International Association IAGC of Geophysical Contractors
energy API
NOIA America's Offshore Energy Industry

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VIA ELECTRONIC SUBMISSION (www.regulations.gov)

Ms. Jolie Harrison
Chief, Permits and Conservation Division
Office of Protected Resources
National Marine Fisheries Service
1315 East West Highway
Silver Spring, MD 20910

Re: Comments on Proposed Marine Mammal Incidental Take Regulations for Geophysical Surveys in the Gulf of Mexico (NMFS-2018-0043)

Dear Ms. Harrison:

This letter provides the comments of the International Association of Geophysical Contractors (IAGC), the American Petroleum Institute (API), the National Ocean Industries Association (NOIA), and the Offshore Operators Committee (OOC) (collectively, the "Associations") in response to the National Marine Fisheries Service's (NMFS) request for comments on its proposed marine mammal incidental take regulations for geophysical surveys in the Gulf of Mexico (GOM) (the "Proposed ITR"). *See* 83 Fed. Reg. 29,212 (June 22, 2018). The Associations previously commented on the Bureau of Ocean Energy Management's (BOEM) revised application to NMFS for a marine mammal incidental take regulation (ITR) for these activities (the "Application").[1] We appreciate NMFS's consideration of the comments set forth below on the Proposed ITR.

I. THE ASSOCIATIONS

IAGC is the international trade association representing the industry that provides geophysical services (geophysical data acquisition, processing and interpretation, geophysical information ownership and licensing, and associated services and product providers) to the oil and natural gas industry. IAGC member companies play an integral role in the successful exploration and development of offshore hydrocarbon resources through the acquisition and processing of geophysical data.

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[1] IAGC, API, NOIA, and OOC *Comments on Revised Application for Marine Mammal Incidental Take Regulations for Geophysical Surveys in the Gulf of Mexico* (Jan. 23, 2017).

API is a national trade association representing over 625 member companies involved in all aspects of the oil and natural gas industry. API's members include producers, refiners, suppliers, pipeline operators, and marine transporters, as well as service and supply companies that support all segments of the industry. API and its members are dedicated to meeting environmental requirements, while economically developing and supplying energy resources for consumers.

NOIA is the only national trade association representing all segments of the offshore industry with an interest in the exploration and production of both traditional and renewable energy resources on the U.S.'s outer continental shelf (OCS). NOIA's membership comprises more than 325 companies engaged in a variety of business activities, including production, drilling, engineering, marine and air transport, offshore construction, equipment manufacture and supply, telecommunications, finance and insurance, and renewable energy.

OOC is an organization of 47 producing companies and 61 service providers to the industry that conduct essentially all of the OCS oil and gas exploration and production activities in the GOM. Founded in 1948, the OOC is a technical advocate for the oil and gas industry regarding the regulation of offshore exploration, development, and producing operations in the GOM.[2]

II. OVERVIEW

The GOM Outer Continental Shelf (OCS) is an indispensable source of oil and gas for the Nation's energy supply. The continued development of the GOM cannot effectively or efficiently occur without geophysical survey activities, which are essential to the safe discovery, development, and valuation of OCS resources. This rulemaking and the coincident restrictions it may impose on GOM geophysical activities are therefore of paramount importance to the continued and future exploration and production of domestic oil and gas reserves in the GOM. Accordingly, the Associations have fully participated in this rulemaking process and in all related regulatory processes. We will continue to collaborate in a process that is transparent, provides certainty to the regulated community, and does not undermine the development of the U.S. OCS or U.S. energy security. Any final rule must be consistent with the Administration's stated policy that "America must put the energy needs of American families and businesses first and continue implementing a plan that ensures energy security and economic vitality for decades to come."[3]

In general, the Proposed ITR is a well-structured and thorough document that appropriately concludes—consistent with decades of uncontroverted data (including data from current mitigation efforts), research results, and agency findings—that geophysical activities in the

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[2] By submitting this letter, the Associations do not intend to limit the ability of their individual member companies to submit separate comments or present their own views on the issues discussed in this letter.

[3] Presidential Executive Order Implementing an America-First Offshore Energy Strategy (Apr. 28, 2017), http://www.presidency.ucsb.edu/ws/index.php?pid=123867.

GOM have no more than a *negligible impact* on marine mammal populations. We appreciate NMFS's effort in preparing the Proposed ITR and its consideration of some of the Associations' previous comments. Although we agree with the general conclusions of the Proposed ITR, there remain significant and important shortcomings that must be fixed in order to ensure a transparent, fair, and lawful regulatory process. Unfortunately, the Proposed ITR carries forward several of the significant flaws contained in BOEM's Application and includes several new flaws that must be addressed. Our concerns and general comments are summarized as follows:

- As addressed in Section III.B below, NMFS continues to inappropriately dismiss the best available science regarding the potential impacts of seismic surveys on marine mammal populations. In so doing, NMFS fails to premise the Proposed ITR on the best available science, as required by law, and, instead, proposes restrictions based on an assumption that seismic surveys in the GOM will have future negative impacts of a nature and magnitude that to date have *never* been observed.

- Although the Proposed ITR appropriately provides practicability analyses for some proposed mitigation measures, it fails to meet NMFS's statutory and regulatory obligations because it does not include a practicability analysis for many of the proposed mitigation measures. In addition, the practicability analyses that are provided in the Proposed ITR fail to consider costs and impacts beyond the immediate survey work for all analyzed geophysical survey types, ignoring the critical broader purpose of the surveys. *See* Sections III.A and III.C below.

- Certain mitigation measures in the Proposed ITR are impracticable, unnecessary, without factual or legal support, and inconsistent with measures that have been required for other geophysical surveys, and impose costs and safety risks far in excess of any foreseeable benefits. Our concerns with these specific proposed measures are set forth in detail in Section III.D below. We request that NMFS devote special attention to these genuine concerns as the problems we identify pose serious threats to the viability and effectiveness of geophysical surveys in the GOM and, if adopted, will undermine U.S. energy policies.

- Although we agree with NMFS's negligible impact determination, we recommend certain improvements to NMFS's approach to this determination in Section III.E below. We similarly agree with NMFS's approach to determining "small numbers," and offer legal support and suggestions in that regard in Section III.F below.

- The Letter of Authorization (LOA) applications requested under the final ITR will substantially multiply the number of incidental take authorization applications NMFS typically receives in a given year. We strongly recommend that the final ITR clearly address how NMFS plans to process voluminous LOA applications in a timely and efficient manner that does not hamper the exploration and production of GOM oil and gas resources. In this vein, we encourage NMFS to retain flexibility in the final ITR for the development of efficient and effective LOA processes through workshops or other engagement with BOEM and the regulated community. *See* Section III.G below.

- NMFS continues to substantially overestimate the number of incidental takes predicted to result from the activities described in the Proposed ITR. We have explained at length, and with detail, that the modeling used to estimate the anticipated number of incidental takes is improperly and intentionally designed to *overestimate* takes and impacts. Notwithstanding NMFS's strong disagreement on this issue, we maintain our position because it is supported by the record facts, the best available science, the agencies' own statements, and the modeling used by both BOEM and NMFS. *See* Section III.H below.

- The Regulatory Impact Analysis (RIA) makes many incorrect assumptions and unsupported conclusions. This analysis is essential to the public's understanding of the true impact of the contemplated regulation, and, accordingly, these flaws must be addressed in the final analysis. *See* Section III.J below.

Although we encourage NMFS to proceed with this rulemaking on a schedule that is compliant with court-ordered deadlines, we urge NMFS to do so in a manner that comports with the Marine Mammal Protection Act's (MMPA) requirements and the best scientific data available. To accomplish this, NMFS must incorporate in its final ITR the comments and recommendations outlined below. We sincerely appreciate NMFS's consideration of these comments, which are intended to be productive and to improve the quality and defensibility of the final ITR.

III. COMMENTS

A. Geophysical surveys play a critical role in the safe and orderly development of the oil and gas resources of the GOM.

1. Legal context.

To issue an ITR under Section 101(a)(5) of the MMPA, NMFS must find that the specified activity for which take will be authorized is limited to a "specified geographical region," has no more than a "negligible impact" on a marine mammal species or stock, and does not result in an unmitigable adverse impact on the availability of such species or stock for taking for subsistence uses. 16 U.S.C. § 1371(a)(5)(A)(I). In addition, NMFS must prescribe permissible methods of taking and other means of effecting the least practicable impact on the affected species or stocks. *See id.* § 1371(a)(5)(A)(II). Here, the geophysical activities to which the Proposed ITR would apply are authorized by BOEM pursuant to the Outer Continental Shelf Lands Act (OCSLA). *See* 43 U.S.C. § 1340.

OCSLA calls for the "expeditious and orderly development" of the OCS "subject to environmental safeguards." 43 U.S.C. § 1332(3); *see California v. Watt*, 668 F.2d 1290, 1316 (D.C. Cir. 1981) (OCSLA's primary purpose is "the expeditious development of OCS resources"). Congress enacted OCSLA to "achieve national economic and energy policy goals, assure national security, reduce dependence on foreign sources, and maintain a favorable balance of payments in world trade." 43 U.S.C. § 1802(1). Congress expressly intended to "make [OCS] resources available to meet the Nation's energy needs as rapidly as possible." *Id.* § 1802(2)(A). Consistent with this Congressional policy, in 2017, the President signed an Executive Order expressly stating that it "shall be the policy of the United States to encourage energy exploration and production, including on the Outer Continental Shelf ... while ensuring that any such activity is safe and environmentally responsible."[4] Neither OCSLA nor the MMPA requires an applicant to obtain an incidental take authorization under the MMPA for geophysical activities.[5] However, unlawful incidental takes of marine mammals may be subject to MMPA-based penalties. *See* 16 U.S.C. § 1375. Marine mammal incidental take authorizations for geophysical activities in the GOM have rarely, if ever, been issued by NMFS. Applications for an ITR for GOM geophysical activities have been pending or in various stages of preparation since 2002.

Notwithstanding the lack of GOM-specific ITRs, industry operators have for years complied with measures imposed under the terms of geophysical activity authorizations to protect marine mammals. *See* Joint Notice to Lessees (NTL) No. 2016-G02 (previously NTL No. 2012-G02 and NTL No. 2007-G02). By all accounts, these measures have been effective. Based on the best available scientific information, there has been no demonstration of any biologically significant negative impacts to marine life from geophysical activities in the GOM.[6]

On June 30, 2010, a consortium of environmental advocacy groups filed a federal lawsuit challenging BOEM's determination that the authorization of geophysical activities in the GOM does not require the preparation of an environmental impact statement (EIS). *See NRDC et al. v. Jewell et al.*, No. 2:10-cv-01882, Dkt. 1 (E.D. La.) ("*NRDC v. Jewell*"). The claims asserted in *NRDC v. Jewell* were resolved through a settlement agreement dated June 18, 2013, as amended by stipulations dated February 8, 2016 and September 26, 2017 (referred to collectively as the "Settlement and Stipulations"). *See NRDC v. Jewell*, Dkts. 118-2, 127-2, and 143-2; *see also id.*, Dkts. 119, 128, and 144 (court orders granting approval of Settlement and Stipulations).

The Settlement and Stipulations address, *inter alia*, BOEM's application for an ITR for GOM geophysical activities and programmatic National Environmental Policy Act analysis of the potential effects of such activities. Under the terms of the Settlement and Stipulations, geophysical operators are required to implement a suite of "interim" mitigation measures that substantially expand upon the mitigation measures traditionally required under NTLs. However, the parties to the Settlement and Stipulations did *not* agree, and there has otherwise been no demonstration, that the mitigation measures imposed pursuant to the Settlement and Stipulations are feasible, appropriate, supported by the best available science, or otherwise required by law.[7]

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[4] Presidential Executive Order Implementing an America-First Offshore Energy Strategy, http://www.presidency.ucsb.edu/ws/index.php?pid=123867.

[5] BOEM has elected to condition its permits on the applicant obtaining an MMPA authorization in some instances, including in the GOM.

[6] *See* BOEM, Science Notes, http://www.boem.gov/BOEM-Science-Note-August-2014/ (Aug. 22, 2014); *see also* BOEM, Science Notes, https://www.boem.gov/BOEM-Science-Note-March-2015/ (Mar. 9, 2015).

[7] *See NRDC v. Jewel*, Dkt. 118-2, Section IX ("Intervenor-Defendants do not agree that all of the measures described in paragraph IX.A and IX.B are feasible or appropriate. Intervenor-

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The Associations' members have performed the terms of the Settlement and Stipulations in good faith. The Associations have also constructively participated in the regulatory processes pertaining to the Application and the development of related environmental documents and decisions.

2. Operational context.

In 2017, the GOM OCS region was responsible for 18% of the total U.S. crude oil production and 4% of dry natural gas production.[8] Likewise, GOM OCS leases are an important source of federal revenues, generating substantial bonuses, rentals, and royalties paid to the U.S. Since 2008, lessees have paid over $11 billion in bonus bids for lease sales in the GOM OCS.[9] Total oil and gas royalty revenues from the GOM OCS amounted to almost $4 billion in fiscal year 2017 alone.[10] Moreover, BOEM has recently estimated the net economic value of future GOM leasing to be as high as $197 billion.[11] Geophysical activities are critical to the discovery, development, and valuation of OCS resources that lead to such production. Geophysical activities are temporary and transitory, and seismic surveying is the least intrusive and most cost-effective means to determine the likely locations of recoverable oil and gas resources in the GOM.

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Defendants shall be free to challenge any such measures should one or more of the Federal Defendants develop and implement them."); *id*. at Dkt. 127-2, Section G ("The terms of this Stipulation have been agreed to for purposes of compromise. No party concedes by entering into this Stipulation that any of the permit requirements described above are warranted by scientific evidence or should be imposed after the Stay expires, or that these requirements are sufficient to achieve legal compliance or reduce biological risk over the long term."); *id.* at Dkt. 143-2 ("The terms of this Second Stipulation have been agreed to for purposes of compromise. No party concedes by entering into this Second Stipulation that any permit requirements heretofore agreed to are warranted by scientific evidence or should be imposed after the Stay expires, or that these requirements are sufficient to achieve legal compliance or reduce biological risk over the long term.").

[8] *See* U.S. Energy Information Administration, *Gulf of Mexico Fact Sheet* (July 17, 2018), https://www.eia.gov/special/gulf_of_mexico/data.php.

[9] *See* BOEM, *Outer Continental Shelf Lease Sale Statistics, Gulf of Mexico Oil and Gas Lease Offerings* (Jan. 25, 2018), https://www.boem.gov/Outer-Continental-Shelf-Lease-Sale-Statistics/.

[10] *See* DOI, Office of Natural Resources Revenue, Statistical Information, http://statistics.onrr.gov/ReportTool.aspx (Reported Revenues [Single Year Only], FY2015, Accounting Year, Federal Offshore, Offshore Gulf).

[11] *See* BOEM, *2017-2022 OCS Oil and Gas Leasing Proposed Final Program*, at Table 5-8 (Nov. 2016), https://www.boem.gov/2017-2022-OCS-Oil-and-Gas-Leasing-PFP.

Geophysical surveying has been and continues to be essential to achieving OCSLA's goals because it is the only feasible technology available to accurately image the subsurface of the OCS before a single well is drilled and to monitor a well during its production. Industry has made significant improvements in acquisition efficiency in recent years. Using standard hardware, the industry now acquires more and better quality data due to advancements in vessels, configurations, acquisition planning and execution, and data processing. For certain categories of geophysical surveys (high resolution, or "HRG"), this includes the use of autonomous technologies to conduct surveys without ships, which reduces the cost of surveys and, more importantly, improves safety by requiring fewer people to be on the water. The use of autonomous technology is becoming standard across many ocean industries—not just oil and gas. Additional advancements in geophysical technology—including seismic reflection and refraction, gravity, magnetics, and electromagnetics—afford industry significant precision in subsurface imaging and will continue to provide more realistic estimates of potential resources. By utilizing these tools and applying increasingly accurate and effective interpretation practices, industry can better locate and safely dissect prospective areas for exploration.

Furthermore, modern geophysical imaging *reduces risk* by increasing the likelihood that exploratory wells will successfully tap hydrocarbons and by decreasing the number of wells that need to be drilled in a given area, thereby reducing associated safety and environmental risks as well as the overall environmental footprint for exploration. For example, subsurface imaging can predict potentially hazardous over-pressurized zones in a reservoir and thus allow an operator to better design a well to reduce its associated types and levels of risk. As technology advances, the geophysical industry can continue to reduce drilling risk and increase potential production. Just as physicians today may use MRI technology to image an area that previously had been imaged by X-ray technology, geophysical experts are actively using and enhancing modern technology to make improved evaluations. In addition to these advancements, the industry has been committed to research aimed at better understanding the potential effects of geophysical activities on marine mammals and informing the development of best mitigation practices and potential alternative technologies.[12]

Finally, deep penetration seismic air sources remain the most effective, commercially available technology to obtain necessary, accurate sub-surface data. Although alternative technologies, including marine vibroseis, continue to be explored, such technology is not yet commercialized and has not yet been shown to provide comparable seismic data quality. The substantial cost to modify vessels and to use vibroseis requires a significant market demand to make the technology

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[12] *See* E&P Sound and Marine Life Joint Industry Programme, www.soundandmarinelife.org; *see also* http://www.brahss.org.au/; *see also, e.g.*, Verfuss, U. et al. 2018. Comparing methods suitable for monitoring marine mammals in low visibility conditions during seismic surveys. Marine Pollution Bulletin 126 (2018), 1-18.

commercially viable. Moreover, the hypothetical environmental benefits of alternative technologies have not yet been demonstrated.

B. NMFS inappropriately dismisses the best available science and interprets the MMPA in a manner that is contrary to Congressional intent.

For over 40 years, the federal government and academic scientists have studied the potential impacts of geophysical activities on marine mammal populations and have concluded that any such potential impacts are *insignificant*. This conclusion has been publicly reaffirmed on multiple occasions by BOEM:

> To date, there has been no documented scientific evidence of noise from air guns used in geological and geophysical (G&G) seismic activities adversely affecting marine animal populations or coastal communities. This technology has been used for more than 30 years around the world. It is still used in U.S. waters off of the Gulf of Mexico with no known detrimental impact to marine animal populations or to commercial fishing.

BOEM, Science Notes, http://www.boem.gov/BOEM-Science-Note-August-2014/ (Aug. 22, 2014); *see also* BOEM, Science Notes, https://www.boem.gov/BOEM-Science-Note-March-2015/ (Mar. 9, 2015) (there has been "no documented scientific evidence of noise from air guns used in geological and geophysical (G&G) seismic activities adversely affecting animal populations").[13] Most recently, BOEM confirmed that "there are multiple factors that indicate the potential for repeated [seismic sound] exposures is unlikely to result in reduced fitness in individuals or populations" and that "G&G surveys have been ongoing in the northern GOM for many years, *with no direct information indicating reduced fitness in individuals or populations*."[14]

Indeed, the history of formal assessments of offshore seismic activities demonstrates that levels of actual incidental take are far smaller than even the most balanced pre-operation estimates of incidental take.[15] More than four decades of worldwide seismic surveying and scientific research

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[13] Copies of both of these BOEM Science Notes are provided with this letter as Appendix A, for NMFS's consideration and for inclusion in the administrative record.

[14] Final Programmatic Environmental Impact Statement, Gulf of Mexico OCS, Proposed Geological and Geophysical Activities ("GOM PEIS") at 4-53 (emphasis added).

[15] *See, e.g.*, BOEM, *Final EIS for Gulf of Mexico OCS Oil and Gas Eastern Planning Area Lease Sales 225 and 226*, at 2-22 (2013), http://www.boem.gov/BOEM-2013-200-v1/ ("Within the CPA, which is directly adjacent to the EPA, there is a long-standing and well developed OCS Program (more than 50 years); there are no data to suggest that activities from the preexisting OCS Program are significantly impacting marine mammal populations."); BOEM, *Final EIS for Gulf of Mexico OCS Oil and Gas Western Planning Area (WPA) Lease Sales 229, 233, 238, 246, and 248 and Central Planning Area (CPA) Lease Sales 227, 231, 235,*

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indicate that the risk of physical injury to marine life from seismic survey activities is extremely low. For example, as BOEM concludes in its recently released GOM PEIS, “within the GOM, there is a long-standing and well-developed OCS [oil and gas] Program (more than 50 years) and there are no data to suggest that activities from the previous OCS Program are significantly impacting marine mammal populations.” GOM PEIS at 4-75.

