt, C. E. (2010). Frequency-dependent and longitudinal changes in noise-induced hearing loss in a bottlenose dolphin (*Tursiops truncatus*). *Journal of the Acoustical Society of America, 128*(2), 567-570.

Finneran, J. J., & Schlundt, C. E. (2011). Subjective loudness level measurements and equal loudness contours in a bottlenose dolphin (*Tursiops truncatus*). *Journal of the Acoustical Society of America, 130*(5), 3124-3136.

Finneran, J. J., Schlundt, C. E., Branstetter, B., & Dear, R. L. (2007a). Assessing temporary threshold shift in a bottlenose dolphin (*Tursiops truncatus*) using multiple simultaneous auditory evoked potentials. *Journal of the Acoustical Society of America, 122*(2), 1249-1264. doi: 10.1121/1.2749447

Finneran, J. J., Schlundt, C. E., Branstetter, B., & Dear, R. L. (2007b). Assessing temporary threshold shift in a bottlenose dolphin (*Tursiops truncatus*) using multiple simultaneous auditory evoked potentials. *The Journal of the Acoustical Society of America, 122*(2), 1249-1264. doi: 10.1121/1.2749447

Finneran, J. J., Schlundt, C. E., Carder, D. A., Clark, J. A., Young, J. A., Gaspin, J. B., & Ridgway, S. H. (2000). Auditory and Behavioral Responses of Bottlenose Dolphins (Tursiops truncatus) and a Belga Whale (Delphinapterus leucas) to Impulsive Sounds

Resembling Distant Signatures of Underwater Explosions. *Journal of the Acoustical Society of America, 108*(1), 417-431.

Finneran, J. J., Schlundt, C. E., Carder, D. A., & Ridgway, S. H. (2002). Auditory filter shapes for the bottlenose dolphin (Tursiops truncatus) and the white whale (Delphinapterus leucas) derived with notched noise. *The Journal of the Acoustical Society of America, 112*(1), 322-328.

Finneran, J. J., Schlundt, C. E., Dear, R., Carder, D. A., & Ridgway, S. H. (2000a). Masked temporary threshold shift (MTTS) in odontocetes after exposure to single underwater impulses from a seismic watergun. *Journal of the Acoustical Society of America, 108*(5 2), 2515.

Finneran, J. J., Schlundt, C. E., Dear, R., Carder, D. A., & Ridgway, S. H. (2000b). Masked temporary threshold shift (MTTS) in odontocetes after exposure to single underwater impulses from a seismic watergun. *The Journal of the Acoustical Society of America, 108*(5 2), 2515.

Finneran, J. J., Schlundt, C. E., Dear, R., Carder, D. A., & Ridgway, S. H. (2002). Temporary shift in masked hearing thresholds in odontocetes after exposure to single underwater impulses from a seismic watergun. *Journal of the Acoustical Society of America, 111*(6), 2929-2940.

Fischer, J. B. (1829). *Synopsis Mammalium*. Stuttgart: J.G. Cottae.

Fletcher, E. R., Yelverton, J. T., & Richmond, D. R. (1976). The thoracoabdominal system's response to underwater blast. Final Technical Report. Albuquerque, New Mexico: Lovelace Foundation for Medical Education and Research.

Florida Power and Light Company St. Lucie Plant. (2002). Annual environmental operating report 2001. Juno Beach, Florida: Florida Power and Light Company St. Lucie Plant.

Foley, A. M., Dutton, P. H., Singel, K. E., Redlow, A. E., & Teas, W. G. (2003). The first records of olive ridleys in Florida, USA. *Marine Turtle Newsletter, 101*, 23-25.

Foote, A. D., Osborne, R. W., & Hoelzel, A. R. (2004). Whale-call response to masking boat noise. *Nature, 428*, 910.

Foote, A. D., Osborne, R. W., & Hoelzel., A. R. (2004). Whale-call response to masking boat noise. *Nature, 428*(6986), 910.

Forcada, J., Trathan, P. N., Reid, K., & Murphy., E. J. (2005). The effects of global climate variability in pup production of Antarctic fur seals. (Arctocephalus gazella). *Ecology, 86*(9), 2408-2417.

Ford, J. K. B., & Reeves, R. R. (2008). Fight or flight: Antipredator strategies of baleen whales. *Mammal Review, 38*(1), 50-86.

Forney, K. A. (2007). Preliminary estimates of cetacean abundance along the U.S. west coast and within four national marine sanctuarnes during 2005.: NOAA Technical Memorandum NMFS-SWFSC-406. 33p.

Forney, K. A., Baird, R. W., & Oleson, E. M. (2010). Rationale for the 2010 revision of stock boundaries for the Hawaii insular and pelagic stocks of false killer whales, Pseudorca crassidens.

Forney, K. A., Baird, R. W., Oleson, E. M., & Center, S. F. S. (2010). *Rationale for the 2010 Revision of Stock Boundaries for the Hawai'i Insular and Pelagic Stocks of False Killer Whales, Pseudorca Crassidens*: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Forney, K. A., & Brownell Jr., R. L. (1996). Preliminary report of the 1994 Aleutian Island marine mammal survey. La Jolla, California: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Forney, K. A., & Kobayashi, D. R. (2007). Updated estimates of mortality and injury of cetaceans in the Hawaii-based longline fishery, 1994-2005.

Forney, K. A., & Kobayashi., D. R. (2007). Updated estimates of mortality and injury of cetaceans in the Hawaii-based longline fishery, 1994-2005.: NOAA Technical Memorandum NMFS-SWFSC-412. 33p.

Forney, K. A., R. L. Brownell, J., & Fiedler., P. C. (1995). *The distribution of marine mammals along the Aleutian Islands in 1994 - where have all the blue whales gone?* Paper presented at the Eleventh Biennial Conference on the Biology of Marine Mammals, 14-18 December 1995 Orlando FL. p.39.

Fossette, S., Girard, C., Bastian, T., Calmettes, B., Ferraroli, S., Vendeville, P., . . . Georges, J. Y. (2009). Thermal and trophic habitats of the leatherback turtle during the nesting season in French Guiana. *Journal of Experimental Marine Biology and Ecology, 378*(1-2), 8-14. doi: DOI 10.1016/j.jembe.2009.06.021

Foti, M., Giacopello, C., Bottari, T., Fisichella, V., Rinaldo, D., & Mammina, C. (2009). Antibiotic resistance of gram negatives isolates from loggerhead sea turtles (*Caretta caretta*) in the central Mediterranean Sea. *Marine Pollution Bulletin, 58*(9), 1363-1366. doi: 10.1016/j.marpolbul.2009.04.020

Fox, C. G., Matsumoto, H., & Lau, T.-K. A. (2001). Monitoring Pacific Ocean seismicity from an autonomous hydrophone array. *J. Geophys. Res., 106*(B3), 4183-4206. doi: 10.1029/2000jb900404

Francis, R. C., & Hare, S. R. (1997). Regime scale climate forcing of salmon populations in the northeast Pacific--Some new thoughts and findings. In R. L. Emmett & M. H. Schiewe (Eds.), *Estuarine and ocean survival of northeastern Pacific salmon: Proceedings of the workshop* (Vol. NOAA Technical Memorandum NMFS-NWFSC-29, pp. 113-127): NOAA.

Francour, P., Ganteaume, A., & Poulain, M. (1999). Effects of boat anchoring in *Posidonia oceanica* seagrass beds in the Port-Cros National Park (north-western Mediterranean Sea). *Aquatic Conservation: Marine and Freshwater Ecosystems, 9*(4), 391-400.

Frankel, A. S. (2009). Sound production. In W. F. Perrin, B. Wursig & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals. Second edition* (pp. 1056-1071). San Diego: Academic Press.

Frankel, A. S., & Clark, C. W. (2000). Behavioral responses of humpback whales (*Megaptera novaeangliae*) to full-scale ATOC signals. *Journal of the Acoustical Society of America, 108*(4), 1930-1937.

Frantzis, A. (1998). Does acoustic testing strand whales? *Nature, 392*(6671), 29.

Fraser, W. R., & Hofmann, E. E. (2003). A predator's perspective on causal links between climate change, physical forcing and ecosystem response. *Marine Ecology Progress Series, 265*, 1-15.

Frazer, L. N., & Mercado III, E. (2000). A sonar model for humpback whale song. *IEEE Journal of Oceanic Engineering, 25*(1), 160-182.

Frazer, N. B., & Ehrhart, L. M. (1985). Preliminary growth models for green, *Chelonia mydas*, and loggerhead, *Caretta caretta*, turtles in the wild. *Copeia, 1985*, 73-79.

Frazer, N. B., Limpus, C. J., & Greene, J. L. (1994). *Growth and estimated age at maturity of Queensland loggerheads*. Paper presented at the Proceedings of the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation, Hilton Head, South Carolina.

Frazier, J., Arauz, R., Chevalier, J., Formia, A., Fretey, J., Godfrey, M. H., . . . Shanker, K. (2007). Human-turtle interactions at sea. In P. T. Plotkin (Ed.), *Biology and conservation of Ridley sea turtles* (pp. 253-295). Baltimore, Maryland: Johns Hopkins University Press.

Frazier, J., Arauz, R., Chevalier, J., Formia, A., Fretey, J., Godfrey, M. H., . . . Shanker, K. (2007). Human-turtle interactions at sea. In P. T. Plotkin (Ed.), *Biology and Conservation of Ridley Sea Turtles* (pp. 253-295). Baltimore, Maryland: Johns Hopkins University Press.

Freeland, H. J. (1990). Sea surface temperatures along the coast of British Columbia: Regional evidence for a warming trend. *Canadian Journal of Fisheries and Aquatic Sciences, 47*, 346-350.

Fretey, J. (1999). Distribution of the turtles of the genus *Lepidochelys* Fitzinger, 1843. I. The western Atlantic. *Biogeographica, 75*(3), 97-111.

Fretey, J. (2001a). Biogeography and conservation of marine turtles of the Atlantic Coast of Africa (pp. Bonn, Germany). UNEP/CMS Secretariat: CMS Technical Series Publication No. 6.

Fretey, J. (2001b). Biogeography and conservation of marine turtles of the Atlantic coast of Africa *CMS Technical Series Publication* (Vol. 6). Bonn, Germany: UNEP/CMS Secretariat.

Fretey, J., Formia, A., Tomas, J., Dontaine, J. F., Billes, A., & Angoni, H. (2005). *Presence, nesting, and conservation of Lepidochelys olivacea in the Gulf of Guinea.* Paper presented at the Twenty-first Annual Symposium on Sea Turtle Biology and Conservation.

Frid, A. (2003). Dall's sheep responses to overflights by helicopter and fixed-wing aircraft. *Biological Conservation, 110*(3), 387-399.

Frid, A., & Dill, L. (2002). Human-caused disturbance stimuli as a form of predation risk. *Conservation Ecology, 6*(1). doi: 11

Fristrup, K. M., Hatch, L. T., & Clark, C. W. (2003). Variation in humpback whale (Megaptera novaeangliae) song length in relation to low-frequency sound broadcasts. *Journal of the Acoustical Society of America, 113*(6), 3411-3424. doi: 10.1121/1.1573637

Fritts, T. H., Hoffman, W., & McGehee, M. A. (1983). The distribution and abundance of marine turtles in the Gulf of Mexico and nearby Atlantic waters. *Journal of Herpetology, 17*(4), 327-344.

Fritts, T. H., Stinson, M. L., & Márquez, R. (1982). Status of sea turtle nesting in southern Baja California, Mexico. *Bulletin of the Southern California Academy of Sciences, 81*, 51-60.

Fromentin, J.-M., & Planque, B. (1996). Calanus and environment in the eastern North Atlantic. 2. Role of the North Atlantic Oscillation on Calanus finmarchicus and C. helgolandicus. *Marine Ecology Progress Series, 134*, 11-118.

Fuentes, M., Hamann, M., & Limpus, C. J. (2009). Past, current and future thermal profiles of green turtle nesting grounds: Implications from climate change. *Journal of Experimental Marine Biology and Ecology, 383*(1), 56-64. doi: 10.1016/j.jembe.2009.11.003

Fuentes, M. M. P. B., & Abbs, D. (2010). Effects of projected changes in tropical cyclone frequency on sea turtles. *Marine Ecology Progress Series, 412*, 283-292.

Fuentes, M. M. P. B., Hamann, M., & Limpus, C. J. (2009). Past, current and future thermal profiles of green turtle nesting grounds: Implications from climate change. *Journal of Experimental Marine Biology and Ecology, in press*(in press), in press.

Fuentes, M. M. P. B., Limpus, C. J., & Hamann, M. (2010). Vulnerability of sea turtle nesting grounds to climate change. *Global Change Biology, in press*(in press), in press.

Fuentes, M. M. P. B., Maynard, J. A., Guinea, M., Bell, I. P., Werdell, P. J., & Hamann, M. (2009). Proxy indicators of sand temperature help project impacts of global warming on sea turtles in northern Australia. *Endangered Species Research, 9*, 33-40.

Fujihara, J., Kunito, T., Kubota, R., & Tanabe, S. (2003). Arsenic accumulation in livers of pinnipeds, seabirds and sea turtles: Subcellular distribution and interaction between arsenobetaine and glycine betaine. *Comparative Biochemistry and Physiology C-Toxicology & Pharmacology, 136*(4), 287-296.

Futuymda, D. J. (1986). *Evolutionary biology* (Second ed. ed.). Sunderland, Massachussetts: Sinauer

Associates, Inc.

Gagnon, G. J., & Clark, C. W. (1993). *The use of U.S. Navy IUSS passive sonar to monitor the movements of blue whale. (Balaenoptera musculus).* Paper presented at the Tenth Biennial Conference on the Biology of Marine Mammals, 11-15 November Galveston TX. p.50.

Gallagher, M. L., & Hall., J. D. (1993). *A comparison of the robustness of the Kolmogorov-Smirnov goodness of fit test and the nearest-neighbor analysis to determine changes in patterns of distribution of migrating bowhead (Balaena mysticetus) whales in the presence of industrial activity in Cam*. Paper presented at the Tenth Biennial Conference on the Biology of Marine Mammals, Galveston, Texas.

Gambaiani, D. D., Mayol, P., Isaac, S. J., & Simmonds, M. P. (2009). Potential impacts of climate change and greenhouse gas emissions on Mediterranean marine ecosystems and cetaceans. *Journal of the Marine Biological Association of the United Kingdom, 89*(1), 179-201.

Gambell, R. (1985). Fin Whale Balaenoptera physalus (Linnaeus, 1758) *Handbook of Marine Mammals. Vol. 3: The Sirenians and Baleen Whales* (pp. 171-192). London, U.K.: Academic Press.

Gambell, R. (1985). Sei whale, *Balaenoptera borealis* Lesson, 1828. In S. H. Ridway & S. R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 3: The Sirenians and Baleen Whales, pp. 155-170). London: Academic Press.

Garcia-Fernandez, A. J., Gomez-Ramirez, P., Martinez-Lopez, E., Hernandez-Garcia, A., Maria-Mojica, P., Romero, D., . . . Bellido, J. J. (2009). Heavy metals in tissues from loggerhead turtles (Caretta caretta) from the southwestern Mediterranean (Spain). *Ecotoxicology and Environmental Safety, 72*(2), 557-563. doi: 10.1016/j.ecoenv.2008.05.003

García-Fernández, A. J., Gómez-Ramírez, P., Martínez-López, E., Hernández-García, A., María-Mojica, P., Romero, D., . . . Bellido, J. J. (2009). Heavy metals in tissues from loggerhead turtles (*Caretta caretta*) from the southwestern Mediterranean (Spain). *Ecotoxicology and Environmental Safety, 72*(2), 557-563.

Gardner, S. C., Fitzgerald, S. L., Vargas, B. A., & Rodriguez, L. M. (2006). Heavy metal accumulation in four species of sea turtles from the Baja California Peninsula, Mexico. *Biometals, 19*(1), 91-99.

Gardner, S. C., Fitzgerald, S. L., Vargas, B. A., & Rodriguez, L. M. (2006). Heavy metal accumulation in four species of sea turtles from the Baja California peninsula, Mexico. *Biometals, 19*, 91-99.

Gardner, S. C., Pier, M. D., Wesselman, R., & Juarez, J. A. (2003). Organochlorine contaminants in sea turtles from the Eastern Pacific. *Marine Pollution Bulletin, 46*, 1082-1089.

Garofalo, L., Mingozzi, T., Mico, A., & Novelletto, A. (2009). Loggerhead turtle (*Caretta caretta*) matrilines in the Mediterranean: Further evidence of genetic diversity and connectivity. *Marine Biology, 156*(10), 2085-2095. doi: 10.1007/s00227-009-1239-5

Garrett, C. (2004). Priority Substances of Interest in the Georgia Basin - Profiles and background information on current toxics issues *Technical Supporting Document* (pp. 402): Canadian Toxics Work Group Puget Sound/Georgia Basin International Task Force.

Gaskin, D. E. (1973). Sperm whales in the western South Pacific. (Physeter catodon). *New Zealand Journal of Marine and Freshwater Research, 7-Jan*(2-Jan), 1-20.

Gaspin, J. B. (1975). Experimental investigations of the effects of underwater explosions on swimbladder fish, I. 1973 Chesapeake Bay tests. Final Report. Silver Spring, Marland: Naval Surface Weapons Center, White Oak Laboratory.

Gauthier, J., & Sears, R. (1999). Behavioral response of four species of balaenopterid whales to biopsy sampling. *Marine Mammal Science, 15*(1), 85-101.

Gauthier, J. M., Metcalfe, C. D., & Sears, R. (1997). Chlorinated organic contaminants in blubber biopsies from northwestern Atlantic balaenopterid whales summering in the Gulf of St Lawrence. *Marine Environmental Research, 44*(2), 201-223.

Gende, S. M., Hendrix, A. N., Harris, K. R., Eichenlaub, B., Nielsen, J., & Pyare, S. (2011). A Bayesian approach for understanding the role of ship speed in whale-ship encounters. *Ecological Applications, 21*(6), 2232-2240.

George, J. C. (2010). *Some recent biological findings on bowhead whales: Implications to management and offshore industrial act.* Paper presented at the Alaska Marine Science Symposium, Anchorage, Alaska.

George, R. H. (1997). Health Probles and Disease of Sea Turtles. In P. l. Lutz & J. A. Musick (Eds.), *The Biology of Sea Turtles*. Washington D.C.: CRC Press.

Geraci, J. R., & Aubin, D. J. S. (1990). *Sea Mammals and Oil: Confronting the Risks*. San Diego, CA: Academic Press.

Geraci, J. R., & Lounsbury, V. J. (2005). *Marine Mammals Ashore: A Field Guide for Strandings* (Second ed.). Baltimore, Maryland: National Aquarium in Baltimore.

Geraci, J. R., & Lounsbury, V. J. (2005). *Marine mammals ashore: A field guide for strandings. 2nd ed.* Baltimore, Maryland: National Aquarium in Baltimore.

Gero, S., Engelhaupt, D., Rendell, L., & Whitehead., H. (2009). Who Cares? Between-group variation in alloparental caregiving in sperm whales. *Behavioral Ecology, 20*(4), 838-843.

Gerrodette, T., & DeMaster, D. P. (1990). Quantitative determination of optimum sustainable population level. *Marine Mammal Science, 6*(1), 1-16.

Giese, B. S., & Carton, J. A. (1999). Interannual and decadal variability in the tropical and midlatitude Pacific Ocean. *Journal of Climate, 12*(3402-3418).

Giese, M. (1996). Effects of human activity on adelie penguin *Pygoscelis adeliae* breeding success. *Biological Conservation, 75*(2), 157-164.

Gill, J. A., & Sutherland, W. J. (2001). Predicting the consequences of human disturbance from behavioral decisions. In L. M. Gosling & W. J. Sutherland (Eds.), *Behavior and Conservation* (pp. 51-64). Cambridge: Cambridge University Press.

Gilman, E. L. (2009). Guidelines to reduce sea turtle mortality in fishing operations. Rome: FAO.

Gilmartin, W. G. (1983). Recovery plan for the Hawaiian monk seal, *Monachus schauinslandi*: National Marine Fisheries Service.

Gilpatrick, J. W., Jr., Perryman, W. L., R. L. Brownell, J., Lynn, M. S., & Deangelis., M. L. (1997). Geographical variation in North Pacific and Southern Hemisphere blue whales (Balaenoptera musculus). Unpublished paper to the IWC Scientific Committee. 33 pp. Bournemouth, September (SC/49/O9).

Girard, C., Tucker, A. D., & Calmettes, B. (2009). Post-nesting migrations of loggerhead sea turtles in the Gulf of Mexico: dispersal in highly dynamic conditions. *Marine Biology, 156*(9), 1827-1839. doi: 10.1007/s00227-009-1216-z

Gitschlag, G. R., & Herczeg, B. A. (1994). Sea turtle observations at explosive removals of energy structures. *Marine Fisheries Review, 56*(2), 1-8.

Glass, A. H., Cole, T. V. N., & Garron, M. (2010). Mortality and serious injury determinations for baleen whale stocks along the United States and Canadian Eastern Seaboards, 2004-2008. (pp. 27): NMFS.

Gless, J. M., Salmon, M., & Wyneken, J. (2008). Behavioral responses of juvenile leatherbacks *Dermochelys coriacea* to lights used in the longline fishery. *Endangered Species Research, 5*, 239-247.

Glockner-Ferrari, D. A., Ferrari, M. J., & McSweeney., D. (1987). *Occurrence of abnormalities, injuries, and strandings of humpback whales in Hawaiian waters.* Paper presented at the Seventh Biennial Conference on the Biology of Marine Mammals, 5-9 December Miami Florida. p.26.

Glockner, D. A., & Venus, S. C. (1983). Identification, growth rate, and behavior of humpback whale (*Megaptera novaeangliae*) cows and calves in the waters off Maui, Hawaii, 1977-79. In R. Payne (Ed.), *Communication and Behavior of Whales* (pp. 223-258). Boulder, CO: Westview Press.

Goddard, P. C., & Rugh, D. J. (1998). A group of right whales seen in the Bering Sea in July 1996. *Marine Mammal Science, 14*(2), 344-349.

Goddard, P. D., & Rugh, D. J. (1998). A group of right whales seen in the Bering Sea in July 1996. *Marine Mammal Science, 14*(2), 344-349.

Godley, B. J. (1999). Do heavy metal concentrations pose a threat to marine turtles from the Mediterranean Sea? *Marine Pollution Bulletin, 38*(6), 497 - 502.

Godley, B. J., Thompson, D. R., & Furness, R. W. (1999). Do heavy metal concentrations pose a threat to marine turtles from the Mediterranean Sea? *Marine Pollution Bulletin, 38*, 497-502.

Goff, G. P., & Lien, J. (1988). Atlantic leatherback turtles, Dermochelys coriacea, in cold water off Newfoundland and Labrador. *The Canadian Field-Naturalist, 102*, 1-5.

Goldbogen, J. A., Southall, B. L., Deruiter, S. L., Calambokidis, J., Friedlaender, A. S., Hazen, E. L., . . . Tyack, P. L. (2013). Blue whales respond to simulated mid-frequency military sonar. *Proceedings of the Royal Society of London Series B Biological Sciences, 280*(1765), Article 20130657.

Goold, J. C., & Jones, S. E. (1995). Time and frequency domain characteristics of sperm whale clicks. *Journal of the Acoustical Society of America, 98*(3), 1279-1291.

Goold, J. C., Whitehead, H., & Reid, R. J. (2002). North Atlantic sperm whale, Physeter macrocephalus, strandings on the coastlines of the British Isles and Eastern Canada. *The Canadian Field-Naturalist, 116*, 371–388.

Gordon, J., Moscrop, A., Carlson, C., Ingram, S., Leaper, R., Matthews, J., & Young, K. (1998). Distribution, movements and residency of sperm whales off the Commonwealth of Dominica, eastern Caribbean: Implications for the development and regulation of the local whalewatching industry. *Report of the International Whaling Commission, 48*, 551-557.

Gordon, J. C. D. (1987). *The behaviour and ecology of sperm whales off Sri Lanka. (Physeter macrocephalus).* (Ph.D. dissertation), University of Cambridge, Cambridge. 347 pp.

Gorga, M. P., Neely, S. T., Dorn, P. A., Dierking, D., & Cyr, E. (2002). Evidence of upward spread of suppression in DPOAE measurements. *The Journal of the Acoustical Society of America, 112*(6), 2910-2920.

Gosho, M. E., Rice, D. W., & Breiwick., J. M. (1984). The sperm whale, Physeter macrocephalus. *Marine Fisheries Review, 46*(4), 54-64. the status of Endangered Whales. 100Pgs.

Gotelli, N. J. (2001). *A Primer of Ecology*. Sunderland, Massachusetts: Sinauer Associates, Inc.

Graff, D. (1995). Nesting and hunting survey of the marine turtles of the island of Sao Tome (pp. 38): ECOFAC Componente de Sao Tome e Principe.

Grant, S. C. H., & Ross, P. S. (2002). Southern Resident killer whales at risk: toxic chemicals in the British Columbia and Washington environment *Canadian Technical Report of Fisheries and Aquatic Sciences 2412* (pp. 124). Sidney, B.C.: Fisheries and Oceans Canada.

Green, D., & Ortiz-Crespo, F. (1982). Status of sea turtle populations in the central eastern Pacific. In K. A. Bjorndal (Ed.), *Biology and Conservation of Sea Turtles* (pp. 221-233). Washington, D. C.: Smithsonian Institution Press.

Green, G. A., Brueggerman, J. J., Grotefendt, R. A., Bowlby, C. E., Bonnell, M. L., & Balcomb III, K. C. (1992). Cetacean distribution and abundance off Oregon and Washington, 1989-1990 *Oregon and Washington Marine Mammal and Seabird Surveys. Minerals Management Service Contract Report 14-12-0001-30426*.

Gregory, P., & Rowden, A. A. (2001). Behaviour patterns of bottlenose dolphins (*Tursiops truncatus*) relative to tidal state, time-of-day, and boat traffic in Cardigan Bay, West Wales. *Aquatic Mammals, 27*(2), 105-114.

Gregr, E. J., & Coyle., K. O. (2009). The biogeography of the North Pacific right whale (Eubalaena japonica). *Progress in Oceanography, 20-Mar*(4-Mar), 188-198.

Gregr, E. J., Nichol, L., Ford, J. K. B., Ellis, G., & Trites, A. W. (2000). Migration and population structure of northeastern Pacific whales off coastal British Columbia: An analysis of commercial whaling records from 1908-1967. *Marine Mammal Science, 16*(4), 699-727.

Gregr, E. J., & Trites, A. W. (2001). Predictions of critical habitat for five whale species in the waters of coastal British Columbia. *Canadian Journal of Fisheries and Aquatic Sciences, 58*(7), 1265-1285.

Groombridge, B. (1982). The IUCN Amphibia - Reptilia Red Data Book *Part 1. Testudines, Crocodylia, Rhynchocephalia. International Union Conservation Nature and Natural Resources*.

Groombridge, B., & Luxmoore, R. (1989). *The Green Turtle and Hawksbill (Reptilia: Cheloniidae): World Status, Exploitation and Trade*. Lausanne, Switzerland: Secretariat of the Convention on International Trade in Endangered Species of Wild Fauna and Flora.

Guerra, A., Gonzalez, A. F., & Rocha, F. (2004). A review of the records of giant squid in the north-eastern Atlantic and severe injuries in *Architeuthis dux* stranded after acoustic explorations. *International Council for the Exploration of the Seas (ICES) Annual Science Conference, 22-25 September 2004, Vigo, Spain. ICES CM 2004/CC:29*.

Hain, J. H. W., Carter, G. R., Kraus, S. D., Mayo, C. A., & Winn., H. E. (1982). Feeding behavior of the humpback whale, Megaptera novaeangliae, in the western North Atlantic. *Fishery Bulletin, 80*(2), 259-268.

Hain, J. H. W., Ellis, S. L., Kenney, R. D., Clapham, P. J., Gray, B. K., Weinrich, M. T., & Babb, I. G. (1995). Apparent bottom feeding by humpback whales on Stellwagen Bank. *Marine Mammal Science, 11*(4), 464-479.

Hain, J. H. W., Hyman, M. A. M., Kenney, R. D., & Winn, H. E. (1985). The role of cetaceans in the shelf-edge region of the Northeastern United States. *Marine Fisheries Review, 47*(1), 13-17.

Hain, J. H. W., Ratnaswamy, M. J., Kenney, R. D., & Winn, H. E. (1992). The fin whale, *Balaenoptera physalus*, in waters of the northeastern United States continental shelf. *Reports of the International Whaling Commission, 42*, 653-669.

Hallegraeff, G. M. (2010). Ocean climate change, phytoplankton community responses, and harmful algal blooms: a formidable predictive challenge. *Journal of Phycology, 46*, 220-235.

Halvorsen, M. B., Wysocki, L. E., & Popper, A. N. (2006a). Effects of high-intensity sonar on fish. *The Journal of the Acoustical Society of America, 119*(5), 3283.

Halvorsen, M. B., Wysocki, L. E., & Popper, A. N. (2006b). Effects of high-intensity sonar on fish. *Journal of the Acoustical Society of America, 119*(5), 3283.

Hamilton, P. K., Stone, G. S., & Martin., S. M. (1997). Note on a deep humpback whale (Megaptera novaeangliae) dive near Bermuda. *Bulletin of Marine Science, 61*(2), 491-494.

Haraguchi, K., Hisamichi, Y., & Endo., T. (2006). Bioaccumulation of naturally occurring mixed halogenated dimethylbipyrroles in whale and dolphin products on the Japanese market. *Archives of Environmental Contamination and Toxicology, 51*(1), 135-141.

Hare, S. R., & Mantua, N. J. (2000). Empirical evidence for North Pacific regime shifts in 1977 and 1989. *Progress in Oceanography, 47*, 103-145.