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241, and 247, at 4-203 (v.1) (2012), http://www.boem.gov/Environmental-Stewardship/Environmental-Assessment/NEPA/BOEM-2012-019_v1.aspx (WPA); *id*. at 4-710 (v.2), http://www.boem.gov/Environmental-Stewardship/Environmental-Assessment/NEPA/BOEM-2012-019_v2.aspx (CPA) (“Although there will always be some level of incomplete information on the effects from routine activities under a WPA proposed action on marine mammals, there is credible scientific information, applied using acceptable scientific methodologies, to support the conclusion that any realized impacts would be sublethal in nature and not in themselves rise to the level of reasonably foreseeable significant adverse (population-level) effects.”); BOEM, *Final Supplemental EIS for Gulf of Mexico OCS Oil and Gas WPA Lease Sales 233 and CPA Lease Sale 231*, at 4-30, 4-130 (2013), http://www.boem.gov/uploadedFiles/BOEM/BOEM_Newsroom/Library/Publications/2013/BOEM%202013-0118.pdf (reiterating conclusions noted above); MMS, *Final Programmatic EA, G&G Exploration on Gulf of Mexico OCS*, at III-9, II-14 (2004), http://www.nmfs.noaa.gov/pr/pdfs/permits/mms_pea2004.pdf (“There have been no documented instances of deaths, physical injuries, or auditory (physiological) effects on marine mammals from seismic surveys.”); *id*. at III-23 (“At this point, there is no evidence that adverse behavioral impacts at the local population level are occurring in the GOM.”); LGL Ltd., *Environmental Assessment of a Low-Energy Marine Geophysical Survey by the US Geological Survey in the Northwestern Gulf of Mexico*, at 30 (Apr.-May 2013), http://www.nmfs.noaa.gov/pr/pdfs/permits/usgs_gom_ea.pdf (“[T]here has been no specific documentation of TTS let alone permanent hearing damage, i.e., PTS, in free-ranging marine mammals exposed to sequences of airgun pulses during realistic field conditions.”); 75 Fed. Reg. 49,759, 49,795 (Aug. 13, 2010) (issuance of IHA for Chukchi Sea seismic activities (“[T]o date, there is no evidence that serious injury, death, or stranding by marine mammals can occur from exposure to airgun pulses, even in the case of large airgun arrays.”)); MMS, *Draft Programmatic EIS for OCS Oil & Gas Leasing Program, 2007-2012*, at V-64 (Apr. 2007) (citing 2005 NRC Report), http://www.boem.gov/Oil-and-Gas-Energy-Program/Leasing/Five-Year-Program/5and6-ConsultationPreparers-pdf.aspx (MMS agreed with the National Academy of Sciences’ National Research Council that “there are no documented or known population-level effects due to sound,” and “there have been no known instances of injury, mortality, or population level effects on marine mammals from seismic exposure”).

BOEM's findings are further supported by a 2016 report from the National Academy of Sciences, Ocean Studies Board (the "NAS Report"),[16] which makes the following findings regarding marine sound from seismic acoustic sources:

- "The National Research Council report Marine Mammal Populations and Ocean Noise (NRC, 2005) noted that: 'No scientific studies have conclusively demonstrated a link between exposure to sound and adverse effects on a marine mammal population.' <u>That statement is still true</u>…." (NAS Report at 16 (emphasis in original));
- "Evidence of the effects of noise on marine mammal populations is largely circumstantial or conjectural." (*id.* at 28);
- "The probability of marine mammals experiencing PTS [injury] from anthropogenic activities will likely be sufficiently low as to preclude any population-level effects." (*id.* at 35); and
- "Miller et al. (2009) conducted controlled approaches of a commercial seismic survey vessel to make pass-by's of sperm whales in the Gulf of Mexico. The whales, which were exposed to received levels varying from 120-147 dBRMS at ranges varying from 1.4-12.8 km, did not change their direction of travel or behavioral state in response to exposure, but did decrease the energy they put into swimming and showed a trend for reduced foraging." (*id.* at 56).

Consistent with BOEM's GOM-related findings and the NAS Report's findings, there are well-documented examples of long-term exposures of acoustically sensitive species in which no biologically significant chronic or cumulative impacts have occurred. For example, oil and gas seismic exploration activities have been regularly conducted in the Beaufort and Chukchi Seas of the Arctic Ocean for decades, with regular monitoring and reporting to NMFS under the auspices of MMPA incidental take authorizations issued since the early 1990s. During this lengthy period of acoustic exposures, the Arctic bowhead whale population has consistently increased in abundance to the point that it now falls within the range of historical pre-whaling abundance estimates.[17] Similarly, no effects of geophysical activities have been observed in Arctic ice seal populations.[18]

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[16] National Academies of Sciences, Engineering, and Medicine. 2016. Approaches to Understanding the Cumulative Effects of Stressors on Marine Mammals. Washington, DC: The National Academies Press. doi: 10.17226/23479. https://www.nap.edu/download/23479#. A copy of the NAS Report is provided as Appendix B to this letter, for NMFS's consideration and for inclusion in the administrative record.

[17] *See* Muto, M. M., et al. 2016 Stock Assessment Reports, Bowhead Whale (Western Arctic Stock). NOAA-TM-AFSC-355. http://www.nmfs.noaa.gov/pr/sars/pdf/stocks/alaska/2016/ak2016_bowhead.pdf; *see, e.g.*, 84 Fed. Reg. 25,829, 25,834 (May 1, 2012) ("Bowhead whales have continued to travel to the

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This summary accurately reflects the current best available science. NMFS states, in the Proposed ITR, that it is "aware of public statements that there is no scientific evidence that geophysical survey activities have caused adverse consequences to marine mammal stocks or populations, and that there are no known instances of injury to individual marine mammals as a result of such activities." 83 Fed. Reg. at 29,264. However, despite the well-established record, NMFS dismisses these statements (made by BOEM, NAS, NMFS itself, and others) because "conclusive statements regarding population-level consequences of acoustic stressors cannot be made due to insufficient investigation, as such studies are exceedingly difficult to carry out and no appropriate study and reference populations have yet been established." *Id*. Because no such conclusive statements can be made, NMFS premises many of its decisions in the Proposed ITR on the idea that NMFS *must* act "conservatively" because adverse effects—that to date have not been observed over decades of performing, monitoring, and reporting on geophysical activities in the GOM—*could occur in the future* and therefore must be precautionarily assessed and mitigated.

NMFS misconstrues its legal obligations. NMFS is required to objectively use the best available scientific information, applying the standards of Section 101(a)(5), when issuing an ITR under the MMPA. 50 C.F.R. §§ 216.102(a), 216.104(c); *see also* 16 U.S.C. § 1371(a)(3)(A). Although Congress arguably intended the MMPA to conservatively protect marine mammals, it did so *by establishing the Section 101(a)(5) standards themselves* (*e.g.*, "negligible impact")—not by establishing an implied assumption that the MMPA's standards would be applied with an

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eastern Beaufort Sea each summer despite seismic exploration in their summer and autumn range for many years (Richardson *et al.* 1987), and their numbers have increased notably (Allen and Angliss 2010). Bowheads also have been observed over periods of days or weeks in areas ensonified repeatedly by seismic pulses (Richardson *et al.* 1987; Harris *et al.* 2007)."); *id*. at 25,837 ("There is no specific evidence that exposure to pulses of air-gun sound can cause PTS [physical injury] in any marine mammal, even with large arrays of air-guns."); *id*. at 25,838 ("To date, there is no evidence that serious injury, death, or stranding by marine mammals can occur from exposure to air-gun pulses, even in the case of large air-gun arrays."); *id*. at 25,839 ("Thus, the proposed activity is not expected to have any habitat-related effects on prey species that could cause significant or long-term consequences for individual marine mammals or their populations."); 75 Fed. Reg. 49,760, 49,795 (Aug. 13, 2010) ("To date, there is no evidence that serious injury, death or stranding by marine mammals can occur from exposure to air-gun pulses, even in the case of large air-gun arrays.").

[18] *See* Reichmuth, C., Ghoul, A., Sills, J., Rouse, A. and B. Southall. 2016. Low-frequency temporary threshold shift not observed in spotted or ringed seals exposed to single air gun impulses, J. Acoust. Soc. Am., 140: 2646-2658 ("There was no evidence that these single seismic exposures altered hearing – including in the highest exposure condition, which matched previous predictions of temporary threshold shift (TTS) onset…. The absence of observed TTS confirms that regulatory guidelines (based on M-weighting) for single impulse noise exposures are conservative for seals.").

additional layer of precautionary bias in favor of marine mammal protection. *See* H.R. Rep. No. 92-907 (1971), *reprinted in* 1972 U.S.C.C.A.N. 4144, 4148 (stating that Congress "endeavored to build such a conservative bias *into the legislation*" (emphasis added)). The MMPA's standards are therefore already biased in favor of marine mammal protection, and Congress intended for NMFS to apply those standards *objectively*, based upon the best available science.

Accordingly, the law does not allow NMFS to speculate about what the science may or may not demonstrate in the future when it makes decisions. NMFS is required to utilize the best *available* scientific information (even if it is not conclusive)—not the best hypothetical information—objectively applying the standards established by Congress.[19] As described above, the best available science applicable here reflects that there have been no observations of any population-level impacts by seismic survey activities on marine mammal populations in the GOM or in other regions where incidental take has been authorized. This current state of the available science is absolutely relevant to NMFS's consideration of the potential impacts of the activities addressed in the Proposed ITR (which are not significantly different than those that have occurred in the GOM for decades) and any measures that may be required to mitigate the impacts of those activities. By dismissing current scientific findings, and premising decisions on hypothesized future impacts, NMFS violates the MMPA's best available science requirement and contradicts many of its past MMPA determinations for seismic surveys.[20]

C. The Proposed ITR's practicability analyses do not satisfy the regulatory requirement.

The Proposed ITR must describe the "availability and feasibility (economic and technological) of equipment, methods, and manner of conducting such activity or other means of effecting the least practicable adverse impact upon the affected species or stocks...." 50 C.F.R. § 216.104(a)(11); *see* 16 U.S.C. § 1371(a)(5)(A)(i)(II)(aa) (NMFS must specify the methods to achieve the least "practicable" adverse impacts). The Proposed ITR acknowledges that, in evaluating measures to ensure the "least practicable adverse impact," NMFS must carefully

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[19] *See Bldg. Indus. Ass'n of Superior Cal. v. Norton*, 247 F.3d 1241, 1246 (D.C. Cir. 2001) ("the Service must utilize the 'best scientific ... data available,' not the best scientific data possible"); *Blue Water Fishermen's Ass'n v. NMFS*, 226 F. Supp. 2d 330, 338 (D. Mass. 2002) ("[I]mperfections in the available data do not doom any agency conclusion...."); *see also, e.g., Brower v. Evans*, 257 F.3d 1058, 1070-1071 (9th Cir. 2001) ("Scientific findings in marine mammal conservation area are often necessarily made from incomplete or imperfect information.").

[20] The Associations do not contend that the science of seismic sound effects on marine mammals is conclusively established. Science, by definition, is never conclusively established. This is why the Associations and their members spend millions of dollars each year supporting longstanding research into the potential effects of our activities. Rather, our point is that NMFS has effectively required *conclusive* scientific proof that seismic surveys do not impact marine mammal populations and, absent that conclusive proof, will unlawfully persist in hypothetically assuming impacts that have never been observed, contrary to the best available science.

consider the cost, impact on operations, personnel safety, and feasibility and practicality of implementation. 83 Fed. Reg. at 29,264, 29,266. A measure will be considered to have a “higher impact” in terms of practicability if it would “completely impede the operator’s ability to acquire necessary data,” and will be considered “lower impact” if it results in incremental delays that increase operational costs but allow the activity to be conducted. *Id.*at 29,265.

We appreciate NMFS’s recognition of its legal duty to consider the practicability of any proposed mitigation measures and its accurate recitation of the factors relevant to the practicability analysis. We also appreciate NMFS’s inclusion of draft practicability analyses for certain proposed mitigation measures. However, in other key respects, the Proposed ITR does not meet NMFS’s statutory and regulatory obligations because it fails to include practicability analyses for many of the proposed mitigation measures. In addition, the practicability analyses that are provided in the Proposed ITR fail to adequately estimate levels of current and future geophysical work or consider costs and impacts beyond the immediate survey work, ignoring the purpose of the surveys and their critical purpose in the industry.[21] Finally, the analyses provided create false choices between undertaking the proposed mitigation measure or proceeding with no mitigation measure, apparently ignoring mitigation options that may be equally protective but have a lower overall cost and impact, as described below.

1. The Proposed ITR fails to provide a practicability analysis for many mitigation and monitoring measures.

Despite the Proposed ITR’s acknowledgement that a practicability analysis is required, the Proposed ITR contains no practicability analysis for general exclusion zones, shutdown requirements, visual and acoustic monitoring requirements, pre-clearance and ramp-up requirements, power down requirements, entanglement avoidance requirements, vessel strike avoidance requirements, or protected species observer (PSO) eligibility and qualification standards.[22] The Proposed ITR appears to provide practicability analyses only for mitigation measures not included in the Settlement and Stipulations; however, the MMPA’s practicability requirement is not limited to the evaluation of new and novel mitigation measures and, even if it were, the approaching expiration of the Settlement and Stipulations means that NMFS is obligated to consider—and evaluate in the Proposed ITR—the practicability of each of the mitigation measures it seeks to impose through LOAs, both individually and collectively.

Moreover, the Proposed ITR imposes what appear to be “standard” mitigation measures (*e.g.*, visual monitoring, shutdowns, etc.) but applies them in a manner that extends their geographic and temporal scope or to circumstances where they are unnecessary or impossible to implement.

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[21] *See also* Section III.J *infra* (demonstrating that the agency’s RIA erroneously underestimates the costs and overestimates the benefits associated with the Proposed ITR).

[22] Practicability analyses are only provided in the Proposed ITR for the dolphin shutdown and power-down options (83 Fed. Reg. at 29,274), shutdown requirements outside the exclusion and buffer zones for certain species and scenarios (*id.* at 29,276-77), and time/area restrictions (*id.* at 29,279-83).

Without a practicability analysis for each of the proposed mitigation measures, NMFS cannot say that it has "carefully considered" the cost of such measures or how they may impact operations, compromise personnel safety, or be impractical to implement.

To be clear, the Proposed ITR's RIA is not an adequate substitute for conducting a practicability analysis in the Proposed ITR itself. The RIA, prepared under a wholly different legal authority (Executive Order 12866), includes assessments of likely costs but does not evaluate other critical aspects of practicability. In sum, NMFS's failure to consider the practicability of mitigation on an individual and aggregate basis in the Proposed ITR is contrary to the MMPA, NMFS's own regulations, and its own statements in the Proposed ITR that it will "carefully consider" practicability. 83 Fed. Reg. at 29,264-65.

2. The Proposed ITR's practicability analyses fail to consider compounding impacts to the industry and the U.S. economy.

To the extent the Proposed ITR provides practicability analyses, those analyses fail to consider impacts beyond immediate operational impacts, such as how restrictions on geophysical surveys will lead to limitations on the number of wells that can be drilled, thus negatively impacting production, government revenue, gross domestic product, and employment. There are at least 5,350 active leases in the geographic area that would be subject to the Proposed ITR's mitigation measures. And yet, the Proposed ITR's practicability analyses do not consider the likely economic impacts (*i.e.*, lost revenue) that could result from the combination of mitigation measures being proposed. Survey data is essential to allow companies to identify and narrow exploration and production targets, thereby reducing the operational costs and environmental impacts of unnecessary drilling. The final ITR must consider not only the practicability of these measures on a vessel-by-vessel or survey-by-survey basis, but also how the proposed measures—individually and collectively—will impact the industry and economy more broadly across the entire lifetime of exploration and production. As described in Section III.A.2 above, geophysical surveys are essential to the determination of where and when to drill.[23] Regulatory hurdles that delay or prevent surveys from timely occurring can not only compromise a single well but also an entire exploration plan, ultimately placing at risk the successful exploration and production of a reservoir.

3. The Proposed ITR's practicability analyses create a false choice by failing to consider equally protective alternatives.

The Proposed ITR's practicability analyses create false choices between the proffered mitigation and *no* mitigation, deeming the costs warranted in light of the purported impacts to marine mammals in the absence of *any* such mitigation proposal. But determining that a measure is

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[23] Advanced geophysics and re-imaging can lead to continued discoveries in the GOM. *See, e.g.*, https://www.shell.com/media/news-and-media-releases/2018/shell-announces-large-deep-water-discovery-in-gulf-of-mexico.html; https://www.shell.us/about-us/features-and-highlights/shell-confirms-major-deep-water-gulf-of-mexico-discovery-.html.

practicable necessarily involves evaluation of identifiable alternative measures to determine whether they would provide equally sufficient marine mammal protections at a lower overall cost and with fewer operational impacts. Although the Associations do not believe that NMFS is obligated to go in search of novel or obscure measures, the agency cannot ignore obvious alternatives (such as smaller restriction areas) if those measures would provide equally sufficient protections at a lower cost or with fewer impacts to data collection. By considering only the practicability of a proposed mitigation measure against the impacts to marine mammals from no measure at all,[24] the Proposed ITR overlooks potential alternatives that would be less costly, have fewer operational impacts, and avoid personnel safety issues. In its final ITR, NMFS must reconsider its proposed mitigation measures in the context of known, standard, and effective alternatives.

D. Certain mitigation measures in the Proposed ITR are impracticable, unnecessary, and without support.

The best available scientific data and information demonstrate that standard mitigation programs can and do effectively minimize and avoid the incidental take of marine mammals associated with offshore geophysical survey operations.[25] Insofar as we are aware, no seismic activities that

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[24] For example, for deep penetration surveys, the practicability analysis considers requiring passive acoustic monitoring (PAM) at all times in waters greater than 100 meters, but does not identify or evaluate whether requiring PAM only in low visibility conditions or under other more limited conditions would be less costly or otherwise reduce practicability concerns while providing similar benefits. 83 Fed. Reg. at 29,269-70.

[25] A study of more than a decade's worth of marine mammal observation data performed by the Joint Nature Conservation Committee demonstrates that mitigation measures significantly reduce the effects of seismic activities on marine mammals. *See* http://jncc.defra.gov.uk/page-6985; *see also* Mary Jo Barkaszi et al., *Seismic Survey Mitigation Measures and Marine Mammal Observer Reports* (2012); A. Jochens et al., *Sperm Whale Seismic Study in the Gulf of Mexico: Synthesis Report*, at 12 (2008) ("There appeared to be no horizontal avoidance to controlled exposure of seismic airgun sounds by sperm whales in the main SWSS study area."); 78 Fed. Reg. 11,821, 11,827, 11,830 (Feb. 20, 2013) ("it is unlikely that the proposed project [a USGS seismic project] would result in any cases of temporary or permanent hearing impairment, or any significant non-auditory physical or physiological effects"; "The history of coexistence between seismic surveys and baleen whales suggests that brief exposures to sound pulses from any single seismic survey are unlikely to result in prolonged effects."); 79 Fed. Reg. 14,779, 14,789 (Mar. 17, 2014) ("There has been no specific documentation of temporary threshold shift let alone permanent hearing damage[] (i.e., permanent threshold shift, in free ranging marine mammals exposed to sequences of airgun pulses during realistic field conditions."); 79 Fed. Reg. 12,160, 12,166 (Mar. 4, 2014) ("To date, there is no evidence that serious injury, death, or stranding by marine mammals can occur from exposure to air gun pulses, even in the case of large air gun arrays."); 84 Fed. Reg. 25,829, 25,837 (May 1, 2012) ("There is no specific evidence that exposure to pulses of air-gun sound can cause PTS [physical injury] in any marine mammal, even with large arrays of air-guns."); *id*. at 25,838 ("To date, there is no evidence that serious

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have received MMPA incidental take authorizations have caused any impacts beyond a temporary change in behavior for individual animals or any adverse consequences to marine mammal species or stocks. Despite this evidence, which constitutes the best available science, the Proposed ITR would impose mitigation measures that exceed what is warranted to avoid and minimize adverse impacts to marine mammals. As described below, these measures will have significant cost and operational impacts on geophysical surveys and the oil and gas exploration and development these surveys support.

Additionally, as a general matter, the Proposed ITR fails to explain how and why the proposed mitigation and monitoring measures are consistent with, or differ from, the measures required by incidental take authorizations issued for other geophysical surveys, such as those issued to the U.S. Geological Survey (USGS). It appears that the Proposed ITR differs in significant respects from the terms typically included in USGS authorizations, such as exclusion zones, visual monitoring, and acoustic monitoring requirements. The final ITR must provide a rational basis for any departure from comparable established practices.

The following subsections detail the Associations' significant concerns regarding unwarranted, impracticable, and unsafe mitigation measures.

1. The Area 1 four-month restriction is not supported by the best available science and would result in significant economic and operational impacts.

The Proposed ITR would impose a four-month restriction on all seismic surveys in "Area 1," an area shoreward of the 20-meter isobath depicted in Figure 5. 83 Fed. Reg. at 29,307; *see id.* at 29,280 (Fig. 5). The Proposed ITR would also require source vessels to maintain a standoff distance of 13 kilometers from Area 1 (the "buffer area"). *Id.* at 29,307. The Proposed ITR states that the Area 1 closure is intended to "avoid additional stressors to bottlenose dolphin populations during the time period believed to be of greatest importance as a reproductive

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(. . . continued)
injury, death, or stranding by marine mammals can occur from exposure to air-gun pulses, even in the case of large air-gun arrays."); *id.* at 25,839 ("Thus, the proposed activity is not expected to have any habitat-related effects on prey species that could cause significant or long-term consequences for individual marine mammals or their populations."); 75 Fed. Reg. 49,760, 49,795 (Aug. 13, 2010) ("To date, there is no evidence that serious injury, death or stranding by marine mammals can occur from exposure to air-gun pulses, even in the case of large air-gun arrays."); Reichmuth, C., Ghoul, A., Sills, J., Rouse, A. and B. Southall. 2016. Low-frequency temporary threshold shift not observed in spotted or ringed seals exposed to single air gun impulses, J. Acoust. Soc. Am., 140: 2646-2658 ("There was no evidence that these single seismic exposures altered hearing – including in the highest exposure condition, which matched previous predictions of temporary threshold shift (TTS) onset…. The absence of observed TTS confirms that regulatory guidelines (based on M-weighting) for single impulse noise exposures are conservative for seals.").

period." [26] *Id.* at 29,279. However, as explained below, the Area 1 seasonal closure is not supported by the best available science, will increase exposure estimates for other marine mammal stocks, and will have significant adverse economic and operational consequences that are entirely ignored in NMFS's two-sentence practicability assessment.

The genesis of the Area 1 seasonal closure proposal is a term in the Settlement and Stipulations. Although the Associations disagree that this nearshore restriction was appropriate or necessary,[27] the rationale for the restriction was in response to coastal bottlenose dolphin strandings and mortalities (*i.e.*, the northern Gulf of Mexico unusual mortality event (UME)). Additionally, as the Proposed ITR acknowledges, "none of the dolphin strandings or deaths have been attributed to airgun survey activities…." 83 Fed. Reg. at 29,279; *see also* GOM PEIS at 2-13.[28] Instead, recent research demonstrates that seismic impulses at even higher thresholds fail to induce even temporary threshold shifts in dolphin hearing. *See* Finneran et al. (2015). Moreover, there is no evidence that sound from seismic surveys contributes directly or cumulatively to dolphin late-term pregnancy complications or perinatal and postnatal responses that would lead to increased calf mortality or UMEs. *See* Litz et al. (2014); Venn-Watson et al. (2015). The Proposed ITR feebly suggests that the broad understanding that "marine mammals react to underwater noise" is sufficient to impose sweeping seasonal area closures. 83 Fed. Reg. at 29,297. In fact, no relevant scientific evidence supports the restriction of seismic surveys in coastal areas or suggests that

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[26] BOEM proposed a similar coastal restriction in the Application and GOM PEIS, and the Associations commented at length on the lack of scientific evidence supporting this proposed closure and the significant adverse economic impacts that will result. *See* Letter to Jolie Harrison (NMFS) from Nikki Martin (IAGC) et al. re: *Comments on Revised Application for Marine Mammal Incidental Take Regulations for Geophysical Surveys in the Gulf of Mexico* at 24-25 (Jan. 23, 2017); Letter to Hon. Ryan Zinke (Department of the Interior) from Nikki Martin et al. re: *Final Programmatic Environmental Impact Statement for Geological & Geophysical Activities on Gulf of Mexico Outer Continental Shelf* at 11-14 (Aug. 25, 2017).

[27] *See supra* note 7.

[28] NMFS's suggestion that seismic surveys are similar to mid-frequency sonar (which has been implicated in strandings) simply because seismic signatures include a mid-frequency component is inaccurate. 83 Fed. Reg. at 29,236. Mid-frequency sonar is very different than seismic sound and has a tonal, narrow band that is 10 times longer than seismic. Specifically, Navy tactical sonar (AN/SQS-53-C, 56) has its peak sound levels at 2.6 to 8.2 kHz. In this band the sound levels are greater than 220 dB at 1 meter from the source. Conversely, seismic sources are designed to have most (ca. 75%) of their energy output as low frequency, *i.e.*, below 100 Hz. Furthermore, seismic energy in the mid-frequency range drops by 30 dB per decade; in other words, between 100 to 1000 Hz, the dB drops by 30 dB and between 1 kHz to 10 kHz, the dB drops by another 30 dB (*i.e.*, at this point it is cumulatively 60 dB lower). The implication that seismic surveys are similar to mid-frequency sonar is inapt and must be removed from the final ITR.

such a restriction would result in any meaningful benefit to coastal bottlenose dolphin populations, and no contrary evidence or meaningful response is provided in the Proposed ITR.[29]

Critically, an Area 1 closure of any kind would have substantial negative economic and operational consequences that are not considered in NMFS's practicability assessment, which states only that, "[g]iven survey operators' ability to plan around these seasonal restrictions, we believe it is unlikely that the restrictions will affect oil and gas productivity in the GOM." 83 Fed. Reg. at 29,279. There are many unleased blocks within the area covered by the Area 1 seasonal closure. Because existing seismic data in these areas is outdated and inadequate to inform decisions regarding future lease sales, the closure will impede industry's and BOEM's evaluations of blocks for future lease sales. As addressed below, the Area 1 closure would significantly increase the likelihood that seismic surveys will not be completed within a one-year permit term, thereby increasing the overall number of surveys that will need to be conducted, increasing costs, and decreasing overall efficiency.

Survey effort is unlikely to simply shift to other months in which the coastal areas are available for exploration. The enormous, mostly unexplored area covered by the Area 1 closure requires certain specialized surveys: full azimuth, long offset, and deep data seismic. The coastal offshore areas of Louisiana and Eastern Texas, in particular, require very specialized equipment: light ocean bottom nodes and ocean bottom cables.[30] Regular marine streamer crews will not be able to collect sufficient data or achieve the required spatial sampling to adequately image the targeted section. Specialized node and ocean bottom cable crews are not designed for deeper, open-water exploration and must be used in coastal areas. Moreover, the vessels used in shallow water are often smaller and have shallower vessel drafts. Such vessels cannot be taken easily or safely into deep open-water environments. In short, the specialized operations required for the areas covered by the Area 1 closure cannot simply be shifted to other areas that do not require the same specialized operations.

Additionally, modern seismic imaging requires an entire aperture to be recorded before imaging can be performed. Essentially, all data for a particular data project must be gathered before the final steps are performed to create the data image. This means that, when surveys are terminated early as a result of the four-month restriction, data collected will not be usable until the crew is

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[29] There are no data to suggest that seismic-generated sound negatively impacts the bottlenose dolphin population in general or the mother-calf pairs in particular, and it is equally, if not more, plausible that the animals are completely unaffected by the sound. The fact that these populations may be affected by coastal pollution, vessel traffic in the estuaries, or endemic diseases is not a basis for restricting an activity that has no demonstrated adverse effect.

[30] Based on the limited information that is available, it is likely that coastal areas offshore Louisiana and East Texas contain very large quantities of natural gas. For example, just one prospect indicates recoverable reserves exceeding 1 trillion cubic feet. *See* http://www.offshore-mag.com/articles/print/volume-70/issue-6/Gulf_of_Mexico/davy-jones-a-new-era-for-gom-shelf-exploration.html. This is an area with significant potential, with infrastructure in place both to bring the gas onshore and to distribute it around the country.

able to return to complete the survey. Moreover, even if crews are able to move to locations outside of the closure area (which will be difficult for the reasons stated above), it is very unlikely that those projects will last for exactly four months, which means that the delays to surveys in Area 1 are likely to last for much longer than four months, not including the substantial time required for mobilization and demobilization. Thus, the delay to actually obtain a data image from a survey that is interrupted because of the four-month closure could be six months or more.

Moreover, the four closed months are the most operationally productive months in the GOM because poor winter conditions (including higher sea states and unpredictable wind patterns) have ended and the summer tropical storms have not yet begun. Accordingly, the cost to operate in Area 1 will be substantially higher than other areas and result in increased and inefficient survey effort overall,[31] as well as increasing safety concerns due to adverse weather and ocean conditions. These issues will discourage interest and the ability to identify prospects in coastal areas, undermining efforts that BOEM has taken to incentivize shallow-water GOM prospects, such as lowering the royalty rate for shallow-water production in an effort to improve the economic case for drilling in those areas.[32]

The MMPA's practicability requirement, reflected in NMFS's regulations and described in the Proposed ITR, necessarily must begin with an accurate depiction of the relevant details of the specified activity. The Associations have commented on a proposed coastal closure numerous times, including to NMFS with respect to the Application.[33] These comments have demonstrated that a seasonal closure would impose unreasonable substantial costs and operational burdens, with broad consequences, on the oil and gas industry. The Proposed ITR lacks any analysis of these factors, concluding that operators would simply "plan around" the closure. Before such a closure can be adopted, the MMPA requires NMFS to undertake a robust practicability analysis that appropriately considers the operational impact—as described in the Associations' numerous comments on this issue—of the seasonal closure.

For the reasons described above, the proposed Area 1 closure is not supported by the best available science, will not benefit marine mammals, will result in overall increased survey effort at a much higher cost to operators and with a corresponding increase in safety concerns, and will hamper the ability of the U.S. to develop nationally strategic natural gas reserves contrary to established federal policy. The final ITR should include a practicability analysis that considers the costs and impacts to the seismic and larger oil and gas industry, and should conclude that an Area 1 closure is not warranted given the lack of benefit to marine mammals and significant

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[31] Based on calculations from one of our member companies, the cost of shutting down a single crew for the proposed four-month closure season could be in the range of $7 million. Based on those same calculations, lost revenues due to operating around a four-month closure over a 10-year period could range from $3 million to $9 million.

[32] *See* https://www.boem.gov/note07062017/.

[33] *See supra* note 26.

practicability concerns. Should the final ITR include closures of any kind, it should also provide for multi-year LOAs that cover the duration of the specified survey work to avoid requiring multiple LOAs for the same survey.

2. The Area 3 restrictions are not supported by the best available science and would impose significant operational limits and costs.

The Proposed ITR would impose a three-month restriction on all seismic surveys in "Area 3," which is an area bounded by the 100- and 400-meter isobaths in the eastern GOM, and a buffer area of 6 kilometers from Area 3. 83 Fed. Reg. at 29,307; *id.* at 29,280 (Fig. 5). The Proposed ITR also requests comments on whether Area 3 should be closed to such surveys year-round instead of seasonally, or not at all. *Id.* at 29,281. The Proposed ITR also requests comments on whether to require BOEM or members of the oil and gas industry to provide real-time Bryde's whale detection through use of a moored listening array to initiate shutdowns when whales are within 6 kilometers. *Id.*

Neither the three-month restriction nor the year-round closure in Area 3 is supported by the best available science. The Area 3 closure is intended to ensure that areas of expected importance to Bryde's whales are not ensonified even once under any circumstances by levels of sound above 160 dB rms SPL, a level of sound exposure for which even repeated exposures at the specified level are unlikely to produce adverse consequences.[34] NMFS states that it also expects the broader exclusion zone to "be helpful" at reducing the severity of behavioral responses at given distances. 83 Fed. Reg. at 29,271 (citing Ellison et al. (2012)). However, the idea that severity scales with distance is largely hypothetical and not supported by data showing more severe responses closer to the sound than further from it. The Associations are aware of no evidence that mother-calf separation occurs in the presence of noise, or of any reported scaled response as a source was nearer or farther away.[35] NMFS may not impose significant new mitigation measures where there is simply no data indicating that the measures are needed to avoid an adverse effect.

The Proposed ITR also posits that Bryde's whales may have once used more of the GOM but have abandoned those other areas due to energy exploration and production activities. 83 Fed. Reg. at 29,280. This speculation does not constitute the best available science. Neither whaling records nor historical or recent stock assessment data suggest any such change in the Bryde's whale's range, nor are there other areas of the world where there is evidence that oil and gas activities have caused whales to abandon habitat. There is no real evidence that a seasonal or

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[34] The Proposed ITR itself states that distances for exceedance of group-specific peak injury thresholds are 65 meters (low frequency), 18 meters (medium frequency) and 457 meters (high frequency). 83 Fed. Reg. at 29,272.

[35] The language in Ellison et al. (2012) illustrates this point, noting that "if the sound were encountered often enough it *might* lead to mother-calf separation" and "if the animal was closer to the source it *might* have reacted more strongly." (Emphases added.)

year-round closure will benefit Bryde's whales in any way.[36] Furthermore, as with the proposed Area 1 closure, a seasonal or year-round closure of Area 3 could result in higher exposure numbers for marine mammal stocks outside the closed area. *See supra* Section III.D.2.

In addition to relying on questionable science, the Proposed ITR's Area 3 practicability analysis is inadequate. With regard to potential impacts on the oil and gas industry, the analysis states that the GOM Energy Security Act moratorium on leasing activity in the eastern GOM will continue for most of the five-year ITR period, and that there are only two active leases in Area 3, each of which would be exempted from the Area 3 closure requirements. 83 Fed. Reg. at 29,281. As NMFS is well aware, however, a temporary moratorium on leasing does not limit survey work and exploration planning. The Proposed ITR's delay of such surveys would have direct impacts on the industry's ability to prepare for leasing that is expected to occur once the moratorium is lifted.

The ITR also repeats BOEM's speculation that there will be "very low activity levels" in Area 3 over the next 10 years. *Id.*; *see also id.* at Table 3. This information is outdated and must be updated in the final ITR. In fact, an Area 3 closure of any kind would have substantial negative economic and operational consequences. The Associations are aware that companies are keenly interested in potential for development in Area 3. The bid round for the March 2018 lease sale saw several leases in the Central GOM being acquired close to Area 3, supporting a conclusion that there is likely to be much greater interest in this area for surveys than BOEM anticipated.[37] A precedential year-round closure would have significant consequences, effectively closing this area to all oil and gas activity for the foreseeable future contrary to U.S. energy security priorities and resulting in substantial economic and operational impacts.

An annual three-month closure of Area 3, as proposed in the regulatory text, would also have serious negative economic and operational consequences that NMFS must consider in its practicability analysis. The geophysical data in Area 3 are outdated and inadequate to inform decisions regarding exploration and production planning and decision-making. A three-month closure each year would impede both the industry's and BOEM's ability to evaluate future lease sales. A seasonal closure would significantly increase the likelihood that seismic surveys will be terminated early as a result of the three-month restriction, thereby increasing the overall number of surveys that will need to be conducted, increasing costs, and decreasing overall efficiency. In addition, because all data must be gathered before final steps are performed to create the data image, data collected would not be usable until the crew is able to return to complete the survey. Limitations on the ability to create a full data image are likely to discourage interest and the ability to identify prospects within Area 3.

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[36] Appendix C demonstrates why, at a minimum, the proposed Bryde's whale area closure should be reduced, consistent with the best available science and with no meaningful benefit to the species.

[37] *See* https://www.boem.gov/Sale-250-Bid-Distribution-Map/. Active leases can be viewed at https://www.boem.gov/Gulf-of-Mexico-Region-Lease-Map/.

Finally, NMFS requests comments on its proposal to require use of a moored listening array in Area 3 to provide real-time detection of Bryde's whales. 83 Fed. Reg. at 29,281. NMFS explains that it "do[es] not consider towed passive acoustic monitoring to be sufficient to ensure detection of the Bryde's whale," but provides no explanation for this conclusion, making it impossible to meaningfully comment on whether such measures are warranted and based on sound science. *Id*. Moreover, NMFS fails to discuss towed PAM improvements that might address any such concerns, such as modifying the depth or distance of the array from vessels. In fact, the reliability and success of a moored listening array would be limited by ambient noise and the complexity of Bryde's whale vocalizations as well as such practical considerations as recording capacity, maintenance, and the significant logistical challenges and costs of retrieving data "real time." Because of these challenges and limits on the usefulness of any data collected, a moored array is simply not practicable.

In sum, the final ITR should impose no Area 3 restrictions. However, if restrictions are imposed, seasonal closures will have significant impacts but are preferable to a year-round closure. The area of any such closure should be reduced, as described in Appendix C, and the final ITR should provide for multi-year LOAs that cover the duration of the specified survey work to avoid requiring multiple LOAs for the same survey. Additionally, the final ITR should not impose a moored array requirement because the limits inherent in such data are outweighed by the impracticability of such arrays.

3. The proposed Area 4 year-round closure is not based on science and would have significant economic and operation impacts that have not been considered.[38]

The Proposed ITR would impose a year-round restriction on seismic surveys in "Area 4," which is an area bounded by the 200- and 2,000-meter isobaths in the southeastern GOM, and a buffer area of 9 kilometers from Area 4. 83 Fed. Reg. at 29,307; *id.* at 29,280 (Fig. 5). The stated purpose of this closure is to benefit sperm and beaked whales based on "very dense" sightings and the possibility that sperm whales may use this area for calving. *Id.* at 29,281. However,

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[38] The Associations agree with NMFS's conclusion that a closure in the Central Planning Area is unwarranted. Among other reasons, a recently published study found that sperm whale vulnerability to seismic sound in the Gulf of Mexico is low, with no horizontal response movement to the presence of an active seismic array. Winsor, M.H., L.M. Irvine, and B.R. Mate. 2017. Analysis of the Spatial Distribution of Satellite-Tagged Sperm Whales (*Physeter macrocephalus*) in Close Proximity to Seismic Surveys in the Gulf of Mexico. Journal of Aquatic Mammals, Vol. 43, pages 439-446. Additionally, NMFS states that Miller et al. (2009) concluded that exposure to sound from seismic sources could impact sperm whale foraging behavior. However, Miller et al. (2009) found that none of the eight sperm whales in their study in the GOM changed the whales' behavioral state (seven foraging, one resting) when exposed to one to two hours of seismic sound.

NMFS relies on personal communications for these assertions and does not present data showing that Area 4 is unusually significant to sperm or beaked whales. NMFS also cites unspecified "additional benefits" to other marine mammal species, but such general arguments that are not based in science cannot be used to support a year-round closure and should be removed from the final ITR. *See id*. at 29,282.

An Area 4 closure would result in operational and economic impacts that NMFS has not considered. Specifically, NMFS states that BOEM has projected no survey activity in this area over the next 10 years. *Id.* at 29,282. In fact, however, the Associations are aware of significant interest among our members in the potential exploration and development of the Eastern GOM. The precedential closure of Area 4 would prevent such exploration and effectively limit the ultimate development of this area for many years, contrary to U.S. energy security priorities. For these reasons, the final ITR should not include an Area 4 closure.

4. Buffer areas are not supported by the best available science and will unduly restrict operations.

As noted above, the Proposed ITR's Area 1, 3, and 4 restriction measures would include "buffer areas" of 13, 6, and 9 kilometers from the subject areas, respectively. 83 Fed. Reg. at 29,307. The Proposed ITR would require source vessels to remain outside of these areas, effectively closing not just Areas 1, 3, and 4, but larger areas around them as well. None of these buffer areas are substantively evaluated or described in the Proposed ITR, nor are they depicted in Figure 5. The Proposed ITR does not describe the basis for the specific distances chosen, including why they differ, in order to facilitate meaningful public review and comment. *See id.* at 29,279 (describing Area 1 and then stating, cryptically, "buffered by 13 km (Mathews *et al.*, 2016)"); *id.* at 29,281 (stating only: "The designated area [Area 3] would then be buffered by 6 km."); *id.* at 29,282 (stating only: "The defined area [Area 4] would be buffered by 9 km (rounded up from the 8.4 km distance provided by Matthews *et al.* (2016)[39] for the Dry Tortugas area).").

Furthermore, the practicability analyses related to the area closures do not discuss the impact of requiring vessels to maintain this standoff area in addition to the impacts of the proposed closures themselves. Every one of the significant economic and operational impacts described above for the Area 1, 3, and 4 closure areas would be exacerbated by the addition of these buffer zones, which would increase the areas off limits to survey work during any closure times. The Proposed ITR fails to consider these impacts and provides no biological basis for imposing them. Accordingly, the final ITR must eliminate buffer areas from further consideration.

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[39] NMFS's materials do not include Matthews et al. 2016, and neither Matthews et al. 2015 nor Matthews et al. 2017 describe this 8.4 kilometer distance. *See* Appendices C and D for additional comments.

5. The Proposed ITR's combined visual and acoustic observation requirements compromise personnel safety, cannot be effectively implemented, and are unnecessary and unsupported.

The Proposed ITR would require a minimum of two PSOs conducting visual observations at all times during daylight hours and 30 minutes prior to and during nighttime airgun array ramp-ups. 83 Fed. Reg. at 29,305. It would require another PSO to monitor the towed PAM system at least 30 minutes prior to ramp-up and at all times during active survey work. *Id.* In addition, the Proposed ITR would prescribe limits on the duration that PSOs can be on duty. Visual PSOs would be limited to two hours of watch time followed by a one-hour break, and no more than 12 hours of observation in a 24-hour period. *Id.* Acoustic PSOs would be limited to a maximum of four consecutive hours of monitoring followed by a two-hour break, and no more than 12 hours of observation in a 24-hour period. *Id.* at 29,306.

The combination of these proposed PSO requirements and watch limitations means that each survey vessel would be required to carry between *six and eight PSOs* on board at all times. This represents an approximate 30% increase over current practice. Adding that many personnel to every survey vessel raises serious safety exposure and logistical concerns. Critically, as the number of people increase, so does the risk of injuries, illnesses, and evacuation for medical reasons, increasing the cost of these activities many fold, reducing safety of all personnel, and hindering operations.

Moreover, many vessels are space-limited, and will not have the ability to accommodate up to eight PSOs in addition to critical personnel to ensure safe operations, making this requirement technically infeasible for those vessels. For example, smaller 2D and 3D vessels tend to have between 45 and 60 beds. A number of these may be in four-bed cabins, however, and requirements under the Maritime Labour Convention of 2006 and industry best practice limit the use of four-bed cabins in a number of situations to two people, reducing the total beds available by four to eight beds. Such a vessel may have approximately 13 to 15 marine crew, 25 to 30 seismic crew, and three to seven client representatives. This type of vessel may already be at maximum capacity with three PSOs. Similarly, source-only vessels tend to be older vessels with fewer than 50 beds, including a number of four-person cabins that now accommodate just two people. With approximately 13 to 15 marine crew, 20 to 23 seismic crew, and one to three client representatives, these vessels may already be required to reduce necessary crew to accommodate three PSOs.[40]

In these situations, the addition of another three to five PSOs beyond what is normally required could force a reduction in necessary marine, client, or seismic crew, putting management of the vessel and equipment at risk and increasing the time to acquire necessary data by 15% to 20%.