Hare, S. R., Mantua, N. J., & Francis, R. C. (1999). Inverse production regimes: Alaska and west coast Pacific salmon. *Fisheries Habitat, 24*(1), 6-14.

Harington, C., Ross, R., Mathewes, R., Stewart, K., & Beattie, O. (2004). A late Pleistocene Steller sea lion (Eumetopias jubatus) from Courtenay, British Columbia: its death, associated biota, and paleoenvironment. *Canadian Journal of Earth Sciences, 41*, 13.

Hartwell, S. I. (2004). Distribution of DDT in sediments off the central California coast. *Marine Pollution Bulletin, 49*(4), 299-305.

Harwood, L. A., & Stirling., I. (1987). *Patterns of aggregation in ringed seals, bearded seals and bowhead whales in the Beaufort Sea during late summer.* Paper presented at the Seventh Biennial Conference on the Biology of Marine Mammals, 5-9 December Miami Florida. p.29.

Harwood, M. B., & Hembree., D. (1987). Incidental catch of small cetaceans in the offshore gillnet fishery in northern Australian waters: 1981-1985. *Report of the International Whaling Commission, 37,* 363-367.

Hashagen, K. A., Green, G. A., & Adams., B. (2009). Observations of humpback whales, Megaptera novaeangliae, in the Beaufort Sea, Alaska. *Northwestern Naturalist, 90*(2), 160-162.

Hastings, M., & Carlson, T. (2008). Barotrauma in aquatic animals. *Bioacoustics, 17-Jan*(3-Jan), 307-310. Special Issue on the International Conference on the Effects of Noise on Aquatic Life. Edited By A. Hawkins, A. N. Popper & M. Wahlberg.

Hastings, M. C., Popper, A. N., Finneran, J. J., & Lanford, P. J. (1996). Effects of low-frequency underwater sound on hair cells of the inner ear and lateral line of the teleost fish *Astronotus ocellatus*. *Journal of the Acoustical Society of America, 99,* 8.

Hatase, H., Kinoshita, M., Bando, T., Kamezaki, N., Sato, K., Matsuzawa, Y., . . . Sakamoto, W. (2002). Population structure of loggerhead turtles, *Caretta caretta,* nesting in Japan: Bottlenecks on the Pacific population. *Marine Biology, 141,* 299-305.

Hatase, H., Matsuzawa, Y., Sakamoto, W., Baba, N., & Miyawaki, I. (2002). Pelagic habitat use of an adult Japanese male loggerhead turtle *Caretta caretta* examined by the Argos satellite system. *Fisheries Science, 68,* 945-947.

Hatase, H., & Sakamoto, W. (2004). Forage-diving behaviour of adult Japanese female loggerhead turtles (*Caretta caretta*) inferred from Argos location data. *Journal of the Marine Biological Association of the United Kingdom, 84,* 855-856.

Hatase, H., Takai, N., Matsuzawa, Y., Sakamoto, W., Omuta, K., Goto, K., . . . Fujiwara, T. (2002). Size-related differences in feeding habitat use of adult female loggerhead turtles *Caretta caretta* around Japan determined by stable isotope analyses and satellite telemetry. *Marine Ecology Progress Series, 233,* 273-281.

Hatch, L. T., Clark, C. W., Parijs, S. M. V., Frankel, A. S., & Ponirakis, D. W. (2012). Quantifying loss of acoustic communication space for right whales in and around a US. National Marine Sanctuary. *Conservation Biology, 26*(6), 983-994.

Hatch, L. T., & Clark., C. W. (2004). Acoustic differentiation between fin whales in both the North Atlantic and North Pacific Oceans, and integration with genetic estimates of divergence.: Unpublished paper to the IWC Scientific Committee. 37 pp. Sorrento, Italy, July (SC/56/SD6).

Hatch, L. T., & Wright, A. J. (2007). A brief review of anthropogenic souond in the oceans. *International Journal of Comparative Psychology, 201*(2-3), 121-133.

Hauser, D. D. W., Holst, M., & Moulton, V. D. (2008). Marine mammal and sea turtle monitoring during Lamont-Doherty Earth Observatory's marine seismic program in the Eastern Tropical Pacific, April–August 2008. King City, Ontario: LGL, Ltd.

Haviland-Howell, G., Frankel, A. S., Powell, C. M., Bocconcelli, A., Herman, R. L., & Sayigh, L. S. (2007). Recreational boating traffic: A chronic source of anthropogenic noise in the Wilmington, North Carolina Intracoastal Waterway. *Journal of the Acoustical Society of America, 122*(1), 151-160.

Hawkes, L. A., Broderick, A., Godfrey, M. H., & Godley, B. J. (2007). The potential impact of climate change on loggerhead sex ratios in the Carolinas - how important are North Carolina's males? *P.153 in: Frick, M.; A. Panagopoulou; A.F. Rees; K. Williams (compilers), 27th Annual Symposium on Sea Turtle Biology and Conservation [abstracts]. 22-28 February 2007, Myrtle Beach, South Carolina. 296p.*

Hawkes, L. A., Broderick, A. C., Coyne, M. S., Godfrey, M. H., & Godley, B. J. (2007). Only some like it hot - quantifying the environmental niche of the loggerhead sea turtle. *Diversity and Distributions, 13*, 447-457.

Hawkes, L. A., Broderick, A. C., Godfrey, M. H., & Godley, B. J. (2007). Investigating the potential impacts of climate change on a marine turtle population. *Global Change Biology, 13*, 1-10.

Hayward, T. L. (2000). El Niño 1997-98 in the coastal waters of southern California: A timeline of events. *CalCOFl Reports, 41*, 98-116.

Hazel, J. (2009). Evaluation of fast-acquisition GPS in stationary tests and fine-scale tracking of green turtles. *Journal of Experimental Marine Biology and Ecology, 374*(1), 58-68. doi: 10.1016/j.jembe.2009.04.009

Heithaus, M. R. (2001). Predator-prey and competitive interactions between sharks (order Selachii) and dolphins (suborder Odontoceti): a review. *Journal of Zoology, 253*, 53-68.

Henderson, D. (1990). Gray whales and whalers on the China coast in 1869. (Eschrichtius robustus). *Whalewatcher, 24*(4), 14-16.

Henderson, D. (2008). Creation of noise standards for man: 50 Years of research. *Bioacoustics, 17*(1-3), 10-12. doi: 10.1080/09524622.2008.9753747

Henderson, D. A. (1984). 19th century gray whaling - grounds, catches and kills, practices and depletion of the whale population *The Gray Whale, Eschrichtius robustus. M. L. Jones, S. L. Swartz and S. Leatherwood (eds.). p.159-186. Academic Press, New York.*

Henry, J., & Best, P. B. (1983). Organochlorine residues in whales landed at Durban, South Africa. *Marine Pollution Bulletin, 14*(6), 223-227.

Herman, L. M. (1979). Humpback whales in Hawaiian waters: A study in historical ecology. (Megaptera novaeangliae). *Pacific Science, 33*(1), 1-16.

Herman, L. M., Baker, C. S., Forestell, P. H., & Antinoja, R. C. (1980). Right whale, Balaena glacialis, sightings near Hawaii: A clue to the wintering grounds? *Marine Ecology Progress Series, 2*, 271-275.

Hernandez-Garcia, V. (2002). Contents of the digestive tract of a false killer whale (*Pseudorca crassidens*) stranded in Gran Canaria (Canary Islands, Central East Atlantic). *Bulletin of Marine Science, 71*(1), 367-369.

Hewitt, R. P. (1985). Reaction of dolphins to a survey vessel: Effects on census data. *Fishery Bulletin, 83*(2), 187-194.

Heyning, J. E., & Lewis, T. D. (1990). Entanglements of baleen whales in fishing gear off southern California. (*Eschrichtius robustus, Balaenoptera acutorostrata, Megaptera novaeangliae*). *Report of the International Whaling Commission, 40*, 427-431.-Sc/441/Ps414).

Higdon, J. W., & Ferguson, S. H. (2009). Loss of Arctic sea ice causing punctuated change in sightings of killer whales (Orcinus orca) over the past century. *Ecological Applications, 19*(5), 1365-1375.

Hildebrand, H. (1987). A reconnaissance of beaches and coastal waters from the border of Belize to the Mississippi River as habitats for marine turtles: National Oceanic and Atmospheric

Administration, National Marine Fisheries Service, Southeast Fisheries Center, Panama City Laboratory.

Hildebrand, H. H. (1983). Random notes on sea turtles in the western Gulf of Mexico. *Western Gulf of Mexico Sea Turtle Workshop Proceedings, January 13-14, 1983*, 34-41.

Hildebrand, J. (2005). Impacts of anthropogenic sound. In J. E. Reynolds III, W. F. Perrin, R. R. Reeves, S. Montgomery & T. J. Ragen (Eds.), *Marine Mammal Research: Conservation Beyond Crisis* (pp. 101-123). Baltimore: Johns Hopkins University Press.

Hildebrand, J. (2005). Impacts of anthropogenic sound. *Marine Mammal Research: Conservation Beyond Crisis. J. E. Reynolds III, W. F. Perrin, R. R. Reeves, S. Montgomery, T. J. Ragen (eds.). p.101-123. Johns Hopkins University Press, Baltimore. 223pp.*

Hildebrand, J. A. (2005). Annex K: Report of the standing working group on environmental concerns. Appendix 3. Introduction to acoustics. *Journal of Cetacean Research and Management, 7*((Supplement)), 284 - 286.

Hildebrand, J. A. (2005). Impacts of anthropogenic sound. In J. E. Reynolds, W. F. Perrin, R. R. Reeves, S. Montgomery & a. T. J. Ragen (Eds.), *Marine mammal research: Conservation beyond crisis*. Baltimore, Maryland: Johns Hopkins University Press.

Hill, P. S., & DeMaster, D. P. (1998a). Alaska marine mammal stock assessments, 1998 *NOAA Technical Memorandum NMFS-AFSC-97*.

Hill, P. S., & DeMaster, D. P. (1998b). Alaska Marine Mammal Stock Assessments, 1998 *NOAA Technical Memorandum* (pp. 168): U.S. Department of Commerce.

Hill, P. S., Laake, J. L., & Mitchell., E. (1999). Results of a pilot program to document interactions between sperm whales and longline vessels in Alaska waters.: NOAA Technical Memorandum NMFS-AFSC-108. 51p.

Hill, R. D. (1985). Investigation of lightning strikes to water surface. *Journal of Acoustical Society of America, 78*(6), 2069-2099.

Hodge, R. P., & Wing, B. L. (2000). Occurrences of marine turtles in Alaska waters: 1960-1998. *Herpetological Review, 31*(3), 148-151.

Hoegh-Guldberg, O., & Bruno, J. F. (2010). The impact of climate change on the world's marine ecosystems. *Science, 328*(5985), 1523-1528. doi: 10.1126/science.1189930

Holberton, R. L., Helmuth, B., & Wingfield, J. C. (1996). The corticosterone stress response in gentoo and king penguins during the non-fasting period. *The Condor, 98*(4), 850-854.

Holst, M., & Beland, J. (2010). Marine mammal and sea turtle monitoring during Lamont-Doherty Earth Observatory's Shatsky Rise marine seismic program in the Northwest Pacific Ocean, July – September 2010: Lamont-Doherty Earth Observatory of Columbia University.

Holst, M., & Smultea, M. A. (2008). Marine mammal and sea turtle monitoring during Lamont-Doherty Earth Observatory's marine seismic program off Central America, Feburary-April 2008 (pp. 133). Palisades, New York: Lamont-Doherty Earth Observatory of Columbia University.

Holst, M., Smultea, M. A., Koski, W. R., & Haley, B. (2005). Marine mammal and sea turtle monitoring during Lamont-Doherty Earth Observatory's marine seismic program in the Eastern Tropical Pacific Ocean off Central America, November–December 2004. *Report from LGL Ltd., King City, Ontario, for Lamont-Doherty Earth Observatory of Columbia Univ., Palisades, NY, and National Marine Fisheries Service, Silver Spring, MD. Report TA2822-30. 125 p.*

Holt, M., Veirs, V., & Veirs, S. (2008). Investigating noise effects on the call amplitude of endangered Southern Resident killer whales (*Orcinus orca*). *Journal of the Acoustical Society of America, 123*(5 Part 2), 2985.

Holt, M. M., Noren, D. P., Veirs, V., Emmons, C. K., & Veirs, S. (2009). Speaking up: Killer whales (Orcinus orca) increase their call amplitude in response to vessel noise. *Journal of the Acoustical Society of America, 125*(1), EL27-EL32. doi: 10.1121/1.3040028

Holt, M. M., Veirs, V., & Veirs, S. (2008). Noise effects on the call amplitude of southern resident killer whales (*Orcinus orca*). *Bioacoustics, 117*(1-3), 164-166.

Hood, L. C., Boersma, P. D., & Wingfield, J. C. (1998). The adrenocortical response to stress in incubating Magellanic Penguins (Spheniscus magellanicus). *The Auk, 115*(1), 76-84.

Hooker, S. K., Baird, R. W., & Fahlman, A. (2009). Could beaked whales get the bends? Effect of diving behaviour and physiology on modelled gas exchange for three species: Ziphius cavirostris, Mesoplodon densirostris and Hyperoodon ampullatus. *Respiratory Physiology & Neurobiology, 167*(3), 235-246.

Hooker, S. K., Fahlman, A., Moore, M. J., de Soto, N. A., de Quirós, Y. B., Brubakk, A., . . . Falke, K. (2012). Deadly diving? Physiological and behavioural management of decompression stress in diving mammals. *Proceedings of the Royal Society B: Biological Sciences, 279*(1731), 1041-1050.

Horrocks, J. A., & Scott, N. M. (1991). Nest site location and nest success in the hawksbill turtle *Eretmochelys imbricata* in Barbados, West Indies. *Marine Ecology Progress Series, 69*, 1-8.

Horwood, J. W. (1987). *The sei whale: Population biology, ecology, and management*. Kent, England: Croom Helm Ltd.

Hotchkin, C., & Parks, S. (2013). The Lombard effect and other noise-induced vocal modifications: Insight from mammalian communication systems. *Biological Reviews, 88*(4), 809-824.

Houser, D. S., Dankiewicz, L. A., Stockard, T. K., & Ponganis., P. J. (2008). Ultrasound inspection for intravascular bubbles in a repetitively diving dolphin. *Bioacoustics, 17-Jan*(3-Jan), 310-312. Special Issue on the International Conference on the Effects of Noise on Aquatic Life. Edited By A. Hawkins, A. N. Popper & M. Wahlberg.

Howell, E. A., Kobayashi, D. R., Parker, D. M., Balazs, G. H., & Polovina, J. J. (2008). TurtleWatch: a tool to aid in the bycatch reduction of loggerhead turtles Caretta caretta in the Hawaii-based pelagic longline fishery. *Endangered Species Research, 5*, 267-278.

Hughes, D. A., & Richard, J. D. (1974). The nesting of the Pacific ridley turtle Lepidochelys olivacea on Playa Nancite, Costa Rica. *Marine Biology, 24*, 97-107.

Hulin, V., Delmas, V., Girondot, M., Godfrey, M. H., & Guillon, J. M. (2009). Temperature-dependent sex determination and global change: Are some species at greater risk? *Oecologia, 160*(3), 493-506. doi: 10.1007/s00442-009-1313-1

Hurrell, J. W. (1995). Decadal trends in the North Atlantic oscillation. *Science, 269*, 676-679.

Hutchinson, B. J., & Dutton, P. (2007). Modern genetics reveals ancient diversity in the loggerhead *SWOT report, volume II. State of the World's Sea Turtles*.

ICES. (2005). Report of the Ad-hoc Group on Impacts of Sonar on Cetaceans and Fish (AGISC) (pp. 25): International Council for the Exploration of the Sea, Advisory Committee on Ecosystems.

Ichihara, T. (1966). The pygmy blue whale, Balaenoptera musculus brevicauda, a new subspecies from the Antarctic. *Whales, Dolphins and Porpoises. K. S. Norris (ed.). University of California Press, Berkeley, CA. p.79-113.*

Insley, S. J. (2000). Long-term vocal recognition in the northern fur seal. (Callorhinus ursinus). *Nature, 406*(6794), 404-405.

IPCC (Ed.). (2000). *Land use, land-use change, and forestry*. Cambridge, England: Cambridge University Press.

IPCC (Ed.). (2001a). *Climate change 2001: Impacts, adaptation and vulnerability, contribution of working group II to the third assessment report of the intergovernmental panel on climate change*. Cambridge, England: Cambridge University Press.

IPCC (Ed.). (2001b). *Climate change 2001: The scientific basis, contribution of working group I to the third assessment report of the intergovernmental panel of climate change*. Cambridge, England: Cambridge University Press.

IPCC (Ed.). (2002). *Climate change and biodiversity*.

Isaac, J. L. (2008). Effects of climate change on life history: Implications for extinction risk in mammals. *Endangered Species Research*.

Ischer, T., Ireland, K., & Booth, D. T. (2009). Locomotion performance of green turtle hatchlings from the Heron Island Rookery, Great Barrier Reef. *Marine Biology, 156*(7), 1399-1409. doi: 10.1007/s00227-009-1180-7

Issac, J. L. (2009). Effects of climate change on life history: Implications for extinction risk in mammals. *Endangered Species Research, 7*(2), 115-123.

IUCN. (2010). IUCN Red List of threatened species. Version 2010.4. Retrieved Downloaded on 20 January 2011, from <www.iucnredlist.org>

Iwamoto, T., Ishii, M., Nakashima, Y., Takeshita, H., & Itoh, A. (1985). Nesting cycles and migrations of the loggerhead sea turtle in Miyazaki, Japan. *Japanese Journal of Ecology, 35*(4), 505-511.

Iwata, H., Tanabe, S., Sakai, N., & Tatsukawa, R. (1993). Distribution of persistent organochlorines in the oceanic air and surface seawater and the role of ocean on their global transport and fate. *Environmental Science and Technology, 27*(6), 1080-1098.

IWC. (2005). Annex K: Report of the standing working group on environmental concerns (pp. 22-23): International Whaling Commission.

IWC. (2006). Scientific permit whaling: Information on scientific permits, review procedure guidelines, and current permits in effect. *International Whaling Commission, http://www.iwcoffice.org/conservation/permits.htm Accessed: 3/14/2007.*

IWC. (2009). Report of the SOWER abundance workshop: International Whaling Commission.

IWC. (2012). International Whaling Commission: Whaling. from http://www.iwcoffice.org/whaling

Jahoda, M., Lafortuna, C. L., Biassoni, N., Almirante, C., Azzellino, A., Panigada, S., . . . Di Sciara, G. N. (2003). Mediterranean fin whale's (*Balaenoptera physalus*) response to small vessels and biopsy sampling assessed through passive tracking and timing of respiration. *Marine Mammal Science, 19*(1), 96-110.

James, M. C., Sherrill-mix, S. A., Martin, K., & Myers, R. A. (2006). Canadian waters provide critical foraging habitat for leatherback sea turtles. *Biological Conservation, 133*(3), 347-357.

Janik, V. M., & Thompson, P. M. (1996). Changes in surfacing patterns of bottlenose dolphins in response to boat traffic. *Marine Mammal Science, 12*(4), 597-602.

Jansen, G. (1998). *Health concepts and noise effects. In Noise as a Public Health Problem*. Paper presented at the Noise Effects '98 Conference, Sydney, Australia.

Jaquet, N., & Gendron, D. (2009). The social organization of sperm whales in the Gulf of California and comparisons with other populations. *Journal of the Marine Biological Association of the United Kingdom, 89*(5), 975-983.

Jasny, M., Reynolds, J., Horowitz, C., & Wetzler, A. (2005). Sounding the depths II: The rising toll of sonar, shipping and industrial ocean noise on marine life. New York, New York: Natural Resources Defense Council.

Jefferson, T. A., Stacey, P. J., & Baird., R. W. (1991). A review of killer whale interactions with other marine mammals: Predation to co-existence. (Orcinus orca). *Mammal Review, 21*(4), 151-180.

Jefferson, T. A., Webber, M. A., & Pitman, R. L. (2008). Marine Mammals of the World: A Comprehensive Guide to their Identification. *Academic Press, Elsevier. London, U.K.*

Jensen, A. S., & Silber, G. K. (2003). Large whale ship strike database. U.S. Department of Commerce, NOAA Technical Memorandum. NMFS-OPR.

Jensen, A. S., & Silber, G. K. (2004). Large Whale Ship Strike Database *NOAA Technical Memorandum* (pp. 37): U.S. Department of Commerce.

Jepson, P. D., Arbelo, M., Deaville, R., Patterson, J. A. P., Castro, P., Baker, J. R., . . . Fernández, A. (2003). Gas-bubble lesions in stranded cetaceans. *Nature, 425*.

Jepson, P. D., & Deaville, R. (2009). Investigation of the common dolphin mass stranding event in Cornwall, 9th June 2008. from http://randd.defra.gov.uk/Document.aspx?Document=WC0601_8031_TRP.pdf

Jepson, P. D., Houser, D. S., Crum, L. A., Tyack, P. L., & Fernández, A. (2005). *Beaked whales, sonar, and the "Bubble Hypothesis"*. Paper presented at the 16th Biennial Conference on the Biology of Marine Mammals, San Diego, California.

Jessop, T. S., Tucker, A. D., Limpus, C. J., & Whittier, J. M. (2003). Interactions between ecology, demography, capture stress, and profiles of corticosterone and glucose in a free-living population of Australian freshwater crocodiles. *General and Comparative Endocrinology, 132*(1), 161-170. doi: 10.1016/s0016-6480(03)00078-9

Jochens, A. E., Biggs, D. C., Engelhaupt, D., Gordon, J., Jaquet, N., Johnson, M. L., . . . Würsig, B. (2006). Sperm whale seismic study in the Gulf of Mexico; Summary report 2002-2004 (pp. 325). New Orleans, Louisiana: U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.

Johnson, A. M., & Kridler, E. (1983). Interisland movement of Hawaiian monk seals. *'Elepaio, 44*, 43-45.

Johnson, C. S. (1967). *Sound detection thresholds in marine mammals*. Paper presented at the Marine Bio-acoustics, Lerner Marine Laboratory, Bimini, Bahamas.

Johnson, C. S. (1971). Auditory masking of one pure tone by another in the bottlenosed porpoise. *Journal of the Acoustical Society of America, 49*(4 Part 2), 1317-1318.

Johnson, J. H., & Wolman, A. A. (1984). The humpback whale, *Megaptera novaeangliae*. *Marine Fisheries Review, 46*(4), 30-37.

Johnson, S. A., & Ehrhart, L. M. (1994). *Nest-site fidelity of the Florida green turtle*. Paper presented at the Proceedings of the Thirteenth Annual Symposium on Sea Turtle Biology and Conservation.

JOI/USSSP, C. G. E. a. (2003). The interplay of collisional tectonics and late Cenozoic glacial climate in Alaska and the northeastern Pacific Ocean. Austin, Texas: Continental

Dynamics Program-Earth Sciences Division-National Science Foundation and Joint Oceanographic Institutions/U.S. Science Support Program.

Jones, D. M., & Broadbent, D. E. (1998). Chapter 24: Human performance and noise. *In: Harris, C.M. (ed), Handbook of Acoustical Measurements and Noise Control. Acoustical Society of America, Woodbury, New York.*

Jones, M. L., & Swartz., S. L. (2009). Gray whale: Eschrichtius robustus. In W. F. Perrin, B. Wursig & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals. Second edition* (pp. 503-511). San Diego: Academic Press.

Josephson, E., Smith, T. D., & Reeves, R. R. (2008). Historical distribution of right whales in the North Pacific. *Fish and Fisheries, 9*, 155–168.

Jurasz, C. M., & Jurasz, V. (1979). Feeding modes of the humpback whale, Megaptera novaeangliae, in southeast Alaska. *Scientific Reports of the Whales Research Institute, Tokyo, 31*, 69-83.

Kalb, H., & Owens, D. (1994). *Differences between solitary and arribada nesting olive ridley females during the internesting period.* Paper presented at the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation. .

Kalb, H., Valverde, R. A., & Owens, D. (1995). *What is the reproductive patch of the olive ridley sea turtle?* Paper presented at the Twelfth Annual Workshop on Sea Turtle Biology and Conservation. .

Kamezaki, N. (1989). The nesting sites of sea turtles in the Ryukyu Archipelago and Taiwan. In M. Matsui, T. Hikida & R. C. Goris (Eds.), *Current Herpetology in East Asia* (pp. 342-348). Kyoto: Herpetological Society of Japan.

Kamezaki, N., Matsuzawa, Y., Abe, O., Asakawa, H., Fujii, T., Goto, K., . . . Wakabayashi, I. (2003). Loggerhead Turtles Nesting in Japan. In A. B. Bolten & B. E. Witherington (Eds.), *Loggerhead Sea Turtles* (pp. 210-217): Smithsonian Institution.

Kamminga, C., & Van Velden, J. (1987). Investigations on cetacean sonar VIII Sonar signals of Pseudorca erassidens in comparison with Tursiops. *Aquatic Mammals, 13*, 43-49.

Kane, A. S., Song, J., Halvorsen, M. B., Miller, D. L., Salierno, J. D., Wysocki, L. E., . . . Popper, A. N. (2010). Exposure of fish to high-intensity sonar does not induce acute pathology. *Journal of Fish Biology, 76*(7), 1825-1840. doi: 10.1111/j.1095-8649.2010.02626.x

Kannan, K., Blankenship, A. L., Jones, A. L., & Giesy, J. P. (2000). Toxicity reference values for the toxic effects of polychlorinated biphenyls to aquatic mammals. *Human and Ecological Risk Assessment, 6*(1), 181-201.

Karnad, D., Isvaran, K., Kar, C. S., & Shanker, K. (2009). Lighting the way: Towards reducing misorientation of olive ridley hatchlings due to artificial lighting at Rushikulya, India. *Biological Conservation, 142*(10), 2083-2088. doi: 10.1016/j.biocon.2009.04.004

Kaschner, K., Quick, N. J., Jewell, R., Williams, R., & Harris, C. M. (2012). Global coverage of cetacean line-transect surveys: status quo, data gaps and future challenges. *PLoS ONE, 7*(9), e44075.

Kastak, D., & Schusterman, R. J. (1996). Temporary threshold shift in a harbor seal (Phoca vitulina) (80). *Journal of the Acoustical Society of America, 100*(3), 4.

Kastak, D., & Schusterman, R. J. (1998). Low-frequency amphibious hearing in pinnipeds: Methods, measurements, noise, and ecology. *The Journal of the Acoustical Society of America, 103*(4), 13.

Kastak, D., & Schusterman, R. J. (2002). Changes in auditory sensitivity with depth in a free-diving California sea lion (Zalophus californianus). *The Journal of the Acoustical Society of America, 112*(1), 5.

Kastak, D., Southall, B. L., Schusterman, R. J., & Kastak, C. R. (2005). Underwater temporary threshold shift in pinnipeds: Effects of noise level and duration. *The Journal of the Acoustical Society of America, 118*, 3154.

Kastelein, R., & Jennings, N. (2012). Impacts of anthropogenic sounds on Phocoena phocoena (harbor porpoise) *The Effects of Noise on Aquatic Life* (pp. 311-315): Springer.

Kastelein, R. A., Heul, S. V. D., Verboom, W. C., Jennings, N., Veen, J. V. D., & Haan., D. D. (2008). Startle response of captive North Sea fish species to underwater tones between 0.1 and 64 kHz. *Marine Environmental Research, 65*(5), 369-377.

Kasuya, T., & Miyashita, T. (1988). Distribution of sperm whale stocks in the North Pacific. *Scientific Reports of the Whales Research Institute, Tokyo, 39*, 31-75.

Kato, H., & Kasuya., T. (2002). Some analyses on the modern whaling catch history of the western North Pacific stock of gray whales (Eschrichtius robustus), with special reference to the Ulsan whaling ground. *Journal of Cetacean Research and Management, 4*(3), 277-282.

Kato, H., Miyashita, T., & Shimada., H. (1995). Segregation of the two sub-species of the blue whale in the Southern Hemisphere. (Balanenoptera musculus). *Report of the International Whaling Commission, 45*, 273-283.-Sc/246/Sh210).

Kato, H., & Tokuhiro, Y. (1997). A sighting of gray whale off Kochi, southwest Japan in 1997, with some notes on its possible migration in adjacent waters of Japan. (pp. 8): IWC Scientific Committee.

Katona, S. K., & Whitehead, H. P. (1981). Identifying humpback whales using their natural markings. (*Megaptera novaeangliae*). *Polar Record, 20*(128), 439-444.

Kaufman, G. D., & Osmond, M. (1987). *Aspects of population dynamics of East Australian humpback whales*. Paper presented at the Seventh Biennial Conference on the Biology of Marine Mammals, 5-9 December Miami Florida. p.36.

Kawamura, G., Naohara, T., Tanaka, Y., Nishi, T., & Anraku, K. (2009). Near-ultraviolet radiation guides the emerged hatchlings of loggerhead turtles *Caretta caretta* (Linnaeus) from a nesting beach to the sea at night. *Marine and Freshwater Behaviour and Physiology, 42*(1), 19-30. doi: 10.1080/10236240802663580

Kelez, S., Velez-Zuzzo, X., Angulo, F., & Manrique, C. (2009). Olive ridley *Lepidochelys olivacea* nesting in Peru: The southernmost records in the eastern Pacific. *Marine Turtle Newsletter, 126*, 5-9.

Keller, J. M., Kannan, K., Taniyasu, S., Day, R. D., Arendt, M. D., Segars, A. L., & Kucklick, J. R. (2005). Perfluorinated compounds in the plasma of loggerhead and Kemp's ridley sea turtles from the southeastern coast of the United States. *Environmental Science and Technology, 39*(23), 9101-9108.

Keller, J. M., Kannan, K., Taniyasu, S., Yamashita, N., Day, R. D., Arendt, M. D., . . . Kucklick, J. R. (2005). Perfluorinated compounds in the plasma of loggerhead and Kemp's ridley sea turtles from the southeastern coast of the United States. *Environmental Science & Technology, 39*(23), 9101-9108. doi: Doi 10.1021/Es050690c

Keller, J. M., Kucklick, J. R., Harms, C. A., & McClellan-Green, P. D. (2004). Organochlorine contaminants in sea turtles: Correlations between whole blood and fat. *Environmental Toxicology and Chemistry, 23*(3), 726-738.

Keller, J. M., Kucklick, J. R., & McClellan-Green, P. D. (2004). Organochlorine contaminants in loggerhead sea turtle blood: Extraction techniques and distribution among plasma, and red blood cells. *Archives of Environmental Contamination and Toxicology, 46*, 254-264.

Keller, J. M., Kucklick, J. R., Stamper, M. A., Harms, C. A., & McClellan-Green, P. D. (2004). Associations between organochlorine contaminant concentrations and clinical health parameters in loggerhead sea turtles from North Carolina, USA. *Environmental Health Parameters, 112*(10), 1074-1079.

Keller, J. M., McClellan-Green, P. D., Kucklick, J. R., Keil, D. E., & Peden-Adams, M. M. (2006). Effects of organochlorine contaminants on loggerhead sea turtle immunity: Comparison of a correlative field study and *in vitro* exposure experiments. *Environmental Health Perspectives, 114*(1), 70-76.

Keller, J. M., McClellan-Green, P. D., Kucklick, J. R., Keil, D. E., & Peden-Adams, M. M. (2006). Turtle immunity: Comparison of a correlative field study and in vitro exposure experiments. *Environmental Health Perspectives, 114*(1), 70-76.

Keller, J. M., Stamper, M. A., Kucklick, J. R., & McClellan-Green, P. (2004). *Assessment of immune function and presence of contaminants in the loggerhead sea turtle (Caretta caretta)*. Paper presented at the Twenty-First Annual Symposium on Sea Turtle Biology and Conservation.

Kemper, C., Gibbs, P., Obendorf, D., Marvanek, S., & Lenghaus., C. (1994). A review of heavy metal and organochlorine levels in marine mammals in Australia. *Science of the Total Environment, 25*(2-3), 129-139.

Kenney, R. D. (2002). North Atlantic, North Pacific, and southern right whales. In W. F. Perrin, B. Würsig & J. G. M. Thewissen (Eds.), *Encyclopedia of marine mammals* (pp. 806-813). San Diego, California: Academic Press.

Kenney, R. D., & Winn, H. E. (1987). Cetacean biomass densities near submarine canyons compared to adjacent shelf/slope areas. *Continental Shelf Research, 7*(2), 107-114.

Kenyon, K. W., & Rice, D. W. (1959). Life history of the Hawaiian monk seal. *Pacific Science, 53*(4), 215-252.

Ketten, D. R. (1997). Structure and function in whale ears. *Bioacoustics-the International Journal of Animal Sound and Its Recording, 8*, 103-135.

Ketten, D. R. (1998). Marine Mammal Auditory Systems: A Summary of Audiometroc and Anatomical Data and its Implications for Underwater Acoustic Impacts *NOAA Technical Memorandum* (pp. 74): U.S. Department of Commerce.

Ketten, D. R. (2000). Cetacean ears. In W. W. L. Au, A. N. Popper & R. R. Fay (Eds.), *Hearing by Whales and Dolphins* (pp. 43-108). New York: Springer-Verlag.

Ketten, D. R. (2000). Cetacean ears. In W. W. L. Au, A. N. Popper & R. R. Fay (Eds.), *Hearing by Whales and Dolphins* (pp. 43-108). New York: Springer-Verlag.

Ketten, D. R. (2012). Marine mammal auditory system noise impacts: Evidence and incidence. In A. N. Popper & A. Hawkings (Eds.), *The Effects of Noise on Aquatic Life* (pp. 6): Springer Science.

Kidd, G., Jr., Mason, C., Richards, V. M., Gallun, F. J., & Durlach, N. I. (2007). Informational masking. In A. N. P. W.A. Yost, R.R. Fay (Ed.), *Springer Handbook of Auditory Research: Auditory perception of sound sources* (Vol. 29, pp. 143-190). New York, New York: Springer.

Kintisch, E. (2006). As the seas warm: Researchers have a long way to go before they can pinpoint climate-change effects on oceangoing species. *Science, 313*, 776-779.

Kjeld, J. M. (1982). Hormones, electrolytes and other blood constituents in large whales.: Unpublished paper to the IWC Scientific Committee. 4 pp. Cambridge, June (SC/34/O12).

Klinck, H., Mellinger, D. K., Klinck, K., Bogue, N. M., Luby, J. C., Jump, W. A., . . . Schorr, G. S. (2012). Near-real-time acoustic monitoring of beaked whales and other cetaceans using a Seaglider™. *PLoS ONE, 7*(5), e36128.

Kloepper, L. N., Nachtigall, P. E., Quintos, C., & Vlachos, S. A. (2012). Single-lobed frequency-dependent beam shape in an echolocating false killer whale (*Pseudorca crassidens*). *Journal of the Acoustical Society of America, 131*(1), 577-581.

Klumov, S. K. (1962). The right whales in the Pacific Ocean. *Trudy Instituta Okeanologii, 58*, 202–297.

Kobayashi, D. R., Polovina, J. J., Parker, D. M., Kamezaki, N., Cheng, I.-J., Uchida, I., . . . Balazs, G. H. (2008). Pelagic habitat characteriation of loggerhead sea turtles, Caretta caretta, in the North Pacific Ocean (1997-2006): insights from satellite tag tracking and remotely sensed data. *Journal of Marine Biology and Ecology, 356*, 96-114.

Kopitsky, K., Pitman, R. L., & Plotkin, P. (2000). *Investigations on at-sea mating and reproductive status of olive ridleys, Lepidochelys olivacea, captured in the eastern tropical Pacific.* Paper presented at the Nineteenth Annual Symposium on Sea Turtle Biology and Conservation. .

Kraght, P. E. (1995). Fronts and storms. In D. M. Considine (Ed.), *Van Nostrand's scientific encyclopedia. Eight Edition*. Reinhold, NY: Van Nostrand

Krahn, M. M., Hanson, M. B., Baird, R. W., Boyer, R. H., Burrows, D. G., Emmons, C. K., . . . Collier, T. K. (2007). Persistent organic pollutants and stable isotopes in biopsy samples (2004/2006) from Southern Resident killer whales. (Orcinus orca). *Marine Pollution Bulletin, 54*(12), 1903-1911.

Krahn, M. M., Hanson, M. B., Schorr, G. S., Emmons, C. K., Burrows, D. G., Bolton, J. L., . . . Ylitalo., G. M. (2009). Effects of age, sex and reproductive status on persistent organic pollutant concentrations in Southern Resident killer whales. *Marine Pollution Bulletin, 58*(10), 1522-1529.

Krausman, P. R., Harris, L. K., Blasch, C. L., Koenen, K. K. G., & Francine, J. (2004). Effects of military operations on behavior and hearing of endangered Sonoran pronghorn. *Wildlife Monographs*(157), 1-41.

Kritzler, H., & Wood, L. (1961). Provisional audiogram for the shark, *Carcharhinus leucas*. *Science, 133*, 1480-1482.

Kruse, S. (1991). The interactions between killer whales and boats in Johnstone Strait, B.C. In K. Pryor & K. Norris (Eds.), *Dolphin Societies - Discoveries and Puzzles*. Berkeley, California: University of California Press.

Kryter, K. (1985). *The effects of noise on man. 2nd ed*. New York, New York: Academic Press.

Kryter, K. D. (1994). *The handbook of hearing and the effects of noise: Physiology, psychology, and public health*. San Diego, California: Academic Press.

Kvadsheim, P. H., Miller, P. J. O., Tyack, P. L., Sivle, L. L. D., Lam, F.-P. A., & Fahlman, A. (2012). Estimated tissue and blood N2 levels and risk of in vivo bubble formation in deep-, intermediate- and shallow diving toothed whales during exposure to naval sonar. *Frontiers in Physiology, 3*. doi: 10.3389/fphys.2012.00125

Lafortuna, C. L., Jahoda, M., Azzellino, A., Saibene, F., & Colombini., A. (2003). Locomotor behaviours and respiratory pattern of the Mediterranean fin whale (Balaenoptera physalus). *European Journal of Applied Physiology, 303*(3-4), 387-395.

Lagerquist, B. A., Stafford, K. M., & Mate, B. R. (2000). Dive characteristics of satellite-monitored blue whales (*Balaenoptera musculus*) off the Central California coast. *Marine Mammal Science, 16*(2), 375-391.

Lagueux, C. J. (1998). *Marine Turtle fishery of Caribbean Nicaragua: human Use Patterns and Harvest Trends.* (Codtor of Philosophy Dissertation), University of Florida.

Laist, D. W., Knowlton, A. R., Mead, J. G., Collet, A. S., & Podesta, M. (2001). Collisions between ships and whales. *Marine Mammal Science, 17*(1), 35-75.

Lambertsen, R. H. (1983). *Crassicaudiasis of the North Atlantic fin whale (Balaenoptera physalus): Prevalence, pathogenesis, transmission, and life cycle.* Paper presented at the Fifth Biennial Conference on the Biology of Marine Mammals, 27 November-1 December New England Aquarium Boston MA. p.59.

Lambertsen, R. H. (1986). Disease of the common fin whale (*Balaenopters physalus*): Crassicaudiosis of the urinary system. *Journal of Mammalogy, 67*(2), 353-366.

Lambertsen, R. H. (1992). Crassicaudosis: A parasitic disease threatening the health and population recovery of large baleen whales. (*Balaenoptera musculus, Balaenoptera physalus, Megaptera novaeangliae*). *Revue Scientifique Et Technique Office International Des Epizooties, 11*(4), 1131-1141.

Lambertsen, R. H., Kohn, B. A., Sundberg, J. P., & Buergelt, C. D. (1987). Genital papillomatosis in sperm whale bulls. *Journal of Wildlife Diseases, 23*(3), 361-367.

Lammers, M. O., Fisher-Pool, P. I., Au, W. W., Meyer, C. G., Wong, K. B., & Brainard, R. E. (2011). Humpback whale Megaptera novaeangliae song reveals wintering activity in the Northwestern Hawaiian Islands. *Marine Ecol. Prog. Ser, 423*, 261-268.

Landry, A. M., Jr., & Costa, D. (1999). Status of sea turtle stocks in the Gulf of Mexico with emphasis on the Kemp's ridley. In H. Kumpf, K. Steidinger & K. Sherman (Eds.), *The Gulf of Mexico large marine ecosystem: Assessment, sustainability, and management* (pp. 248-268). Malden, Massachusetts: Blackwell Science.

Landsberg, J. H., Balazs, G. H., Steidinger, K. A., Baden, D. G., Work, T. M., & Russell, D. J. (1999). The potential role of natural tumor promoters in marine turtle fibropapillomatosis. *Journal of Aquatic Animal Health, 11*(3), 12.

Lankford, S. E., Adams, T. E., Miller, R. A., & Cech, J. J. (2005). The cost of chronic stress: Impacts of a nonhabituating stress response on metabolic variables and swimming performance in sturgeon. *Physiological and Biochemical Zoology, 78*(4), 599-609.

Latishev, V. M. (2007). Scientific report from factory ships "Vladivostok" and "Dalniy Vostok" in 1967. In Y. V. Ivashchenko, P. J. Clapham & R. L. Brownell Jr. (Eds.), *Scientific reports of Soviet whaling expeditions in the North Pacific, 1955-1978.* (Vol. NOAA Technical Memorandum NMFS-AFSC-175, pp. 16-17). Seattle, Washington: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center.

Law, R. J., Stringer, R. L., Allchin, C. R., & Jones, B. R. (1996). Metals and organochlorines in sperm whales (*Physetes macrocephalus*) stranded arounf the North Sea during the 1994/1995 winter. *Marine Pollution Bulletin, 32*(1), 72-77.

Lazar, B., & Gračan, R. (2010). Ingestion of marine debris by loggerhead sea turtles, Caretta caretta, in the Adriatic Sea. *Marine Pollution Bulletin*. doi: 10.1016/j.marpolbul.2010.09.013

Le Boeuf, B. J., & Crocker, D. E. (2005). Ocean climate and seal condition. *BMC Biology, 3*, 9.

Learmonth, J. A., Macleod, C. D., Santos, M. B., Pierce, G. J., Crick, H. Q. P., & Robinson, R. A. (2006). Potential effects of climate change on marine mammals. *Oceanography and Marine Biology: an Annual Review, 44*, 431-464.

Learmonth, J. A., Macleod, C. D., Santos, M. B., Pierce, G. J., Crick, H. Q. P., & Robinson, R. A. (2006). Potential effects of climate change on marine mammals. *Oceanography and Marine Biology: An Annual Review, 44*, 431-464.

Leatherwood, S., Awbrey, F. T., & Thomas, J. A. (1982). Minke whale response to a transiting survey vessel. *Report of the International Whaling Commission, 32*, 795-802.

Leatherwood, S., Caldwell, D. K., & Winn., H. E. (1976). Whales, dolphins, and porpoises of the western North Atlantic: A guide to their identification.: NOAA Technical Report NMFS CIRCULAR No. 396. 176p.

Leatherwood, S., Reeves, R. R., Perrin, W. F., & Evans, W. E. (1982). Whales, dolphins and porpoises of the eastern north Pacific and adjacent Arctic waters. A guide to their Identification *NOAA Technical Memorandum Report* (Vol. Circular 444, pp. 245): National Marine Fisheries Service.

LeDuc, R., Weller, D., Hyde, J., Burdin, A., Rosel, P., Brownell, R., . . . Dizon, A. (2002). Genetic differences between western and eastern North Pacific gray whales (Eschrichtius robustus). *Journal of Cetacean Research And Management, 4*(1), 1-6.

Lee Long, W. J., Coles, R. G., & McKenzie, L. J. (2000). Issues for seagrass conservation management in Queensland. *Pacific Conservation Biology, 5*(4), 321-328.

Lenhardt, M. L., Bellmund, S., Byles, R. A., Harkins, S. W., & Musick, J. A. (1983). Marine turtle reception of bone conducted sound. *The Journal of auditory research, 23*, 119-125.

Lenhardt, M. L., Moein, S. E., Musick, J. A., & Barnard, D. E. (1994). Evaluation of the Response of Loggerhead Sea Turtles (Caretta caretta) to a Fixed Sound Source. *Draft Final Report Submitted to the U.S. Army Corps of Engineers, Waterways Experiment Station*, 13.

Lesage, V., Barrette, C., Kingsley, M. C. S., & Sjare, B. (1999). The effect of vessel noise on the vocal behavior of Belugas in the St. Lawrence River estuary, Canada. *Marine Mammal Science, 15*(1), 65-84.

Lewison, R. L., Freeman, S. A., & Crowder, L. B. (2004). Quantifying the effects of fisheries on threatened species: The impact of pelagic longlines on loggerhead and leatherback sea turtles. *Ecology Letters, 7*, 221-231.

LGL Ltd. (2007). Environmental Assessment of a Marine Geophysical Survey by the *R/V Marcus G. Langseth* off Central America, January–March 2008. *Prepared for the Lamont-Doherty Earth Observatory, Palisades, NY, and the National Science Foundation, Arlington, VA, by LGL Ltd., environmental research associates, Ontario, Canada. LGL Report TA4342-1*.

Lien, J. (1994). Entrapments of large cetaceans in passive inshore fishing gear in Newfoundland and Labrador (1979-1990). *Report of the International Whaling Commission*(Special Issue 15), 149–157.

Limpus, C., & Chaloupka, M. (1997). Nonparametric regression modeling of green sea turtle growth rates (southern Great Barrier Reef). *Marine Ecology Progress Series, 149*, 23-34.

Limpus, C. J. (1985). *A study of the loggerhead turtle, Caretta caretta, in eastern Australia.* (Ph.D.), University of Queensland, Brisbane, Australia.

Limpus, C. J., & Limpus, D. J. (2003). Loggerhead turtles in the equatorial, and southern Pacific Ocean: A species in decline. In A. B. Bolten & B. E. Witherington (Eds.), *Loggerhead sea turtles* (pp. 199-209). Washington D.C.: Smithsonian Books.

Linnæus, C. (1758). *Systema Naturæ per Regna Tria Naturæ, Secundum Classes, Ordines, Genera, Species, Cum Characteribus, Differentiis, Synonymis, Locis* (Vol. Tomus I): Salvius.

Lino, S. P. P., Gonçalves, E., & Cozens, J. (2010). The loggerhead sea turtle (Caretta caretta) on Sal Island, Cape Verde: nesting activity and beach surveillance in 2009. *Arquipelago, 27*, 59-63.

Lockwood, B. L., & Somero, G. N. (2011). Invasive and native blue mussels (genus Mytilus) on the California coast: The role of physiology in a biological invasion☆. *Journal of Experimental Marine Biology and Ecology*. doi: 10.1016/j.jembe.2011.02.022

Lohmann, K. J., Putman, N. F., & Lohmann, C. M. F. (2012). The magnetic map of hatchling loggerhead sea turtles. *Current Opinion in Neurobiology, 22*(2), 336-342. doi: 10.1016/j.conb.2011.11.005

Lohoefener, R. R., Hoggard, W., Mullin, K., Roden, C., & Rogers, C. (1990). Association of sea turtles with petroleum platforms in the north-central Gulf of Mexico. *OCS Study, MMS 90-0025*, 90 pp.

Loop, K. A., Miller, J. D., & Limpus, C. J. (1995). Nesting by the Hawksbill Turtle (Eretmochelys Imbricata) on Milman Island, Great-Barrier-Reef, Australia. *Wildlife Research, 22*(2), 241-252.

Lucke, K., Siebert, U., Lepper, P. A., & Blanchet, M.-A. (2009). Temporary shift in masked hearing thresholds in a harbor porpoise (*Phocoena phocoena*) after exposure to seismic airgun stimuli. *Journal of the Acoustical Society of America, 125*(6), 4060-4070.

Luschi, P., Hays, G. C., & Papi, F. (2003). A review of long-distance movements by marine turtles, and the possible role of ocean currents. *Oikos, 103*(2), 293-302.

Luschi, P., Lutjeharms, J. R. E., Lambardi, P., Mencacci, R., Hughes, G. R., & Hays, G. C. (2006). A review of migratory behaviour of sea turtles off southeastern Africa. *South African Journal of Science, 102*, 51-58.

Lusseau, D. (2004). The hidden cost of tourism: Detecting long-term effects of tourism using behavioral information. *Ecology and Society, 9*(1), 2.

Lusseau, D., Williams, R., Wilson, B., Grellier, K., Barton, T. R., Hammond, P. S., & Thompson, P. M. (2004). Parallel influence of climate on the behaviour of Pacific killer whales and Atlantic bottlenose dolphins. *Ecology Letters, 7*, 1068-1076.

Lutcavage, M. E., & Lutz, P. L. (1997). Diving physiology *The Biology of Sea Turtles* (pp. 277-295). Boca Raton, Florida: CRC Press.

Lutcavage, M. E., Plotkin, P., Witherington, B. E., & Lutz, P. L. (1997). Human impacts on sea turtle survival. In P. L. Lutz & J. A. Musick (Eds.), *Biology of sea turtles* (pp. 387-409). New York, New York: CRC Press.

Lyrholm, T., & Gyllensten, U. (1998). Global matrilineal population structure in sperm whales as indicated by mitochondrial DNA sequences. *Proceedings of the Royal Society of London Series B, 265*(1406), 1679-1684.

Lyrholm, T., Leimar, O., & Gyllensten., U. (1996). Low diversity and biased substitution patterns in the mitochondrial DNA control region of sperm whales: Implications for

estimates of time since common ancestry. (Physeter macrocephalus). *Molecular Biology and Evolution, 13*(10), 1318-1326.

Lyrholm, T., Leimar, O., Johanneson, B., & Gyllensten, U. (1999). Sex-biased dispersal in sperm whales: Contrasting mitochondrial and nuclear genetic structure of global populations. *Transactions of the Royal Society of London, Series B: Biological Sciences, 266*(1417), 347-354.

MacCall, A., Batchelder, H., King, J., Mackas, D., Mantua, N., McFarlane, G., . . . Schwing, F. (2005). Recent ecosystem changes in the California Current System. In J. King (Ed.), *Report of the Study Group on Fisheries and Ecosystem Responses to Recent Regime Shifts* (Vol. PICES Scientific Report No. 28, pp. 162).

Mackintosh, N. A. (1965). Blue and fin whales *The Stocks of Whales* (pp. 174-182): Fishing News.

Macleod, C. D. (2009). Global climate change, range changes and potential implications for the conservation of marine cetaceans: A review and synthesis. *Endangered Species Research, 7*(2), 125-136.

Macleod, C. D., Bannon, S. M., Pierce, G. J., Schweder, C., Learmonth, J. A., Herman, J. S., & Reid, R. J. (2005). Climate change and the cetacean community of north-west Scotland. *Biological Conservation, 124*(4), 477-483.

Macleod, C. D., Santos, M. B., Reid, R. J., Scott, B. E., & Pierce., G. J. (2007). Linking sandeel consumption and the likelihood of starvation in harbour porpoises in the Scottish North Sea: Could climate change mean more starving porpoises? *Biology Letters, 3*(2), 185-188.

Madsen, P. T., Carder, D. A., Au, W. W. L., Nachtigall, P. E., Møhl, B., & Ridgway, S. H. (2003). Sound production in neonate sperm whales (L). *Journal of the Acoustical Society of America, 113*(6), 2988-2991.

Magalhaes, S., Prieto, R., Silva, M. A., Goncalves, J., Afonso-Dias, M., & Santos, R. S. (2002). Short-term reactions of sperm whales (*Physeter macrocephalus*) to whale-watching vessels in the Azores. *Aquatic Mammals, 28*(3), 267-274.

Magalhaes, S., Prieto, R., Silva, M. A., Goncalves, J., Afonso-Dias, M., & Santos., R. S. (2002). Short-term reactions of sperm whales (Physeter macrocephalus) to whale-watching vessels in the Azores. *Aquatic Mammals, 28*(3), 267-274.

Malme, C. I., Miles, P. R., Clark, C. W., Tyack, P., & Bird, J. E. (1983). Investigations of the potential effects of underwater noise from petroleum industry activities on migrating gray whale behavior. Final report for the period of 7 June 1982 - 31 July 1983.: Report No. 5366. For U.S. Department of the Interior, Minerals Management Service, Alaska OCS Office, Anchorage, AK 99510. 64pp.

Malme, C. I., Miles, P. R., Clark, C. W., Tyack, P., & Bird, J. E. (1984). Investigations of the potential effects of underwater noise from petroleum industry activities on migrating gray whale behavior: Phase II: January 1984 migration (pp. 357): U.S. Department of Interior, Minerals Management Service, Alaska OCS Office.

Malme, C. I., Miles, P. R., Miller, G. W., Richardson, W. J., Roseneau, D. G., Thomson, D. H., & Green Jr., C. R. (1989). Analysis and ranking of the acoustic disturbance potential of petroleum industry activities and other sources of noise in the environment of marine mammals in Alaska (Vol. Contract No. 14-12-0001-30365, pp. 304): Minerals Management Service, U.S. Department of the Interior.

Mansfield, A. W. (1985). Status of the blue whale, Balaenoptera musculus, in Canada. *The Canadian Field-Naturalist, 99*(3), 417-420.

Mansfield, K. L., Saba, V. S., Keinath, J. A., & Musick, J. A. (2009). Satellite tracking reveals a dichotomy in migration strategies among juvenile loggerhead turtles in the Northwest Atlantic. *Marine Biology, 156*(12), 2555-2570.

Mantua, N. (2002). Pacific-Decadal Oscillation (PDO). In M. C. MacCracken & J. S. Perry (Eds.), *Encyclopedia of global environmental change. Volume 1: The earth system: Physical and chemical dimensions of global environmental change* (pp. 592-594). Chichister, England: John Wiley & Sons, Ltd.

Mantua, N. J., & Hare, S. R. (2002). The Pacific decadal oscillation. *Journal of Oceanography, 58*(1), 35-44.

Mantua, N. J., & Hare, S. R. (2002). The Pacific Decadal Oscillation. *Journal of Oceanography, 58*, 35-44.

Mantua, N. J., Hare, S. R., Zhang, Y., Wallace, J. M., & Francis, R. C. (1997). A Pacific interdecadal climate oscillation with impacts on salmon production. *Bulletin of the American Meteorological Society, 78*(6), 1069-1079.

Marcano, L. A., & Alió-M, J. J. (2000). *Incidental capture of sea turtles by the industrial shrimping fleet off northwestern Venezuela*. Paper presented at the 18th International Sea Turtle Symposium.

Marcovaldi, M. A., & Chaloupka, M. (2007). Conservation status of the loggerhead sea turtle in Brazil: An encouraging outlook. *Endangered Species Research, 3*, 133-143.

Margaritoulis, D., Argano, R., Baran, I., Bentivegna, F., Bradai, M. N., Camiñas, J. A., . . . Lazar, B. (2003). Loggerhead turtles in the Mediterranean Sea: Present knowledge and conservation perspectives. In A. B. Bolten & B. E. Witherington (Eds.), *Loggerhead sea turtles* (pp. 175-198). Washington, D. C.: Smithsonian Books.

Márquez, M. R. (1990). Sea turtles of the world. An annotated and illustrated catalogue of sea turtle species known to date. *FAO Species Catalog, FAO Fisheries Synopsis 11(125):81p.*

Márquez, M. R., Carrasco, M. A., Jimenez, M. C., S., C. P.-., & Bravo-G, R. (2005). *Kemp's, and olive ridley sea turtles populations status.* Paper presented at the Twenty-first Annual Symposium on Sea Turtle Biology and Conservation. .

Márquez, M. R., Peñaflores, C., & Vasconcelos, J. (1996). Olive ridley turtles (Lepidochelys olivacea) show signs of recovery at La Escobilla, Oaxaca. *Marine Turtle Newsletter, 73*, 5-7.

Marshall, C. D., Moss, A. L., & Guzman, A. (2009). Loggerhead sea turtle (*Caretta caretta*) feeding on mackerel-baited longline hooks. *Integrative and Comparative Biology, 49*, E266-E266.

Marsili, L., & Focardi, S. (1996). Organochlorine levels in subcutaneous blubber biopsies of fin whales (*Balaenoptera physalus*) and striped dolphins (*Stenella coeruleoalba*) from the Mediterranean Sea. *Environmental Pollution, 91*(1), 1-9.

Martin, K. J., Alessi, S. C., Gaspard, J. C., Tucker, A. D., Bauer, G. B., & Mann, D. A. (2012). Underwater hearing in the loggerhead turtle (Caretta caretta): a comparison of behavioral and auditory evoked potential audiograms. *The Journal of Experimental Biology, 215*(17), 3001-3009.

Masaki, Y. (1976). Biological studies on the North Pacific sei whale. *Bulletin of Far Seas Fishery Research, 14*, 1-104.

Masaki, Y. (1977). The separation of the stock units of sei whales in the North Pacific. *Report of the International Whaling Commission*(Special Issue 1), 71-79.

Maser, C., Mate, B. R., Franklin, J. F., & Dyrness, C. T. (1981). Natural history of Oregon coast mammals *General Technical Report* (Vol. PNW-133, pp. 524). Portland, OR: U.S. Department of Agriculture, Forest Service.

Mate, B. R., Lagerquist, B. A., & Calambokidis., J. (1998). *The movements of North Pacific blue whales off southern California and their southern fall migration.* Paper presented at the The World Marine Mammal Science Conference, 20-24 January Monaco. p.87-88. (=Twelth Biennial Conference on the Biology of Marine Mammals).

Mate, B. R., Nieukirk, S., Mesecar, R., & Martin., T. (1992). Application of remote sensing methods for tracking large cetaceans: North Atlantic right whales (Eubalaena glacialis). Unpublished report for contract #14-12-0001-30411 to Minerals Management Service, U.S. Department of the Interior, Reston VA 22091. 167pp.

Matkin, C. O., & Saulitis, E. (1997). Restoration notebook: killer whale (Orcinus orca). *Exxon Valdez Oil Spill Trustee Council, Anchorage, Alaska.*

Matkin, C. O., Saulitis, E. L., Ellis, G. M., Olesiuk, P., & Rice, S. D. (2008). Ongoing population-level impacts on killer whales *Orcinus orca* following the '*Exxon Valdez*' oil spill in Prince William Sound, Alaska. *Marine Ecology Progress Series, 356*, 269-281.

Mattson, M. C., Thomas, J. A., & Aubin, D. S. (2005). Effects of boat activity on the behavior of bottlenose dolphins (*Tursiops truncatus*) in waters surrounding Hilton Head Island, South Carolina. *Aquatic Mammals, 31*(1), 133-140.

Maury, M. F. (1852). Whale chart of the world, series F (wind, and current charts), sheet 1 (Washington, D.C. 1852), sheets 2–4 (no date).

Maury, M. F. (1853). A chart showing the favourite resort of the sperm and right whale *Constructed from Maury's whale chart of the world.*

May-Collado, L. J., & Wartzok, D. (2008). A comparison of bottlenose dolphin whistles in the Atlantic Ocean: Factors promoting whistle variation. *Journal of Mammalogy, 89*(5), 1229-1240.

Mazaris, A. D., Kallimanis, A. S., Sgardelis, S. P., & Pantis, J. D. (2008). Do long-term changes in sea surface temperature at the breeding areas affect the breeding dates and reproduction performance of Mediterranean loggerhead turtles? Implications for climate change. *Journal of Experimental Marine Biology and Ecology*.

Mazaris, A. D., Kallimanis, A. S., Tzanopoulos, J., Sgardelis, S. P., & Pantis, J. D. (2009). Sea surface temperature variations in core foraging grounds drive nesting trends and phenology of loggerhead turtles in the Mediterranean Sea. *Journal of Experimental Marine Biology and Ecology*.