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[40] Even larger vessels such as X-bow seismic and C-class vessels are space limited and may be at full capacity with the average crew of approximately 56, four engineers (in single cabins), and the usual PSO staff.

Meeting the proposed PSO requirements could therefore materially increase the cost of survey activities and extend the time period during which marine mammals would be exposed to the acoustic source. The Proposed ITR considers none of these costs or the technical infeasibility of adding six to eight people to every survey vessel, because it does not analyze the practicability of this mitigation measure, contrary to applicable law.

In addition to the sheer number of PSOs being impracticable, the Proposed ITR's requirement for visual monitoring during nighttime ramp-ups is fundamentally flawed and inconsistent with the Proposed ITR's own conclusions. Specifically, the Proposed ITR would require that visual PSOs be on duty and conducting visual observations 30 minutes prior to and during nighttime ramp-ups. However, visibility is significantly reduced due to night lighting required under U.S. and international maritime law, reducing the effectiveness of visual observations at night. As the Proposed ITR itself notes, "there can be no expectation that any animal would be detected at night" using visual monitoring. 83 Fed. Reg. at 29,267; *see also id.* ("visual monitoring is only effective during periods of good visibility and when animals are available for detection"). Additionally, the Proposed ITR concludes that PAM is "an effective detection system, supplanting visual monitoring during periods of poor visibility." *Id.* The Proposed ITR provides no rationale for requiring visual monitoring at night when there is "no expectation" that it will be effective and when PAM is an effective detection system that "supplant[s] visual monitoring" during low visibility, such as at night. *See id.* To avoid imposing a measure that increases costs and requires additional PSO duty hours without benefitting marine mammals, the final ITR must remove this requirement.

To ensure the final ITR is practicable and avoids unnecessarily prolonging survey work, it must include visual and acoustic PSO requirements that add no more than three to four non-crew personnel to each survey vessel, consistent with current practice. The requirement should be sufficiently flexible to provide for situations in which smaller vessels can only accommodate three PSOs, but allow for four PSOs where possible. In addition, the final ITR should require visual PSOs during daylight surveys only, consistent with the Proposed ITR's finding that PAM is effective (and visual monitoring is not) for detecting marine mammals during periods of poor visibility.[41]

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[41] Relatedly, NMFS provides no justification (or practicability analysis) for the proposed requirement that operators using ocean-bottom nodes (OBN) employ a PSO for the sole purpose of documenting entanglements with the OBN cable. *See* 83 Fed. Reg. at 29,397. First, many OBN surveys do not use cables at all. Second, entanglements during OBN surveys that use cables are *extremely rare* and must be reported by the operator. Third, OBN surveys may not necessarily use "negatively buoyant coated wire-core tether cable," the practicability of which is entirely unexplained. *Id*. These OBN-related requirements must be removed altogether from the final ITR.

6. The proposed combined exclusion and buffer zones for pre-clearance are excessive and not supported by the best available science, and would result in delay and other operational impacts.

For deep penetration surveys, the Proposed ITR would impose a 500-meter buffer zone in addition to the 500-meter exclusion zone and would require PSO monitoring for 30 minutes prior to ramp-up to confirm that no marine mammals are observed in the entire 1,000 meter zone before ramp-up may begin. 83 Fed. Reg. at 29,306. For shallow penetration and non-airgun surveys, the exclusion zone would be 200 meters but would also include another 200-meter buffer to be monitored during pre-clearance. *Id.* at 29,307.

There is no scientific basis for monitoring a zone larger than the exclusion zones, which are already precautionary based on the best available science developed over a decade of observations and experience. Moreover, the additional buffer zones would result in increased risk of delays for all survey operations, and in particular would result in significant delays in ramp-up during deep penetration surveys. This, in turn, would result in surveys taking longer periods to complete, which would increase costs and the risk to personnel safety. The final ITR should eliminate this requirement, which would create delays that extend survey work and increase the overall exposure of marine mammal populations to seismic surveys without providing any known benefit to those same populations.

7. Non-airgun high-resolution geophysical surveys should not be subjected to pre-clearance and shutdown requirements.

For non-airgun, high-resolution geophysical (HRG) surveys, the 200-meter exclusion zone for shutdowns and the 400-meter buffer zone pre-clearance requirement is excessive, unnecessary, and impracticable. Although the sound profile of HRG equipment can vary considerably, the acoustic footprint of most surveys will simply be too small to warrant pre-clearance requirements at 400 meters or shutdown at 200 meters.[42] Animals observed at the surface are generally outside the beam and not receiving sound, and there is no evidence or reason to believe they would be deterred from approaching geophysical survey vessels any differently than any other moving vessel in the GOM. Indeed, the federal government's own HRG survey work is conducted

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[42] For example, multibeam echosounders (MBES) or sub-bottom profilers beam patterns can be very narrow and directed, and energy levels are often low enough to make Level A take of marine mammals highly unlikely. Level B exposures, as estimated with models, can be very low for this type of equipment as well, and animals observed at the surface, even next to the vessel, may not be in an exposure zone at all. A Kongsberg EM 302 MBES has a beam pattern for which very little sound energy propagates near the surface, and the depth at which a Level B harassment exposure could occur (based on thresholds) becomes greater as the sound travels farther from the vessel. The National Ocean Service (NOS) (2013) has determined that the acoustic energy of echosounders is limited by the downward-facing beam, particularly for single-beam echosounders, which NOS states have beam widths that would "be barely noticeable among a background of standard depth sounders found on almost all small and large vessels."

without pre-clearance or shutdown requirements, demonstrating that such measures are not necessary or appropriate.[43]

In addition, although the Proposed ITR appears to recognize that HRG surveys are "sometimes conducted using autonomous underwater vehicles (AUV) equipped with multiple acoustic sources,"[44] it appears to apply many of the mitigation measures to such surveys without any consideration of the relevance or feasibility of those mitigation measures. For example, because AUVs typically run 30 to 40 meters above the seafloor, a visual PSO will not be able to make species observations effectively. The final ITR should clarify that visual monitoring and associated exclusion, ramp-up, and shutdown requirements do not apply to HRG surveys conducted using AUVs.

Finally, the proposed non-airgun HRG measures present serious safety and cost concerns that have not been fully vetted. For example, as explained above, adding PSOs to the smaller vessels used for non-airgun HRG surveys in waters deeper than 200 meters will be challenging, increase costs an estimated 5% to 20%, and present safety risks due to having more people onboard these smaller vessels. Unlike larger seismic surveys, these HRG surveys can occur as frequently as monthly, compounding the increased expense and resulting in millions of dollars of added cost. Because these measures are not expected to reduce exposures in any meaningful way, the resulting delays and associated costs are overly burdensome and impracticable. The final ITR should exempt HRG surveys from pre-clearance and shutdown requirements.[45]

8. The final ITR should not require shutdowns or power-downs for dolphins of any size.

The Proposed ITR would impose shutdown requirements when marine mammals are visually or acoustically detected in the exclusion area.[46] 83 Fed. Reg. at 29,306. However, NMFS requests

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[43] HRG equipment is commonly used by NOS, NMFS, and USGS. In LOAs for NMFS science centers, there are no clearance or shutdown requirements for using this equipment (*e.g.*, active LOAs for Northeast Fisheries Science Center issued 2016 and Southwest Fisheries Science Center issued 2015). The government's policies regarding its own use of this type of equipment suggest that clearance and shutdown requirements are not considered necessary during government activities that employ HRG equipment.

[44] 83 Fed. Reg. at 29,219; *see also id.* at 29,220 (referencing AUVs in two places).

[45] HRG in the GOM is highly variable across survey types, locations, and operators. *See* https://www.boem.gov/High-Resolution-Geophysical-Survey-Application/.

[46] The Associations note that the proposed regulatory text is unnecessarily confusing with regard to whether general shutdown requirements are triggered by acoustic detection within an exclusion zone, because no zone is specified. Specifically, the text states: "If a marine mammal (excluding delphinids) is detected acoustically, the acoustic source must be shut down." Proposed ITR at 29,306 (proposed section (8)(ii)). Based on the description of the shutdown requirements in earlier sections of the Proposed ITR, the Associations understand that NMFS

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comments on two possible exceptions to the shutdown criteria. Under the first proposal, the acoustic source would be powered down to the smallest single element of the array when a small dolphin enters the exclusion zone and remain powered down until it is seen leaving the exclusion zone or 15 minutes after the last observation. *Id.* at 29,273. Under the second proposal, there would be no shutdown or power-down required for small dolphins entering the exclusion zone. *Id.*

The best available science does not support imposing shutdown or power-down requirements in the event a dolphin enters the exclusion zone, regardless of its size. Although the Associations agree that small dolphins are more likely to bow-ride than large dolphins, historic PSO reports indicate that dolphins transiting survey vessels at full power do not exhibit behavior that would indicate a disturbance, regardless of their size.[47] Imposing a shutdown for large dolphins is just as unnecessary and presents the same practicability concerns as the Proposed ITR describes with respect to small dolphins. There is also no evidence that larger dolphins will benefit from a shutdown requirement.[48]

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means to limit this requirement to when a marine mammal (other than delphinids) is detected acoustically *within the applicable exclusion zone*, except as provided in the sections regarding detection of baleen whale, beaked whale, *Kogia* spp., or large whales with calves. *See id.* at 29,271 ("PSOs must establish a minimum exclusion zone with a 500-m radius…."); *id.* at 29,268 ("when we use 'PSO' without a qualifier, the term refers to either visual PSOs or PAM operators (acoustic PSOs)"). The Associations request that NMFS modify this text to clarify that, except as provided in the subsections that follow the statement, shutdown is only required upon acoustic detection in an applicable exclusion zone.

[47] *See* Schlundt et al., 2013, *Auditory effects of Multiple Impulses from a Seismic Air Gun on Bottlenose Dolphins (Tursiops truncates)* ("Bottlenose dolphins exposed to impulses from seismic airguns show that the potential for seismic surveys using air guns to cause auditory effects are lower than previously predicted. No injury took place and no significant behavioral reaction was observed. Dolphins may show little reaction to airgun impulses, even at range as close as 3.9 m and with the air gun operating at 150^3 and 200 psi (cumulative Sound Exposure Levels (SELs) of 189-195 db re $1upa^2s$); these sound levels did <u>not</u> produce clear, reliable TTS in any dolphins tested."). It is also important to note that the Proposed ITR's reference to Gray and Van Waerebeek (2010) is misleading. In that study, a pantropical spotted dolphin was reportedly observed spyhopping or vigorously keeping its body out of the water—a behavior never seen before or since. No conclusions can be reasonably drawn about acoustic surveys from this observation, which constitutes an unverified anecdote of suspect origin more than scientific evidence. In that case, a dolphin was reported to keep its body above the water for five minutes continuously, something that is highly improbable physically. This aberration was attributed to a seismic source 600 meters away, but no similar behavior has ever been observed from this or other species at ranges comparable to or even closer to an array than the reported event.

[48] The Proposed ITR states that including large dolphins in the shutdown requirement will "simplif[y] somewhat the total array of decision-making for PSOs." 83 Fed. Reg. at 29,274. In

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Powering down for dolphin presence is operationally difficult and commercially devastating, and would only serve to delay and prolong survey work. The Proposed ITR states that, based on vessel speeds of 4.5 knots, NMFS expects operators will not need to reshoot to infill any gap in data acquisition lost during a 15-minute power-down. 83 Fed. Reg. at 29,273. This assumes, however, that any power-down will be no more than 15 minutes because either (1) dolphins that enter the zone to bow-ride will appear only once, or (2) dolphins will bow-ride for a few minutes and then immediately and obviously exit the exclusion zone. It is far more likely that dolphins will move in and out of the exclusion zone repeatedly, resulting in repeated power-downs over extended periods that seriously interfere with a vessel's ability to collect the necessary data.

Historical PSO and PAM data from over 32,000 survey activity hours conducted in the GOM between 2007 and 2017 indicate that dolphins spent a total of 5,312 minutes in the exclusion zone, with the duration ranging from less than one minute to 350 minutes.[49] Dolphins (including one sighting of a false killer whale and one mixed-species sighting of pygmy killer whales) were observed to bow-ride from less than one minute to up to 106 minutes, averaging 18 minutes. The Proposed ITR's assumptions are not consistent with these data, which represent the best available science. In fact, using the same calculation methods as the RIA at Exhibit 4-5, power-downs for visual observations of small dolphins would equate to between 0.4 to 15 additional days to a WAZ survey,[50] generating between $316,000 to $13 million in additional survey costs, including an estimated $8,000 to $300,000 in PSO and equipment costs. Requiring power-downs for visual observations of large and unidentified dolphins would add approximately six survey days,[51] generating an estimated $5 million in additional survey costs, including an estimated $120,000 in PSO and equipment costs. In summary, mitigation for dolphins observed visually in the exclusion zone would add 20 days to a WAZ survey, generating up to $18 million in survey costs, including more than $420,000 in monitoring costs. This cost is three to 16 times greater

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fact, PSOs are required to make decisions regarding species identification at the genus level regardless of whether large dolphins are included in shutdown requirements; excluding them will not make a PSO's task any more complicated.

[49] Small dolphins were visually observed in the exclusion zone 2.18% of the total activity survey days, large dolphins were visually observed for 2.76% of the total activity days, and unidentified dolphins were visually observed for 2.76% of the total activity days.

[50] This estimate assumes that, at an average vessel speed of 4.5 knots (or 8.3 kilometers per hour), a downtime period of up to .3 hours (18 minutes) will create a data gap of up to approximately 2.5 kilometers and a small number of these gaps would likely be tolerable, while greater gaps would not be. However, this is dependent on the survey type, the cumulative downtime, and the operator's terms for the percentage of tolerable data gaps.

[51] This estimate assumes that, at an average vessel speed of 4.5 knots, any downtime period of over .82 hours would likely require infill, depending on the survey type, cumulative downtime, and operator's terms for the percentage of tolerable data gaps.

than the RIA predicts. This does not include the acoustic detection of dolphins in the exclusion zone for 57.65% of survey activity days.

Maintaining full survey capability when dolphins enter the exclusion zone would allow PSOs to record valuable behavioral data to allow confirmation of the absence of impacts over time as part of the long-term monitoring plan. In any event, because even a power-down requirement has significant operational impacts that are costly and cause substantial delay, and because the best science available today indicates that small and large dolphins are not adversely impacted by these operations, the final ITR should impose no shutdown or power-down requirements for small or large dolphins.

9. Proposed shutdown requirements for other species would be ineffective and impracticable, and are not supported by the best available science.

The Proposed ITR also includes proposals to shut down an acoustic source at any distance (proposal 1) or within 1 kilometer of the source (proposal 2) in the event of visual or acoustic observation of a baleen whale, beaked whale, or *Kogia spp.*, or upon visual observation of a large whale with calf. 83 Fed. Reg. at 29,306-07. The Proposed ITR states that additional shutdown protections are important to reduce effects to these species, but the requirements proposed by NMFS exceed its authority. Requiring shutdowns "at any distance" is arbitrary and unlawful on its face because it contemplates shutdowns in circumstances in which no disturbance or harassment will occur. There is no reasonable basis in the MMPA for requiring surveys to stop when whales are acoustically detected beyond the point at which they may experience adverse impacts.

In addition to this fatal legal shortcoming, detecting these species "at any distance" will be difficult or impossible to implement effectively.[52] Visual observation beyond 1 kilometer is unlikely to be successful unless environmental conditions (sea state and glare) are ideal, which is not generally the case. PSOs typically cannot confidently visually identify beaked whales as close as 400 meters away, and it will be difficult to determine the presence of calves from a distance. Consequently, PSOs are likely to make frequent "precautionary" shutdown calls for uncertain observations "at any distance." NMFS has recognized that such circumstances "simply displace seismic activity in time and increase the total duration of acoustic influence as well as total sound energy in the water." 82 Fed. Reg. 26,244, 26,254 (June 6, 2017).

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[52] NMFS should also consider that requiring PSOs to monitor beyond the exclusion zones will cause implementation problems because observers are only required to monitor out to 1 kilometer during deep penetration survey pre-clearance and ramp-ups, and otherwise are focused on 500 meters or closer. If an observer is required to monitor beyond the exclusion zone—indeed, to monitor *everywhere* under proposal 1—that long-distance monitoring will almost certainly undermine the effectiveness of their monitoring of the 500-meter exclusion zone during operations.

Similarly, acoustic detection distances will vary depending on a variety of parameters, including aspects of the PAM equipment itself, environmental conditions, and animal-related variables such as call frequency, call rate, and the direction the animal is facing relative to the PAM system, among other things. Proposal 1, in particular, would result in an inordinate number of shutdowns without benefiting marine mammals. However, because both the visual and acoustic monitoring is significantly less effective at distances beyond 500 meters, both proposals would lead to unnecessary shutdowns that prolong overall survey duration, increase risks to personnel and increase costs significantly. NMFS lacks sufficient evidence and support to justify either requirement. The well-established 500-meter shutdown distance should be applied to all species subject to shutdowns for both visual and acoustic observations.

10. Vessel strike restrictions are not supported by existing data.

The Proposed ITR would require that all vessels observe a 10 knot restriction in Area 3 or when mother-calf pairs, pods, or large assemblages of whales are observed. 83 Fed. Reg. at 29,307. It would require vessels to maintain distances of 500 yards from baleen whales, 100 yards from sperm whales, and 50 yards from all other marine mammals (except those that approach the vessel). *Id.* In addition, except when a vessel is towing gear, the Proposed ITR would require moving vessels to reduce speed and shift to neutral if a whale is sighted in close proximity. *Id.*

The Associations are aware of no whale strike event ever occurring in the course of vessels conducting or supporting seismic surveys in the GOM. Nevertheless, the Proposed ITR would require vessels to observe these speed and minimum separation restrictions without evidence that these mitigation measures will benefit marine mammals. This is contrary to the MMPA's requirement to utilize the best available science, which indisputably demonstrates that geophysical survey vessels do not strike marine mammals. Indeed, the Associations are aware of no other vessels in the GOM subject to such restrictions.

In addition to these general objections, there are practical concerns with these mitigation measures that must be addressed in the final ITR, if these measures persist. Specifically, the Proposed ITR would exempt vessels towing gear from the requirement to shift to neutral, appropriately recognizing safety concerns associated with reducing speed quickly. *Id.* On those same grounds, the final ITR should extend the exemption to *all* separation and avoidance requirements to avoid requiring vessels towing gear to move abruptly in a manner that could jeopardize the safety of the towing operation. Vessels towing gear are generally operating at well under 10 knots, which means that there is no meaningful risk of a vessel strike under towing conditions.

In addition, the Proposed ITR's speed restrictions appear to apply to escort and support vessels that may need to move quickly to intercept and communicate with fishing vessels, remove marine debris posing a safety hazard, or for other reasons. The final ITR should explicitly exempt escort and support vessels from the speed, separation, and avoidance restrictions, or, alternatively, should include a blanket exception for activities that, in the discretion of the captain, are necessary to protect human safety, property, or the environment.

11. The Proposed ITR's PSO training and experience standards are impracticable.

In general, the Associations agree that it is helpful to have training requirements and reasonable standards for PSOs. Operators have a vested interest in hiring the most experienced PSOs capable of conducting accurate mitigation and monitoring. However, the Proposed ITR includes PSO training requirements that may be difficult, if not impossible, to achieve. For example, it is infeasible to require that PSOs have a minimum of 90 days of at-sea experience with no more than 18 months elapsed since the conclusion of that experience, or that all PSOs or PAM operators have bachelor's degrees. Given the high PSO turnover and reduced marine geophysical activities in the U.S., the Associations are very concerned that the pool of PSOs that meet these requirements will not be sufficient to support the industry's activities. The final ITR should state that these are preferred training standards that should be met whenever possible but should recognize that, in the absence of PSOs that meet those criteria, LOA holders would not be prevented from conducting surveys with otherwise qualified PSOs.[53]

12. Some monitoring and reporting requirements are unreasonable, not supported by science, or will result in inaccurate reporting.

The Proposed ITR would impose several monitoring and reporting requirements that warrant revision because they are not supported by science, are overly burdensome, or would result in inaccurate reporting. For example, the Proposed ITR would require that vessel operators provide "Bigeye" binoculars. 83 Fed. Reg. at 29,307. Bigeye binoculars are an expensive and maintenance-intensive piece of equipment. Although they can be useful at monitoring ranges of 2 to 7 kilometers, they are not useful for monitoring the 500-1,000 meter range that is most relevant during survey operations. Expert PSOs prefer to scan that range with 8x to 10x binoculars or the naked eye in order to optimally cover the monitoring zone.[54] In addition, installing Bigeye binoculars would require the additional installation of a pedestal or other solid mount on the bridge or flying bridge, which will require welding and drilling on each vessel. The final ITR should not include this requirement, which will be costly, will require permanent alteration of the vessels, and will not be useful for PSOs in detecting marine mammals in the exclusion areas defined in the Proposed ITR.

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[53] The Associations have previously commented on observer standards and have offered constructive solutions to ensure that these standards are workable, accurate, and appropriate before they are imposed. *See* Letter from Andy Radford et al. to Kyle Baker (May 2, 2014). We urge NMFS to consider those comments and modify the PSO standards in the final ITR accordingly.

[54] While Bigeye binoculars may be a regular tool of dedicated marine mammal surveys in which the vessel may be directed off-track to confirm long-range species identifications and to assess group sizes, the narrow field of view and cumbersome searching process for the large, heavy Bigeye binoculars is detrimental to the intended purpose of visual mitigation monitoring for timely and effective shutdowns or other mitigative actions.

The Proposed ITR would also require PSO reporting on factors that the PSO perceives to be contributing to "impaired observations." 83 Fed. Reg. at 29,308. This should be removed from the reporting requirements in the final ITR because it would require PSOs to speculate in a manner that can result in the reporting of unverifiable and often incorrect data. In addition, this could encourage PSOs—who are not trained in vessel operations, personnel safety, human resources, or any of the many other aspects of geophysical survey operations—to speculate about information that could have serious commercial, legal, and reputational impacts.

The Proposed ITR would require a PSO to report on the estimated number of animals by cohort (adults, yearlings, juveniles, calves, etc.). *Id.* This is overly complicated and impractical, and is only likely to lead to inaccurate reporting and the drawing of even more inaccurate conclusions. The Associations also question whether NMFS, BOEM, or BSEE have the necessary staff to review and evaluate this level of detail. The final ITR should require the recording of juveniles and adults only.

The Proposed ITR includes confusing language regarding daily reports that the final ITR should clarify. Specifically, after discussing a requirement for daily reports, the Proposed ITR states that these reports "would include … corrected numbers of marine mammals 'taken.'" Proposed ITR at 29,287. It is not clear what the Proposed ITR intends to "correct" in the extrapolation of an estimated number of takes from a documented number of sightings. It is not feasible for PSOs to complete daily line-transect analyses, nor would it provide any benefit to conduct such an analysis on an almost real-time basis, given the variability of marine mammal distribution and detection on a day-to-day basis. NMFS may have meant that comprehensive reports should include estimates of take numbers *based on* the daily reports. In any case, the final ITR should clarify that there is no expectation for daily reports to include estimated take numbers since such calculations would be virtually meaningless given the large statistical uncertainty produced by using the small numbers of sightings encountered on a given day.

Finally, and importantly, the Proposed ITR reporting requirements include a process recommended by the Marine Mammal Commission (MMC) for estimating marine mammal takes from PSO observations. The industry has consulted with several experts in the field of PSO monitoring and distance sampling and is concerned that the proposed process as currently written is not practicable. Specifically, reporting on a daily, monthly, or end-of-permit basis is unlikely to yield a sufficient number of observations to enable estimation of takes with reasonable statistical confidence even if *a priori* f(0) and g(0) values, such as those from Barlow *et al.* (2015), are used. For example, even NMFS's sighting surveys—which are generally longer and cover more area than a seismic survey—are hampered by low sample sizes after many weeks or months of survey effort, while using an optimized and time-tested sampling protocol very unlike a seismic survey vessel track.

The differences between the kinds of observation protocols upon which Barlow et al. (2015) are based and the constraints of the seismic survey protocols offer serious challenges to confident estimation of takes from PSO observations, even with adequate sample sizes. The constraints include (1) seismic survey vessel speeds that are less than half the speed of a dedicated NMFS marine mammal survey vessel; (2) differences in search area and observer coverage of that area (PSOs are looking relatively close to the vessel to cover the mitigation zone whereas NMFS observers look forward but not behind the vessel and make extensive use of Bigeye binoculars to evenly cover an area out to the horizon); (3) the coil, racetrack, or back-and-forth patterns of seismic surveys create a greater probability of re-encountering individual animals whereas the NMFS line transects are designed to minimize repeated sightings of the same individuals; and (4) NMFS surveys typically move off-track to approach animals in order to confirm species identifications and group sizes, whereas this is not possible during a seismic survey. These and likely other differences will need to be factored into the take estimates derived from PSO data, including generation of refined f(0) and g(0) values, and will require a sufficient sample size to support estimates of correction factors for the issues mentioned above. The choice of statistical approach should be made based on a contextual consideration of, *inter alia*, numbers of sightings, conditions, vessel speed, track-line shape, and observer protocols.

We therefore suggest that the MMC-proposed distance-based protocol be applied at the end of a period long enough to accumulate sufficient data, such as the end of the first year of the ITR, when the challenges of applying the protocol noted above can be undertaken with sufficient data. At that time, NMFS and LOA holders could evaluate the practicability of this approach and validity of the results presented as part of the Annual Monitoring Report process described in the ITR. The efficacy of the MMC-proposed distance-based protocol should be evaluated as part of the adaptive management process, and with the engagement of distance methodology experts and independent PSOs, to further improve and develop recommendations for appropriate extrapolation methods that ensure the extrapolated data are based on the best science and the best possible statistical approach. In sum, the process described in the Proposed ITR, while conceptually reasonable, would not work as intended for the reasons addressed above. The industry proposes a way forward by which the concept can be translated into a practicable and meaningful process for take estimation from observer data.

E. The Proposed ITR will have no more than a negligible impact on marine mammal species and stocks.

Based, in part, on the extensive record of agency findings, observational data, and research regarding the potential effects of seismic survey activities on marine mammals in the GOM—in which no significant effects on *any* marine mammal species or stock have been observed—the Associations concur with NMFS's finding that the incidental taking allowed under the Proposed ITR will have a negligible impact on marine mammal species and stocks.

We also emphasize that NMFS's negligible impact determination is based upon highly conservative assumptions about the potential effects of seismic survey activities in the GOM. For example, as addressed in more detail in Section III.H below, NMFS's estimates of the numbers of potential takes by the proposed surveys are substantially inflated as a result of overly conservative modeling assumptions, and NMFS acknowledges that incidental takes will be reduced as a result of mitigation requirements. Consequently, NMFS's modeling of potential impacts presumes that far more numbers of animals will be incidentally taken than will actually be taken, based on past and recent observations in the field for similar permitted activities.

We commend NMFS for developing a thoughtful approach to the assessment in support of its negligible impact determination. As described in Appendix D, and summarized below, we recommend certain improvements to NMFS's approach to the negligible impact determination in the spirit of proactively enhancing NMFS's assessment.

First, the framework developed by the "expert working group" (EWG) was applied here without following all of the originally recommended steps, such as conducting expert elicitation to drive risk functions for species that do not have parameterized Population Consequence of Disturbance (PCOD) models.[55] As a result, the professional judgments regarding the vulnerability and severity rankings were made by the authors of Southall et al. (2017) rather than developed through a formal process involving independent experts. The Associations recommend that NMFS seek input and advice on the framework and its conclusions from independent experts.

Second, based on the EWG's framework, NMFS makes overly conservative "severity of effect" ratings (such as "very high," "high," or "moderate") for certain marine mammal stocks or species in certain areas that cannot be rationally reconciled with the best available scientific data and information. *See* 83 Fed. Reg. at 29,293-94. We are aware of no findings by any agency, including NMFS, that a seismic survey had anything more than an insignificant effect or a negligible impact on a marine mammal species or stock, and certainly no effects with "very high," "high," or "moderate" levels of impact on marine mammal populations. We recognize that these are defined values, but the implication that, for example, seismic surveys will have "very high" severity effects on sperm whales GOM-wide is simply not compatible with the multi-decade history of offshore seismic exploration in the GOM or the broader U.S. OCS. Accordingly, although the Associations concur with NMFS's conclusion that the take allowed under the ITR will result in no more than a negligible impact on marine mammal species or stocks, we disagree with the implications of NMFS's "severity" ratings, which are not consistent with the best available science.[56]

Third, and relatedly, NMFS's use of the "potential biological removal" (PBR) metric in its negligible impact assessment is inappropriate. Although we agree with NMFS's caveats in using

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[55] The Proposed ITR does not include any meaningful discussion of the PCOD model, which is very relevant to the assessments contained in the Proposed ITR. Appendix D describes the PCOD model and provides associated references. We request that the PCOD model and associated references be carefully considered and incorporated into the agency's assessments in support of the final ITR.

[56] The Associations' position that there are currently no demonstrated adverse effects from seismic surveys on marine mammal populations does not preclude our taking a proactive and environmentally responsible approach by actively investigating legitimate concerns raised by subject matter authorities, and doing so in the best traditions of independent, peer-reviewed scientific study. *See* E&P Sound and Marine Life Joint Industry Programme, www.soundandmarinelife.org. We appreciate NMFS's acknowledgement and summary of the studies conducted under this program. *See* 83 Fed. Reg. at 29,300.

the PBR metric (*e.g.*, that Level A harassment does not result in mortality or "removals" from PBR), it has no relevance in the context of permitting incidental take by *harassment* under MMPA Section 101(a)(5). *See* 83 Fed. Reg. at 29,290. Level A harassment is defined as having the "potential to injure." 16 U.S.C. § 1362(18). PBR refers to the number of animals "not including natural mortalities that may be *removed* from a marine mammal stock." *Id*. § 1362(20) (emphasis added). Under the MMPA, NMFS is required to compare the amount of "serious injury and mortality" from commercial fisheries against a stock's PBR to determine whether measures must be taken under the MMPA's take reduction planning provisions to reduce the rate of serious injury and mortality by commercial fisheries. *See id.* §§ 1386, 1387. "Serious injury" is defined by regulation as "any injury that will likely result in mortality." 50 C.F.R. § 216.3. By definition, Level A harassment does *not* include "serious injury" or "mortality" and, therefore, it is inappropriate to assess the merit of a Level A harassment authorization by comparing it against a metric (PBR) that is far more narrow, is not referenced at all in Section 101(a)(5), and has no applicability in the incidental take authorization context. It is a misguided apples-to-oranges comparison.[57]

Fourth, there is little scientific support for the elevated "high risk" rankings for sperm and beaked whales exposed to seismic survey sounds (*see* Miller (2009); Madsen (2002)). Although there is considerable evidence for strong beaked whale response to mid-frequency *military sonars*, there is no evidence that the very different impulse sounds produced by seismic surveys elicit a similar response.[58] Indeed, NMFS admits that there "has been no direct evaluation of beaked whale sensitivity to noise from airguns." 83 Fed. Reg. at 29,248.

Fifth, to the extent NMFS has relied upon Appendix K (the "CCE report") to the GOM PEIS (*id*. at 29,243), we object to that reliance and incorporate by reference our criticisms of Appendix K, as stated in our November 29, 2016 comment letter addressing the draft GOM PEIS.[59] Concepts such as "soundscape," "communication space," or "acoustic footprint" have no basis in any existing statutory or regulatory authorities, and are therefore inapplicable to this rulemaking. *See also* Appendix D.

Finally, we wish to emphasize a few points of agreement with NMFS's conclusions (other than our agreement with the negligible impact determination itself). We agree with NMFS's conclusion that "that Level A harassment will [not] play a meaningful role in the overall degree of impact experienced by marine mammal populations as a result of the projected survey activity." *Id*. at 29,296. We also agree that *no* Level A harassment is "likely to actually occur for mid-frequency cetaceans." *Id*. at 29,290. These determinations are well-supported by the best

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[57] For the same reasons, the inclusion of PBR values on Table 3 is inappropriate.

[58] *See* Tyack P. et al. 2011. Beaked whales respond to simulated and actual navy sonar. PloS One, 6(3): e17009.

[59] Letter from Nikki Martin et al. to Jill Lewandowski (Nov. 29, 2016). We also assume, and hereby expressly request, as necessary, that all of the Associations' comments on the draft and final GOM PEIS will be considered and included in the administrative record for the final ITR.

available science, as is NMFS's overarching determination that the incidental take proposed to be allowed by NMFS will have a negligible impact on any affected marine mammal species and stocks in the GOM.[60]

F. The Associations support NMFS's proposed approach for determining "small numbers."

The Associations generally agree with NMFS's proposed approach to Section 101(a)(5)(A)'s "small numbers" provision. Specifically, the Associations agree that, when issuing an ITR, NMFS may "determine[] that the small numbers finding should be applied to the annual take authorized in each LOA" rather than to the ITR itself. 83 Fed. Reg. at 29,298. As addressed below, this approach is consistent with the MMPA's plain language and intent, and with case law.

The MMPA directs that the Secretary "shall *allow*, during periods of not more than five consecutive years each, the incidental, but not intentional, taking by citizens while engaging in that activity within that region of *small numbers* of marine mammals...." 16 U.S.C. § 1371(a)(5)(A)(i) (emphases added). Under the federal government's long-established two-tiered process for authorizing incidental take under Section 101(a)(5)(A), no take is "allowed" when an ITR is issued. Rather, the ITR simply establishes a programmatic regulatory framework for the subsequent authorization of incidental take, but does not actually authorize the incidental take associated with the specific underlying activity. Incidental take is only authorized in the second step of the process—*i.e.*, when NMFS issues LOAs to individual operators carrying out the activities contemplated by the ITR.[61] The MMPA requires a small numbers finding at the second stage, when incidental take is "allowed."

In contrast, the only substantive determinations that NMFS must make at the ITR stage are whether the total of such taking over a five-year period will have a negligible impact and will not have an unmitigable adverse impact on the availability of such species or stock for taking for subsistence uses. 16 U.S.C. § 1371(a)(5)(A)(I). The MMPA further states that NMFS shall allow the incidental taking of "small numbers" of marine mammals only when it determines that the statute's substantive determinations have been satisfied. Had Congress intended otherwise, it

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[60] We further agree that the use of Wood et al. (2012) step function and accounting for differential hearing sensitivity of marine mammal hearing groups for Level B take estimation is the best available science and appreciate the thorough treatment by NMFS in evaluating, and documenting the problems with, Nowacek et al. (2015). We also appreciate the consideration of the Wood et al. (2012) framework in the specific context of the GOM, and the appropriate removal of the risk factor associated with migratory baleen whales (as the Bryde's whale in the GOM is not known to be migratory).

[61] *See* 50 C.F.R. pt. 216, subpt. I. As these implementing regulations make clear, the ITR does not authorize any operator to incidentally take marine mammals and no such take may lawfully occur until if and when an operator obtains an LOA from NMFS.

would have expressly required a "small numbers" finding for the "total of such taking" at the ITR stage.

Additionally, NMFS's interpretation does not compromise species protections. It is well-recognized that the MMPA's "negligible impact" standard is extraordinarily protective and, for example, is "more conservative" and "stricter than" even the Endangered Species Act's standard for the authorization of incidental take.[62] As NMFS's recognizes, and as the Associations agree, the "negligible impact" standard has primary biological significance—*not* the "small numbers" finding—and is the touchstone for incidental take authorization under the MMPA. *See* 83 Fed. Reg. at 29,299 ("the small numbers standard has little biological relevance").

Moreover, the law is clear that NMFS need not support its "small numbers" determinations for LOAs with quantified assessments.[63] Here, NMFS's proposed approach for authorizing small numbers of incidental take at the LOA stage is generally consistent with applicable law, but arguably goes above and beyond legal requirements by establishing a quantitative standard (one-third of a marine mammal stock size) in certain instances when sufficient data are available. If NMFS retains the one-third standard for these specific circumstances, the Associations recommend that NMFS provide a detailed and thorough explanation in the final ITR in support of the one-third standard.[64]

Finally, two related points warrant particular emphasis. *First*, it is arbitrary and inappropriate for NMFS to establish a hard requirement that *any* deviation from NMFS's recommended modeling approach (Zeddies et al. (2015)) for estimating incidental takes will *require* public notice. *See* 83 Fed. Reg. at 29,301. Such a requirement is contrary to the legal requirement that NMFS base its authorization of incidental take under the MMPA on the best available science. There may very

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[62] *See Ctr. for Biological Diversity v. Salazar*, 695 F.3d 893, 913 (9th Cir. 2012) (Section 101(a)(5)(A) standard is "more conservative than the ESA standard"); *In re Polar Bear Endangered Species Act Listing & 4(d) Rule Litig.*, 818 F. Supp. 2d 214, 233 (D.D.C. 2011) (agreeing with government that "the MMPA is comparable to, or even stricter than, the take provisions of the ESA in most respects").

[63] *See Ctr. for Biological Diversity*, 695 F.3d at 907 ("The Service need not quantify the number of marine mammals that would be taken under the regulations, so long as the agency reasonably determines through some other means that the specified activity will result in take of only 'small numbers' of marine mammals."); *see also* H.R. Rep. No. 97-229 (1981), *reprinted in* 1981 U.S.C.C.A.N. 1458, 1459 ("The Committee recognizes the imprecision of the term 'small numbers', but was unable to offer a more precise formulation because the concept is not capable of being expressed in absolute numerical limits.").

[64] *See, e.g.*, *Ctr. for Biological Diversity*, 695 F.3d at 905-06 (plaintiffs argued that Section 101(a)(5)(A) "requires the Service to quantify in absolute terms the number of mammals that would be taken"); *Native Vill. of Chickaloon v. NMFS*, 947 F. Supp. 2d 1031, 1052-53 (D. Alaska 2013) (plaintiffs argued that NMFS improperly "categorically" established 10% as a "small number," among other challenges to the small numbers finding).

well be better and more accurate modeling available during the five-year period of the ITR for estimating incidental take. LOA applicants should not be penalized for using the best available science. Moreover, given that mitigation and monitoring measures will have already been prescribed and total take for the ITR will have been determined to be negligible (through a thorough public review process), additional public review of LOA applications adds unnecessary time and investment by both the government and regulated community. *Second*, it is imperative that NMFS use the same abundance of each population for purposes of take estimation and purposes of determining whether incidental take levels for a requested in LOA application constitute "small numbers." It would be arbitrary and unreasonable if NMFS were to use densities in take modeling that resulted in a population abundance that is different than the abundance against which incidental take is evaluated for small numbers.

G. NMFS must clarify how it intends to evaluate and process LOA applications.

The Proposed ITR does not address how NMFS intends to process the numerous LOA applications it will receive under the ITR in a timely manner. We understand that NMFS's Protected Resources permitting department has been, and continues to be, understaffed in comparison to the number of incidental take applications it receives. The LOAs requested under the GOM ITR will significantly multiply the number of applications NMFS typically receives in a given year. The applications related to HRG surveys alone will be very substantial, as companies execute different strategies to exploration and production drilling for their individual assets. It is not clear how NMFS will review and approve applications on strict timelines, given the agency staffing constraints and the substantial number of companies operating in the GOM. Additionally, completing the application process for numerous activities (providing the 14 pieces of information for each activity under 50 C.F.R. § 216.104(a)) will be a significant and repetitious paperwork burden for applicants. Moreover, there will likely be short periods of time during the five-year ITR period (*e.g.*, immediately upon promulgation of the ITR) in which NMFS receives a high volume of LOA applications that cannot be processed in a timely manner, thus delaying critical and time-sensitive activities.

We therefore strongly recommend that the final ITR clearly address how NMFS plans to process voluminous LOA applications in a timely and efficient manner. In that vein, we encourage NMFS to retain flexibility in the final ITR for the development of efficient and effective LOA processes through workshops or other engagement with BOEM and the regulated community.

We also wish to emphasize that there is no legal justification for NMFS to use the ITR as a mechanism to limit the number of activities that may occur in the GOM because authorization of the activities themselves are subject to BOEM's jurisdiction. Accordingly, we strongly disagree with NMFS's suggestion that the amount of incidental takes listed in the ITR serves as a "cap on the number of authorizations that could be issued." 83 Fed. Reg. at 29,301. Whether NMFS may continue to authorize incidental take under an ITR is not determined by the amount of take projected in the ITR, or by the amount of activity projected in the ITR, but rather upon NMFS's determination as to whether the actual "total of such taking" allowed under the ITR has a negligible impact. 16 U.S.C. § 1371(a)(5)(A)(i)(I). It may very well be, for example, that the amount of projected take under an ITR is exceeded by the LOAs collectively authorized under an

ITR, or that projections for certain categories of activities are exceeded, but that the type or degree of such taking is not as severe as projected in the ITR and, consequently, that the allowed take continues to have no more than a negligible impact on the affected marine mammal stocks. NMFS should not and cannot restrain its own discretion by placing a “cap” on incidental take, particularly when doing so is contrary to Section 101(a)(5)(A).

H. NMFS substantially overestimates the number of incidental takes predicted to result from the Proposed ITR.

In our comments on BOEM’s Application, we explained that the modeling used to estimate the anticipated number of incidental takes is improperly and intentionally designed to *overestimate* takes and impacts. In the Proposed ITR, NMFS states that it “strongly disagrees” with our comments. 83 Fed. Reg. at 29,259. Notwithstanding NMFS’s strong disagreement on this issue, we maintain our position because it is supported by the record facts, the best available science, the agencies’ own statements, and the modeling used by both BOEM and NMFS.

The model used by NMFS in the Proposed ITR for estimating incidental take is essentially the same model used by BOEM and NMFS in the environmental documentation supporting incidental take authorization in the Atlantic and the GOM. The Associations have provided criticism of that modeling process in numerous comment letters, supported by many pages of detailed technical data and explanation, and legal authorities. We also provided a technical assessment prepared by BOEM’s contractor in support of our comments, which NMFS inexplicably dismisses as being provided too late despite the fact that it was provided to NMFS *11 months ago* (at approximately the same time Southall et al. (2017) was incorporated into the agency’s analysis).

Our comments have criticized the use of extremely conservative or “precautionary” data values in multiple places within the modeling. The use of such data values result in estimates of potential exposures and derived estimated incidental take levels far higher than those that would reasonably be expected to occur. The gist of the agencies’ errors is that their approach to take estimation is based upon a modeling exercise that uses conservatively biased assumptions for many model variables. These conservatively biased assumptions, each contributing relatively modest overestimates of effect, lead to multiplicatively accumulating bias as the conservative assumptions interact with each other to multiply uncertainty toward unlikely statistical probabilities, with the result that the modeled take estimates are not representative of realistic conditions. Consequently, the result of BOEM and NMFS persistently using worst case values for many variables within the model becomes little more than an improbable, highly implausible worst case scenario—*not* a fair or best estimate simulation of likely consequences.