Mazaris, A. D., Matsinos, G., & Pantis, J. D. (2009). Evaluating the impacts of coastal squeeze on sea turtle nesting. *Ocean & Coastal Management, 52*(2), 139-145. doi: 10.1016/j.ocecoaman.2008.10.005

McCarthy, A. L., Heppell, S., Royer, F., Freitas, C., & Dellinger, T. (2010). Identification of likely foraging habitat of pelagic loggerhead sea turtles (*Caretta caretta*) in the North Atlantic through analysis of telemetry track sinuosity. *Progress in Oceanography*.

McCauley, R. D., Fewtrell, J., Duncan, A. J., Jenner, C., Jenner, M.-N., Penrose, J. D., . . . McCabe, K. (2000). Marine seismic surveys: Analysis and propagation of air-gun signals; and effects of air-gun exposure on humpback whales, sea turtles, fishes and squid (pp. 203). Western Australia: Curtin University of Technology.

McCauley, R. D., Fewtrell, J., Duncan, A. J., Jenner, C., Jenner, M.-N., Penrose, J. D., . . . McCabe, K. (2000b). Marine seismic surveys: analysis and propagation of air-gun signals; and effects of air-gun exposure on humpback whales, sea turtles, fishes and squid. *Prepared for the Australian Petroleum Production Exploration Association by the Centre for Marine Science and Technology, Project CMST 163, Report R99-15. 203p.*

McCauley, R. D., Fewtrell, J., Duncan, A. J., Jenner, C., Jenner, M.-N., Penrose, J. D., . . . McCabe, K. (2000a). Marine seismic surveys - a study of environmental implications. *Australian Petroleum Production & Exploration Association (APPEA) Journal, 40*, 692-708.

McCauley, R. D., Fewtrell, J., & Popper, A. N. (2003). High intensity anthropogenic sound damages fish ears. *The Journal of the Acoustical Society of America, 113*(1), 638-642.

McCracken, M., & Forney, K. (2010). Preliminary assessment of incidental interactions with marine mammals in the Hawaii longline deep and shallow set fisheries. *NMFS, Pacific Islands Fisheries Science Center Working Paper WP-10-001. 27p.*

McCracken, M. L., & Forney, K. A. (2010). Preliminary assessment of incidental interactions with marine mammals in the hawaii longline deep and shallow set fisheries: NOAA, NMFS.

McDonald, M. A., Calambokidis, J., Teranishi, A. M., & Hildebrand, J. A. (2001). The acoustic calls of blue whales off California with gender data. *Journal of the Acoustical Society of America, 109*(4), 1728-1735.

McDonald, M. A., Hildebrand, J. A., Webb, S., Dorman, L., & Fox, C. G. (1993). Vocalizations of blue and fin whales during a midocean ridge airgun experiment. *Journal of the Acoustical Society of America, 94*(3 pt.2), 1849.

McDonald, M. A., Hildebrand, J. A., & Webb, S. C. (1995). Blue and fin whales observed on a seafloor array in the northeast Pacific. *Journal of the Acoustical Society of America, 98*(2 Part 1), 712-721.

McDonald, M. A., Hildebrand, J. A., & Wiggins, S. M. (2006). Increases in deep ocean ambient noise in the Northeast Pacific west of San Nicolas Island, California. *Journal of the Acoustical Society of America, 120*(2), 711-718.

McDonald, M. A., Hildebrand, J. A., Wiggins, S. M., Thiele, D., Glasgow, D., & Moore, S. E. (2005). Sei whale sounds recorded in the Antarctic. *Journal of the Acoustical Society of America, 118*(6), 3941-3945.

McEwen, B. S., & Wingfield, J. C. (2003). The concept of allostasis in biology and biomedicine. *Hormones and Behavior, 43*(1), 2-15. doi: 10.1016/s0018-506x(02)00024-7

McFarlane, G. A., & Beamish, R. J. (1992). Climatic influence linking copepod production with strong yearclasses in sablefish, *Anoplopoma fimbria*. *Canadian Journal of Fisheries and Aquatic Sciences, 49*, 743-753.

McKenzie, C., Godley, B. J., Furness, R. W., & Wells, D. E. (1999). Concentrations and patterns of organochlorine contaminants in marine turtles from Mediterranean and Atlantic waters. *Marine Environmental Research, 47*, 117-135.

McMahon, C. R., Bradshaw, C. J. A., & Hays, G. C. (2007). Satellite tracking reveals unusual diving characteristics for a marine reptile, the olive ridley turtle *Lepidochelys olivacea*. *Marine Ecology Progress Series, 329*, 239-252.

McMahon, C. R., & Burton, H. R. (2005). Climate change and seal survival: Evidence for environmentally mediated changes in elephant seal, Mirounga leonina, pup survival.

Proceedings of the Royal Society of London Series B Biological Sciences, 272(1566), 923-928.

McMahon, C. R., & Burton., H. R. (2005). Climate change and seal survival: Evidence for environmentally mediated changes in elephant seal, Mirounga leonina, pup survival. *Proceedings of the Royal Society of London Series B Biological Sciences, 272*(1566), 923-928.

McMahon, C. R., & Hays, G. C. (2006). Thermal niche, large-scale movements and implications of climate change for a critically endangered marine vertebrate. *Global Change Biology, 12*(7), 1330-1338. doi: 10.1111/j.1365-2486.2006.01174.x

Melcon, M. L., Cummins, A. J., Kerosky, S. M., Roche, L. K., Wiggins, S. M., & Hildebrand, J. A. (2012). Blue whales respond to anthropogenic noise. *PLoS ONE, 7*(2), e32681.

Mellinger, D. K., & Clark, C. W. (2003). Blue whale (Balaenoptera musculus) sounds from the North Atlantic. *Journal of the Acoustical Society of America, 114*(2), 1108-1119. doi: 10.1121/1.1593066

Metcalfe, C., Koenig, B., Metcalfe, T., Paterson, G., & Sears, R. (2004). Intra- and inter-species differences in persistent organic contaminants in the blubber of blue whales and humpback whales from the Gulf of St. Lawrence, Canada. *Marine Environmental Research, 57*, 245–260.

Meylan, A. B., Bowen, B. W., & Avise, J. C. (1990). A genetic test of the natal homing versus social facilitation models for green turtle migration. *Science, 248*, 724-727.

Meylan, A. B., Schroeder, B. A., & Mosier, A. (1995). Sea turtle nesting activity in the State of Florida 1979-1992. *Florida Department of Environmental Protection (In press Florida Marine Research Publications)*(52), 63.

Miksis-Olds, J. L., Donaghay, P. L., Miller, J. H., Tyack, P. L., & Reynolds, J. E. (2007). Simulated vessel approaches elicit differential responses from manatees. *Marine Mammal Science, 23*(4), 1001-1001.

Miller, E. H., & Job, D. A. (1992). Airborne acoustic communication in the Hawaiian monk seal, Monachus schauinslandi *Marine Mammal Sensory Systems* (pp. 485-531): Springer.

Miller, P. J., Kvadsheim, P. H., Lam, F.-P. A., Wensveen, P. J., Antunes, R., Alves, A. C., . . . Sivle, L. D. (2012). The Severity of Behavioral Changes Observed During Experimental Exposures of Killer (Orcinus orca), Long-Finned Pilot (Globicephala melas), and Sperm (Physeter macrocephalus) Whales to Naval Sonar. *Aquatic Mammals, 38*(4), 362-401.

Miller, P. J. O., Biassoni, N., Samuels, A., & Tyack, P. L. (2000a). Whale songs lengthen in response to sonar. *Nature, 405*(6789), 903-903.

Miller, P. J. O., Biassoni, N., Samuels, A., & Tyack, P. L. (2000b). Whale songs lengthen in response to sonar. *Nature, 405*(6789), 903.

Mills, S. K., & Beatty, J. H. (1979). The propensity interpretation of fishes. *Philosophy of Science, 46*(2), 263-286.

Minobe, S. (1997). A 50-70 year climatic oscillation over the North Pacific and North America. *Geophysical Research Letters, 24*(6), 683-686.

Minobe, S. (1999). Resonance in bidecadal and pentadecadal climate oscillations over the North Pacific: Role in climatic regime shifts. *Geophysical Research Letters, 26*(7), 855-858.

Mitchell, E. (1974). Present status of northwest Atlantic fin and other whale stocks *The Whale Problem: A Status Report* (pp. 108-169). Cambridge, Massachusetts: Harvard University Press.

Miyashita, T., Kato, H., & Kasuya, T. (1995). *Worldwide map of cetacean distribution based on Japanese sighting data (Volume 1).* Shizuoka: National Research Institute of Far Seas Fisheries.

Mizroch, S. A., Rice, D. W., & Breiwick, J. M. (1984). The blue whale, Balaenoptera musculus. *Marine Fisheries Review, 46*(4), 15-19.

Moberg, G. P. (1987). Influence of the adrenal axis uponthe gonads. In J. Clarke (Ed.), *Oxford reviews in reproductive biology* (pp. 456 - 496). New York, New York: Oxford University Press.

Moberg, G. P. (2000). Biological response to stress: implications for animal welfare. In G. P. Moberg & J. A. Mench (Eds.), *The biology of animal stress* (pp. 1 - 21). Oxford, United Kingdom: Oxford University Press.

Mobley Jr, J. R. (2001). Results of 2001 aerial surveys of humpback whales north of Kauai (pp. 8): North Pacific Acoustic Laboratory (NPAL) Program, Scripps Institution of Oceanography.

Mobley Jr, J. R. (2006). Results of 2006 aerial surveys of humpback whales north of Kauai. *Marine Mammal Research Consultants, Ltd. Annual report submitted to the North Pacific Acoustic Laboratory (NPAL), Scripps Oceanographic Institution, La Jolla, CA.*

Mobley Jr., J. R. (2003). Results of 2003 aerial surveys of humpback whales north of Kauai. (pp. 18): North Pacific Acoustic Laboratory (NPAL) Program, Scripps Oceanographic Institution.

Mobley Jr., J. R. (2004). Results of marine mammal surveys on US Navy underwater ranges in Hawaii and Bahamas: Office of Naval Research.

Mobley Jr., J. R. (2005). Results of 2005 aerial surveys of humpback whales north of Kauai (pp. 18): Scripps Oceanographic Institution, North Pacific Acoustic Laboratory (NPAL) Program.

Moein Bartol, S., & Musick, J. A. (2003). Sensory biology of sea turtles. In P. L. Lutz, J. A. Musick & J. Wyneken (Eds.), *The Biology of Sea Turtle* (Vol. II, pp. 90-95). Boca Raton, FL.: CRC Press.

Mohl, B. (1968). Hearing in seals. *The Behavior and Physiology of Pinnipeds. p.172-195. R. J. Harrison, R. C. Hubbard, R. S. Peterson, C. E. Rice and R. J. Schusterman (eds.). Appleton-Century-Crofts, New York, NY.*

Moline, M. A., Claustre, H., Frazer, T. K., Schofields, O., & Vernet, M. (2004). Alterations of the food web along the Antarctic Peninsula in response to a regional warming trend. *Global Change Biology, 10*, 1973-1980.

Monagas, P., Oros, J., Anana, J., & Gonzalez-Diaz, O. M. (2008). Organochlorine pesticide levels in loggerhead turtles (*Caretta caretta*) stranded in the Canary Islands, Spain. *Marine Pollution Bulletin, 56*, 1949-1952.

Monzon-Arguello, C., Rico, C., Carreras, C., Calabuig, P., Marco, A., & Lopez-Jurado, L. F. (2009). Variation in spatial distribution of juvenile loggerhead turtles in the eastern Atlantic and western Mediterranean Sea. *Journal of Experimental Marine Biology and Ecology, 373*(2), 79-86. doi: 10.1016/j.jembe.2009.03.007

Mooney, T. A., Nachtigall, P. E., Breese, M., Vlachos, S., & Au, W. W. L. (2009). Predicting temporary threshold shifts in a bottlenose dolphin (*Tursiops truncatus*): The effects of noise level and duration. *Journal of the Acoustical Society of America, 125*(3), 1816-1826.

Mooney, T. A., Nachtigall, P. E., & Vlachos, S. (2009). Sonar-induced temporary hearing loss in dolphins. *Biology Letters, 5*(4), 565-567. doi: 10.1098/rsbl.2009.0099

Moore, J. E., & Barlow, J. P. (2013). Declining Abundance of Beaked Whales (Family Ziphiidae) in the California Current Large Marine Ecosystem. *PLoS ONE, 8*(1), e52770.

Moore, M. J., Bogomolni, A. L., Dennison, S. E., Early, G., Gamer, M. M., Hayward, B. A., . . . Rotstein., D. S. (2009). Gas bubbles in seals, dolphins, and porpoises entangled and drowned at depth in gillnets. *Veterinary Pathology, 46*(3), 536-547.

Moore, M. J., & Early, G. A. (2004). Cumulative sperm whale bone damage and the bends. *Science, 306*(24 December 2004), 2215.

Moore, S. E., & Clark, J. T. (2002). Potential impact of offshore human activities on gray whales (*Eschrichtius robustus*). *Journal of Cetacean Research and Management, 4*(1), 19-25.

Moore, S. E., Stafford, K. M., Dahlheim, M. E., Fox, C. G., Braham, H. W., Polovina, J. J., & Bain, D. E. (1998). Seasonal variation in reception of fin whale calls at five geographic areas in the north Pacific. *Marine Mammal Science, 14*(3), 617-627.

Moore, S. K., Mantua, N. J., & Salathé, E. P. (2011). Past trends and future scenarios for environmental conditions favoring the accumulation of paralytic shellfish toxins in Puget Sound shellfish. *Harmful Algae, 10*(5), 521-529. doi: 10.1016/j.hal.2011.04.004

Morimitsu, T., Nagai, T., Ide, M., Kawano, H., Naichuu, A., Koono, M., & Ishii, A. (1987). Mass stranding of Odontoceti caused by parasitogenic eighth cranial neuropathy. *journal of wildlife diseases, 23*(4), 586-590.

Morreale, S. J., Standora, E. A., Paladino, F. V., & Spotila, J. R. (1994). *Leatherback migrations along deepwater bathymetric contours*. Paper presented at the Proceedings of the Thirteenth Annual Symposium on Sea Turtle Biology and Conservation.

Mortimer, J. A. (1982). Factors influencing beach selection by nesting sea turtles. In K. Bjorndal (Ed.), *Biology and Conservation of Ridley Sea Turtles* (pp. 45-51). Washington, D. C.: Smithsonian Institution Press.

Mortimer, J. A., & Donnelly, M. (2007). Marine turtle specialist group review draft 2007 IUCN red list status assement hawksbill turtle (*Eretmochelys imbricata*) (pp. 117): International Union for Conservation of Nature and Natural Resources.

Mrosovsky, N. (1972). Spectrographs of the sounds of leatherback turtles. *Herpetologica*, 256-258.

Mrosovsky, N., Hopkins-Murphy, S. R., & Richardson, J. I. (1984). Sex ratio of sea turtles: Seasonal changes. *Science, 225*(4663), 739-741.

Müllner, A., Eduard Linsenmair, K., & Wikelski, M. (2004). Exposure to ecotourism reduces survival and affects stress response in hoatzin chicks (*Opisthocomus hoazin*). *Biological Conservation, 118*(4), 549-558.

Mundy, P. R., & Cooney, R. T. (2005). Physical and biological background. In P. R. Mundy (Ed.), *The Gulf of Alaska: Biology and oceanography* (pp. 15-23). Fairbanks, Alaska: Alaska Sea Grant College Program, University of Alaska.

Mundy, P. R., & Olsson, P. (2005). Climate and weather. In P. R. Mundy (Ed.), *The Gulf of Alaska: Biology and oceanography* (pp. 25-34). Fairbanks, Alaska: Alaska Sea Grant College Program, University of Alaska.

Murakawa, S. K. K., Balazs, G. H., Ellis, D. M., Hau, S., & Eames, S. M. (2000). *Trends in fibropapillomatosis among green turtles stranded in the Hawaiian Islands, 1982-98.* Paper presented at the Nineteenth Annual Symposium on Sea Turtle Biology and Conservation.

Murphy, T. M., & Hopkins, S. R. (1984). Aerial and ground surveys of marine turtle nesting beaches in the southeast region. *Final Report to NOAA/NMFS/SEFC, U.S. Department of Commerce, 73p.*

Murray, S. O., Mercado, E., & Roitblat, H. L. (1998). Characterizing the graded structure of false killer whale (Pseudorca crassidens) vocalizations. *The Journal of the Acoustical Society of America, 104*, 1679.

Musick, J. A., & Limpus, C. J. (1997). Habitat utilization, and migration in juvenile sea turtles. In P. L. Lutz & J. A. Musick (Eds.), *The biology of sea turtles* (pp. 137-163). Boca Raton, Florida: CRC Press.

Nachtigall, P. E., Pawloski, J. L., & Au, W. W. L. (2003a). Temporary threshold shifts and recovery following noise exposure in the Atlantic bottlenosed dolphin (Tursiops truncatus). *The Journal of the Acoustical Society of America, 113*(6), 5.

Nachtigall, P. E., Pawloski, J. L., & Au, W. W. L. (2003b). Temporary threshold shifts and recovery following noise exposure in the Atlantic bottlenosed dolphin (*Tursiops truncatus*). *Journal of the Acoustical Society of America, 113*(6), 3425-3429.

Nachtigall, P. E., Supin, A. Y., Pawloski, J., & Au, W. W. L. (2004). Temporary threshold shifts after noise exposure in the bottlenose dolphin (Tursiops truncatus) measured using evoked auditory potentials. *Marine Mammal Science, 20*(4), 15.

Nachtigall, P. E., Supin, A. Y., Pawloski, J. L., & Au, W. W. L. (2004). Temporary threshold shifts after noise exposure in the bottlenose dolphin (*Tursiops truncatus*) measured using evoked auditory potentials. *Marine Mammal Science, 20*(4), 672-687.

Napp, J. M., & G.L. Hunt, Jr. (2001). Anomalous conditions in the southeastern Bering Sea, 1997: linkages among climate, weather, ocean, and biology. *Fisheries and Oceanography, 10*, 61-68.

Nasu, K. (1974). Movements of baleen whales in relation to hydrographic conditions in the northern part of the North Pacific Ocean, Bering Sea. In D. W. Hood & E. J. Kelley (Eds.), *Oceanography of the Bering Sea* (pp. 345-361). Fairbanks, Alaska: University of Alaska.

Navy. (2001a). *Biological Assessment for the Employment of Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar*. Department of the Navy, Chief of Naval Operations.

Navy. (2001b). *Final Overseas Environmental Impact Statement and Environmental Impact Statement for Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar Volume 1 of 2*. Department of the Navy Chief of Naval Operations Retrieved from http://www.surtass-lfa-eis.com/EIA/index.htm.

Navy. (2001c). *Final Overseas Environmental Impact Statement and Environmental Impact Statement for Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar Volume 2 of 2*. Department of the Navy Chief of Naval Operations Retrieved from http://www.surtass-lfa-eis.com/EIA/index.htm.

Navy. (2003). Annual Report No. 1: Operation of the Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar Onboard the R/V Cory Chouest Under the National Marine Fisheries Services Letter of Authorization of 16 August 2002. Washington, D.C.: Department of the Navy, Chief of Naval Operations.

Navy. (2004). Annual Report No. 2: Operation of the Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar Onboard the R/V Cory Chouest and USNS IMPECCABLE (T-AGOS 23) Under the National Marine Fisheries Services

Letter of Authorization of 16 August 2003. Washington, D.C.: Department of the Navy, Chief of Naval Operations.

Navy. (2005a). Annual Report No. 3: Operation of the Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar Onboard the R/V Cory Chouest and USNS IMPECCABLE (T-AGOS 23) Under the National Marine Fisheries Services Letter of Authorization of 16 August 2004. Washington, D.C.: Department of the Navy, Chief of Naval Operations.

Navy. (2005b). Operation of the Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar onboard the R/V Cory Chouest and USNS IMPECCABLE (T-AGOS 23) under the National Marine Fisheries Service Letters of Authorization of 13 August 2004. Annual Report No. 3. : Department of the Navy Chief of Naval Operations.

Navy. (2006a). Annual Report No. 4: Operation of the Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar Onboard the R/V Cory Chouest and USNS IMPECCABLE (T-AGOS 23) Under the National Marine Fisheries Services Letter of Authorization of 16 August 2005. Washington, D.C.: Department of the Navy, Chief of Naval Operations.

Navy. (2006b). *Biological Assessment for the Employment of Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar*. Department of the Navy Chief of Naval Operations.

Navy. (2007a). Annual Report No. 5: Operation of the Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar Onboard the R/V Cory Chouest and USNS IMPECCABLE (T-AGOS 23) Under the National Marine Fisheries Services Letter of Authorization of 16 August 2006. Washington, D.C.: Department of the Navy, Chief of Naval Operations.

Navy. (2007b). Final Comprehensive Report for the Operation of the Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar Onboard the R/V Cory Chouest and USNS IMPECCABLE (T-AGOS 23) Under the National Marine Fisheries Service Regulations 50 CFR 216 Subpart Q. Washington, D.C.: Department of the Navy, Chief of Naval Operations.

Navy. (2007c). *Final Supplemental Environmental Impact Statement for Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar*. Department of the Navy Retrieved from http://www.surtass-lfa-eis.com/EIA/index.htm.

Navy. (2008a). Annual Report No. 1: Operation of the Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar Onboard the R/V Cory Chouest and USNS IMPECCABLE (T-AGOS 23): Department of the Navy Chief of Naval Operations.

Navy. (2008b). Annual Report No. 1: Operation of the Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar Onboard the R/V Cory Chouest and USNS IMPECCABLE (T-AGOS 23) Under the National Marine Fisheries Services Letter of Authorization of 16 August 2007. Washington, D.C.: Department of the Navy, Chief of Naval Operations.

Navy. (2009a). Annual Report No. 2: Operation of the Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar Onboard the USNS ABLE (T-AGOS 20) and USNS IMPECCABLE (T-AGOS 23): Department of the Navy Chief of Naval Operations.

Navy. (2009b). Annual Report No. 2: Operation of the Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar Onboard the USNS ABLE (T-AGOS 20) and USNS IMPECCABLE (T-AGOS 23) Under the National Marine Fisheries Services Letter of Authorization of 15 August 2008. Washington, D.C.: Department of the Navy, Chief of Naval Operations.

Navy. (2010a). Annual Report No. 3: Operation of the Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar Onboard the USNS ABLE (T-AGOS 20) and USNS IMPECCABLE (T-AGOS 23): Department of the Navy Chief of Naval Operations.

Navy. (2010b). Annual Report No. 3: Operation of the Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar Onboard the USNS ABLE (T-AGOS 20) and USNS IMPECCABLE (T-AGOS 23) Under the National Marine Fisheries Services Letter of Authorization of 13 August 2009. Washington, D.C.: Department of the Navy, Chief of Naval Operations.

Navy. (2011a). Annual Report No. 4: Operation of the Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar Onboard the USNS ABLE (T-AGOS 20), USNS EFFECTIVE, and USNS IMPECCABLE (T-AGOS 23) Under the National Marine Fisheries Services Letter of Authorization of 13 August 2010. Washington, D.C.: Department of the Navy, Chief of Naval Operations.

Navy. (2011b). *Biological Assessment for the Employment of Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar*. Department of the Navy, Chief of Naval Operations.

Navy. (2012a). *Final Supplemental Environmental Impact Statement/Supplemental Overseas Environmental Impact Statement for Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar*. Department of the Navy Chief of Naval Operations.

Navy. (2012b). Marine Species Monitoring for the U.S. Navy's Hawaii Range Complex 2012: Department of the Navy, U.S. Pacific Fleet.

Navy. (2013 -a). Unclassified Quartely Mission Report for the USNS Able (T-AGOS 20) For the Quarter Ending 14 May 2013. June 13, 2013. (pp. 4 pp.).

Navy. (2013 -b). Unclassified Quartely Mission Report for the USNS Effective (T-AGOS 21) For the Quarter Ending 14 May 2013. June 13, 2013. (pp. 4 pp.).

Navy. (2013 -c). Unclassified Quartely Mission Report for the USNS Impeccable (T-AGOS 23) For the Quarter Ending 14 May 2013. June 13, 2013. (pp. 4 pp.).

Navy. (2013 -d). Unclassified Quartely Mission Report for the USNS Victorious (T-AGOS 19) For the Quarter Ending 14 May 2013. June 13, 2013. (pp. 4 pp.).

Nelson, D., & Gruber, S. (1963). Sharks: Attraction by low-frequency sounds. *Science, 142*, 975-977.

Nelson, M., Garron, M., Merrick, R. L., Pace III, R. M., & Cole, T. V. N. (2007). Mortality and serious injury determinations for baleen whale stocks along the United States eastern seaboard and adjacent Canadian Maritimes, 2001-2005 *Reference Document 07-05* (pp. 26): U.S. Department of Commerce, NOAA, Northeast Fisheries Science Center.

Nemoto, T. (1957). Foods of baleen whales in the northern Pacific. *Scientific Reports of the Whales Research Institute Tokyo, 12*, 33-89.

Nemoto, T. (1964). School of baleen whales in the feeding areas. *Scientific Reports of the Whales Research Institute Tokyo, 18*, 89-110.

Nerini, M. K. (1984). A review of gray whale feeding ecology. (Eschrichtius robustus). In M. L. Jones, S. L. Swartz & S. Leatherwood (Eds.), *The Gray Whale, Eschrichtius robustus* (pp. 423-450). New York: Academic Press.

Nichols, W. J. (2005). *Following redwood logs, rubber ducks, and drift bottles: Transoceanic developmental migrations of loggerhead turtles in the North Pacific Ocean*. Paper presented at the Proceedings of the Twenty-First Annual Symposium on Sea Turtle Biology and Conservation.

Nichols, W. J., Resendiz, A., Seminoff, J. A., & Resendiz, B. (2000). Transpacific migration of a loggerhead turtle monitored by satellite telemetry. *Bulletin of Marine Science, 67*(3), 937-947.

Nishemura, W., & Nakahigashi, S. (1990). Incidental capture of sea turtles by Japanese research and training vessels: Results of a questionnaire. *Marine Turtle Newsletter, 51*, 1-4.

Nishimura, W. (1994). Internesting interval and nest frequency for loggerhead turtles on Inakahama Beach, Yakushima Island, Japan. *Marine Turtle Newsletter, 67*, 21-22.

Nishiwaki, M. (1966). Distribution and migration of the larger cetaceans in the North Pacific as shown by Japanese whaling results *Whales, Dolphins and Porpoises. K. S. Norris (ed.). University of California Press, Berkeley, CA. p.171-191.*

Nishiwaki, S., Tohyama, D., Ishikawa, H., Otani, S., Bando, T., Murase, H., . . . Komatsu, W. (2006). Cruise Report of the Second Phase of the Japanese Whale Research Program under Special Permit in the Antarctic (JARPAII) in 2005/2006 -Feasibility study *Paper SC/58/O7 presented to the IWC Scientific Committee, June 2006* (pp. 21). St Kitts and Nevis, WI.

Nitta, E., & Henderson, J. R. (1993). A reveiw of interactions between Hawaii's fisheries and protected species. *Marine Fisheries Review, 55*(2), 83-92.

NMFS. (1991a). Final recovery plan for the humpback whale (*Megaptera novaeangliae*). Silver Spring, Maryland: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources.

NMFS. (1991b). Recovery Plan for the Northern Right Whale (Eubalaena glacialis). *Prepared by the Right Whale Recovery Team for the National Marine Fisheries Service, Silver Spring, Maryland*.

NMFS. (1998a). *Draft Recovery Plan for the Fin Whale (Balaenoptera physalus) and the Sei Whale (Balaenoptera borealis)*. Prepared by R.R. Reeves, G.K. Silber, and P.M. Payne for the National Marine Fisheries Service, National Oceanic and Atmospheric Administration, Silver Spring, Maryland.

NMFS. (1998b). Recovery plan for the blue whale (*Balaenoptera musculus*). In R. L. Reeves, P. J. Clapham, R. L. Brownell Jr. & G. K. Silber (Eds.), (pp. 42). Silver Spring, Maryland: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

NMFS. (2001). Stock assessments of loggerhead and leatherback sea turtles and an assessment of the impact of the pelagic longline fishery on the loggerhead and leatherback sea turtles of the western North Atlantic.

NMFS. (2002). *Final biological opinion on the U.S. Navy's Surveillance Towed Array Sensor System Low Frequency Active Sonar (SURTASS LFA)*. Silver Spring, Maryland: Office of Protected Resources, Endangered Species Division.

NMFS. (2003). *Final biological opinion on the proposed letter of authorization to authorize the Navy to take marine mammals incidental to its employment of Surveillance Towed Array Sensor System Low Frequency Active Sonar for the period August 16, 2003, through*

August 15, 2004. Silver Spring, Maryland: Office of Protected Resources, Endangered Species Division.

NMFS. (2004a). Biological Opinion on the Proposed Marine Seismic Survey in the Southeast Caribbean Sea and adjacent Atlantic Ocean, and the Proposed Issuance of an MMPA authorization to Lamont-Doherty Earth Observatory [F/FPR/2003/01492]. *National Marine Fisheries Service, Office of Protected Resources, Silver Spring, Maryland. 69p.*

NMFS. (2004b). Endangered Species Act section 7 consultation on the proposed regulatory amendments to the FMP for the pelagic fisheries of the western Pacific region. *Biological Opinion, February 23, 2004.*

NMFS. (2004c). *Final biological opinion on the proposed letter of authorization to authorize the Navy to take marine mammals incidental to its employment of Surveillance Towed Array Sensor System Low Frequency Active Sonar for the period August 16, 2004, through August 15, 2005*. Silver Spring, Maryland: Office of Protected Resources, Endangered Species Division.