These criticisms are supported by numerous agency statements, such as:

- “Even as defined to include the sensitive threshold of Level B harassment, the numbers estimated for incidental take are *higher than BOEM expects would actually occur*.... They do not, for example, take into account most of the mitigation measures incorporated into Alternative B because the

> effect of those measures cannot be quantified with statistical confidence at this time."[65]

- "[T]he take estimates are based on acoustic and impact models that are by design conservative, which results in an over-estimate of take. Each of the inputs into the models is *purposely developed to be conservative, and conservative assumptions accumulate throughout the analysis*." *Id.*

- "The existing modeling largely does not account for uncertainty in the data inputs and also selects highly conservative data inputs. This bias often produces unrealistically high exposure numbers and 'takes' that *exponentially increase uncertainty throughout each step of the modeling*." Draft GOM PEIS at 4-47 (emphasis added).

- "This estimate [of marine mammals exposed to sound] alone does not reflect BOEM's determination of the actual expected physical or behavioral impacts to marine mammals but rather an *overly conservative upper limit* because none of the mitigations examined in this Programmatic EIS were modeled." *Id.* (emphasis added).

- The model "requires accepting a *worst-case scenario,* which ultimately overestimates the numbers of 'take' under the MMPA by equating those numbers with the exposures identified in the modeling rather than real world conditions." *Id*. at 1-19 (emphasis added).[66]

NMFS's own statements in the Proposed ITR substantiate BOEM's characterizations. Specifically, NMFS acknowledges that its Level B harassment modeling likely "leads to *substantial overestimates* of the numbers of individual potentially disturbed [and] ... to an overestimation of the population-level consequences of the estimated exposures" and that, even with the application of a correction factor, the modeling still represents an "overestimate." 83 Fed. Reg. at 29,291 (emphasis added); *see id*. (NMFS admission that its modeling choices are "purposely conservative"). Indeed, the Proposed ITR is replete with examples of NMFS's use of conservative overestimates instead of the most likely values.[67] NMFS's decision to

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[65] Record of Decision, Atlantic OCS Proposed Geological and Geophysical Activities, Mid-Atlantic and South Atlantic Planning Areas, Final Programmatic Environmental Impact Statement ("Atlantic ROD"), at 12 (emphasis added).

[66] BOEM's decision not to include some of these definitive statements in the GOM PEIS does not undermine their value because BOEM provided no rational explanation for doing so. The precision and clarity with which BOEM has described the modeling in the Atlantic documentation and in the Draft GOM PEIS accurately describes the admittedly unrealistic results of the modeling.

[67] *See, e.g., id*. at 29,248 (expressly rejecting the best available science [Finneran (2016) Type III filters], which NMFS admits "are better designed to predict the onset of auditory injury," and,

(continued . . .)

conservatively estimate certain variables to overestimate incidental take is directly contrary to NMFS's own statement that "a decision about the appropriateness of a particular function should be based on *how well it reflects the best available information, rather than on how it affects the resulting number of takes*." 83 Fed. Reg. at 29,249 (emphasis added).

We appreciate that NMFS has made some modifications to partially mitigate the inaccurate and overly conservative results produced by the model (*e.g.*, modifying PBR to be more consistent with the inflated population values applied in the model, and incorporating aversion in the risk assessment), but we respectfully maintain that a better approach would be to use the best and most likely values for all of the input variables to the model in the first place, which NMFS, by admission, has not done. Although the format of the model itself correctly captures the relevant variables needed to estimate sound exposure and thus a derived risk metric, the handling of uncertainty about which values to enter into the model is not mathematically correct. We recognize that our knowledge of some of the values used in the model is not perfect and assumptions must be made, but it is important to acknowledge that when conservative assumptions are used across multiple variables within the model those conservative assumptions do not average out or add up, but interact multiplicatively, resulting in an substantial overestimate of exposures and associated incidental takes. Conservatism due to uncertainty about the values entered into the model must properly be handled separately, after modeling to most likely outcome, as is widely demonstrated and well-known for a variety of similar risk models such as weather models, economic models, and medical diagnostic and treatment models.[68]

To further illustrate these problems, IAGC and API requested and received permission from both BOEM and NMFS to engage the same contractor that performed the GOM PEIS modeling (JASCO Applied Sciences) to run the same model, with the same data, but with certain alterations. This new analysis included alterations to only four or five variables to illustrate the

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(. . . continued)
instead, "as a conservative measure [retaining] Type I filters ... for use in evaluating potential behavioral disturbance in conjunction with the Wood *et al.* (2012) probabilistic response function"), 29,251 ("the single airgun results were used as a conservative substitute for the boomer"; using a "a conservative but reasonable approximation to simplify the variability across all HRG sources"), 29,252 ("When necessary, the choices were made to be conservative so as not to ultimately underestimate potential marine mammal exposures to noise."), 29,257 ("A conservative estimate of +/- 3 dB standard deviation was used to investigate the effects of source level variance on SEL injury exposure estimates."), 29,261 ("this method of correction still overestimates the numbers of individuals affected across the year, as it does not consider the additional repeated takes of individuals during surveys that are longer than 30 days or by multiple surveys"), and 29,262 ("we believe that while some amount of Level A harassment is likely, the lack of aversion within the animal movement modeling process results in overestimates of potential injurious exposure").

[68] *See, e.g.*, Slingo, J. and T. Palmer. 2011. Uncertainty in weather and climate prediction. Phil. Trans. R. Soc. A(2011) 369: 4751-4767.

dramatic consequences of redundantly applied precaution in a large, complex, multivariate model. We provided this analysis to BOEM and NMFS on August 25, 2017. However, NMFS states in the Proposed ITR that the results "were not made available to NMFS in time to fully consider them in preparing these proposed regulations." 83 Fed. Reg. at 29,259. Given that the analysis was provided to NMFS *11 months ago*, NMFS's failure to fully consider it is not credible and, in conjunction with this notice and comment regulatory process, we again request that NMFS consider the analysis and make appropriate changes to the final ITR.[69] We reiterate our comments provided with the analysis on August 25, 2017, and specifically address some of NMFS's preliminary statements about the analysis as follows.[70]

Array Size. An 8,000-in^3 array is *not* "a reasonable representation of the arrays that may be used in the future." 83 Fed. Reg. at 29,260. The JASCO re-analysis plainly documents the average GOM array size based on the best available information. A review of more than 2,000 surveys in the GOM, spanning a period of more than a decade, yielded mean or median array sizes of 4,000 to 6,000 cubic inches, depending on the data source (from multiple survey companies and an archived databased maintained by a PSO service provider). Equally important, in terms of estimating the sound produced by an array, is the number of elements in the array. The array used in the BOEM/NMFS model (8,000 cubic inches from 72 elements) was a double array—most arrays have only 16-40 elements. The array chosen by BOEM and NMFS to represent "all deep penetration surveys" was therefore approximately 7 dB louder than average, which is a value that translates into exposures and takes that would be about four times the number the model would have predicted if an average array were used in the model. We had previously offered an array size of 4,130-in^3, which falls within the median array size used in the GOM, and with a number of elements within the common range. Other array metrics in this range are available if NMFS does not agree with the choice of array for the JASCO re-modeling exercise.[71] If NMFS does not apply a smaller array size, then NMFS must explain how it addresses the potentially significant differences in area for exposures zones for the 67% of surveys conducted with smaller arrays.

Mitigation Measures. We continue to disagree with NMFS's decision not to take the known beneficial effects of mitigation measures into account in its take modeling. *See* 83 Fed. Reg. at 29,260. We acknowledge that precise quantification of these beneficial effects is difficult given the inherent uncertainty in the best available information. However, this is the case with essentially all of the variables involved with estimating offshore seismic impacts on marine mammals and for which NMFS readily (and precautionarily) assigns values. That mitigation effects may be different for low- and high-frequency species should also be no impediment to reasonably estimating those effects. For instance, the take estimation process uses different hearing groups, dual Level A criteria, species-specific distribution, diving, and other parameters

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[69] The cover letter and enclosed analysis is provided again as Appendix E to this comment letter.

[70] Some of these points are addressed in more detail in Appendix D.

[71] 2,500 cubic inches is more representative for vertical seismic profiles (VSP).

to distinguish between low- and high-frequency species based on uncertain information. Additionally, there is no dispute that required mitigation measures will have *some* beneficial effect.[72] In other parts of its analysis, NMFS has admittedly chosen conservative numerical values to assess allegedly uncertain variables to overestimate adverse effects. But, NMFS does not assign *any* value to, and thereby treats as entirely irrelevant, beneficial mitigation effects that are known to occur. This is a classic example of arbitrary decision-making.

In sum, we again request that NMFS carefully consider the analysis that was provided by the Associations in August 2017, and make appropriate changes to its modeling of estimated takes. The MMPA requires NMFS to use the best available science to determine the best and most accurate impact estimate and does not permit NMFS to overestimate the potential impacts based upon speculative assumptions that are not supported by the best available information. The law also does not allow NMFS to arbitrarily treat categories of uncertain information differently—*i.e.*, to conservatively estimate inflated numerical values for some factors to estimate adverse effects while simultaneously assigning *zero* value to other factors showing known beneficial effects.

I. The Associations generally support NMFS's proposed approach to "comprehensive monitoring" and adaptive management.

As we stated in our comments on BOEM's Application, the Associations have a strong interest in environmental monitoring—both to better understand the environment in which our members work and to mitigate potential risks of activities to living marine resources. We continue to support efforts that improve the quantity and quality of information related to determining the nature and magnitude of the potential effects of offshore geophysical activities on marine mammals. In this light, the Associations support both ongoing and future research endeavors by independent third-party researchers, supported by industry. This industry-sponsored independent research helps industry, regulators, and the public better understand the nature and magnitude of the potential effects from offshore activities and develop ways to mitigate potential effects of geophysical activities on marine mammals in the GOM. We also support agency efforts to improve the collection and use of the best available science consistent with the requirements and limits of the MMPA.

Additionally, we have explained that the MMPA does not authorize NMFS to *require* as a condition of an incidental take authorization the preparation or development of a large-scale,

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[72] *See* 83 Fed. Reg. at 29,257; *see id*. at 29,257-58 (recognizing that "a very conservative estimate of mitigation effectiveness should be used" for deep-diving species but concluding, without explanation, that "[u]ltimately ... quantification of mitigation effectiveness was not incorporated into the Phase II modeling effort (*i.e.*, is not reflected in the modeled exposure estimates)"; Atlantic ROD at 12 ("Although all mitigation measures cannot be effective 100 percent of the time, these measures undoubtedly will contribute to species protection, and they will be refined as environmental impacts are evaluated in environmental review for site-specific authorizations, including ESA and MMPA consultations.").

expansive monitoring plan that reaches beyond the time and area in which site-specific activities are undertaken or the performance of actions related to such a plan. We appreciate NMFS's apparent consideration of those comments and, relatedly, generally support the Proposed ITR's approach to "comprehensive monitoring" and adaptive management. Although we continue to believe that NMFS, as the regulating agency, has the responsibility to collect, organize, and assess all of the data reported to NMFS under the terms of LOAs, the Associations are nonetheless willing to participate in the annual assessment process described in the Proposed ITR, subject to any legal impediments.

We agree that annual assessment and adaptive management is essential to reducing both potential impacts on marine mammals and operational and cost impacts to the regulated community. We also agree that any research or long-term monitoring outside of the data required to be gathered under LOAs can only be performed by industry on a voluntary basis. In sum, the Associations look forward to working collaboratively with NMFS and BOEM on an annual assessment and adaptive management plan that is both legally compliant and operationally effective.

J. The RIA makes a number of incorrect assumptions and unsupported conclusions.

The RIA included in the Proposed ITR is an important, required assessment that must transparently demonstrate the impact of the proposed regulation on the regulated community. The Associations appreciate NMFS's preparation of the draft RIA, but identify here numerous incorrect assumptions and unsupported conclusions that result in an underestimate of costs and overestimate of benefits:

- The RIA incorrectly assumes that the costs of closures and other measures are simply "delays." RIA at ES-8. In fact, such measures may render some survey proposals economically unattractive to the point at which prospects will not be explored. In addition, closures will likely be assumed to be permanent and thus result in decreased exploration interest, which, in turn, will cause firms to examine alternative investments in other competing regions of the world. Additionally, the timing of the Eastern GOM restrictions poses a serious risk of failing to take advantage of existing nearby infrastructure that may not be available if new exploration is delayed.

- The RIA incorrectly assumes that the costs of closures are highly uncertain or even low because geologic potential of some areas is low. RIA at ES-9. History is replete with examples of areas thought to have low potential turning out to contain enormous reserves following the results of geophysical surveys, which is precisely why geophysical surveys are so essential. In fact, the deep water GOM was considered to have negligible resource potential until the early 1990s.

- The RIA wrongly assumes that the GOMESA moratorium prevents exploration of the Eastern GOM. In fact, the moratorium expires in 2022 and the Administration has indicated that it is interested in including much of the Eastern GOM in the new Five-Year OCS Plan. As a result, the RIA seriously misleads readers about the costs of closure and increased restrictions in the Eastern GOM. The conclusion wrongly judges that the closure impact is only “moderate.” Because high-potential resources may underlie Eastern GOM areas, the cost of closure must be “high” for the Eastern GOM just as it is for the Central GOM.

- The RIA fails to account for how geophysical technology allows companies to narrow exploration targets, thereby reducing environmental impacts associated with unnecessary drilling. Accordingly, the RIA fails to account for the loss of this environmental benefit.

- The RIA incorrectly assumes that current geophysical data for the Eastern GOM is “suitable.” RIA at ES-20. In fact, there is high demand for state-of-the-art new data for Eastern GOM frontier areas where older data is considered unsuitable to support new investment.

- The RIA fails to account for possible increased industry interest in Eastern GOM geophysical surveys. The use of old statistics on survey interest is therefore inappropriate for estimating costs. *See* RIA at 4-0.

IV. CONCLUSION

We appreciate the opportunity to submit these comments, which are focused on improving the scientific and legal integrity of a final ITR as well as ensuring effective, streamlined, and reasonable implementation of the final ITR and subsequent LOA processes. As addressed in detail in the sections above, and summarized in Section II above, although the Proposed ITR is a well-structured and thorough document that appropriately concludes that geophysical activities in the GOM have no more than a *negligible impact* on marine mammal populations, it is essential that NMFS address the Proposed ITR’s significant shortcomings to ensure the transparency, predictability, and legal suitability of the final ITR. Should you have any questions, please do not hesitate to contact Nikki Martin (713.957.8080) or Andy Radford (202.682.8584).

[continued on next page]

Sincerely,

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APPENDIX A

BOEM
Bureau of Ocean Energy Management

SCIENCE NOTES

Applied science for informed decision making

March 9, 2015

Dear Reader:

In August 2014, BOEM published a *Science Note* addressing a few fundamentals about impacts of seismic air gun surveys on marine mammal populations. The surveys are used to characterize sub-seabed geology, including oil and gas resources but are also used for our marine minerals program and renewable energy. One sentence in the *Science Note* has generated some dialogue: "To date, there has been no documented scientific evidence of noise from air guns used in geological and geophysical (G&G) seismic activities adversely affecting animal populations."

BOEM's conclusion regarding the impact of these surveys is in stark contrast with public statements citing BOEM research and asserting that many thousands of marine mammals will be killed or injured through these surveys. For example, one web posting states that "Seismic air gun testing currently being proposed in the Atlantic will injure 138,000 whales and dolphins and disturb millions more, according to government estimates." This characterization of our conclusion, however, is not accurate; that is actually not what we estimate. I hope that providing background and discussion on BOEM's conclusion and the numbers may help those who follow this issue to understand our position. I'll begin with an overview of a few key legal terms.

Terms of the Marine Mammal Protection Act (MMPA)

Three MMPA terms are key to this conversation. First, a "take" of a marine mammal under the MMPA is defined as follows: "to harass, hunt, capture, or kill, or attempt to harass, hunt, capture, or kill any marine mammal." The MMPA defines the term "harassment" to mean

> "[A]ny act of pursuit, torment, or annoyance which - (i) has the potential to injure a marine mammal or marine mammal stock in the wild [referred to in the MMPA as 'Level A harassment']; or (ii) has the potential to disturb a marine mammal or marine mammal stock in the wild by causing disruption of behavioral patterns, including but not limited to, migration, breathing, nursing, breeding, feeding, or sheltering [referred to in the MMPA as 'Level B harassment']." MMPA Sec. 3 (18).

In other words, a "take" can mean an act that kills or injures a marine mammal, but it can also mean an act that does no more than have the potential to disturb a marine mammal.

Second, it is important to recognize that the MMPA prohibits the take of marine mammals as a result of permitted activities - referred to in the statute as "incidental take" -- unless that take will have no more than "negligible impact." In particular, section 101 (5) of the MMPA prohibits incidental "taking" of a marine mammal, including Level A and Level B harassment, unless the Secretary of Commerce, acting through the National Oceanic and Atmospheric Administration (NOAA), determines that the taking will have no more than "negligible impact" on the species or stocks affected. NOAA regulations define negligible impact to mean "an impact resulting from the specified activity that cannot be reasonably expected to, and is not reasonably likely to, adversely affect the species or stock through effects on annual rates of recruitment or survival." By definition, then, the impact analysis is measured on the "species or stock," not on an individual animal.

Our bureau has stated publicly that it will not consider issuing any air gun seismic survey permits in the Atlantic unless applicants have first obtained an MMPA authorization from NOAA, including the required finding of no adverse effect on marine mammal species or stocks.

"Optimum sustainable population" or OSP is a third key MMPA concept. Obtaining optimum sustainable populations is a stated goal of the MMPA, and OSP is defined by the statute to mean, "with respect to any population stock, the number of animals which will result in the maximum productivity of the population or the species, keeping in mind the carrying capacity of the habitat and the health of the ecosystem of which they form a constituent element." OSP is about populations, not individuals.

No Documented Scientific Evidence of Adverse Effects on Population Sustainability

With these three terms in mind, it is critically important to understand that BOEM's conclusion in our August 2014 *Science Note, and* its *Programmatic Environmental Impact Statement (PEIS),* refers to effects on population sustainability, rather than effects on individual animals. We know from studies by BOEM and others that marine mammals can react to sound, sometimes moving away and sometimes changing their vocalizations. One prominent concern is whether anthropogenic sounds may "mask" communications between some marine mammals. However, as BOEM concluded in the PEIS, and reiterated in the 2014 *Science Note*, potential links between these effects and the sustainability of species or stocks have not been demonstrated. For example, because of its abundance, the bottlenose dolphin heads the class in number of potential exposures to air gun sound levels with potential effects on behavior. Yet Federal stock assessments for the dolphin do not identify air gun seismic surveys as adversely impacting stock sustainability in the Gulf of Mexico, where air gun surveys are routine.

It is also important to understand that BOEM does not expect that 138,000 individual marine mammals, or anything close to that number, will have their hearing injured by air guns if seismic surveys are permitted on the Atlantic Outer Continental Shelf. BOEM published numbers for potential air gun survey "takings" of marine mammals in its PEIS. The highest numbers estimated for a particular species are for the bottlenose dolphin, as noted above, and in its case the PEIS estimated potential for Level A takings of up to 11,748 individual bottlenose dolphins a year from air gun surveys and potential for up to 1,151,442 Level B takings. But the number of modeled "takes" in the PEIS is by design highly over-estimated to err on the side of protection, and it does not consider key mitigation measures that will be required to prevent "taking." One such requirement, for example, is that seismic survey vessels maintain "exclusion zones" around vessels whose boundaries are set to avoid any injury to marine mammal hearing. If a marine mammal enters the zone, or appears on a course to enter, trained observers call for immediate shut down of the air guns until the animals are clear of the area. Therefore, even those numbers included in the PEIS are far in excess of those takes we anticipate, given the mitigation measures that will be employed.

Need for More Research

A final point warrants mention. BOEM does not and should not assume that lack of evidence for adverse population-level effects of air gun surveys means that those effects may not occur. What we know is a function of the effort and intelligence put into evaluating effects as well as what is actually happening in nature. Since 1998, BOEM has invested over $50 million on protected species and noise-related research, including marine mammals. We have also convened workshops for acoustic experts to help us identify questions for future research. But BOEM needs to keep looking -- hard and well -- for adverse effects of offshore oil and gas activities on the environment, including sound. And we have asked our environmental studies program to make this a priority.

I'll conclude by noting that BOEM's 2014 *Science Note* has been cited publicly by both industry and environmental NGOs alike in presenting their respective positions on seismic surveys. BOEM is responsible for providing environmental safeguards in development of offshore resources, and our Science Note was intended to help the public understand our thinking on that task. I hope this follow-on *Science Note* is a helpful explanation.

As always, your feedback is important to us, so please feel free to contact us.

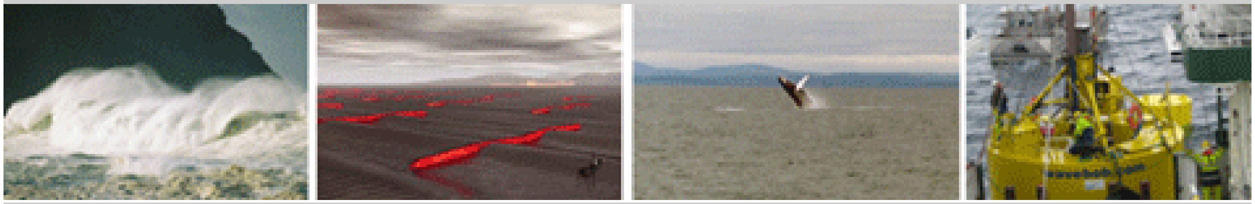
Sincerely,

William Y. Brown

Chief Environmental Officer, Bureau of Ocean Energy Management

———————————————————————

The Bureau of Ocean Energy Management (BOEM) promotes energy independence, environmental protection and economic development through responsible, science-based management of offshore conventional and renewable energy resources.