NMFS. (2004d). *Recovery plan for the North Atlantic right whale (Eubalaena glacialis)*. Silver Spring, Maryland: Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration.

NMFS. (2005a). Biological Opinion on the Issuance of ESA Section 10(a)(1)(A) Permit No. 1451 to the National Marine Fisheries Service - Office of Sustainable Fisheries for Research on Sea Turtles. National Marine Fisheries Service, Office of Protected Resources, Silver Spring, Maryland. 48p.

NMFS. (2005b). *Final biological opinion on the proposed letter of authorization to authorize the Navy to take marine mammals incidental to its employment of Surveillance Towed Array Sensor System Low Frequency Active Sonar for the period August 16, 2005, through August 15, 2006*. Silver Spring, Maryland: Office of Protected Resources, Endangered Species Division.

NMFS. (2006a). Biological opinion on the issuance of section lO(a)(l)(A) permits to conduct scientific research on the southern resident killer whale (*Orcinus orca*) distinct population segment and other endangered or threatened species (pp. 92). Seattle, Washington: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Region.

NMFS. (2006b). Draft recovery plan for the sperm whale (*Physeter macrocephalus*) (pp. 92). Silver Spring, Maryland: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources.

NMFS. (2006c). National Marine Fisheries Service, Office of Protected Resources website: Http://www.nmfs.noaa.gov/pr/.

NMFS. (2006d). Review of the status of the right whales in the North Atlantic and North Pacific Oceans: NOAA, National Marine Fisheries Service.

NMFS. (2006e). Biological Opinion on Permitting Structure Removal Operations on the Gulf of Mexico Outer Continental Shelf and the Authorization for Take of Marine Mammals Incidental to Structure Removals on the Gulf of Mexico Outer Continental Shelf. *National Marine Fisheries Service, Silver Spring, Maryland. 131p.*

NMFS. (2007a). *Final biological opinion on the proposed employment of Surveillance Towed Array Sensor System Low Frequency Active Sonar from 2007-2012 and proposed regulations to authorize the Navy to "take" marine mammals incidental to its employment of the Surveillance Towed Array Sensor System Low Frequency Active Sonar System*

from 2007-2012. Silver Spring, Maryland: National Marine Fisheries Service Office of Protected Resources, Endangered Species Division.

NMFS. (2007b). *Final biological opinion on the proposed letter of authorization to authorize the Navy to take marine mammals incidental to its employment of Surveillance Towed Array Sensor System Low Frequency Active Sonar for the period August 16, 2007, through August 15, 2008*. Silver Spring, Maryland: Office of Protected Resources, Endangered Species Division.

NMFS. (2007c). Recovery plan for the Hawaiian Monk Seal (*Monachus schauinslandi*). Silver Spring, Maryland.

NMFS. (2007d). Recovery plan for the Hawaiian monk seal (*Monachus schauinslandi*) (pp. 165): National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

NMFS. (2008). *Final biological opinion on the proposed letter of authorization to authorize the Navy to take marine mammals incidental to its employment of Surveillance Towed Array Sensor System Low Frequency Active Sonar for the period August 16, 2008, through August 15, 2009*. Silver Spring, Maryland: Office of Protected Resources, Endangered Species Division.

NMFS. (2009a). Environmental assessment on issuance of a permit for field research and enhancement activities on the endangered Hawaiian monk seal (Monachus schauinslandi) (pp. 52). Silver Spring, Maryland: Office of Protected Resources.

NMFS. (2009b). Fed, Researchers and Industry Tackle Pacific False kIller Whale Issue. www.wpcouncil.org/news/PressRelease_2009Apr15_Marine%20Mammal%20Advisory%20Committee.pdf

NMFS. (2009c). *Final biological opinion on the proposed letter of authorization to authorize the Navy to take marine mammals incidental to its employment of Surveillance Towed Array Sensor System Low Frequency Active Sonar for the period August 16, 2009, through August 15, 2010*. Silver Spring, Maryland: Office of Protected Resources, Endangered Species Division.

NMFS. (2009d). Status of U.S. Fisheries. Second quarter update. *http://www.nmfs.noaa.gov.sfa.statusoffisheries/2009/secondquarter/fsso_non_stock_status_cy_q2_2009.pdf*.

NMFS. (2010a). *Final biological opinion on the proposed letter of authorization to authorize the Navy to take marine mammals incidental to its employment of Surveillance Towed Array Sensor System Low Frequency Active Sonar for the period August 16, 2010, through August 15, 2011*. Silver Spring, Maryland: Office of Protected Resources, Endangered Species Division.

NMFS. (2010b). *Final Recovery plan for the fin whale (Balaenoptera physalus)*. Silver Spring, Maryland: Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration.

NMFS. (2010c). *Final Recovery plan for the sperm whale (Physeter macrocephalus)*. Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, Silver Spring, Maryland.

NMFS. (2011). *Final biological opinion on the proposed letter of authorization to authorize the Navy to take marine mammals incidental to its employment of Surveillance Towed Array Sensor System Low Frequency Active Sonar for the period August 16, 2011, through*

August 15, 2012. Silver Spring, Maryland: Office of Protected Resources, Endangered Species Division.

NMFS. (2012a). *Final biological opinion on the proposed letter of authorization to authorize the Navy to take marine mammals incidental to its employment of Surveillance Towed Array Sensor System Low Frequency Active Sonar for the period August 15, 2012, through August 14, 2013*. Silver Spring, Maryland: Office of Protected Resources, National Marine Fisheries Service.

NMFS. (2012b). *Final Biological Opinion on the U.S. Navy's proposed use of the Surveillance Towed Array Sensor System Low Frequency Active Sonar from August 2012 through August 2017*. Silver Spring, Maryland: National Marine Fisheries Service.

NMFS, & USFWS. (1991). Recovery Plan for U.S. Population of Atlantic Green Turtle *Chelonia mydas*. Washington, D.C.: National Marine Fisheries Service and U.S. Fish and Wildlife Service.

NMFS, & USFWS. (1993). Recovery Plan for the hawksbill turtle in the U.S. Caribbean Sea, Atlantic Ocean, and Gulf of Mexico *National Marine Fisheries Service and U.S. Fish and Wildlife Service*. St. Petersburg, Florida.

NMFS, & USFWS. (1995). Status reviews for sea turtles listed under the Endangered Species Act of 1973. Silver Spring, Maryland: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources.

NMFS, & USFWS. (1998a). Recovery Plan for the U.S. Pacific Populations of the Leatherback Turtles (*Dermochelys coriacea*). *National Marine Fisheries Service and U.S. Fish and Wildlife Service*. Silver Spring, Maryland.

NMFS, & USFWS. (1998b). Recovery plan for U.S. Pacific populations of the green turtle (*Chelonia mydas*). Silver Spring, Maryland: National Marine Fisheries Service.

NMFS, & USFWS. (1998c). Recovery Plan for U.S. Pacific Populations of the Green Turtle (*Chelonia mydas*) *National Marine Fisheries Service and U.S. Fish and Wildlife Service*. Silver Spring, Maryland.

NMFS, & USFWS. (1998d). Recovery Plan for U.S. Pacific Populations of the Hawksbill Turtle (*Eretmochelys imbricata*) *National Marine Fisheries Service and U.S. Fish and Wildlife Service*. Silver Spring, Maryland.

NMFS, & USFWS. (1998e). Recovery plan for U.S. Pacific populations of the olive ridley turtle (*Lepidochelys olivacea*). Silver Spring, Maryland: National Marine Fisheries Service.

NMFS, & USFWS. (2007a). Green Sea Turtle (*Chelonia mydas*) 5-Year Review: Summary and Evaluation (pp. 105). Silver Spring, MD: National Marine Fisheries Service and U.S. Fish and Wildlife Service.

NMFS, & USFWS. (2007b). Hawsbill sea turtle (Eretmochelys imbricata) 5-year review: Summary and evaluation (pp. 92 pp.). Silver Spring, Maryland Jacksonville, Florida: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources U.S. Department of the Interior, U.S. Fish and Wildlife Service, Southeast Region, Jacksonville Ecological Services Field Office.

NMFS, & USFWS. (2007c). Leatherback sea turtle (*Dermochelys coriacea*) 5-year review: Summary and evaluation (pp. 81). Silver Spring, Maryland: National Marine Fisheries Service and U.S. Fish and Wildlife Service.

NMFS, & USFWS. (2007d). Loggerhead Sea Turtle (*Caretta caretta*) 5-Year Review: Summary and Evaluation (pp. 67): National Marine Fisheries Service and U.S. Fish and Wildlife Service.

NMFS, & USFWS. (2007e). Olive ridley sea turtle (Lepidochelys olivacea) 5-year review: Summary and evaluation. Silver Spring, Maryland Jacksonville, Florida: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources U.S. Department of the Interior, U.S. Fish and Wildlife Service, Southeast Region, Jacksonville Ecological Services Field Office.

NMFS, & USFWS. (2008). Draft recovery plan for the northwest Atlantic population of the loggerhead sea turtle (*Caretta caretta*): Second revision (pp. 306). Silver Spring, Maryland: National Marine Fisheries Service and U.S. Fish and Wildlife Service.

NMFS and USFWS. (1991b). Recovery Plan for U.S. Population of Loggerhead Turtle (*Caretta caretta*). *National Marine Fisheries Service, Washington, D.C.*

NMFS and USFWS. (1998). Recovery plan for U.S. Pacific populations of the loggerhead turtle (Caretta caretta). Silver Spring, Maryland

Portland, Oregon: NMFS and USFWS.

NMFS and USFWS. (1998d). Recovery Plan for U.S. Pacific Populations of the Loggerhead Turtle (*Caretta caretta*). *National Marine Fisheries Service, Silver Spring, MD*.

NMFS and USFWS. (2007f). Olive Ridley Sea Turtle (*Lepidochelys olivacea*) 5-Year Review: Summary and Evaluation. *U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Silver Spring, Maryland; and U.S. Department of the Interior, U.S. Fish and Wildlife Service, Southeast Region, Jacksonville Ecological Services Field Office, Jacksonville, Florida. 67p.*

NMFS and USFWS. (2010). Final draft report: Summary report of a meeting of the NMFS/USFWS cross-agency working group on joint listing of North Pacific and northwest Atlantic loggerhead turtle distinct population segments. Washington, D.C.: NMFS and USFWS.

Noren, D. P., Johnson, A. H., Rehder, D., & Larson, A. (2009). Close approaches by vessels elicit surface active behaviors by southern resident killer whales. *Endangered Species Research, 8*(3), 179-192.

Norman, S. A., Bowlby, C. E., Brancato, M. S., Calambokidis, J., Duffield, D., Gearin, P. J., . . . Scordino, J. (2004). Cetacean strandings in Oregon and Washington between 1930 and 2002. *Journal of Cetacean Research and Management, 6*(1), 87-99.

Norman, S. A., C. E. Bowlby, M. S. Brancato, J. Calambokidis, D. Duffield, P. J. G., T. A. Gornall, . . . Scordino, J. (2004). Cetacean strandings in Oregon and Washington between 1930 and 2002. *Journal of Cetacean Research and Management, 6*(1), 87-99.

Norrgard, J. (1995). *Determination of stock composition and natal origin of a juvenile loggerhead turtle population (Caretta caretta) in Chesapeake Bay using mitochondrial DNA analysis.* (Masters Master's thesis), College of William and Mary, Williamsburg, Virginia.

Norris, K. S., & Harvey., G. W. (1972). A theory for the function of the spermaceti organ of the sperm whale (Physeter catodon L.). *Animal Orientation and Navigation. S. R. Galler, T.*

Schmidt-Koenig, G. J. Jacobs and R. E. Belleville (eds.). p.397-417. National Air and Space Administration, Washington, DC.

Norris, K. S., & Prescott, J. H. (1961). Observations on Pacific cetaceans of California and Mexican waters. *University of California Publications in Zoology, 63*(4), iv + 291-401.

Norris, T. F., McDonald, M., & Barlow, J. (1999). Acoustic detections of singing humpback whales (Megaptera novaeangliae) in the eastern North Pacific during their northbound migration. *The Journal of the Acoustical Society of America, 106*, 506.

Northrop, J., Cummings, W. C., & Norrison, M. F. (1971). Underwater 20-Hz signals recorded near Midway Island. *Journal of the Acoustical Society of America, 49*(6, pt. 2), 1909-1910.

Nowacek, D. P., Johnson, M. P., & Tyack, P. L. (2004). North Atlantic right whales (Eubalaena glacialis) ignore ships but respond to alerting stimuli. *Proceedings of the Royal Society of London Series B-Biological Sciences, 271*(1536), 227-231. doi: 10.1098/rspb.2003.2570

Nowacek, D. P., Johnson, M. P., & Tyack, P. L. (2004). North Atlantic right whales (*Eubalaena glacialis*) ignore ships but respond to alerting stimuli. *Proceedings of the Royal Society of London Series B Biological Sciences, 271*(1536), 227-231.

Nowacek, D. P., Thorne, L. H., Johnston, D. W., & Tyack, P. L. (2007). Responses of cetaceans to anthropogenic noise. *Mammal Review, 37*(2), 81-115.

NRC. (1990). Decline of the Sea Turtles: Causes and Prevention. National Academy of Sciences, National Academy Press, Washington, D.C.

NRC. (2000). *Marine Mammals and Low-Frequency Sound: Progress Since 1994*. Washington, D. C.: National Academy Press.

NRC. (2003a). *Ocean Noise and Marine Mammals*. Washington, D.C.: National Academy Press.

NRC. (2003b). *Ocean noise and marine mammals*. Washington, D.C.: National Research Council National Academies Press.

NRC. (2005). *Marine Mammal Populations and Ocean Noise: Determining when noise causes biologically significant effects*. Washington, D.C.: National Research Council of the National Academies.

O'Hara, J., & Wilcox, J. R. (1990). Avoidance responses of loggerhead turtles, *Caretta caretta*, to low frequency sound. *Copeia*(2), 564-567.

O'Hara, T. M., & Rice, C. (1996). Polychlorinated biphenyls. In A. Fairbrother, L.Locke & G. Hoff (Eds.), *Noninfectious Diseases of Wildlifeeds* (2nd ed., pp. 71-86). Ames, Iowa: Iowa State University Press.

Odell, D. K., Asper, E. D., Baucom, J., & Cornell, L. H. (1980). A recurrent mass stranding of the false killer whale, *Pseudorce crassidens*, in Florida. *Fishery Bulletin, 78*(1), 171-177.

Ohsumi, S., & Masaki, Y. (1975). Japanese whale marking in the North Pacific, 1963-1972. *Bulletin of the Far Seas Fisheries Research Laboratory, 12*, 171-219.

Ohsumi, S., & Masaki., Y. (1972). Report of the Scientific Committee, Annex J. Eighth memorandum on the results of Japanese stock assessment of whales in the North Pacific. *Report of the International Whaling Commission, 22*, 91-95.-Sc/23/95).

Ohsumi, S., & Wada., S. (1972). *Stock assessment of blue whales in the North Pacific. Working Paper for the 24th Meeting of the International Whaling Commission. 20 pp.*

Oleson, E. M., Boggs, C. H., Forney, K. A., Hanson, M. B., Kobayashi, D. R., Taylor, B. L., . . . Ylitalo, G. M. (2010). Status Review of Hawaiian Insular False Killer Whales (Pseudorca crassidens) under the Endangered Species Act *NOAA Technical Memorandum* (Vol. NMFS-PIFSC-22, pp. 140): Pacific Islands Fisheries Science Center, National Marine

Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce.

Omura, H. (1958). North Pacific right whale *Scientific Reports of the Whales Research Institute of Japan 13*.

Omura, H. (1974). Possible migration route of the gray whale on the coast of Japan. (Eschrichtius robustus). *Scientific Reports of the Whales Research Institute, 26*, 1-14.

Omura, H. (1984). History of gray whales in Japan. (Eschrichtius robustus) *The Gray Whale, Eschrichtius robustus. M. L. Jones, S. L. Swartz and S. Leatherwood (eds.). p.57-78. Academic Press, New York.*

Omura, H. (1988). Distribution and migration of the western Pacific stock of the gray whale (Eschrichtius robustus). *Scientific Reports of the Whales Research Institute, 39*, 1-10.

Omura, H., Ichihara, T., & Kasuya., T. (1970). Osteology of pygmy blue whale with additional information on external and other characteristics. (Balaenoptera musculus brevicauda). *Scientific Reports of the Whales Research Institute Tokyo, 22*, 1-27, +25Pls.

Omura, H., Ohsumi, S., Nemoto, K. N., & Kasuya, T. (1969). Black right whales in the north Pacific *Scientific Reports of the Whales Research Institute of Japan 21*.

Oros, J., Gonzalez-Diaz, O. M., & Monagas, P. (2009). High levels of polychlorinated biphenyls in tissues of Atlantic turtles stranded in the Canary Islands, Spain. *Chemosphere, 74*(3), 473-478. doi: 10.1016/j.chemosphere.2008.08.048

Palacios, D. M., & Mate, B. R. (1996). Attack by false killer whales (*Pseudorca crassidens*) on sperm whales (*Physeter macrocephalus*) in the Galapagos Islands. *Marine Mammal Science, 12*(4), 582-587.

Palka, D. (1996). Effects of Beaufort sea state on the sightability of harbor porpoises in the Gulf of Maine. *Reports of the International Whaling Commission, 46*, 575-582.

Pandav, B., & Choudhury, B. C. (1999). An update on the mortality of the olive ridley sea turtles in Orissa, India. *Marine Turtle Newsletter, 83*, 10-12.

Panigada, S., Pesante, G., Zanardelli, M., Capoulade, F., Gannier, A., & Weinrich, M. T. (2006). Mediterranean fin whales at risk from fatal ship strikes. *Marine Pollution Bulletin, 52*(10), 1287-1298. doi: 10.1016/j.marpolbul.2006.03.014

Papastavrou, V., Smith, S. C., & Whitehead, H. (1989). Diving behaviour of the sperm whale, Physeter macrocephalus, off the Galápagos Islands. *Canadian Journal of Zoology, 67*(4), 839-846.

Papi, F., Liew, H. C., Luschi, P., & Chan, E. H. (1995). Long-range migratory travel of a green turtle tracked by satellite: evidence for navigational ability in the open sea. *Marine Biology, 122*(2), 171-175. doi: 10.1007/bf00348929

Parker, D. M., Dutton, P. H., Kopitsky, K., & Pitman, R. L. (2003). *Movement and dive behavior determined by satellite telemetry for male and female olive ridley turtles in the Eastern Tropical Pacific*. Paper presented at the Proceedings of the Twenty-second Annual Symposium on Sea Turtle Biology and Conservation. .

Parks, S. E. (2009). Assessment of acoustic adaptations for noise compensation in marine mammals (pp. 3): Office of Naval Research.

Parks, S. E., Clark, C. W., & Tyack, P. L. (2007). Short- and long-term changes in right whale calling behavior: The potential effects of noise on acoustic communication. *Journal of the Acoustical Society of America, 122*(6), 3725-3731. doi: 10.1121/1.2799904

Parrish, F. A., & Abernathy, K. (2006). Movements of monk seals relative to ecological depth zones in the lower Northwestern Hawaiian Islands. *Atoll Research Bulletin, 543*, 115-130.

Parrish, F. A., Abernathy, K., Marshall, G. J., & Buhleier, B. M. (2002). Hawaiian monk seals (*Monachus schauinslandi*) foraging in deep-water coral beds. *Marine Mammal Science, 18*(1), 15.

Parrish, F. A., Craig, M. P., Ragen, T. J., Marshall, G. J., & Buhleier, B. M. (2000). Identifying diurnal foraging habitat of endangered Hawaiian monk seals using a seal-mounted video camera. *Marine Mammal Science, 16*(2), 392.

Patterson, B., & Hamilton, G. R. (1964). *Repetitive 20 cycle per second biological hydroacoustic signals at Bermuda*. Paper presented at the Marine Bio-acoustics, W N Tavolga ed. Pergamon Press Oxford. p.125-145. Proceedings of a Symposium held at the Lerner Marine Laboratory Bimini Bahamas April.

Patterson, P. D. (1966). Hearing in the turtle. *J. Auditory Res, 6*, 453.

Payne, K. (1985). Singing in humpback whales. *Whalewatcher, 19*(1), 3-6.

Payne, P., Nicholas, J., O'Brien, L., & Powers, K. (1986). The distribution of the humpback whale, Megaptera novaeangliae, on Georges Bank and in the Gulf of Maine in relation to densities of the sand eel, Ammodytes americanus. *Fisheries Bulletin, 84*, 271-277.

Payne, P. M., Wiley, D. N., Young, S. B., Pittman, S., P.J. Clapham, & Jossi, J. W. (1990). Recent fluctuations in the abundance of baleen whales in the southern Gulf of Maine in relation to changes in selected prey. *Fishery Bulletin, 88*, 687-696.

Payne, R., & Webb., D. (1971). Orientation by means of long range acoustic signaling in baleen whales. *Annals of the New York Academy of Sciences, 188*(1), 110-141.

Payne, R. S., & McVay, S. (1971). Songs of humpback whales. Humpbacks emit sounds in long, predictable patterns ranging over frequencies audible to humans. *Science, 173*(3997), 585-597.

Peckham, S. H., Maldonado-Diaz, D., Koch, V., Mancini, A., Gaos, A., Tinker, M. T., & Nichols, W. J. (2008). High mortality of loggerhead turtles due to bycatch, human consumption and strandings at Baja California Sur, Mexico, 2003 to 2007. *Endangered Species Research, 5*, 171-183.

Peckham, S. H., Maldonado-Diaz, D., Tremblay, Y., Ochoa, R., Polovina, J. J., Balazs, G. H., . . . Nichols, W. J. (2011). Demographic implications of alternative foraging strategies in juvenile loggerhead turtles Caretta caretta of the North Pacific Ocean. *Mar Ecol Prog Ser, 425*, 269-280.

Perkins, J. S., & Beamish, P. C. (1979). Net entanglements of baleen whales in the inshore fishery of Newfoundland. *Journal of the Fisheries Research Board of Canada, 36*, 521-528.

Perry, S. L., DeMaster, D. P., & Silber, G. K. (1999). The great whales: History and status of six species listed as endangered under the U.S. Endangered Species Act of 1973. *Marine Fisheries Review, 61*(1), 1-74.

Perry, S. L., Demaster, D. P., & Silber., G. K. (1999). The sperm whales (Physeter macrocephalus). *Marine Fisheries Review, 61*(1), 59-74. W. L. Hobart-Ed.). In the Great Whales History and status of Six Species Listed As Endangered Under the U.S. Endangered Species Act of.

Perryman, W. L., LeDuc, R., & R. L. Brownell, J. (1999). Progress report on eastern North Pacific right whale research during July 1998. *International Whaling Committee Scientific Committee paper SC/51/CAWS36*.

Perugini, M., Giammarino, A., Olivieri, V., Guccione, S., Lai, O. R., & Amorena, M. (2006). Polychlorinated biphenyls and organochlorine pesticide levels in tissues of Caretta caretta from the Adriatic Sea. *Diseases of Aquatic Organisms, 71*(2), 155-161.

Petersen, S. L., Honig, M. B., Ryan, P. G., Nel, R., & Underhill, L. G. (2009). Turtle bycatch in the pelagic longline fishery off southern Africa. *African Journal of Marine Science, 31*(1), 87-96. doi: 10.2989/ajms.2009.31.1.8.779

Peterson, C. H., Rice, S. D., Short, J. W., Esler, D., Bodkin, J. L., Ballachey, B. E., & Irons, D. B. (2003). Long-term ecosystem response to the Exxon Valdez oil spill. *Science, 302*(5653), 2082-2086.

Philippart, C. J. M., Anadón, R., Danovaro, R., Dippner, J. W., Drinkwater, K. F., Hawkins, S. J., . . . Reid, P. C. (2011). Impacts of climate change on European marine ecosystems: Observations, expectations and indicators☆. *Journal of Experimental Marine Biology and Ecology*. doi: 10.1016/j.jembe.2011.02.023

Pike, G. C., & Macaskie, I. B. (1969). Marine mammals of British Columbia. *Bulletin of the Fisheries Research Board of Canada, 171*, 1-54.

Pinela, A. M., Querouil, S., Magalhaes, S., Silva, M. A., Prieto, R., Matos, J. A., & Santos, R. S. (2009). Population genetics and social organization of the sperm whale (*Physeter macrocephalus*) in the Azores inferred by microsatellite analyses. *Canadian Journal of Zoology, 87*(9), 802-813.

Piniak, W. E. D. (2012). *Dissertation: Acoustic Ecology of Sea Turtles: Implications for Conservation.* (Ph.D.), Duke University.

Pinto De Sa Alves, L. C., Andriolo, A., Zerbini, A. N., Pizzorno, J. L. A., & Clapham., P. J. (2009). Record of feeding by humpback whales (Megaptera novaeangliae) in tropical waters off Brazil. *Marine Mammal Science, 25*(2), 416-419.

Pirotta, E., Milor, R., Quick, N., Moretti, D., Di Marzio, N., Tyack, P., . . . Hastie, G. (2012). Vessel noise affects beaked whale behavior: Results of a dedicated acoustic response study. *PLoS ONE, 7*(8), e42535.

Pitman, R. L. (1990). Pelagic distribution and biology of sea turtles in the eastern tropical Pacific. *Pp.143-148 In: T.H. Richardson, J.I. Richardson, and M. Donnelly (compilers), Proceedings of the 10th Annual Symposium on Sea Turtle Biology and Conservation. Department of Commerce, NOAA Technical Memorandum NMFS-SEFSC-278*.

Pitman, R. L. (1991). *Sea turtle associations with flotsam in the eastern tropical Pacific Ocean*. Paper presented at the Proceedings of the Eleventh Annual Workshop on Sea Turtle Biology and Conservation. .

Pitman, R. L. (1993). Seabird associations with marine turtles in the eastern Pacific Ocean. *Colonial Waterbirds, 16*(2), 194-201.

Pitman, R. L., Ballance, L. T., Mesnick, S. I., & Chivers, S. J. (2001). Killer whale predation on sperm whales: Observations and implications. *Marine Mammal Science, 17*(3), 494-507.

Plotkin, P. (2003). Adult migrations and habitat use. In P. L. Lutz, J. A. Musick & J. Wyneken (Eds.), *Biology of sea turtles* (Vol. II, pp. 225-241). Boca Raton, Florida: CRC Press.

Plotkin, P., Byles, R., & Owens, D. (1993). *Migratory and reproductive behaviour of Lepidochelys in the eastern Pacific Ocean*. Paper presented at the 13th Annual Sea Turtle Symposium on Sea Turtle Biology and Conservation.

Plotkin, P. T. (1994). *Migratory and reproductive behavior of the olive ridley turtle, Lepidochelys olivacea (Eschscholtz, 1829), in the eastern Pacific Ocean.* (Ph.D.), Texas A&M University, College Station, Texas.

Plotkin, P. T., Byles, R. A., & Owens, D. W. (1994a). *Migratory and reproductive behavior of Lepidochelys olivacea in the eastern Pacific Ocean*. Paper presented at the Proceedings of the 13th Annual Symposium on Sea Turtle Biology and Conservation, Jekyll Island, Georgia.

Plotkin, P. T., Byles, R. A., & Owens, D. W. (1994b). Post-breeding movements of male olive ridley sea turtles *Lepidochelys olivacea* from a nearshore breeding area. In K. A. Bjorndal, A. B. Bolten, D. A. Johnson & P. J. Eliazar (Eds.), *Proceedings of the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation.* (pp. 119): Department of Commerce.

Plotkin, P. T., Byles, R. A., Rostal, D. C., & Owens, D. W. (1991). *Arribadas: Social events or simply aggregations? Preliminary results from satellite telemetry*. Paper presented at the Proceedings of the Eleventh Annual Workshop on Sea Turtle Biology and Conservation. .

Plotkin, P. T., Byles, R. A., Rostal, D. C., & Owens, D. W. (1995). Independent versus socially facilitated oceanic migrations of the olive ridley, Lepidochelys olivacea. *Marine Biology, 122*, 137-143.

Plotkin, P. T., Owens, D. W., Byles, R. A., & Patterson, R. (1996). Departure of male olive ridley turtles (Lepidochelys olivacea) from a nearshore breeding ground. *Herpetologica, 52*(1), 1-7.

Plotkin, P. T., Rostal, D. C., Byles, R. A., & Owens, D. W. (1997). Reproductive and developmental synchrony in female Lepidochelys olivacea. *Journal of Herpetology, 31*(1), 17-22.

Polacheck, T., & Thorpe, L. (1990). The swimming direction of harbor porpoise in relatinship to a survey vessel. *Report of the International Whaling Commission, 40*, 463-470.

Pollack, I. (1975). Auditory informational masking. *Journal of the Acoustical Society of America, 57*((S1):S5-S5).

Poloczanska, E. S., Limpus, C. J., & Hays, G. C. (2009). Vulnerability of marine turtles in climate change *Advances in Marine Biology* (Vol. 56, pp. 151-211). New York: Academic Press.

Polovina, J., Uchida, I., Balazs, G., Howell, E. A., Parker, D., & Dutton, P. (2006). The Kuroshio Extension Bifurcation Region: a pelagic hotspot for juvenile loggerhead sea turtles. *Deep-Sea Research II, 53*, 362-339.

Polovina, J. F., Mitchum, G. T., Graham, N. E., Craig, M. P., Demartini, E. E., & Flint, E. N. (1994). Physical and biological consequences of a climate event in the central North Pacific. *Fish. Oceanography, 3*, 15-21.

Polovina, J. J., Balazs, G. H., Howell, E. A., Parker, D. M., Seki, M. P., & Dutton, P. H. (2004). Forage and migration habitat of loggerhead (Caretta caretta) and olive ridley (Lepidochelys olivacea) sea turtles in the central North Pacific Ocean. *Fisheries Oceanography, 13*(1), 36-51.

Polovina, J. J., Howell, E., Parker, D. M., & Balazs, G. H. (2003). Dive-depth distribution of loggerhead (*Carretta carretta*) and olive ridley (*Lepidochelys olivacea*) sea turtles in the central North Pacific: Might deep longline sets catch fewer turtles? *Fishery Bulletin, 101*(1), 189-193.

Polovina, J. J., Kobayashi, D. R., Parker, D. M., Seki, M. P., & Balazs, G. H. (2000). Turtles on the edge: Movement of loggerhead turtles (Caretta caretta) along oceanic fronts, spanning longline fishing grounds in the central North Pacific, 1997-1998. *Fisheries Oceanography, 9*, 71-82.

Popper, A. N. (2005). A review of hearing by sturgeon and lamprey: U.S. Army Corps of Engineers, Portland District.

Popper, A. N., Halvorsen, M. B., Kane, A., Miller, D. L., Smith, M. E., Song, J., . . . Wysocki, L. E. (2007). The effects of high-intensity, low-frequency active sonar on rainbow trout. *Journal of the Acoustical Society of America, 122*(1), 623-635.

Popper, A. N., & Hastings, M. C. (2009a). The effects of anthropogenic sources of sound on fishes. *Journal of Fish Biology, 75*(3), 455-489. doi: 10.1111/j.1095-8649.2009.02319.x

Popper, A. N., & Hastings, M. C. (2009b). The effects of human-generated sound on fish. *Integrative Zoology, 4*(1), 43-52. doi: 10.1111/j.1749-4877.2008.00134.x

Potter, J. R., Thillet, M., Douglas, C., Chitre, M. A., Doborzynski, Z., & Seekings, P. J. (2007). Visual and passive acoustic marine mammal observations and high-frequency seismic source characteristics recorded during a seismic survey. *IEEE Journal of Oceanic Engineering, 32*(2), 469-483.

Pritchard, P. C. H. (1969). *Studies of the systematics and reproductive cycles of the genus Lepidochelys.* (Ph.D.), University of Florida.

Pritchard, P. C. H. (1971). The leatherback or leathery turtle, Dermochelys coriacea. *International Union for the Conservation of Nature, Monograph, 1*, 39 pp.

Pritchard, P. C. H. (1982). Nesting of the leatherback turtle, Dermochelys coriacea, in Pacific Mexico, with a new estimate of the world population status. *Copeia, 4*, 741-747.

Pritchard, P. C. H. (1997). Evolution, phylogeny, and current status. In P. L. Lutz & J. A. Musick (Eds.), *The Biology of Sea Turtles* (pp. 1-28). Boca Raton, Florida: CRC Press.

Pugh, R. S., & Becker, P. R. (2001). Sea turtle contaminants: A review with annotated bibliography. Charleston, South Carolina: U.S. Department of Commerce, National Institute of Standards and Technology, Chemical Science and Technology Laboratory.

Pullin, A. S., & Stewart, G. B. (2006). Guidelines for systematic review in conservation and environmental management. *Conservation Biology, 20*(6), 1647-1656. doi: 10.1111/j.1523-1739.2006.00485.x

Purvis, A., Gittleman, J. L., Cowlishaw, G., & Mace, G. M. (2000). Predicting extinction risk in declining species. *Proceedings of the Royal Society B-Biological Sciences, 267*, 1947-1952.

Ragen, T. J. (1993). Status of the Hawaiian monk seal in 1992.

Rankin-Baransky, K. (1997). Origin of loggerhead turtles (Caretta caretta) in the western North Atlantic Ocean as determined by mtDNA analysis. Masters Thesis submitted to Drexel University, June 1997. 1949p.

Rankin, C. H., Abrams, T., Barry, R. J., Bhatnagar, S., Clayton, D. F., Colombo, J., . . . Thompson, R. F. (2009). Habituation revisited: An updated and revised description of the behavioral characteristics of habituation. *Neurobiology of Learning and Memory, 92*(2), 135-138. doi: DOI: 10.1016/j.nlm.2008.09.012

Rankin, S., & Barlow, J. (2007). Vocalizations of the sei whale *Balaenoptera borealis* off the Hawaiian Islands. *Bioacoustics, 16*(2), 137-145.

Reeves, R. R. (1992). Whale responses to anthropogenic sounds: a literature review. Wellington, New Zealand: New Zealand Department of Conservation.

Reeves, R. R., Leatherwood, S., & Baird, R. W. (2009). Evidence of a Possible Decline since 1989 in False Killer Whales (Pseudorca crassidens) around the Main Hawaiian Islands 1. *Pacific Science, 63*(2), 253-261.

Reeves, R. R., Leatherwood, S., & Baird., R. W. (2009). Evidence of a possible decline since 1989 in false killer whales (Pseudorca crassidens) around the main Hawaiian Islands. *Pacific Science, 63*(2), 253-261.

Reeves, R. R., Leatherwood, S., Karl, S. A., & Yohe, E. R. (1985). Whaling results at Akutan (1912-39) and Port Hobron (1926-37). *Report of the International Whaling Commission, 35*, 441-457.

Reeves, R. R., & Mitchell., E. (1987). Shore whaling for right whales in the northeastern United States.: Final report submitted to National Marine Fisheries Service, Southeast Fisheries Center, Miami, FL in partial fulfillment of contract NA85-WC-C-06194. vi + 108pp.

Reeves, R. R., & Mitchell., E. D. (1988). Current status of the gray whale, Eschrichtius robustus. *Canadian field-naturalist, 102*(2), 369-390.

Reeves, R. R., Read, A. J., & Notarbartolo-di-Sciara, G. (2001). ICRAM - Report of the Workshop on Interactions between Dolphins and Fisheries in the Mediterranean: Evaluation of Mitigation Alternatives. *Istituto Centrale per la Ricerca Applicata al Mare*, 44 pp.

Reeves, R. R., Smith, B. D., Crespo, E. A., & Notarbartolo di Sciara, G. (2003). Dolphins, whales and porpoises: 2002–2010 conservation action plan for the world's cetaceans *IUCN/SSC Cetacean Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK. ix + 139p.*

Reeves, R. R., Stewart, B. S., & Leatherwood., S. (1992). *The Sierra Club handbook of seals and sirenians.*: Sierra Club Books. San Francisco, CA. 359pgs. ISBN 0-87156-656-7.

Reeves, R. R., & Whitehead, H. (1997). Status of the sperm whale, Physeter macrocephalus, in Canada. *The Canadian Field-Naturalist, 111*(2), 15.

Reid, K., & Croxall, J. (2001). Environmental response of upper trophic-level predators reveals a system change in an Antarctic marine ecosystem. *Proceedings of the Royal Society London Series B, 268*, 377–384.

Reid, K. A., Margaritoulis, D., & Speakman, J. R. (2009). Incubation temperature and energy expenditure during development in loggerhead sea turtle embryos. *Journal of Experimental Marine Biology and Ecology, 378*, 62-68.

Reilly, S. B., & Thayer, V. G. (1990). Blue whale (*Balaenoptera musculus*) distribution in the eastern tropical Pacific. *Marine Mammal Science, 6*(4), 265-277.

Reina, R. D., Spotila, J. R., Paladino, F. V., & Dunham, A. E. (2008). Changed reproductive schedule of eastern Pacific leatherback turtles Dermochelys coriacea following the 1997–98 El Niño to La Niña transition. *Endangered Species Research*.

Reiner, F., Dos Santos, M. E., & Wenzel, F. W. (1996). Cetaceans of the Cape Verde archipelago. *Marine Mammal Science, 12*(3), 10.

Resendiz, A., Resendiz, B., Nichols, W. J., Seminoff, J. A., & Kamezaki, N. (1998). First confirmed eastwest transpacific movement of a loggerhead sea turtle, Caretta caretta, released in Baja California, Mexico. *Pacific Science, 52*(2), 151-153.

Reynolds, J. E., DeMaster, D. P., & Silber, G. K. (2002). Endangered species and populations. *Pages 373-382 in: Perrin, W.F., B. Würsig, and J.G.M. Thewissen, editors. Encyclopedia of marine mammals. Academic Press. San Diego, California*.

Rice, D. W. (1960). Population dynamics of the Hawaiian monk seal. *Journal of Mammalogy, 41*, 376-385.

Rice, D. W. (1974). Whales and whale research in the eastern North Pacific. In W. E. Schevill (Ed.), *The Whale Problem: A Status Report* (pp. 170-195). Cambridge, MA: Harvard University Press.

Rice, D. W. (1977). Synopsis of biological data on the sei whale and Bryde's whale in the eastern North Pacific. *Report of the International Whaling Commission*(Special Issue 1), 92-97.

Rice, D. W. (1978). The humpback whale in the North Pacific: distribution, exploitation, and numbers. . In K. S. Norris & R. R. Reeves (Eds.), *Report on a Workshop on Problems Related to Humpback Whales (Megaptera novaeangliae) in Hawaii* (pp. pp. 29–44): U.S. Marine Mammal Commission.

Rice, D. W. (1984). Cetaceans. *Orders and Families of Recent Mammals of the World. S. Anderson AND J. Knox Jones, Jr. (eds.). p.447-490. John Wiley AND Sons, Inc., New York.*

Rice, D. W. (1989). Sperm whale, Physeter macrocephalus (Linnaeus, 1758). In S. H. Ridway & S. R. Harrison (Eds.), *Handbook of Marine Mammals Volume 4: River Dolphins and the Larger Toothed Whales* (Vol. 4, pp. 177-233).

Rice, D. W. (1998). *Marine mammals of the world: systematics and distribution*. Lawrence, KS: Society for Marine Mammalogy.

Rice, D. W., & Wolman, A. A. (1971). *The Life History and Ecology of the Gray Whale (Eschrichtius robustus)* (Vol. 3). Stillwater, Oklahoma: American Society of Mammalogist.

Rice, D. W., Wolman, A. A., Mate, B. R., & Harvey, J. T. (1986). A Mass Stranding of Sperm Whales in Oregon - Sex and Age Composition of the School. *Marine Mammal Science, 2*(1), 64-69.

Richardson, T. H., Richardson, J. I., Ruckdeshel, C., & Dix, M. W. (1978). Remigration patterns of loggerhead sea turtles (Caretta caretta) nesting on Little Cumberland and Cumberland Islands, Georgia. *Florida Marine Research Publications, 33*, 39-44.

Richardson, W. J., Davids, R. A., Evans, C. R., & Norton, P. (1985). Distribution of bowheads and industrial activity. In W. J. Richardson (Ed.), *Behavior, disturbance and distribution of bowhead whales Balaena mysticetus in the eastern Beaufort Sea, 1980-84* (pp. 255-306). Bryan, Texas, and Reston, Virginia: Report from LGL Ecological Research Associates, Inc. for U.S. Minerals Management Service.

Richardson, W. J., Finley, K. J., Miller, G. W., & Davis, R. A. (1995). Feeding, social and migration behavior of bowhead whales, Balaena mysticetus, in Baffin Bay vs the Beaufort Sea - regions with different amounts of human activity. *Marine Mammal Science, 11*(1), 1.

Richardson, W. J., Fraker, M. A., Wursig, B., & Wells, R. S. (1985). Behavior of bowhead whales *Balaena mysticetus* summering in the Beaufort Sea: Reactions to industrial activities. *Biological Conservation, 32*(3), 195-230.

Richardson, W. J., Greene Jr., C. R., Malme, C. I., & Thomson, D. H. (1995). *Marine Mammals and Noise*. San Diego, California: Academic Press.

Richardson, W. J., Jr., C. R. G., Malme, C. I., & Thomson, D. H. (1995). *Marine Mammals and Noise*. San Diego, California: Academic Press, Inc.

Richardson, W. J., McDonald, T. L., Greene, C. R., & Blackwell, S. B. (2004). Acoustic localization of bowhead whales near Northstar, 2001-2003: Evidence of deflection at

high-noise times? In W. J. Richardson & M. T. Williams (Eds.), *Monitoring of Industrial Sounds, Seals, and Bowhead Whales Near BP's Northstar Oil Development, Alaskan Beaufort Sea, 1999-2003*. Anchorage, Alaska: LGL Ltd, Greenridge Science Inc., and WEST Inc.

Richardson, W. J., Wells, R. S., & Würsig, B. (1985). Disturbance responses of bowheads, 1980-84. In W. J. Richardson (Ed.), *Behavior, disturbance and distribution of bowhead whales Balaena mysticetus in the eastern Beaufort Sea, 1980-84* (pp. 89-196). Bryan, Texas, and Reston, Virginia: Report from LGL Ecological Research Associates, Inc. for U.S. Minerals Management Service.

Richardson, W. J., & Williams, M. T. (2003). Monitoring of industrial sounds, seals, and bowhead whales near BP's northstar oil development, Alaskan Beaufort Sea, 1999–2002 (pp. 343). King City, Ontario and Santa Barbara, California: LGL Ltd. and Greeneridge Sciences Inc.

Richardson, W. J., & Williams, M. T. (2004). Monitoring of industrial sounds, seals, and bowhead whales near BP's Northstar oil development, Alaskan Beaufort Sea, 1999-2003. Annual and comprehensive report, Dec 2004 *A report from LGL Ltd., Greenridge Sciences Inc., and WEST Inc. for BP Exploration (Alaska) Inc. LGL Report TA 4001* (pp. 297). Anchorage, AK: BP Exploration (Alaska) Inc.

Richardson, W. J., & Wursig, B. (1995). Significance of responses and noise impacts. In W. J. Richardson, C. R. Greene Jr., C. I. Malme & D. H. Thomson (Eds.), *Marine Mammals and Noise* (pp. 387-424). San Diego: Academic Press.

Richardson, W. J., & Wursig, B. (1997). Influences of man-made noise and other human actions on cetacean behaviour. *MARINE AND FRESHWATER BEHAVIOUR AND PHYSIOLOGY, 29*(1-4), 183-209.

Richardson, W. J., Würsig, B., & Greene Jr, C. R. (1990). Reactions of bowhead whales, Balaena mysticetus, to drilling and dredging noise in the Canadian Beaufort Sea. *Marine Environmental Research, 29*(2), 135-160.

Richardson, W. J., Würsig, B., & Greene Jr., C. R. (1986). Reactions of bowhead whales, Balaena mysticetus, to seismic exploration in the Canadian Beaufort Sea. *The Journal of the Acoustical Society of America, 79*(4), 1117-1128.

Richardson, W. J., & Wursig., B. (1995). Significance of responses and noise impacts. (Chapter 11) *Marine Mammals and Noise. p.387-424. Academic Press, San Diego.*

Richter, C., Dawson, S., & Slooten, E. (2006). Impacts of commercial whale watching on male sperm whales at Kaikoura, New Zealand. *Marine Mammal Science, 22*(1), 46-63.

Richter, C. F., Dawson, S. M., & Slooten., E. (2003). Sperm whale watching off Kaikoura, New Zealand: Effects of current activities on surfacing and vocalisation patterns.: Department of Conservation, Wellington, New Zealand. Science For Conservation 219. 78p.

Ridgway, S. H., Carder, D. A., Smith, R. R., Kamolnick, T., Schlundt, C. E., & Elsberry., W. R. (1997). Behavioral Responses and Temporary Shift in Masked Hearing Threshold of Bottlenose Dolphins, Tursiops truncatus, to 1-second tones of 141 to 201 dB re 1 uPa.: Technical Report 1751. Naval Command, Control and Ocean Surveilance Center, RDT&E Division, San Diego, CA 92152-5001. 32pp.

Ridgway, S. H., Wever, E. G., McCormick, J. G., Palin, J., & Anderson, J. H. (1969). Hearing in the giant sea turtle, Chelonoa mydas. *Proceedings of the National Academies of Science, 64*.

Risch, D., Corkeron, P. J., Ellison, W. T., & Parijs, S. M. V. (2012). Changes in humpback whale song occurrence in response to an acoustic source 200 km away. *PLoS ONE, 7*(1), e29741.

Ritter, F. (2002). Behavioural observations of rough-toothed dolphins (*Steno bredanensis*) off La Gomera, Canary Islands (1995-2000), with special reference to their interactions with humans. *Aquatic Mammals, 28*(1), 46-59.

Rivier, C. (1985). Luteinizing-hormone-releasing hormone, gonadotropins, and gonadol steroids in stress. *Annals of the New York Academy of Sciences, 771*, 187-191.

Robbins, J., & Mattila, D. K. (2001). Monitoring entanglements of humpback whales (*Megaptera novaeangliae*) in the Gulf of Maine on the basis of caudal peduncle scarring. Cambridge, United Kingdom: International Whaling Commission Scientific Committee.

Robinson, R. A., Crick, H. Q. P., Learmonth, J. A., Maclean, I. M. D., Thomas, C. D., Bairlein, F., . . . Visser, M. E. (2008). Travelling through a warming world: climate change and migratory species. *Endangered Species Research*.

Robinson, R. A., Learmonth, J. A., Hutson, A. M., Macleod, C. D., Sparks, T. H., Leech, D. I., . . . Crick, H. Q. P. (2005). Climate change and migratory species *BTO Research Report 414* (pp. 306). Norfolk, U.K. : Defra Research, British Trust for Ornithology.

Rolland, R. M., Parks, S. E., Hunt, K. E., Castellote, M., Corkeron, P. J., Nowacek, D. P., . . . Kraus, S. D. (2012). Evidence that ship noise increases stress in right whales. *Proceedings of the Royal Society B: Biological Sciences, 279*(1737), 2363-2368.

Romero, L. M. (2004). Physiological stress in ecology: lessons from biomedical research. *Trends in Ecology & Evolution, 19*(5), 249-255. doi: 10.1016/j.tree.2004.03.008

Rone, B. K., Douglas, A. B., Zerbini, A. N., Morse, L., Martinez, A., Clapham, P. J., & Calambokidis, J. (2010). Results from the April 2009 Gulf of Alaska Line Transect Survey (GOALS) in the Navy Training Exercise Area (pp. 39): National Marine Fisheries Service.

Rosenbaum, H. C., Brownell Jr., R. L., Brown, M. W., Schaeff, C. M., Portway, V., White, B. N., . . . DeSalle, R. (2000). World-wide genetic differentiation of *Eubalana*: questioning the number of right whale species. *Molecular Ecology, 9*, 1793-1802.

Rosman, I., Boland, G. S., Martin, L., & Chandler, C. (1987). Underwater Sightings of Sea Turtles in the Northern Gulf of Mexico. *OCS Study; MMS 87-0107*, 37.

Ross, D. (2005). Ship sources of ambient noise. *Oceanic Engineering, IEEE Journal of, 30*(2), 257-261. doi: 10.1109/joe.2005.850879

Ross, J. P. (1979). Sea turtles in the Sultanate of Oman.

Ross, J. P. (1998). *Estimations of the nesting population size of loggerhead sea turtles, Caretta caretta, Masirah Island, Sultanate of Oman*. Paper presented at the Proceedings of the Seventeenth Annual Sea Turtle Symposium.

Ross, J. P., & Barwani, M. A. (1982). Review of sea turtles in the Arabian area. In K. A. Bjorndal (Ed.), *Biology and conservation of sea turtles* (pp. 373-383). Washington, D.C.: Smithsonian Institution Press.

Ross, P. S. (2002). The role of immunotoxic environmental contaminants in facilitating the emergence of infectious diseases in marine mammals. *Human and Ecological Risk Assessment, 8*(2), 277-292.

Royall, R. (2004). The likelihood paradigm for statistical evidence. In M. L. Taper & S. R. Lele (Eds.), *The Nature of Scientific Evidence. Statistical, Philosophical, and Empirical Considerations* (pp. 119-152). Chicago, Illinois: University of Chicago Press.

Royer, T. C. (2005). Hydrographic responses at a coastal site in the northern Gulf of Alaska to seasonal and interannual forcing. *Deep-Sea Research Part Ii-Topical Studies in Oceanography, 52*(1-2), 267-288. doi: 10.1016/j.dsr2.2004.09.022

Royer, T. C. (2005). Hydrographic responses at a coastal site in the northern Gulf of Alaska to seasonal and interannual forcing. *Deep-Sea Research II, 52*, 267-288.

Royer, T. C., & Weingartner, T. (1999). *Coastal hydrographic responses in the northern Gulf of Alaska to the 1997-98 ENSO event*. Paper presented at the Proceedings of the 1998 Science Board Symposium on the impacts of the 1997/98 El Niño event on the North Pacific Ocean and its marginal seas.

Ruiz, G. M., Fofonoff, P., & Hines, A. H. (1999). Non-indigenous species as stressors in estuarine and marine communities: Assessing invasion impacts and interactions. *Limnology and Oceanography, 44*(3), 950–972.

Ruud, J. T. (1956). The blue whale. (Balaenoptera musculus). *Scientific American, 195*, 46-50.

Rybitski, M. J., Hale, R. C., & Musick, J. A. (1995). Distribution of organochlorine pollutants in Atlantic sea-turtles

Copeia(2), 379-390.

Rybitski, M. J., Hale, R. C., & Musick, J. A. (1995). Distribution of organochlorine pollutants in Atlantic sea turtles. *Copeia 1995*(2), 379-390.

Saeki, K., Sakakibara, H., Sakai, H., Kunito, T., & Tanabe, S. (2000). Arsenic accumulation in three species of sea turtles. *Biometals, 13*(3), 241-250.

Sahoo, G., Sahoo, R. K., & Mohanty-Hejmadi, P. (1996). Distribution of heavy metals in the eggs and hatchlings of olive ridley sea turtle, Lepidochelys olivacea, from Gahirmatha, Orissa. *Indian journal of marine sciences, 25*(4), 371-372.

Sahoo, G., Sahoo, R. K., & Mohanty-Hejmadi, P. (1996). Distribution of heavy metals in the eggs and hatchlings of olive ridley sea turtles, *Lepidochelys olivacea*, from Gahirmatha, Orissa. *Indian Journal of Marine Sciences, 25*(4), 371-372.

Sakai, H., Saeki, K., Ichihashi, H., Kamezaki, N., Tanabe, S., & Tatsukawa, R. (2000). Growth-related changes in heavy metal accumulation in green turtle (*Chelonia mydas*) from Yaeyama Islands, Okinawa, Japan. *Archives of Environmental Contamination and Toxicology, 39*(3), 378-385.

Salm, R. V. (1991). Turtles in Oman: Status, threats, and management options *Report of the scientific results of the IUCN coastal zone management project CZMP4:F11*.

Salm, R. V., Jensen, R. A. C., & Papastavrou, V. A. (1993). Marine fauna of Oman: Cetaceans, turtles, seabirds, and shallow water corals. Gland, Switzerland: IUCN.

Samaran, F., Guinet, C., Adam, O., Motsch, J. F., & Cansi, Y. (2010). Source level estimation of two blue whale subspecies in southwestern Indian Ocean. *Journal of the Acoustical Society of America, 127*(6), 3800-3808. doi: 10.1121/1.3409479

Samuel, Y., Morreale, S. J., Clark, C. W., Greene, C. H., & Richmond, M. E. (2005). Underwater, low-frequency noise in a coastal sea turtle habitat. *Journal of the Acoustical Society of America, 117*(3), 1465-1472. doi: 10.1121/1.1847993

Santana Garcon, J., Grech, A., Moloney, J., & Hamann, M. (2010). Relative Exposure Index: An important factor in sea turtle nesting distribution. *Aquatic Conservation: Marine and Freshwater Ecosystems, 20*, 140-149.

Sapolsky, R. M. (2006). Stress and the city. *Natural History, 115*(5), 72-72.

Sapolsky, R. M., Romero, L. M., & Munck, A. U. (2000). How do glucocorticoids influence stress responses? Integrating permissive, suppressive, stimulatory, and preparative actions. *Endocrine Reviews, 21*(1), 34.

Sarti, L. M., Eckert, S. A., Garcia, N. T., & Barragan, A. R. (1996). Decline of the world's largest nesting assemblage of leatherback turtles. *Marine Turtle Newsletter, 74*, 2-5.

Sato, K., Matsuzawa, Y., Tanaka, H., Bando, T., Minamikawa, S., Sakamoto, W., & Naito, Y. (1998). Internesting intervals for loggerhead turtles, Caretta caretta, and green turtles, Chelonia mydas, are affected by temperature. *Canadian Journal of Zoology, 76*, 1651-1662.

Scarff, J. E. (1986). Historic and present distribution of the right whale (*Eubalaena glacialis*) in the eastern North Pacific south of 50°N and east of 180°W. *Report of the International Whaling Commission*(Special Issue 10), 43-63.

Scarff, J. E. (1991). Historic distribution and abundance of the right whale (Eubalaena glacialis) in the North Pacific, Bering Sea, Sea of Okhotsk, and Sea of Japan from the Maury Whale Charts. *Reports of the International Whaling Commission, 41*, 467-489.

Scarpaci, C., Bigger, S. W., Corkeron, P. J., & Nugegoda, D. (2000). Bottlenose dolphins (*Tursiops truncatus*) increase whistling in the presence of 'swim-with-dolphin' tour operations. *Journal of Cetacean Research and Management, 2*(3), 183-185.

Scheidat, M., Castro, C., Gonzalez, J., & Williams., R. (2004). Behavioural responses of humpback whales (Megaptera novaeangliae) to whalewatching boats near Isla de la Plata, Machalilla National Park, Ecuador. *Journal of Cetacean Research and Management, 6*(1), 63-68.

Scheifele, P. M., Andrew, S., Cooper, R. A., Darre, M., Musiek, F. E., & Max, L. (2005a). Indication of a Lombard vocal response in the St. Lawrence River beluga. *Journal of the Acoustical Society of America, 117*(3), 1486-1492.

Scheifele, P. M., Andrew, S., Cooper, R. A., Darre, M., Musiek, F. E., & Max, L. (2005b). Indication of a Lombard vocal response in the St. Lawrence River beluga. *Journal of the Acoustical Society of America, 117*(3), 1486-1492.

Schick, R. S., & Urban, D. L. (2000). Spatial components of bowhead whale (Balaena mysticetus) distribution in the Alaskan Beaufort Sea. *Canadian Journal of Fisheries and Aquatic Sciences, 57*(11), 2193-2200.

Schlais, J. F. (1985). Bait snatching porpoises plague Hawaiians. *National Fisherman, 65*(9), 25-26.

Schlundt, C. E., Finneran, J. J., Carder, D. A., & Ridgway, S. H. (2000). Temporary shift in masked hearing thresholds of bottlenose dolphins, *Tursiops truncatus*, and white whales, *Delphinapterus leucas*, after exposure to intense tones. *Journal of the Acoustical Society of America, 107*(6), 3496-3508.

Schlundt, C. R., Finneran, J. J., Carder, D. A., & Ridgway, S. H. (2000). Temporary shift in masked hearing thresholds of bottlenose dolphins, Tursiops truncatus, and white whale, Delphinapterus leucas, after exposure to intense tones. *Journal of the Acoustical Society of America, 107*(6), 3496-3508.

Schoenherr, J. R. (1991). Blue whales feeding on high concentrations of euphausiids around Monterey Submarine Canyon. (Balaenoptera musculus). *Canadian Journal of Zoology, 69*(3), 583-594.

Schofield, G., Bishop, C. M., Katselidis, K. A., Dimopoulos, P., Pantis, J. D., & Hays, G. C. (2009). Microhabitat selection by sea turtles in a dynamic thermal marine environment. *Journal of Animal Ecology, 78*(1), 14-21. doi: 10.1111/j.1365-2656.2008.01454.x

Schultz, J. K., Baker, J. D., Toonen, R. J., Harting, A. L., & Bowen, B. W. (2011). Range-wide genetic connectivity of the Hawaiian monk seal and implications for translocation. *Conservation Biology, 25*(1), 124-132.

Schusterman, R. J. (1978). Vocal communication in pinnipeds. In H. Markowitz & V. J. Stevens (Eds.), *Behavior of Captive Wild Animals* (pp. 247-285, 306-308). Chicago: Nelson-Hall.

Schweder, T., & Host., G. (1992). Integrating experimental data and survey data to estimate g(0): A first approach. *Report of the International Whaling Commission, 42*, 575-582.-Sc/543/O574).

Schweder, T., Oien, N., & Host, G. (1992). Estimates of g(0) for northeastern Atlantic minke whales based on independent observer experiments in 1989 and 1990 found by the hazard probability method. *Report of the International Whaling Commission, 42*, 399-405.

Sea Turtle Association of Japan. (2010). Sea turtles of Japan. from www.umigame.org/E1/turtlepowerstyle2.html

Sears, C. J. (1994). Preliminary genetic analysis of the population structure of Georgia loggerhead sea turtles. *NOAA Technical Memorandum NMFS-SEFSC-351. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Miami, Florida*.

Sears, C. J., Bowen, B. W., Chapman, R. W., Galloway, S. B., Hopkins-Murphy, S. R., & Woodley, C. M. (1995). Demographic composition of the feeding population of juvenile loggerhead sea turtles (Caretta caretta) off Charleston, South Carolina: evidence from mitochondrial DNA markers. *Marine Biology, 123*, 869-874.

Sears, R. (1983). A glimpse of blue whales feeding in the Gulf of St. Lawrence. *Whalewatcher, 17*(3), 12-14.

Sears, R. (1983). A glimpse of blue whales feeding in the Gulf of St. Lawrence. *Whalewatcher, 17*(3), 12-14.

Sears, R. (1987). Study on blue whales - brief description.: Unpublished paper to the IWC Scientific Committee. 2 pp. Bournemouth, June (SC/39/PS20).

Sears, R., & Perrin, W. F. (2009). Blue whale, *Balaenoptera musculus*. In W. F. Perrin, B. Wursig & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals. Second edition* (pp. 120-124). San Diego: Academic Press.

Sedlak-Weinstein, E. (1991). New records of cyamids (Amphipods) from Australian cetaceans. *Crustaceana, 60*(1), 90-104.