BOEM
Bureau of Ocean Energy Management

SCIENCE NOTES

Applied science for informed decision making

August 22, 2014

Dear Reader:

It has been just over a month since BOEM released a Record of Decision -- or ROD -- on the Mid- and South Atlantic Geological and Geophysical (G&G) Activities Programmatic Environmental Impact Statement, or PEIS for short. And there's been a lot of attention on both sides of this complex issue. I wanted to take some time to clear up a few misperceptions about the bureau's decision and what it means.

As a scientist who has spent a good part of my career working in non-governmental environmental organizations and in industry, I understand and appreciate advocacy. At the same time, I believe that everyone benefits by getting the facts right.

BOEM has the legal responsibility to protect marine species and ecosystems from harm by the energy exploration and development which we regulate, and that is a responsibility which I embrace without reservation. Since 1998, BOEM has partnered with academia and other experts to invest more than $50 million on protected species and noise-related research. The bureau has provided critical studies on marine mammals, such as researching seismic survey impacts on sperm whales, and BOEM has conducted many expert stakeholder workshops to discuss and identify information needs on acoustic impacts in the ocean.

As noted below, the bureau's decision requires a set of protective measures that will be used in site-specific permits for any future G&G activities in the Atlantic. BOEM will conduct site-specific environmental reviews for any permit applications. These reviews will include coordination and consultation with federal, state and tribal authorities under a variety of additional statutory requirements. In particular, any "taking" of a marine mammal requires authorization from the National Oceanic and Atmospheric Administration, or NOAA, separately from BOEM, and that authorization requires NOAA to find that there is no more than "negligible impact" and no adverse effects on marine mammal species or stocks.

Below, please find our latest edition of *Science Notes* that I hope will help to clarify the facts on BOEM's recent decision and the science behind it. As always, your feedback is important to us, so please feel free to contact us at boempublicaffairs@boem.gov.

Sincerely,

William Y. Brown

Chief Environmental Officer, Bureau of Ocean Energy Management

The Science Behind the Decision

Answers to Frequently Asked Questions about the Atlantic Geological and Geophysical Activities Programmatic Environmental Impact Statement (PEIS)

Will air guns used in seismic surveys kill dolphins, whales and sea turtles and ruin coastal communities?

Bottlenose dolphin from the Atlantic AMAPPS study.

To date, there has been no documented scientific evidence of noise from air guns used in geological and geophysical (G&G) seismic activities adversely affecting marine animal populations or coastal communities. This technology has been used for more than 30 years around the world. It is still used in U.S. waters off of the Gulf of Mexico with no known detrimental impact to marine animal populations or to commercial fishing.

While there is no documented case of a marine mammal or sea turtle being killed by the sound from an air gun, it is possible that at some point where an air gun has been used, an animal could have been injured by getting too close. Make no mistake, airguns are powerful, and protections need to be in place to prevent harm. That is why mitigation measures -- like required distance between surveys and marine mammals and time and area closures for certain species -- are so critical.

Is it true that the air guns are 100,000 times louder than a jet, and if so, won't they kill or deafen marine life?

A large air gun is loud, although it is not 100,000 times louder than a jet. Measured comparably in decibels, an air gun is about as loud as one jet taking off. Scientists who specialize in acoustics confirm that sounds in water and sounds in air that have the same pressures have very different intensities (which is a measure of energy produced by the source) because the density of water is much greater than the density of air, and because the speed of sound in water is much greater than the speed of sound in air. For the same pressure, the higher density and higher speed make sound in water less intense than sound in air.

We do not know what a whale, dolphin, or turtle actually experiences when it hears an air gun. Many marine mammal species -- but not the baleen whales including North Atlantic right whales -- have reduced sensitivity to sound signals that are in the same frequency range as airplanes and air gun arrays. Some whales appear to move away from surveys, indicating that they probably don't like the noise, but bottlenose dolphins have often been observed swimming toward surveying vessels, and ride bow waves along the vessels.

Is it true that the government's own scientists expect 100,000 injuries or deaths of marine life if seismic surveys go forward?

This statement misrepresents the facts. When our scientists began to look at possible impacts of seismic surveys, they first looked at what might happen if no measures were taken to mitigate or avoid possible injury to marine mammals. Next they began to look at what could be done to avoid harm, such as avoiding migration routes and stopping surveys if vessels get close enough to marine mammals to possibly injure their hearing.

After a thorough, public process, the Department selected a preferred alternative that included the most restrictive mitigation measures that would allow surveys to take place. We expect survey operators to comply with our requirements and, if they do, seismic surveys should not cause any deaths or injuries to the hearing of marine mammal or sea turtles.

Another source of confusion is about what a "take" is. As defined by Federal law, a "take" of a marine mammal, unsurprisingly, includes causing its death. However "take" also includes not only injury to hearing but also any disturbance to an animal that may disrupt its behavior. BOEM has published numbers of potential "takes," and the highest numbers are based on potential for behavioral effects, such as temporarily leaving survey areas. These behavioral effects have not been linked to negative impacts on populations. In fact, the same Federal law defining "take" of a marine mammal prohibits all taking unless the NOAA has determined that the taking will have no more than "negligible impact" and no adverse effects on marine mammal species or stocks.

BOEM cannot authorize air gun surveys which "take" marine mammals unless the surveys are also authorized by NOAA and meet this requirement. BOEM also consulted with both NOAA and the U.S. Fish and Wildlife Service under the Endangered Species Act to develop mitigations that would limit any potential impacts to endangered and threatened species, including baleen whales and sea turtles.

Does this decision mean that the federal government is opening the entire Atlantic coast up for offshore oil and gas drilling?

The decision to authorize G&G activities for all three program areas (oil and gas, renewable energy and marine minerals) does not authorize leasing for oil and gas exploration and development in the Atlantic. Those decisions will be addressed through the development of the next Five Year Program for oil and gas leasing. BOEM is at the beginning of the process to develop that program pursuant to the Outer Continental Shelf Lands Act. The planning process will take two-and-a-half to three years to complete and will offer many opportunities for the public to provide input.

Completion of the PEIS and BOEM's selection of the strongest environmental alternative and its documentation in the decision (ROD) do not themselves authorize any specific activities. Nor does this make any decision about future leasing.

The bureau's decision requires a set of protective measures that will be used in site-specific permits for any future G&G activities in the Atlantic. BOEM will conduct site-specific environmental reviews for any permit applications. These reviews will include coordination and consultation with federal, state and tribal authorities under a variety of additional statutory requirements. In particular, any "taking" of a marine mammal requires authorization from NOAA, separately from BOEM, and that authorization requires NOAA to find that there is no more than "negligible impact" and no adverse effects on marine mammal species or stocks.

Click here for the fact sheet on Atlantic G&G Surveys Record of Decision.

- BOEM -

The Bureau of Ocean Energy Management (BOEM) promotes energy independence, environmental protection and economic development through responsible, science-based management of offshore conventional and renewable energy resources.

APPENDIX B

The National Academies of SCIENCES ENGINEERING MEDICINE

THE NATIONAL ACADEMIES PRESS

Approaches to Understanding the Cumulative Effects of Stressors on Marine Mammals

DETAILS

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Approaches to Understanding the Cumulative Effects of Stressors on Marine Mammals

Committee on the Assessment of the Cumulative Effects of Anthropogenic Stressors on Marine Mammals

Ocean Studies Board

Division on Earth and Life Studies

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Preface

Assessing the cumulative effects of multiple stressors is a top-priority problem in marine ecology. An important marine policy paper by Rudd (2014) surveyed more than 2,000 ocean scientists and policy makers from nearly 100 countries, asking them to prioritize the most important questions for the ocean environment. Out of 67 questions, the top priority was "How will the individual and interactive effects of multiple stressors (e.g., ocean acidification, anoxia, warming, fishing, and pollution) affect the capacity of marine ecosystems and species to adapt to changing oceans?" The topic of cumulative effects was chosen by the federal agencies that funded this report because assessing cumulative effects has been an important part of U.S. regulations protecting marine mammals since the 1970s, but the approaches used have little predictive value. Marine mammal populations are affected by a large number of natural and anthropogenic stressors. This report was tasked with focusing on sound and other stressors when evaluating cumulative effects on marine mammals. If cumulative effects cannot be accounted for, then unexpected adverse impacts from interactions between stressors pose a risk to marine mammal populations and the marine ecosystems on which people and marine mammals depend.

Assessing cumulative effects is not only important, it is also a problem that has proven nearly impossible to solve. Scientists and managers involved in these assessments confront data gaps concerning the dosages of all stressors to which marine mammals are exposed, and a lack of dose–response functions to predict effects of single stressors. For ethical and practical reasons, there are no studies in marine mammals on interactions between stressors. Studies in other marine organisms show that these stressors often interact, but their cumulative effects are extremely difficult to predict.

The audience intended for this report includes stakeholders, managers, policy makers, and scientists. This report has developed approaches to analyze how stressors exert their effects on individuals, populations, and ecosystems to help guide research on cumulative effects in the future. The report aims to help managers decide when cumulative effects are particularly important, and to help guide decisions about which stressors or combinations of stressors to reduce when this is necessary to protect marine mammal populations.

Recognizing that quantitative prediction of cumulative effects of stressors on marine mammals is not currently possible, this committee developed a conceptual framework for assessing the population consequences of multiple stressors. The framework uses indicators of health that integrate short-term effects of different stressors that affect survival and reproduction. The report explores a variety of methods to estimate health, stressor exposure, and responses to stressors. The committee also developed a decision tree for determining when cumulative effects are particularly important for managing a marine mammal population.

Many stressors that affect marine mammals are themselves affected by larger-scale ecological drivers. For example, ocean climate is an ecological driver that changes the exposure of marine life to the stressors of warming and ocean acidification. Similarly predators, prey, and competitors of marine mammals are potential stressors whose distributions are affected by ecological interactions. The committee explored the use of interaction webs to help ensure that important ecological interactions, including indirect interactions, are included in assessments of cumulative effects.

Cumulative effects must be evaluated in environmental assessments of planned activities, but this evaluation is equally important for selecting management actions once populations or ecosystems are found to be at risk of adverse impacts. In this case, the critical issue is to decide what combination of stressors to reduce in order to bring the population or ecosystem into a more favorable state. Whatever increases in stressors may have created the risk, the best management action may require reducing a different combination of stressors. For example, if a persistent toxicant increases mortality of a species but cannot be removed from the ocean, the best management action might involve reducing fishing bycatch, which can be controlled. This broadening of management approaches could be a particularly important result of assessing cumulative effects.

Recognizing difficulties with measuring trends in marine mammal populations, the report explores early warning indicators for adverse impacts, including health and population measures. Measures of health that indicate which stressors caused an effect would be particularly useful for managing the effects. The committee hopes that this report may help direct the development of methods to identify when cumulative effects pose a risk of driving a population or ecosystem into an adverse state, and to develop management strategies that can select stressors whose reduction will minimize this risk. The committee recognizes the enormous scientific challenge posed by these two problems, but their importance justifies significant effort to solve them.

This committee met four times and held a workshop in the National Academies of Sciences, Engineering, and Medicine's Beckman Center in Irvine, California. On behalf of the committee, I would like to thank the speakers invited to the workshop and audience members who shared their insights with the committee. On behalf of the committee, I would also like to thank the study directors who oversaw this report, first Deborah Glickson and then Kim Waddell, and the director of the Ocean Studies Board, Susan Roberts, along with other members of the staff whose contributions were essential for our meetings and development of the report.

Academies reports are designed to address problems that are both important and difficult, but this committee was tasked with a more difficult and broad-ranging problem than I have encountered in previous studies on marine mammals and sound. The committee explored many approaches to evaluating cumulative effects, and, in response to this task, this report is more extensive than the others on marine mammals and sound. The committee members and members of the National Academies staff working on this report not only had to write about and review a large body of information, but were all stretched to work outside of their disciplines. I would like to thank the committee members for their generosity in working together so well to meet the challenge of the statement of task, exploring creative solutions while providing a broad and critical review of the problem of evaluating cumulative effects in marine mammals.

Peter L. Tyack, *Chair*
Committee on the Assessment of the Cumulative
Effects of Anthropogenic Stressors on Marine Mammals

Acknowledgments

This report was greatly enhanced by discussions with participants at the committee's meetings as part of this study. The committee would like to acknowledge, especially, the efforts of those who gave presentations at the committee meetings: Kim Anderson (Oregon State University), Jesse Barber (Boise State University), Steve Beissinger (University of California, Berkeley), Shekhar Bhansali (Florida International University), Tiffini Brookens (Marine Mammal Commission), Mitch Eaton (U.S. Geological Survey), Tim Essington (University of Washington), Jason Gedamke (National Oceanic and Atmospheric Administration), Horst Greczmiel (Council on Environmental Quality), Carrie Kappel (University of California, Santa Barbara), Sara Maxwell (Old Dominion University), Jonna Mazet (University of California, Davis), Jim Price (Bureau of Ocean Energy Management), and Mike Weise (Office of Naval Research).

This report has been reviewed in draft form by individuals chosen for their diverse perspectives and technical expertise. The purpose of this independent review is to provide candid and critical comments that will assist the institution in making its published report as sound as possible and to ensure that the report meets institutional standards for objectivity, evidence, and responsiveness to the study charge. The review comments and draft manuscript remain confidential to protect the integrity of the process. We wish to thank the following individuals for their review of this report:

MICHAEL CASTELLINI, University of Alaska, Fairbanks
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CATHERINE WANNAMAKER, Southern Environmental Law Center

Although the reviewers listed above have provided many constructive comments and suggestions, they were not asked to endorse the report's conclusions or recommendations, nor did they see the final draft of the report before the release. The review of this report was overseen by Andrew R. Solow, Woods Hole Oceanographic Institution Senior Scientist, and John Dowling, Harvard University Professor of Neurosciences. They were responsible for making certain that an independent examination of this report was carried out in accordance with institutional procedures and that all review comments were carefully considered. Responsibility for the final content of this report rests entirely with the authoring committee and the institution.

Contents

Appendixes

Summary

Marine mammals face a large array of stressors, including loss of habitat, chemical and noise pollution, and bycatch in fishing, which alone kills hundreds of thousands of marine mammals per year globally. To discern the factors contributing to population trends, scientists must consider the full complement of threats faced by marine mammals. Once populations or ecosystems are found to be at risk of adverse impacts, it is critical to decide which combination of stressors to reduce to bring the population or ecosystem into a more favorable state. Assessing all stressors facing a marine mammal population also provides the environmental context for evaluating whether an additional activity could threaten it. Under the National Environmental Policy Act of 1969 (NEPA), federal agencies are directed to assess the environmental impacts of their actions, considering direct, indirect, and cumulative effects. Cumulative effects are defined by the U.S. Council on Environmental Quality as "the incremental impact of the action when added to the other past, present and reasonably foreseeable actions" that might interact with a proposed action. Although significant progress has been made in understanding the responses of marine mammals to specific stressors such as noise and toxins, it is not yet possible to provide quantitative estimates of the impact of repeated exposure to a stressor or to predict how different stressors will interact to affect individuals and populations of marine mammals.

The Office of Naval Research, the National Marine Fisheries Service, the Bureau of Ocean Energy Management, and the U.S. Marine Mammal Commission funded the present study in order to review the understanding of cumulative effects of anthropogenic stressors, including sound, on marine mammals and to identify new approaches that may improve the ability to estimate cumulative effects. The statement of task is detailed in Box S.1.

CUMULATIVE EFFECTS

The definition of *cumulative effects* under the implementing regulations for NEPA focuses on the incremental effect of a proposed human action when added to those of other human actions. In contrast, most biologists view cumulative effects similarly to the U.S. Environmental Protection Agency's view of cumulative risk, which focuses on the individual animal or population, with effects accumulating when animals are repeatedly exposed to the same or different stressors. In this ecotoxicology-type approach, a noise source would be considered one of a number of stressors experienced by marine mammals and one component of an overall aggregate exposure to noise. Cumulative risk would derive from the combination of noise and other anthropogenic stressors, such as chemical pollution, marine debris, introduced pathogens, fishing, and warming or lower pH induced by carbon dioxide emissions, as well as natural stressors, such as increased presence of predators, pathogens, parasites, or reduced availability of prey due to natural ecological interactions.

In this report *aggregate exposure* is defined as the combined exposure to one stressor from multiple sources or pathways and *cumulative risk* as the combined risk from exposures to multiple stressors integrated over a defined relevant period: a day, season, year, or lifetime.

Cumulative risk from exposure to multiple stressors cannot be predicted based on existing scientific theory and data for individual marine mammals or their populations. The Committee developed a Population Consequences of Multiple Stressors (PCoMS) model to provide a conceptual framework for the challenging task of assessing the risks associated with aggregate exposures to one kind of stressor, such as sound, and the cumulative exposure associated with sound and other stressors. To broaden the analysis of cumulative effects to include multiple species and ecosystems, the concept of interaction webs was introduced.

The report distinguishes between two kinds of stressors: an *intrinsic stressor* (e.g., fasting), which is an internal factor or stimulus that results in a significant change to an animal's homeostatic set points,[1] and an *extrinsic stressor* (e.g., noise or a pathogen), which is a factor in an animal's external environment that creates stress in an animal. It also distinguishes between stressors, defined by how they influence an individual animal, and ecological drivers, which affect levels of organization from populations to ecosystems. An *ecological driver* is defined as a biotic or abiotic feature of the environment that affects multiple components of an ecosystem directly and/or indirectly by changing exposure to a suite of extrinsic stressors. Ecological drivers for marine mammals include loss of keystone or foundational species, variations in ocean climate (such as El Niño events), and climate change.

BOX S.1
Statement of Task

The National Academies of Sciences, Engineering, and Medicine's Ocean Studies Board has previously convened four highly successful panels on the subject of biological effects of manmade underwater sound, which produced a progressive series of reports published in 1994, 2000, 2003, and 2005, with the latest report focusing on the potential for biologically significant effects on marine mammal populations. Sound, however, is only one of a variety of potential anthropogenic or natural stressors that marine mammals encounter, and it is often evaluated in isolation without consideration of the effects of other stressors (e.g., fishing, climate change, pollution, etc.), or consideration of how these other stressors may affect an animal's response to sound exposure. The committee will conduct a workshop and review the present scientific understanding of cumulative effects of anthropogenic stressors on marine mammals with a focus on anthropogenic sound. The committee will assess current methodologies used for evaluating cumulative effects and identify new approaches that could improve these assessments. The committee will examine theoretical and field methods used to assess the effect of anthropogenic stressors for

- short or infrequent exposure in the context of other known stressors (i.e., multiple stressors, both natural and anthropogenic) and
- chronic exposure in the context of other known stressors.

The review of methodologies will begin by focusing on ways to quantify exposure-related changes in the behavior, health, or body condition of individual marine mammals and assess the potential to use quantitative indicators of health or body condition to estimate changes in vital rates and, in turn, estimate the potential population-level effects.

Effects of Sound

In this study, the committee was asked to place sound in the context of other stressors to which marine mammals may be exposed. The National Research Council (NRC) report *Marine Mammal Populations and Ocean Noise* (NRC, 2005) noted that "[n]o scientific studies have conclusively demonstrated a link between exposure to sound and adverse effects on a marine mammal population." That statement is still true, largely because these impacts are so difficult to demonstrate, but the intervening decade has seen an increasing number of studies showing the effects of ocean noise on individual marine mammals. Under the U.S. Marine Mammal Protection Act (MMPA), regulation of the effects of human activities on marine mammals requires determining the number of individual animals expected to be "taken"[2] lethally, by injury or by harassment. One current method is to set an all-or-nothing threshold at the sound pressure level corresponding with an estimated probability of response of 50% from the dose–response function. However, the radiation of sound from point source emissions typically exposes many more animals at sound levels below this threshold compared with the number exposed to higher sound levels. Hence, using this threshold leads to potentially significant underestimates of the total number of animals taken. An "effective received level" can be calculated that gives a more realistic take estimate. Still, the effects of sound on marine mammals cannot reliably be condensed into a single estimate of the number of animals affected by a given exposure. Changes in transmission patterns of sound in the ocean, distribution of animals, variable responsiveness of individual animals, and temporal, spatial, and social determinants of response all create uncertainty in the number of animals that will respond behaviorally or physiologically to any defined sound stimulus. Including measures of uncertainty, such as confidence intervals for estimates of predicted take, would be more consistent with the state of knowledge than providing a single number for the MMPA take estimates.

Estimating the effect of sound on marine mammals requires understanding the relationship between acoustic dosage and the probability of behavioral or physiological

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[1] Homeostasis is a characteristic of a system that regulates its internal environment and tends to maintain a stable, relatively constant condition of properties. The normal value of a physiological variable is called its set point.