Seminoff, J. A. (2002). *Global status of the green turtle (Chelonia mydas): A summary of the 2001 Status Assessment for the IUCN Red List Programme*. Paper presented at the Western Pacific Sea Turtle Cooperative Research and Management Workshop, Honolulu, Hawaii.

Seminoff, J. A. (2004). 2004 global status assessment: Green turtle (*Chelonia mydas*) (pp. 71): The World Conservation Union (International Union for Conservation of Nature and Natural Resources), Species Survival Commission Red List Programme, Marine Turtle Specialist Group.

Seminoff, J. A., Resendiz, A., & Nichols, W. J. (2002). Diet of east pacific green turtles (*Chelonia mydas*) in the central Gulf of California, Mexico. *Journal of Herpetology, 36*(3), 447-453.

Senko, J., Koch, V., Megill, W. M., Carthy, R. R., Templeton, R. P., & Nichols, W. J. (2010). Fine scale daily movements and habitat use of East Pacific green turtles at a shallow coastal lagoon in Baja California Sur, Mexico. *Journal of Experimental Marine Biology and Ecology, in press*(in press), in press.

Senko, J., Lopez-Castro, M. C., Koch, V., & Nichols, W. J. (2010). Immature East Pacific green turtles (Chelonia mydas) use multiple foraging areas off the Pacific Coast of Baja California Sur, Mexico: First evidence from mark-recapture data. *Pacific Science, 64*(1), 125-130.

Seyle, H. (1950). Stress and the general adaptation syndrome. *The British Medical Journal*, 1383-1392.

Shallenberger, E., Commission, M. M., States, U., & Corporation, M. (1981). The status of Hawaiian cetaceans; Final Report to the U.S. Marine Mammal Comission *Journal of the Marine Biological Association of the U.K.* (pp. 88): U.S. Department of Commerce, National Technical Information Service

Shallenberger, E. W. (1981). The status of Hawaiian cetaceans: Marine Mammal Commission.

Shane, S. H. (1994). Occurrence and habitat use of marine mammals at Santa Catalina Island, California from 1983-91. *Bulletin of the Southern California Academy of Sciences, 93*, 13-29.

Shane, S. H. (1995). Behavior patterns of pilot whales and Risso's dolphins off Santa Catalina Island, California. *Aquatic Mammals, 21*(3), 195-197.

Shane, S. H., Wells, R. S., Wursig, B., & Wursig, B. (1986). Ecology, behavior and social organization of the bottle-nosed dolphin - a review. *Marine Mammal Science, 2*(1), 34-63.

Shanker, K., & Mohanty, B. (1999). Operation Kachhapa: In search of a solution for the olive ridley of Orissa. *Marine Turtle Newsletter, 86*, 1-3.

Sherr, E. B., Sherr, B. F., & Wheeler, P. A. (2005). Distribution of coccoid cyanobacteria and small eukaryotic phytoplankton in the upwelling ecosystem off the Oregon coast during 2001 and 2002. *Deep-Sea Research II, 52*, 317-330.

Shillinger, G. L., Palacios, D. M., Bailey, H., Bograd, S. J., Swithenbank, A. M., Gaspar, P., . . . Block, B. A. (2008). Persistent leatherback turtle migrations present opportunities for conservation. *PLoS Biology, 6*(7), 1408-1416. doi: e171

10.1371/journal.pbio.0060171

Sigler, M. F., Tollit, D. J., Vollenweider, J. J., Thedinga, J. F., Csepp, D. J., Womble, J. N., . . . Trites, A. W. (2009). Steller sea lion foraging response to seasonal changes in prey availability. *Marine Ecology-Progress Series, 388*, 243-261. doi: 10.3354/meps08144

Sigurjonsson, J., Gunnlaugsson, T., & Payne., M. (1989). NASS-87: Shipboard sightings surveys in Icelandic and adjacent waters June-July 1987. *Report of the International Whaling Commission, 39*, 395-409.

Silber, G. K. (1986). The Relationship of Social Vocalizations to Surface Behavior and Aggression in the Hawaiian Humpback Whale (Megaptera novaeangliae). *Canadian Journal of Zoology, 64*(10), 2075-2080.

Simmonds, M. P., & Eliott., W. J. (2009). Climate change and cetaceans: Concerns and recent developments. *Journal of the Marine Biological Association of the United Kingdom, 89*(1), 203-210.

Simmonds, M. P., & Hutchinson, J. D. (1996). *The conservation of whales and dolphins.*: John Wiley and Sons, Chichester, U.K.

Simmonds, M. P., & Isaac, S. J. (2007). The impacts of climate change on marine mammals: Early signs of significant problems. *Oryx, 41*(1), 19-26.

Simmonds, M. P., & Lopez-Jurado, L. F. (1991). Whales and the military. (navel maneuvers off the Canary Islands may have caused whales to strand themselves there). *Nature, 351*(6326), 448.

Sivle, L. D., Kvadsheim, P. H., Ainslie, M. A., Solow, A., Handegard, N. O., Nordlund, N., & Lam, F. P. A. (2012). Impact of naval sonar signals on Atlantic herring (Clupea harengus) during summer feeding. *ICES Journal of Marine Science, 69*(6), 1078-1085. doi: 10.1093/icesjms/fss080

Slijper, E. J. (1962). *Whales.*: English translation Hutchinson & Co. (Publishers). First published in the U.S. by Basic Books Publishing Co., Inc, New York. 475pp.

Smith, A. W., & Latham, A. B. (1978). Prevalence of vesicular exanthema of swine antibodies among feral mammals associated with the southern California coastal zones. *American Journal of Veterinary Research, 39*(2), 291-296.

Smith, M. E., Kane, A. S., & Popper, A. N. (2004). Acoustical stress and hearing sensitivity in fishes: does the linear threshold shift hypothesis hold water? *The Journal of Experimental Biology, 207*(20), 3591 - 3602.

Smith, M. E., Kane, A. S., & Popper, A. N. (2004). Noise-induced stress response and hearing loss in goldfish (*Carassius auratus*). *Journal of Experimental Biology, 207*(3), 427-435. doi: 10.1242/jeb.00755

Smultea, M., & Holst, M. (2003). Marine mammal monitoring during Lamont-Doherty Earth Observatory's seismic study in the Hess Deep area of the eastern equatorial tropical Pacific, July 2003. *Prepared for Lamont-Doherty Earth Observatory, Palisades, New York, and the National Marine Fisheries Service, Silver Spring, Maryland, by LGL Ltd., environmental research associates. LGL Report TA2822-16*.

Smultea, M. A., Jefferson, T. A., & Zoidis, A. M. (2010). Rare sightings of a bryde's whale (*Balaenoptera edeni*) and sei whales (*B. borealis*) (Cetacea: Balaenopteridae) northeast of O'ahu, Hawai'i. *Pacific Science, 64*(3), 449-457.

Sousa-Lima, R. S., & Clark, C. W. (2008). Modeling the effect of boat traffic on the fluctuation of humpback whale singing activity in the Abrolhos National Marine Park, Brazil. *Canadian Acoustics, 36*(1), 174-181.

Southall, B., Calambokidis, J., Tyack, P., Moretti, D., Hildebrand, J., Kyburg, C., . . . Barlow, J. (2011). Biological and behavioral response studies of marine mammals in southern California, 2010 ("SOCAL -10").

Southall, B. L., Bowles, A. E., Ellison, W. T., Finneran, J. J., Gentry, R. L., Greene, C. R., . . . Tyack, P. L. (2007a). Special Issue: Marine Mammal Noise Exposure Criteria Special Issue. *Aquatic Mammals, 33*(4), Iv + 411-521.

Southall, B. L., Bowles, A. E., Ellison, W. T., Finneran, J. J., Gentry, R. L., Greene, C. R., . . . Tyack, P. L. (2007b). Special Issue: Marine Mammal Noise Exposure Criteria Special Issue. *Aquatic Mammals, 33*(4), Iv + 411-521.

Southall, B. L., Bowles, A. E., Ellison, W. T., Finneran, J. J., Gentry, R. L., Greene Jr., C. R., . . . Tyack, P. L. (2007). Marine mammal noise exposure criteria: Initial scientific recommendations. *Aquatic Mammals, 33*(4), 411-521.

Southall, B. L., Moretti, D., Abraham, B., Calambokidis, J., DeRuiter, S. L., & Tyack, P. L. (2012). Marine Mammal Behavioral Response Studies in Southern California: Advances in Technology and Experimental Methods. *Marine Technology Society Journal, 46*(4), 48-59.

Southall, B. L., Schusterman, R. J., & Kastak, D. (2000). Masking in three pinnipeds: Underwater, low-frequency critical ratios. *Journal of the Acoustical Society of America, 108*(3), 1322-1326.

Southall, B. L., Schusterman, R. J., & Kastak, D. (2003). Auditory masking in three pinnipeds: Aerial critical ratios and direct critical bandwidth measurements. *Journal of the Acoustical Society of America, 114*(3), 1660-1666.

Southwood, A. L., Andrews, R. D., Lutcavage, M. E., Paladino, F. V., West, N. H., George, R. H., & Jones, D. R. (1999). Heart rates and diving behavior of leatherback sea turtles in the Eastern Pacific Ocean. *Journal of Experimental Biology, 202*(9), 1115-1125.

Spero, D. (1981). *Vocalizations and associated behavior of northern right whales Eubalaena glacialis*. Paper presented at the Fourth Biennial Conference on the Biology of Marine Mammals, December 1981, San Francisco, California, USA.

Spotila, J. R. (2004). *Sea turtles: a complete guide to their biology, behavior, and conservation*. Baltimore, Maryland: The Johns Hopkins University Press and Oakwood Arts.

Spotila, J. R. (2004). Sea turtles: A complete guide to their biology, behavior, and conservation. *John Hopkins University Press, Baltimore. 227p.*

Spotila, J. R. (2004). Sea turtles: A complete guide to their biology, behavior, and conservation. *The Johns Hopkins University Press and Oakwood Arts, Baltimore, Maryland*.

Spotila, J. R., Dunham, A. E., Leslie, A. J., Steyermark, A. C., Plotkin, P. T., & Paladino, F. V. (1996). Worldwide population decline of *Dermochelys coriacea*: Are leatherback turtles going extinct? *Chelonian Conservation and Biology, 2*(2), 209-222.

Spotila, J. R., Reina, R. D., Steyermark, A. C., Plotkin, P. T., & Paladino, F. V. (2000). Pacific leatherback turtles face extinction. *Nature, 405*, 529-530.

SPREP. (2007). Pacific Islands regional marine species programme 2008–2012. Apia, Samoa: Pacific Islands Regional Marine Species Programme.

Spring, C. S. (1982). Status of marine turtle populations in Papua New Guinea. In K. Bjorndal (Ed.), *Biology and Conservation of Sea Turtles* (pp. 281-289). Washington, D. C.: Smithsonian Institution Press.

Stabeno, P. J., Bond, N. A., Hermann, A. J., Kachel, N. B., Mordy, C. W., & Overland., J. E. (2004). Meteorology and oceanography of the northern Gulf of Alaska. *Continental Shelf Research, 24*, 859-897.

Stabeno, P. J., Bond, N. A., Hermann, A. J., Kachel, N. B., Mordy, C. W., & Overland., J. E. (2004). Meteorology and oceanography of the northern Gulf of Alaska. *Continental Shelf Research, 24-Jan*(8-Jul), 859-897.

Stacey, P. J., Leatherwood, S., & Baird, R. W. (1994). *Pseudorca crassidens*: Mammalian Species No. 456. 6P.

Stafford, K. M. (2003). Two types of blue whale calls recorded in the Gulf of Alaska. *Marine Mammal Science, 19*(4), 682-693.

Stafford, K. M., Nieukirk, S. L., & Fox, C. G. (2001). Geographic and seasonal variation of blue whale calls in the North Pacific. *Journal of Cetacean Research And Management, 3*(1), 65-76.

STAJ. (2002). *Population trends and mortality of Japanese loggerhead turtles, Caretta caretta in Japan*. Paper presented at the Western Pacific Sea Turtle Cooperative Research and Management Workshop, Honolulu, Hawaii.

Standora, E. A., Spotila, J. R., Keinath, J. A., & Shoop, C. R. (1984). Body temperatures, diving cycles, and movement of a subadult leatherback turtle, *Dermochelys coriacea*. *Herpetologica, 40*, 169-176.

Stark, J. D., Banks, J. E., & Vargas, R. (2004). *How risky is risk assessment: The role that life history strategies play in susceptibility of species to stress*.

Stearns, S. C. (1992). *The evolution of life histories*. New York, New York: Oxford University Press.

Stearns, S. C. (1992). The Evolution of Life Histories. *Oxford Press, Oxford. 249*.

Steiger, G. H., Calambokidis, J., Straley, J. M., Herman, L. M., Cerchio, S., Salden, D. R., . . . Barlow, J. (2008). Geographic variation in killer whale attacks on humpback whales in the North Pacific: Implications for predation pressure. *Endangered Species Research, 4*(3), 247-256.

Stensland, E., & Berggren, P. (2007). Behavioural changes in female Indo-Pacific bottlenose dolphins in response to boat-based tourism. *Marine Ecology Progress Series, 332*, 225-234.

Sternberg, J. (1981). The worldwide distribution of sea turtle nesting beaches. Washignton, D. C.: Sea Turtle Rescue Fund, Center for Environmental Education.

Stinson, M. L. (1984). *Biology of sea turtles in San Diego Bay, California, and in northeastern Pacific Ocean.* (M. S.), San Diego State University, San Diego, California.

Stockin, K. A., Lusseau, D., Binedell, V., Wiseman, N., & Orams, M. B. (2008). Tourism affects the behavioural budget of the common dolphin *Delphinus* sp. in the Hauraki Gulf, New Zealand. *Marine Ecology Progress Series, 355*, 287-295.

Stokes, L. W., & Epperly, S. P. (2006). *Lepidochelys olivacea* (olive ridley sea turtle). Western North Atlantic Ocean. *Herpetological Review, 37*(1), 105.

Stone, C. J. (1997). Cetacean observations during seismic survey in 1996: JNCC.

Stone, C. J. (1998). Cetacean observations during seismic surveys in 1997 (pp. 86). Peterborough: Joint Nature Conservation Committee.

Stone, C. J. (2000). Cetacean observations during seismic surveys in 1998 (pp. 62). Peterborough: Joint Nature Conservation Committee.

Stone, C. J. (2001). Cetacean observations during seismic surveys in 1999 (pp. 92). Peterborough: Joint Nature Conservation Committee.

Stone, C. J. (2003). The effects of seismic activity on marine mammals in UK waters, 1998-2000 (pp. 78): Joint Nature Conservation Committee.

Stone, G. S., Katona, S. K., Mainwaring, A., Allen, J. M., & Corbett., H. D. (1992). Respiration and surfacing rates of fin whales (Balaenoptera physalus) observed from a lighthouse tower. *Report of the International Whaling Commission, 42*, 739-745.

Storelli, M., Barone, M. G., & Marcotrigiano, G. O. (2007). Polychlorinated biphenyls and other chlorinated organic contaminants in the tissues of Mediterranean loggerhead turtle Caretta caretta. *Science of the Total Environment, 273*(2-3), 456-463.

Storelli, M., Barone, M. G., Storelli, A., & Marcotrigiano, G. O. (2008). Total and subcellular distribution of trace elements (Cd, Cu and Zn) in the liver and kidney of green turtles (Chelonia mydas) from the Mediterranean Sea. *Chemosphere, 70*(5), 908-913.

Storelli, M. M., Barone, G., & Marcotrigiano, G. O. (2007). Polychlorinated biphenyls and other chlorinated organic contaminants in the tissues of Mediterranean loggerhead turtle *Caretta caretta*. *Science of the Total Environment, 373*(2-3), 456-463.

Storelli, M. M., Barone, G., Storelli, A., & Marcotrigiano, G. O. (2008). Total and subcellular distribution of trace elements (Cd, Cu and Zn) in the liver and kidney of green turtles (*Chelonia mydas*) from the Mediterranean Sea. *Chemosphere, 70*(5), 908-913.

Streever, B., Angliss, R. P., Suydam, R., Ahmaogak, M., Bailey, C., Blackwell, S. B., . . . Richardson., W. J. (2008). Progress through collaboration: A case study examining effects of industrial sounds on bowhead whales. *Bioacoustics, 17-Jan*(3-Jan), 345-347. Special Issue on the International Conference on the Effects of Noise on Aquatic Life. Edited By A. Hawkins, A. N. Popper & M. Wahlberg.

Sutherland, W. J., & Crockford, N. J. (1993). Factors affecting the feeding distribution of red-brested geese *Branta ruficollos* wintering in Romania. *Biological Conservation, 63*(1), 61-65.

Talavera-Saenz, A., Gardner, S. C., Rodriquez, R. R., & Vargas, B. A. (2007). Metal profiles used as environmental markers of green turtle (Chelonia mydas) foraging resources. *Science of the Total Environment, 373*(1), 94-102.

Tarpy, C. (1979). Killer whale attack! (*Orcinus orca*). *National Geographic Magazine, 155*(4), 542-545.

Taylor, A. H., & Stephens, J. A. (1998). The North Atlantic oscillation and the latitude of the Gulf Stream. *Tellus A, 50*(1), 134-142.

TEC. (2009). Cetacean depredation in the Hawaii longline fishery: Interviews of longline vessel owners and captains. Honolulu, Hawaii: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Regional Office.

Teloni, V., Zimmer, W. M. X., Wahlberg, M., & Madsen., P. T. (2007). Consistent acoustic size estimation of sperm whales using clicks recorded from unknown aspects. *Journal of Cetacean Research and Management, 9*(2), 127-136.

Terhune, J. M., & Ronald, K. (1972). The harp seal, Pagophilus groenlandicus (Erxleben, 1777). III. The underwater audiogram. *Canadian Journal of Zoology, 50*(5), 565-569.

Terhune, J. M., & Ronald, K. (1976). The upper frequency limit of ringed seal hearing. *Canadian Journal of Zoology, 54*, 1226-1229.

Terhune, J. M., & Ronald., K. (1985). *Distant and near range functions of harp seal underwater calls.* Paper presented at the Sixth Biennial Conference on the Biology of Marine Mammals, Vancouver, B.C., Canada.

Tershy, B. R., Urbán-R, J., Breese, D., Rojas-B, L., & Findley, L. T. (1993). Are fin whales resident to the Gulf of California. *Rev. Invest. Cient., Univ. Auton. de Baja California Sur, 1*, 69-71.

TEWG. (1998a). *An assessment of the Kemp's ridley (Lepidochelys kempii) and loggerhead (Caretta caretta) sea turtle populations in the Western North Atlantic*. Paper presented at the Proceedings of the Sixteenth Annual Symposium on Sea Turtle Biology and Conservation.

TEWG. (1998b). An assessment of the Kemp's ridley (*Lepidochelys kempii*) and loggerhead (*Caretta caretta*) sea turtle populations in the western North Atlantic. *A report of the Turtle Expert Working Group (TEWG); NOAA Technical Memorandum NMFS-SEFSC-409. 96p.*

Thewissen, J. G. M., & Williams, E. M. (2002). The early radiations of Cetacea (Mammalia): Evolutionary pattern and developmental correlations. *Annual Review of Ecology and Systematics, 33*, 73-90.

Thode, A., Straley, J., Tiemann, C. O., Folkert, K., & O'Connell, V. (2007). Observations of potential acoustic cues that attract sperm whales to longline fishing in the Gulf of Alaska. *Journal of the Acoustical Society of America, 122*(2), 1265-1277. doi: 10.1121/1.2749450

Thomas, A., & Strub, P. T. (2001). Cross-shelf phytoplankton pigment variability in the California Current. *Continental Shelf Research, 21*(11-12), 1157-1190.

Thomas, J. A., Fisher, S. R., & Ferm., L. M. (1986). Acoustic detection of cetaceans using a towed array of hydrophones. *Report of the International Whaling Commission Special Issue, 8*, 139-148.-Sc/137/O133).

Thomas, J. A., Moore, P., Withrow, R., & Stoermer, M. (1990a). Underwater audiogram of a Hawaiian monk seal (Monachus schauinslandi). *Journal of the Acoustical Society of America, 87*(1), 417-420.

Thomas, J. A., Moore, P., Withrow, R., & Stoermer, M. (1990b). Underwater audiogram of a Hawaiian monk seal (Monachus schauinslandi). *Journal of the Acoustical Society of America, 87*(1), 417-420.

Thomas, J. A., & Turl, C. W. (1990). Echolocation characteristics and range detection threshold of a false killer whale (Pseudorca crassidens). *Sensory abilities of cetaceans*, 321-333.

Thompson, P. O., Cummings, W. C., & Ha., S. J. (1986). Sounds, source levels, and associated behavior of humpback whales, Southeast Alaska. *Journal of the Acoustical Society of America, 80*(3), 735-740.

Thompson, P. O., Findley, L. T., & Vidal., O. (1992). 20-Hz pulses and other vocalizations of fin whales, Balaenoptera physalus, in the Gulf of California, Mexico. *Journal of the Acoustical Society of America, 92*(6), 3051-3057.

Thompson, P. O., & Friedl, W. A. (1982). A long term study of low frequency sounds from several species of whales off Oahu, Hawaii. *Cetology, 45*, 1-19.

Thompson, T. J., Winn, H. E., & Perkins., P. J. (1979). Mysticete sounds. In H. E. Winn & B. L. Olla (Eds.), *Behavior of Marine Animals: Current Perspectives in Research Vol. 3: Cetaceans.* (pp. 403-431). New York, NY: Plenum Press.

Tillman, M. F. (1977). Estimates of population size for the North Pacific sei whale. (Balaenoptera borealis). *Report of the International Whaling Commission Special Issue, 1*, 98-106.-Sc/127/Doc 125).

Tomás, J., Gozalbes, P., Raga, J. A., & Godley, B. J. (2008). Bycatch of loggerhead sea turtles: Insights from 14 years of stranding data. *Endangered Species Research, 5*, 161-169.

Tomilin, A. G. (1967). Mammals of the USSR and adjacent countries, Vol. 9, Cetacea.: Akademiya Nauk SSR, Moscow. (Translated from Russian by Israel Program for Scientific Translantions. 717 pgs.).

Tonnessen, J. N., & Johnsen, A. O. (1982). *The history of modern whaling* (T. F. T. N. B. R. I. Christophersen, Trans.). Berkeley, CA: University of California Press.

Townsend, C. H. (1935). The distribution of certain whales as shown by logbook records of American whaleships. *Zoologica 19:1–50*.

Trimper, P. G., Standen, N. M., Lye, L. M., Lemon, D., Chubbs, T. E., & Humphries, G. W. (1998). Effects of low-level jet aircraft noise on the behaviour of nesting osprey. *Journal of Applied Ecology, 35*(1), 122-130. doi: 10.1046/j.1365-2664.1998.00290.x

Trinidad, H., & Wilson, J. (2000). *The bio-economics of sea turtle conservation and use in Mexico: History of exploitation and conservation policies for the olive ridley (Lepidochelys olivacea)*. Paper presented at the Proceedings of the International Institute of Fisheries Economics, and Trade Conference. , Corvallis, Oregon.

Trites, A. W., & Bain, D. E. (2000). Short- and long-term effects of whale watching on killer whales (*Orcinus orca*) in British Columbia (pp. 10): University of British Columbia, Fisheries Center, Marine Mammal Research Unit.

Tucker, A. D. (2009). Nest site fidelity and clutch frequency of loggerhead turtles are better elucidated by satellite telemetry than by nocturnal tagging efforts: Implications for stock estimation. *Journal of Experimental Marine Biology and Ecology, in press*(in press), in press.

Turkozan, O., & Yilmaz, C. (2008). Loggerhead turtles, Caretta caretta, at Dalyan Beach, Turkey: Nesting activity (2004–2005) and 19-year abundance trend (1987–2005). *Chelonian Conservation and Biology, 7*(2), 178-187.

Turnpenny, A. W. H., Thatcher, K. P., & Nedwell, J. R. (1994). The effects on fish and other marine animals of high-level underwater sound. Southampton SO45 ITW. Hampshire, United Kingdom: Fawley Aquatic Research Laboratories, Ltd.

Twiss, S. D., Thomas, C., Poland, V., Graves, J. A., & Pomeroy., P. (2007). The impact of climatic variation on the opportunity for sexual selection. *Biology Letters, 3*(1), 12-15.

Tyack, P. (1981). Interactions between singing Hawaiian humpback whales and conspecifics nearby. (Megaptera novaeangliae). *Behavioral Ecology and Sociobiology, 8*(2), 105-116.

Tyack, P. (1983). Differential response of humpback whales, Megaptera novaeangliae, to playback of song or social sounds. *Behavioral Ecology and Sociobiology, 13*(1), 49-55.

Tyack, P. L. (1999). Communication and cognition. In J. E. R. III & S. A. Rommel (Eds.), *Biology of Marine Mammals* (pp. 287-323). Washington: Smithsonian Institution Press.

Tyack, P. L. (2008). Implications for marine mammals of large-scale changes in the marine acoustic environment. *Journal of Mammalogy, 89*(3), 549-558.

Tyack, P. L., & Clark, C. W. (2000). Communication and acoustic behavior of dolphins and whales *Hearing by Whales and Dolphins. p.156-224. W. W. L. Au, A. N. Popper, R. R. Fay (eds.). Springer-Verlag, New York Inc.*

Tynan, C. T., Ainley, D. G., Barth, J. A., Cowles, T. J., Pierce, S. D., & Spear., L. B. (2005). Cetacean distributions relative to ocean processes in the northern California Current System. *Deep Sea Research Part Ii*, Topical studies in Oceanography 52(51-52)145-167.

Uchida, I., & Nishiwaki, M. (1995). Sea turtles in the waters adjacent to Japan. In K. A. Bjorndal (Ed.), *Biology and Conservation of Sea Turtles* (Vol. Revised Edition, pp. 317-319). Washington, D. C.: Smithsonian Institution Press.

Urick, R. J. (1983). *Principles of Underwater Sound*: McGraw-Hill.

USAF. (1996). Sea turtles in the Gulf: Air Force Material Command, Eglin Air Force Base.

USFWS, N. a. (1998). Recovery Plan for U.S. Pacific Populations of the Loggerhead Turtle (*Caretta caretta*). Silver Spring, Maryland: National Marine Fisheries Service.

Van de Merwe, J. P. V., Hodge, M., Olszowy, H. A., Whittier, J. M., Ibrahim, K., & Lee, S. Y. (2009). Chemical contamination of green turtle (*Chelonia mydas*) eggs in peninsular Malaysia: Implications for conservation and public health. *Environmental Health Perspectives, 117*(9), 1397-1401. doi: 10.1289/ehp.0900813

Van Der Hoop, J. M., Moore, M. J., Barco, S. G., Cole, T. V., DAOUST, P. Y., Henry, A. G., . . . Solow, A. R. (2013). Assessment of management to mitigate anthropogenic effects on large whales. *Conservation Biology, 27*(1), 121-133.

Van Parijs, S. M., & Kovacs, K. M. (2002). In-air and underwater vocalizations of eastern Canadian harbour seals, Phoca vitulina. *Canadian Journal of Zoology, 80*(7), 1173-1179.

vanDam, R. P., & Diez, C. E. (1997). Diving behavior of immature hawksbill turtles (Eretmochelys imbricata) in a Caribbean reef habitat. *Coral Reefs, 16*(2), 133-138.

Vanderlaan, A. S. M., & Taggart, C. T. (2007). Vessel collisions with whales: The probability of lethal injury based on vessel speed. *Marine Mammal Science, 23*(1), 144-156. doi: 10.1111/j.1748-7692.2006.00098.x

Vanderlaan, A. S. M., Taggart, C. T., Serdynska, A. R., Kenney, R. D., & Brown, M. W. (2008). Reducing the risk of lethal encounters: Vessels and right whales in the Bay of Fundy and on the Scotian Shelf. *Endangered Species Research, 4*(3), 283-283.

Viada, S. T., Hammer, R. A., Racca, R., Hannay, D., Thompson, M. J., Balcom, B. J., & Phillips, N. W. (2008a). Review of potential impacts to sea turtles from underwater explosive removal of offshore structures. *Environmental impact assessment review, 128*(4-5), 267-285.

Viada, S. T., Hammer, R. M., Racca, R., Hannay, D., Thompson, M. J., Balcom, B. J., & Phillips, N. W. (2008b). Review of potential impacts to sea turtles from underwater explosive removal of offshore structures. *Environmental impact assessment review, 28*(4–5), 267-285. doi: 10.1016/j.eiar.2007.05.010

Villegas-Amtmann, S., & Costa., D. P. (2010). Oxygen stores plasticity linked to foraging behaviour and pregnancy in a diving predator, the Galapagos sea lion. *Functional Ecology, 24*(4), 785-795.

Visser, I. N., Zaeschmar, J., Halliday, J., Abraham, A., Ball, P., Bradley, R., . . . Pace., D. (2010). First record of predation on false killer whales (Pseudorca crassidens) by killer whales (Orcinus orca). *Aquatic Mammals, 36*(2), 195-204.

Wada, S. (1973). The ninth memorandum on the stock assessment of whales in the North Pacific. *Report of the International Whaling Commission, 23*, 164-169.

Wade, P. M., Heide-Jorgensen, P., Shelden, K., Barlow, J., Carretta, J., Durban, J., . . . Stinchcomb, C. (2006). Acoustic detection and satellite tracking leads to discovery of rare concentration of endangered North Pacific right whales. *Biology Letters, 2*, 417-419.

Wade, P. R., & Gerrodette, T. (1993). Estimates of cetacean abundance and distribution in the Eastern Tropical Pacific. *Report of the International Whaling Commission, 43*(477-493).

Wade, P. R., Kennedy, A., LeDuc, R., Barlow, J., Carretta, J., Shelden, K., . . . Clapham, P. J. (2010). The world's smallest whale population? *Biology Letters*. doi: 10.1098/rsbl.2010.0477

Wade, P. R., Kennedy, A., Leduc, R., Barlow, J., Carretta, J., Shelden, K., . . . Clapham, P. J. (2011). The world's smallest whale population? (Eubalaena japonica). *Biology Letters, 7*(1), 83-85.

Waite, J. M., Wynne, K., & Mellinger, D. K. (2003). Documented sighting of a North Pacific right whale in the Gulf of Alaska and post-sighting acoustic monitoring. *Northwestern Naturalist, 84*, 38-43.

Waite, J. M., Wynne, K., & Mellinger., D. K. (2003). Documented sighting of a North Pacific right whale in the Gulf of Alaska and post-sighting acoustic monitoring. *Northwestern Naturalist, 84*(1), 38-43.

Walker, B. G., Dee Boersma, P., & Wingfield, J. C. (2005). Physiological and behavioral differences in magellanic Penguin chicks in undisturbed and tourist-visited locations of a colony. *Conservation Biology, 19*(5), 1571-1577.

Wallace, B. P., Avens, L., Braun-McNeill, J., & McClellan, C. M. (2009). The diet composition of immature loggerheads: Insights on trophic niche, growth rates, and fisheries interactions. *Journal of Experimental Marine Biology and Ecology, 373*(1), 50-57. doi: 10.1016/j.jembe.2009.03.006

Wallace, B. P., Lewison, R. L., McDonald, S. L., McDonald, R. K., Kot, C. Y., Kelez, S., . . . Crowder, L. B. (2010). Global patterns of marine turtle bycatch. *Convervation Letters, in press*(in press), in press.

Wang, P. (1984). Distribution of the gray whale (Eschrichtius gibbosus) off the coast of China. *Acta Theriologica Sinica, 4*(1), 21-26.

Waring, G. T. (1993). *Spatial patterns of six cetaceans along a linear habitat.* Paper presented at the Tenth Biennial Conference on the Biology of Marine Mammals, 11-15 November Galveston TX. p.2. Symposium: Cetacean Habitats.

Waring, G. T., Josephson, E., Fairfield, C. P., & (Eds)., K. M.-F. (2009). U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments -- 2008.: NOAA Technical Memorandum NMFS-NE-210. 440pp.

Waring, G. T., Josephson, E., Fairfield, C. P., & Maze-Foley, K. (2007). U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments 2006: U.S. Department of Commerce, NOAA, NMFS.

Waring, G. T., Josephson, E., Maze-Foley, K., & P.E.Rosel. (2011). U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments -- 2010 NOAA Tech Memo NMFS NE 219 (pp. 598).

Waring, G. T., Pace, R. M., Quintal, J. M., Fairfield, C. P., & Maze-Foley, K. (2004). U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments - 2003. Woods Hole, Massachusetts.

Wartzok, D., Watkins, W. A., Wursig, B., & Malme., C. I. (1989). *Movements and behaviors of bowhead whales in response to repeated exposures to noises associated with industrial activities in the Beaufort Sea. (Balaena mysticetus)*: Whale Research Report, AMOCO Production Co., Anchorage, AK. 228p.

Watkins, W. A. (1981a). Activities and underwater sounds of fin whales. (Balaenoptera physalus). *Scientific Reports of the Whales Research Institute Tokyo, 33*, 83-118.

Watkins, W. A. (1981b). Radio tagging of finback whales - Iceland, June-July 1980 *Technical Report WHOI-82-1* (pp. 46): Woods Hole Oceanagraphic Institution.

Watkins, W. A. (1981c). Reaction of three species of whales, *Balaenoptera physalus, Megaptera novaeangliae*, and *Balaenoptera edeni* to implanted radio tags. *Deep Sea Research Part I: Oceanographic Research Papers, 28*(6), 589-599.

Watkins, W. A. (1985). Changes observed in the reaction of whales to human activities. (pp. 31): National Marine Fisheries Service.

Watkins, W. A. (1986). Whale Reactions to Human Activities in Cape-Cod Waters. *Marine Mammal Science, 2*(4), 251-262.

Watkins, W. A., Daher, M. A., Fristrup, K. M., Howald, T. J., & Disciara, G. N. (1993). Sperm whales tagged with transponders and tracked underwater by sonar. *Marine Mammal Science, 9*(1), 55-67.

Watkins, W. A., Moore, K. E., Sigujónsson, J., Wartzok, D., & di Sciara, G. N. (1984). Fin Whale (*Balaenoptera physalus*) tracked by radio in the Irminger Sea. *Rit Fiskideildar, 8*, 1-14.

Watkins, W. A., Moore, K. E., Wartzok, D., & Johnson, J. H. (1981). Radio tracking of finback (Balaenoptera physalus) and humpback (Megaptera novaeangliae) whales in Prince William Sound, Alaska. *Deep-Sea Research, 28A*(6), 577-588.

Watkins, W. A., Morre, K. E., & Tyack, P. (1985). Sperm whale acoustic behaviors in the southeast Caribbean. *Cetology, 49*, 1-15.

Watkins, W. A., & Schevill, W. E. (1972). Sound source location by arrival-times on a non-rigid three-dimensional hydrophone array. *Deep-Sea Research, 19*, 691-706.

Watkins, W. A., & Schevill, W. E. (1975). Sperm whales (*Physeter catodon*) react to pingers. *Deep Sea Research and Oceanogaphic Abstracts, 22*(3), 123-129 +121pl.

Watkins, W. A., Tyack, P., Moore, K. E., & Bird, J. E. (1987). The 20-Hz signals of finback whales (*Balaenoptera physalus*). *Journal of the Acoustical Society of America, 82*(6), 1901-1912.

Watson, C. S. (2005). Some comments on informational masking. *Acta Acustica united with Acustica, 91*(3), 502-512.

Watwood, S. L., Miller, P. J. O., Johnson, M., Madsen, P. T., & Tyack, P. L. (2006). Deep-diving foraging behaviour of sperm whales (*Physeter macrocephalus*). *Journal of Animal Ecology, 75*, 814-825.

Waycott, M., Longstaff, B. J., & Mellors, J. (2005). Seagrass population dynamics and water quality in the Great Barrier Reef region: A review and future research directions. *Marine Pollution Bulletin, 51*(1-4), 343-350.

Wearmouth, V. J., & Sims, D. W. (2008). Sexual segregation in marine fish, reptiles, birds and mammals: Behaviour patterns, mechanisms and conservation implications. *Advances in Marine Biology, 54*, 107-170.

Weilgart, L. S., & Whitehead, H. (1993). Coda communication by sperm whales (*Physeter macrocephalus*) off the Galapagos Islands. *Canadian Journal of Zoology, 71*(4), 744-752.

Weilgart, L. S., & Whitehead, H. (1997). Group-specific dialects and geographical variation in coda repertoire in South Pacific sperm whales. (*Physeter macrocephalus*). *Behavioral Ecology and Sociobiology, 40*(5), 277-285.

Weilgart, L. S., Whitehead, H., Carler, S., & Clark., C. W. (1993). *Variations in the vocal repertoires of sperm whales (Physeter macrocephalus) with geographic area and year.* Paper presented at the Tenth Biennial Conference on the Biology of Marine Mammals, 11-15 November Galveston TX. p.112.

Weinrich, M. T., Lambertson, R. H., Belt, C. R., Schilling, M. R., Iken, H. J., & Syrjala, S. E. (1992). Behavioral reactions of humpback whales *Megaptera novaeangliae* to biopsy procedures. *Fishery Bulletin, 90*(3), 588-598.

Weir, C. R., Frantzis, A., Alexiadou, P., & Goold, J. C. (2007). The burst-pulse nature of 'squeal' sounds emitted by sperm whales (Physeter macrocephalus). *Journal of the Marine Biological Association of the U.K., 87*(1), 39-46.

Weir, C. W. (2007). Observations of marine turtles in relation to seismic airgun sound off Angola. *Marine Turtle Newsletter, 116*, 17-20.

Welch, B. L., & Welch, A. S. (1970). *Physiological Effects of Noise*. New York: Plenum Press.

Weller, D., Burdin, A., Wursig, B., Taylor, B., & Robert L. Brownell, J. (2002). The western gray whale: a status review of past exploitation, current status and potential threats. *Journal of Cetacean Res Manage, 4*(1), 7-12.

Weller, D. W., Bradford, A. L., Burdin, A. M., Miyashita, T., Kariya, T., Trukhin, A. M., . . . Doroshenko., N. V. (2002). Photographic recaptures of western gray whales in the Okhotsk Sea.: Unpublished paper to the IWC Scientific Committee. Shimonoseki, Japan, April (SC/54/BRG13).

Weller, D. W., Wursig, B., Bradford, A. L., Burdin, A. M., Blokhin, S. A., Minakuchi, H., & Brownell, R. L. (1999). Gray whales (*Eschrichtius robustus*) off Sakhalin Island, Russia: Seasonal and annual patterns of occurrence. *Marine Mammal Science, 15*(4), 1208-1227.

Weller, D. W., Wursig, B., Whitehead, H., Norris, J. C., Lynn, S. K., Davis, R. W., . . . Brown, P. (1996). Observations of an interaction between sperm whales and short-finned pilot whales in the Gulf of Mexico. *Marine Mammal Science, 12*(4), 588-593.

Wells, J. V., & Richmond, M. E. (1995). Populations, metapopulations, and species populations: What are they and who should care? *Wildlife Society Bulletin, 23*(3), 458-462.

Wenz, G. M. (1962). Acoustic ambient noise in the ocean: spectra and sources. *The Journal of the Acoustical Society of America, 34*, 1936.

Wenzel, F., Mattila, D. K., & Clapham, P. J. (1988). Balaenoptera musculus in the Gulf of Maine. *Marine Mammal Science, 4*(2), 172-175.

Western Australian Department of Industry and Resources. (2002). Petroleum Information Series - Guidelines Sheet 1. Guidelines on minimising acoustic disturbance to marine fauna. *Available online at: http://www.doir.wa.gov.au/documents/mineralsandpetroleum/envsheetmar02.pdf Accessed on: 10/19/2006.*

Wever, E. G., & Vernon, J. A. (1956). The sensitivity of the turtle's ear as shown by its electrical potentials. *Proceedings of the National Academies of Science, 42*.

Wheeler, P. A., & Hill, J. (1999). *Biological effects of the 1997-1998 El Niño event off Oregon: Nutrient and chlorophyll distributions*. Paper presented at the Proceedings of the 1998 Science Board Symposium on the impacts of the 1997/98 El Niño event on the North Pacific Ocean and its marginal seas.

Whitehead, H. (1997). Sea surface temperature and the abundance of sperm whale calves off the Galapagos Islands: Implications for the effects of global warming. *Report of the International Whaling Commission, 47*, 941-944.-Sc/948/O930).

Whitehead, H. (2002). Estimates of the current global population size and historical trajectory for sperm whales. *Marine Ecology Progress Series, 242*, 295-304.

Whitehead, H. (2003). Society and culture in the deep and open ocean: The sperm whale and other cetaceans. In F. B. M. d. Waal & P. L. Tyack (Eds.), *Animal Social Complexity: Intelligence, Culture, and Individualized Societies* (pp. 616): Harvard University Press.

Whitehead, H. (2008). Social and cultural evolution in the ocean: Convergences and contrasts with terrestrial systems. *The Deep Structure of Biology: Is Convergence Sufficiently Ubiquitous to Give a Directional Signal? p.143-160. Simon conway Morris (ed.). Templeton Foundation Press, West Conshohocken, Pennsylvania. ISBN 978-1-59947-138-9. 256pp.*

Whitehead, H., Christal, J., & Dufault., S. (1997). Past and distant whaling and the rapid decline of sperm whales off the Galapagos Islands. (Physeter macrocephalus). *Conservation Biology, 11*(6), 1387-1396.

Whitehead, H., & Mesnick, S. L. (2003). Social structure and effects of differential removals by sex in sperm whales: Methodology: Unpublished paper to the IWC Scientific Committee. 12 pp. Berlin, May (SC/55/O12).

Whitney, F. A., Mackas, D. L., Welch, D. W., & Robert, M. (1999). *Impact of the 1990s El Niños on nutrient supply and productivity of Gulf of Alaska waters*. Paper presented at the Proceedings of the 1998 Science Board Symposium on the impacts of the 1997/98 El Niño event on the North Pacific Ocean and its marginal seas.

Wibbels, T. (2003). Critical approaches to sex determination in sea turtles. In P. L. Lutz, J. A. Musick & J. Wyneken (Eds.), *The Biology of Sea Turtles* (Vol. 2, pp. 103-134). Boca Raton, Florida: CRC Press.

Wilkinson, C. (2000). Status of coral reefs of the world: 2000: Global Coral Reef Monitoring Network, Australian Institute of Marine Science.

Wilkinson, C., & Souter, D. (2008). Status of Caribbean coral reefs after bleaching and hurricanes in 2005. Townsville: Global Coral Reef Monitoring Network and Reef and Rainforest Research Centre.

Williams, R., Bain, D. E., Ford, J. K. B., & Trites., A. W. (2002). Behavioural responses of male killer whales to a “leapfrogging” vessel. (Orcinus orca). *Journal of Cetacean Research and Management, 4*(3), 305-310.

Williams, R., & Noren, D. P. (2009). Swimming speed, respiration rate, and estimated cost of transport in adult killer whales. *Marine Mammal Science, 25*(2), 327-350.

Williams, R., Trites, A. W., & Bain, D. E. (2002). Behavioural responses of killer whales (*Orcinus orca*) to whale-watching boats: Opportunistic observations and experimental approaches. *Journal of Zoology, 256*(2), 255-270.

Williams, R., Trites, r. W., & Bain., D. E. (2002). Behavioural responses of killer whales (Orcinus orca) to whale-watching boats: Opportunistic observations and experimental approaches. *Journal of Zoology, 256*(2), 255-270.

Winn, H. E., Goodyear, J. D., Kenney, R. D., & Petricig, R. O. (1995). Dive patterns of tagged right whales in the Great South Channel. *Continental Shelf Research, 15*(4-5), 593-611.

Winn, H. E., Perkins, P. J., & Poulter, T. C. (1970). *Sounds of the humpback whale*. Paper presented at the Proceedings of the 7th Annual Conference on Biological Sonar and Diving Mammals, Stanford Research Institute Menlo Park CA. p.39-52.

Winn, H. E., & Reichley, N. E. (1985). Humpback whale - *Megaptera novaeangliae*. In S. H. Ridgway & S. R. Harrison (Eds.), *Handbook of Marine Mammals: Vol. 3 The Sirenians and Baleen Whales* (pp. 241-274). London: Academic Press Ltd.

Wise, J. P., Sr., Payne, R., Wise, S. S., Lacerte, C., Wise, J., Christy Gianios, J., . . . Kerr., I. (2009). A global assessment of chromium pollution using sperm whales (Physeter macrocephalus) as an indicator species. *Chemosphere, 75*(11), 1461-1467.

Witherington, B., Kubilis, P., Brost, B., & Meylan, A. (2009). Decreasing annual nest counts in a globally important loggerhead sea turtle population. *Ecological Applications, 19*(1), 30-54. doi: doi:10.1890/08-0434.1

Witherington, B. E. (1992a). Behavioral responses of nesting sea turtles to artificial lighting. *Herpetologica, 48*, 31-39.

Witherington, B. E. (1992b). Behavioral responses of nesting sea turtles to artificial lighting. *Herpetologica, 48*(1), 31-39.

Witherington, B. E., & Bjorndal, K. A. (1991a). Influences of artificial lighting on the seaward orientation of hatchling loggerhead turtles *Caretta caretta*. *Biological Conservation, 55*(2), 139-149.

Witherington, B. E., & Bjorndal, K. A. (1991b). Influences of artificial lighting on the seaward orientation of hatchling loggerhead turtles Caretta caretta. *Biological Conservation, 55*, 139-149.

Witherington, B. E., Herren, R., & Bresette, M. (2006b). *Caretta caretta* – Loggerhead Sea Turtle. *Chelonian Research Monographs, 3*, 74-89.

Witteveen, B. H., Foy, R. J., Wynne, K. M., & Tremblay, Y. (2008). Investigation of foraging habits and prey selection by humpback whales (*Megaptera novaeangliae*) using acoustic tags and concurrent fish surveys. *Marine Mammal Science, 24*(3), 516-534. doi: 10.1111/j.1748-7692.2008.00193.x

Witteveen, B. H., Worthy, G. A. J., Wynne, K. M., & Roth., J. D. (2009). Population structure of North Pacific humpback whales on their feeding grounds revealed by stable carbon and nitrogen isotope ratios. *Marine Ecology Progress Series, 379*, 299-310.

Wright, A. J. (2005). Lunar cycles and sperm whales (*Physeter macrocephalus*) strandings on the North Atlantic coastlines of the British Isles and Eastern Canada. *Marine Mammal Science, 21*(1), 145-149.

Wright, A. J., Soto, N. A., Baldwin, A., Bateson, M., Beale, C., Clark, C., . . . Martin, V. (2007). Anthropogenic noise as a stressor in animals: A multidisciplinary perspective. *International Journal of Comparative Psychology, 201*(2-3), 250-273.

Wursig, B., Lynn, S. K., Jefferson, T. A., & Mullin, K. D. (1998). Behaviour of cetaceans in the northern Gulf of Mexico relative to survey ships and aircraft. *Aquatic Mammals, 24*(1), 41-50.

Wursig, B., & Richardson, W. J. (2009). Noise, effects of. In W. F. Perrin, B. Wursig & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals. Second edition* (pp. 765-773). San Diego: Academic Press.

Wysocki, L. E., Amoser, S., & Ladich, F. (2007). Diversity in ambient noise in European freshwater habitats: Noise levels, spectral profiles, and impact on fishes. *Journal of the Acoustical Society of America, 121*(5), 2559-2566. doi: 10.1121/1.2713661

Yablokov, A. V. (2000). Consequences and perspectives of whaling (instead of a preface) *Soviet Whaling Data (1949-1979)* (pp. 6-10). Moscow: Center for Russian Environmental Policy Marine Mammal Council.

Yablokov, A. V., Zemsky, V. A., Mikhalev, Y. A., Tormosov, V. V., & Berzin, A. A. (1998). Data on Soviet whaling in the Antarctic in 1947–1972 (population aspects). *Russian Journal of Ecology, 29*, 38–42.

Yang, Z., Wang, Z., Tang, C., & Ying, Y. (1996). Biological Effects of Weak Blast Waves and Safety Limits for Internal Organ Injury in the Human Body. *The Journal of Trauma and Acute Care Surgery, 40*(3S), 81S-84S.

Yelverton, J. T., Richmond, D. R., Hicks, W., Saunders, H., & Fletcher, E. R. (1975). The relationship between fish size and their response to underwater blast. Albuquerque, N. M.: Lovelace Foundation for Medical Education Research

Ylitalo, G. M., Baird, R. W., Yanagida, G. K., Webster, D. L., Chivers, S. J., Bolton, J. L., . . . Mcsweeney., D. J. (2009). High levels of persistent organic pollutants measured in blubber of island-associated false killer whales (Pseudorca crassidens) around the main Hawaiian Islands. *Marine Pollution Bulletin, 58*(12), 1932-1937.

Yochem, P. K., & Leatherwood, S. (1985). Blue whale *Balaenoptera musculus* (Linnaeus, 1758). In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals, vol. 3: The Sirenians and Baleen Whales* (pp. 193-240). London: Academic Press.

Yuen, M. M. L., Nachtigall, P. E., Breese, M., & Supin, A. Y. (2005). Behavioral and auditory evoked potential audiograms of a false killer whale (Pseudorca crassidens). *The Journal of the Acoustical Society of America, 118*(4), 8.

Zaitseva, K. A., Morozov, V. P., & Akopian, A. I. (1980). Comparative characteristics of spatial hearing in the dolphin ursiops truncatus and man. *Neuroscience and Behavioral Physiology, 10*(2), 180-182.

Zakarauskas, P., Chapman, D. M. F., & Staal, P. R. (1990). Underwater acoustic ambient noise levels on the eastern Canadian continental shelf. *The Journal of the Acoustical Society of America, 87*, 2064.

Zárate, P. M., Parra, M. A., Robles, M., Dutton, P. H., & Seminoff, J. A. (2010). *Sea turtle strandings and mortality in the Galapagos Archipelago: causes and threats* Paper presented at the 28th Annual Symposiumon Sea Turtle Biology and Conservation.

Zimmer, W. M. X., & Tyack, P. L. (2007). Repetitive shallow dives pose decompression risk in deep-diving beaked whales. *Marine Mammal Science, 23*(4), 888-925.

Zimmerman, B. (1983a). Hawaii- Long log. *Hawaii Fishing News, 8*(3), 25.

Zimmerman, B. (1983b). Hawaii-long log. *Hawaii Fishing News, 8*(3), 25.

Zug, G. R., & Parham, J. F. (1996). Age and growth in leatherback turtles, *Dermochelys coriacea*: A skeletochronological analysis. *Chelonian Conservation and Biology, 2*, 244-249.

Zurita, J. C., Herrera, R., Arenas, A., Torres, M. E., Calderón, C., Gómez, L., . . . Villavicencia, R. (2003). *Nesting loggerhead and green sea turtles in Quintana Roo, Mexico*. Paper presented at the 22nd Annual Symposium on Sea Turtle Biology and Conservation, Miami, FL.

Zylber, M. I., Failla, G., & Bas, A. L. (2002). Stenurus globicephalae Baylis et Daubney, 1925 (Nematoda: Pseudaliidae) from a false killer whale, Pseudorca crassidens (Cetacea: Delphinidae), stranded on the coast of Uruguay. *Memorias Do Instituto Oswaldo Cruz, Rio De Janeiro, 97*(2), 221-225.

REFERENCES NOT IN OPR LIBRARY IN EndNote

Au, W.W.L., and M.C. Hastings. 2008. Principles of marine bioacoustics. Modern acoustics and signal processing. New York, New York: Springer-Verlag.

Banner, A. 1967. Evidence of sensitivity to acoustic displacements in the lemon shark, *Negaprion brevirostris* (Poey). Pages 265 - 273 in P.H. Cahn, ed . Lateral line detectors. Bloomington, Indiana: Indiana University Press.

Banner, A. 1972. Use of sound in predation by young lemon sharks, *Negaprion brevirostris* (Poey). Bulletin of Marine Science 22 (2) :251 - 283.

Bessudo, S., G.A. Soler, A.P. Klimley, J.T. Ketchum, A. Hearn, and R. Arauz. 2011. Residency of the scalloped hammerhead shark *(Sphyrna lewini)* at Malpelo Island and evidence of migration to other islands in the Eastern Tropical Pacific. Environmental Biology o f Fishes 91:165-176.

Casper, B.M., P.S. Lobel, and H.Y . Yan. 2003. The hearing sensitivity of the little skate, *Raja erinacea:* A comparison of two methods. Environmental Biology of Fishes 68(4) :371-379.

Casper, B.M., and D.A. Mann. 2006. Evoked potential audiograms of the nurse shark *(Ginglymostoma cirratum)* and the yellow stingray *(Urobatis jamaicensis).* Environmental Biology of Fishes 76 (1): 101- 108 .

Casper, B.M., and D.A. Mann. 2007. Dipole hearing measurements in elasmobranch fishes. Journal of Experimental Biology 210 (1): 75 - 81.

Clarke, S.C., J.E. Magnussen, D.L. Abercrombie, M.K. McAllister, and M.S . Shivji. 2006. Identification of shark species composition and proportion in the Hong Kong shark fin market based on molecular genetics and trade records. Conservation Biology 20(1) :201-211.

Compagno, L.J.V. 1984. Sharks of the world. An annotated and illustrated catalogue of shark species to date. Part II (Carcharhiniformes). FAO Fisheries Synopsis, FAO, Rome.

Diemer, K.M., B.Q. Mann, N.E. Hussey. 2011. Distribution and movement of scalloped hammerhead *(Sphyrna lewini)* and smooth hammerhead *(Sphyrna zygaena)* sharks along the east coast of Southern Africa. African Journal of Marine Science 33:229-238.

DoN (U.S. Department of the Navy). 2012. Final supplemental environmental impact statement/supplemental overseas environmental impact statement for Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar. Washington, D.C. : Department of the Navy, Chief of Naval Operations.

Halvorsen, M.B., L.E. Wysocki, and A.N. Popper. 2006. Effects of high-intensity sonar on fish. Journal of the Acoustical Society of America 119 : 3283.

Hastings, M.C., A.N. Popper, J.J. Finneran, and P.J. Lanford.1996. Effects of low-frequency underwater sound on hair cells of the inner ear and lateral line of the teleost fish *(Astronotus* ocellatus) . Journal of Acoustical Society of America 99(3) :1759-1766.

Hill, R.J. 2005 . Standardizing the auditory evoked potential technique: Ground-truthing against behavioral conditioning in the goldfish *Carassius auratus.* Ph.D. dissertation, University of South Florida. 175 pages.

Houser, D.S., and J.J. Finneran. 2006. A comparison of underwater hearing sensitivity in bottlenose dolphins *(Tursiops truncatus)* determined by electrophysiological and behavioral methods. Journal of the Acoustical Society of America 120(3) :1713-1722.

Kane, A.S., J. Song, M.B. Halvorsen, D.L. Miller, J.D. Salierno, L.E. Wysocki, D. Zeddies, and A.N. Popper. 2010. Exposure of fish to high-intensity sonar does not induce acute pathology. Journal of Fish Biology 76(7) :1825-1840.

Kelly, J.C., and D.R. Nelson. 1975. Hearing thresholds of the horn shark, *Heterodontus francisci.* Journal of the Acoustical Society of America 58(4) : 905-909 .

Klimley, A.P. 1993. Highly directional swimming by scalloped hammerhead sharks, *Sphyrna lewini*, and subsurface irradiance, temperature, bathymetry, and geomagnetic field. Marine Biology 117:1-22.

Klimley, A.P., and A.A . Myrberg. 1979. Acoustic stimuli underlying withdrawal from a sound source by adult lemon sharks, *Negaprion brevirostris* (Poey). Bulletin of Marine Science 29:447-458.

Kritzler, H., and L. Wood. 1961. Provisional audiogram for the shark, *Carcharhinus leucas* . Science 133 : 1480-1482.

Maguire, J.-J., M. Sissenwine, J. Csirke, R. Grainger, and S. Garcia. 2006. The state of world highly migratory, straddling and other high seas fishery resources and associated species. FAO Fisheries Technical Paper 495. FAO, Rome, Italy . 88 pages.

Miller, M.H., J. Carlson, P. Cooper, D. Kobayashi, M. Nammack, and J. Wilson. 2014. Status review report : Scalloped hammerhead shark *(Sphyrna lewini)* . Final report to the National Marine Fisheries Service, Office of Protected Species. March 2014. 133 pages.

Montgomery, J.C., and M.M. Walker. 2001. Orientation and navigation in elasmobranchs: Which way forward? Environmental Biology of Fishes 60:109-116.

Myrberg, A.A. 1978. Underwater sound-its effect on the behavior of sharks . Pages 391-417 in E.S. Hodgson and R. Mathewson, eds. Sensory biology of sharks, skates, and rays. Arlington, Virgina: Department of the Navy, Office of Naval Research.

Nelson, D.R. 1967. Hearing thresholds, frequency discrimination, and acoustic orientation in the lemon shark, *Negaprion brevirostris* (Poey) . Bulletin of Marine Science 17:741-768.

Nelson, D. R., and S.H. Gruber. 1963. Sharks: Attraction by lowfrequency sounds. Science 142(3594), New Series: 975-977.

Nelson, D.R . , and R. H. Johnson . 1970. Acoustic studies on sharks, Rangiroa Atoll, July 1969. ONR Technical Report 2 , No. N00014 -6 8 -C- 0138. Defense Technical Information Center.

Nelson, D.R., and R.H. Johnson. 1972. Acoustic attraction of Pacific reef sharks: Effects of pulse intermittency and variability. Comparative Biochemistry and Physiology, Part A: Physiology 42(1) :85-95.

Nelson, D.R., and R.H. Johnson. 1976. Some recent observations on acoustic attraction of Pacific reef sharks. Pages 229 – 239 in A. Schuijf and A.D. Hawkins, eds. Sound reception in fish. Developments in aquaculture and fisheries science. Amsterdam: Elsevier.

NOAA (National Oceanic and Atmospheric Administration). 2014. Endangered and threatened wildlife and plants; Threatened and endangered status for distinct population segments of scalloped hammerhead sharks; Final rule. 50 CFR Parts 223 and 224. Federal Register 79(128) : 38214 - 38242.

Olla, B. 1962 . The perception of sound in small hammerhead sharks *Sphyrna lewini*. M.S. thesis, University of Hawaii.

Popper, A.N., M. B. Halvorsen, D. Miller, M.E. Smith, J. Song, L.E. Wysocki, M.C. Hastings, A.S. Kane, and P . Stein. 2005. Effects of Surveillance Towed Array Sensor System (SURTASS) low frequency active sonar on fish. Paper presented at the 149th Meeting of the Acoustical Society of America. Journal of the Acoustical Society of America 117(4) :2440-2440.

Popper, A.N., M.B. Halvorsen, A. Kane, D.L . Miller, M.E. Smith, J. Song, P. Stein, and L.E. Wysocki. 2007. The effects of high-intensity, low-frequency active sonar on rainbow trout. Journal of the Acoustical Society of America 122 (1) :623-635.

Prus, K. 2013. Behavior and conservation of scalloped hammerhead *(Sphyrna lewini)* in the Eastern Tropical Pacific. Natural Sciences, Paper 1. http://commons.pacificu . edu/casns.

Smith, M.E., A.S. Kane, and A. N. Popper. 2004a . Noise-induced stress response and hearing loss in goldfish *(Carassius* auratus). Journal of Experimental Biology 207:427-435.

Smith, M.E., A.S. Kane, and A.N . Popper. 2004b. Acoustical stress and hearing sensitivity in fishes: Does the linear threshold shift hypothesis hold water? Journal of Experimental Biology 207:3591-3602.