[2] A marine mammal "take" is the act of hunting, killing, capture, and/or harassment of any marine mammal, or the attempt at such.

responses of varying degrees of severity. The criterion used under the MMPA for injury induced by sound is noise-induced hearing loss. The distribution of sound exposures that cause permanent hearing loss is estimated from studies of noise levels that cause the onset of temporary shifts in the hearing threshold (temporary threshold shift [TTS] onset) followed by the increase in the amount of TTS with increasing levels of noise. Currently, data on this relationship exist for one species of fur seal, two species of true seals, two species of mid-frequency dolphins, and two species of high-frequency porpoises. Only a few individuals (one to five) of each species have been tested, and within hearing groups there is wide variation in TTS onset and growth with increasing levels of noise. This variation indicates that the physiological effects of sound cannot be generalized based on testing of a few species of marine mammals but will require studies in more individuals of more species. Understanding how the physiological effects of sound become permanent hearing loss requires audiogrametric measurements. Because there are no audiograms available for baleen whales, physiological sound impacts are estimated based on indirect evidence, such as modeling how sound interacts with tissues in the head, estimated historical ocean noise thresholds, and data from other cetacean hearing groups.

For the recommendations that follow, the chapter number is given where supporting text for a particular recommendation can be found.

Recommendation: Uncertainties about animal densities, sound propagation, and effects should be translated into uncertainty on take estimates, for example, through stochastic simulation. (Chapter 2)

Recommendation: Additional research will be necessary to establish the probabilistic relationships between exposure to sound, contextual factors, and severity of response. (Chapter 2)

Significant progress has been made in developing experiments that can estimate acoustic dose–behavioral response relationships in marine mammals. The response criteria selected for dose–response studies have typically had low severity so as not to harm the subjects, but high enough to act as indicators of harassment under the MMPA. However, in the course of these studies some high-severity responses have been observed for signals that were barely audible. The severity levels were established based on assumed effects on individual fitness, and thus severe responses to low sound levels raise concerns regarding population consequences. This will require research to establish (1) the relationship between levels of exposure and the severity of response, (2) the role of behavioral context in determining the dose–response relationship and the response severity, and (3) the most appropriate acoustic dosage measures for sound exposure.

EFFECTS OF MULTIPLE STRESSORS

There is considerable evidence for single-factor stressor effects on marine mammals. Most of these involve physiological and behavioral responses. Dose–response functions have been estimated for a limited number of single stressors. Particular progress has been made in understanding the effects of anthropogenic sound on behavior. Experiments on a few species have estimated dose–response functions, and, once responses have been characterized in this way, monitoring can be used to estimate the scale of effects from sound-producing activities. Studies of effects of pollutants on marine mammal health and reproduction have also estimated dose–response functions, but there are fewer data on dose–response relationships for other stressors.

While the relationship between the dose of a single stressor and the response of an individual animal is relatively straightforward to predict given sufficient data, the addition of a second stressor can add considerable complexity due to the potential for interaction between the stressors or their effects. Stressors may interact in a synergistic or antagonistic manner, where the resulting response is larger or smaller, respectively, than the sum of the individual stressor responses.

Insight about cumulative effects in the individual can be gained by considering mechanisms at the molecular, cellular, and organ system levels. When stressors act through a common pathway, this provides a high potential for interaction because the stressors may provoke physiological perturbations within the same organ or neuroendocrine system. One common assumption of ecotoxicologists is that, if two or more stressors act through a common molecular mechanism, then their doses can be summed to provide a cumulative dose that can then be used with a single dose–response function (dose addition). Many dose–response functions are sigmoidal in shape or are otherwise nonlinear, and in these cases the sum of two doses may produce a response that is greater or less than the added responses to each stressor alone (response addition). A simple example to illustrate the complexity introduced when a dose–response function is nonlinear is discussed below.

Consider two stressors that act through a common molecular mechanism and are therefore considered eligible for dose addition. After correcting for different strengths (e.g., a toxicity factor for chemical stressors), the doses of the two stressors can be added to give a combined dosage and compared to a dose–response function (see Figure S.1). Stressor A has an effect of 0.10 given a dose of 40 units (see Figure S.1a), and stressor B has an effect of 0.20 given a dose of 60 units (see Figure S.1b). If the responses were additive (response addition), then the response to stressor A and B combined is expected to be 0.30. However, due to the sigmoidal shape of the dose–response function, the added doses of the two stressors (100 units) produces an effect of 1.0, more than threefold higher than the sum of the

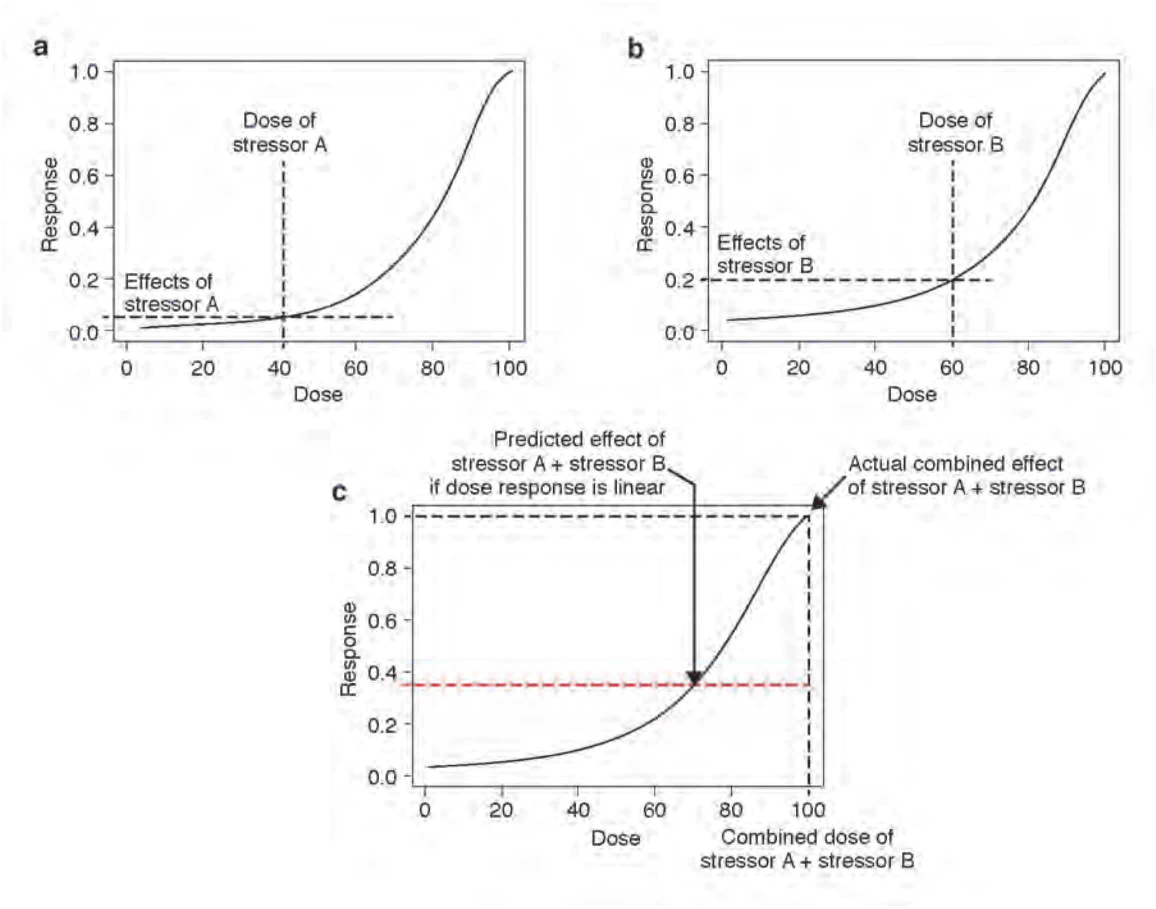

FIGURE S.1 This figure illustrates how the potential for interaction between two stressors (A and B) that share a common mechanism of action depends on the form of the dose–response relationship. (a) Effect of stressor A alone. (b) Effect of stressor B alone. (c) Effect of a combined dose of stressor A and stressor B, obtained by adding the dose from stressor A to that of stressor B (dose addition). The effect predicted from the dose–response relationship shared by the two stressors is more than three times higher than the prediction if their effects are assumed to be additive (red line).

individual responses (see Figure S.1c). Therefore, although these stressors are considered additive in terms of dosage (dose addition), they produce a synergistic response. Note that this same phenomenon could also occur with aggregate exposure to a single stressor. Even for this simple situation, a prediction cannot be made of the effects of most stressors unless the dosages, the relative strengths of the stressors, and the dose–response functions are known.

The interaction of stressors that act through different mechanisms but still involve a common adverse outcome pathway may be more difficult to predict due to the complexities of signaling pathways and the existence of feedback loops. For example, stressors such as noise, prey limitation, and some chemical pollutants can induce responses involving the neuroendocrine system known as the hypothalamic-pituitary-adrenal (HPA) axis that controls reactions to stress and regulates many body processes, albeit potentially through differing mechanisms. Chronic activation or perturbation of the HPA axis may be an important mechanism through which cumulative effects arise, and the nature of these effects will be difficult to predict. In cases such as this where there are common adverse outcome pathways but potentially differing mechanisms, the form of interaction between two stressors could be estimated by determining the dose–response relationships for one stressor at different dosages of the second stressor. However, this type of study would be extremely difficult if not impossible to conduct, particularly when more than two stressors are involved, and mechanistic models may be a more appropriate approach to elucidate potential effects. Unfortunately, mechanistic models generally require a detailed understanding of the biochemical and physiological systems, and this is often lacking for marine mammals.

A review of the literature revealed that many stressors whose effects are mediated through common adverse outcome pathways are therefore more likely to interact. The examination of common adverse outcome pathways underscores the importance of understanding and detecting changes at lower levels of biological organization, such as at the cellular or organ response level, before they exert potentially irreversible effects at individual or population levels. However, it is also imperative to collect information to understand the linkages and processes by which such lower-level responses eventually translate into individual or population-level impacts.

The influences of multiple stressors on marine mammals might be inferred from studies of other species, such as nonmammalian marine species or terrestrial mammals. However, this can be problematic because marine mammals have evolved unique morphologies, behaviors, and physiologies as adaptations for life at sea.

Most existing research on interactions between effects of stressors on marine systems involves factorial experiments with species or systems in settings where treatments can be replicated and controlled. Factorial experiments are useful for detecting the presence of interactions but, because such systems are usually only exposed to one level of each stressor, they rarely provide sufficient information to predict responses at varying levels of stressors present in nature. Meta-analyses of results from studies of multiple stressors on various marine species have been conducted, but no general pattern has emerged for predicting how the effects of stressors will interact. Findings from each specific study were categorized as additive (i.e., noninteractive), synergistic, or antagonistic. One review paper reported that synergy is more common when more than two stressors are added to a system; another study found no evidence of antagonistic interactions between physiological responses. Beyond these generalities, the committee found no information to help predict the influences of multiple stressors on marine mammals. Given the difficulty in predicting interactions, cumulative effects assessments often assume that stressor effects are additive. However, work on other species indicates that this assumption is often wrong.

A rigorous approach for testing interactive effects of multiple stressors involves factorial experiments using a range of levels of each stressor coupled with some tests of mixtures of stressors. But for both practical and ethical reasons, such experimental approaches are often not possible for marine mammals, in which case inferences must be based on quasi-experiments: patterns associated with stressor variation in space or time. Although such data are subject to confounding and thus multiple interpretations, reasonably strong inferences are often possible from time-series analyses and weight of evidence approaches.

One type of single-stressor experimental study design could select subjects from the wild population to sample the cumulative effects of exposure to sound along with the combination of stressors currently found in that population. If this type of study adds one stressor to subjects in the wild whose exposure to other stressors can be documented, the cumulative effects of the single stressor then can be evaluated in the context of the full complement of environmental stressors. The interpretation of these single-stressor experiments in terms of cumulative effects is difficult because the exposures to preexisting stressors are difficult to quantify. Also experimental addition of a stressor is limited for ethical reasons to stressors such as sound, where the added stressor can be controlled in terms of both intensity and duration of exposure. In situations where the current pattern of exposure to stressors is expected to change in the future beyond the levels currently experienced, such as those caused by changes in ocean climate, this approach for studying cumulative effects will be inadequate.

The exposure of marine mammals to stressors has been estimated by mapping stressors in both space and time. However, in order to understand cumulative effects, mapping of stressors needs to be accompanied by mapping the distribution of marine mammal species of concern, because stressors must overlap with the species to exert an effect. Another approach, which is common for chemical stressors, is to sample tissue from a marine mammal to characterize its dosage of the stressor. Biopsies are now a standard remote sampling method for marine mammals that cannot be handled. The development of new methods for remote sampling of blood and other tissues for estimating dosage of stressors from marine mammals at sea are included in a recommendation later in this summary. On-animal dosimeters could also provide a time series of stressor exposure measurements for individual animals.

A MODEL FOR HEALTH AND POPULATION CONSEQUENCES OF MULTIPLE STRESSORS

The PCoMS model (see Figure S.2) developed in this report provides a framework for exploring pathways from stressor exposure to effects on health to effects on populations. Following the general structure of the Population Consequences of Acoustic Disturbance model developed in NRC (2005), PCoMS documents the pathways from exposures to stressors through their effects on physiology, behavior, and health to their effects on vital rates and population dynamics. A key component of this framework is an assessment of the health of individuals. A variety of health indices, including allostatic load, energy stores, immune status, organ status, stress levels, contaminant burden, and parasite load, are discussed. Appropriate health indices integrate the potential effects of physiological and behavioral responses to multiple stressors on fitness over a time scale that is longer than the duration of the responses themselves but shorter than the response time of vital rates. Such indices can provide early indicators of risk of reduced survival and reproduction before an actual alteration in these rates and can increase

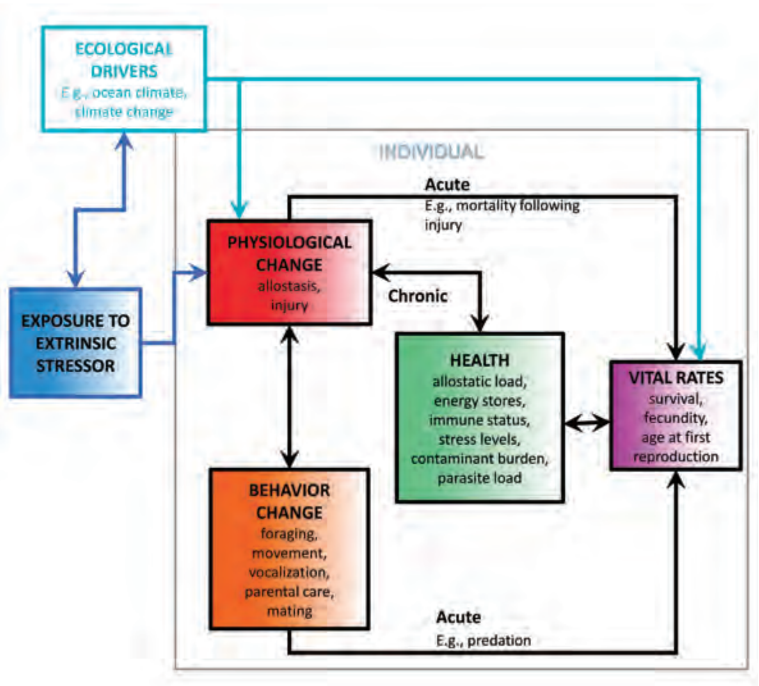

FIGURE S.2 The Population Consequences of Multiple Stressors (PCoMS) framework for a single individual exposed to one stressor. Each compartment in the framework represents one or more quantities (variables) that evolve over time. Compartments are connected by arrows that represent causal flows ("transfer functions" in the terminology of NRC [2005]). For each individual, changes in physiology may result in changes in behavior (such as movement away from a sound source and cessation of feeding), which may in turn affect physiology.

understanding of the mechanisms by which these stressors affect fitness.

The committee developed a number of research recommendations that are designed to address the PCoMS model and measures of stressors and health:

Recommendation: Future research initiatives should include efforts to develop case studies that apply the PCoMS framework to actual marine mammal populations. (Chapter 5)

These studies will need to estimate exposure to multiple stressors, predict changes in behavior and physiology from those stressors, assess health, and measure vital rates in order to parameterize the functional relationships between these components of the framework. Where possible, the data on changes in demography, population size, and the health of individuals collected in these studies should be used to improve estimates of the parameters of the PCoMS model and reduce uncertainty.

Recommendation: Future research initiatives should support evaluation of the range of emerging technologies for sampling and assessing individual health in marine mammals, and identification of a suite of health indices that can be measured for diverse taxa and that best serves to predict future changes in vital rates. (Chapter 8)

Potentially relevant measures include hormones, immune function, body condition, oxidative damage, and indicators of organ status, as well as contaminant burden and parasite load. New technology for remotely obtaining respiratory, blood, and other tissue samples and for remote assessment (e.g., visual assessment of body condition) should also be pursued.

Comprehensive health assessments are not only a critical component of the PCoMS framework, but they can also be used to serve as early warning indicators of risk before the consequences have population-level effects. There are some populations of marine mammals where periodic health assessments can include a sufficient sample of individuals to assess population health. To optimize usefulness for management, there is a need to develop databases of stressors and effects measured using established standards. For species that cannot be handled, methods are not currently available to obtain the samples used to assess health.

Establishing baseline values of health indices and their associations across life history stages in marine mammal species will provide critical information for assessing individual and population health. Cross-sectional sampling and repeated sampling from the same individuals of blood or other tissues during critical life-history phases can help to document exposure to and health effects of extrinsic stressors within the context of annual cycles and life cycles of intrinsic stressors. Long-term studies of known individuals are required for longitudinal studies.

Recommendation: Agencies charged with monitoring and managing the effects of human activities on marine mammals should identify baselines and document exposures to stressors for high-priority populations. (Chapter 8)

High-priority populations should be selected to include those likely to experience extremes (both high and low) of stressor exposure in order to increase the probability of detecting relationships. This will require stable, long-term funding to maintain a record of exposures and responses that could inform future management decisions. Information on baselines and contextual variables is critically important to interpreting responses.

Recommendation: A real-time, nationally centralized system for reporting marine mammal health data should be established. (Chapter 7)

Recommendation: Standards for measurement of stressors should be developed along with national or international databases on exposure of marine mammals to high-priority stressors and associated health measures that are accessible to the research community. (Chapter 8)

Recommendation: Techniques should be developed that will allow historical trajectories of stress responses to be constructed based on the chemical composition of the large number of baleen whale earplugs and baleen samples in museums or similar natural matrices in other species. Artificial matrices should be studied for their potential to absorb materials (hormones or chemical stressors) and thereby provide a record of exposures and responses to stressors. (Chapter 8)

Recent work on baleen whales has shown that some tissues that lay down layers with time, such as baleen or a waxy earplug, can provide a record of stress, reproductive hormones, and some contaminants for up to the entire lifespan. Large archival collections of such tissues could be analyzed to provide time series of data that could yield critical information on the relationships between contaminants, stress, and reproductive intervals in baleen whales. Other materials that lay down semiannual layers, such as teeth, could be assessed for their potential to record stressor and life-history information over long periods of time. In addition, artificial materials could be tested for their capacity to store chemical stressors and hormones over long enough time periods to test the relationship between exposure to the stressors and response in terms of health or vital rates.

ECOSYSTEM-LEVEL EFFECTS

The committee broadened its review from cumulative effects of stressors on marine mammals to consider how interactions among stressors may affect entire ecosystems. The distribution and abundance of species in an ecosystem are determined by the interactions among and between species and abiotic environmental elements, which together define an interaction web (see Figure S.3).

In an interaction web, species or abiotic elements that affect the distribution and abundance of a selected species are called drivers of the recipient species. When a driver affects the recipient directly, for example, when gill nets entangle and kill marine mammals, this is called a direct effect. When a driver affects a second driver that in turn affects the recipient, this is called an indirect effect. For example, human fisheries might reduce the population of a fish species that feeds on the same prey as a marine mammal. If this reduction in the competitor species increased the abundance of prey for the marine mammal species, it might have an indirect positive effect on the recipient species. Known or suspected drivers for marine mammals include ocean climate, prey limitation, predators, fishing bycatch, toxins, and pathogens. Interaction webs can help identify the suite of factors that need to be considered in evaluating cumulative effects on populations and ecosystems. As with the PCoMS model, interaction webs do not provide an algorithm for predicting cumulative effects; they serve primarily to identify the most important components of any comprehensive model of cumulative effects, including indirect effects. Interaction webs and the PCoMS model would need to include mathematical functions that describe the relationships between the different compartments before they could be used to predict those effects. Estimating these functions will be extremely challenging.

MANAGEMENT OF CUMULATIVE EFFECTS

The critical question for predicting risk of cumulative effects asks what combinations of stressors dosages elevate the cumulative effect enough to pose a risk to populations and ecosystems. The committee's review indicates that the strength of effects cannot currently be predicted based on specific levels of exposure to multiple stressors for marine mammals. Once populations or ecosystems are found to be at risk of adverse impacts, the critical issue for selecting management actions is to decide what combination of stressors to reduce in order to bring the population or ecosystem into a more favorable state. The committee concluded that current scientific knowledge is not up to the task of predicting cumulative effects of different combinations of stressors on marine mammal populations. Even though exposure to multiple stressors is an unquestioned reality for marine mammals, the best current approach for management and conservation is to identify which stressor combinations cause the greatest risk. The committee developed a decision tree that can be used to identify situations where a detailed study of potential cumulative effects should be given a high priority (see Figure S.4). The decision tree was applied to three case studies demonstrating its utility.

Recommendation: Situations where studies of cumulative effects should be prioritized can be identified using tools such as the decision tree developed by the committee

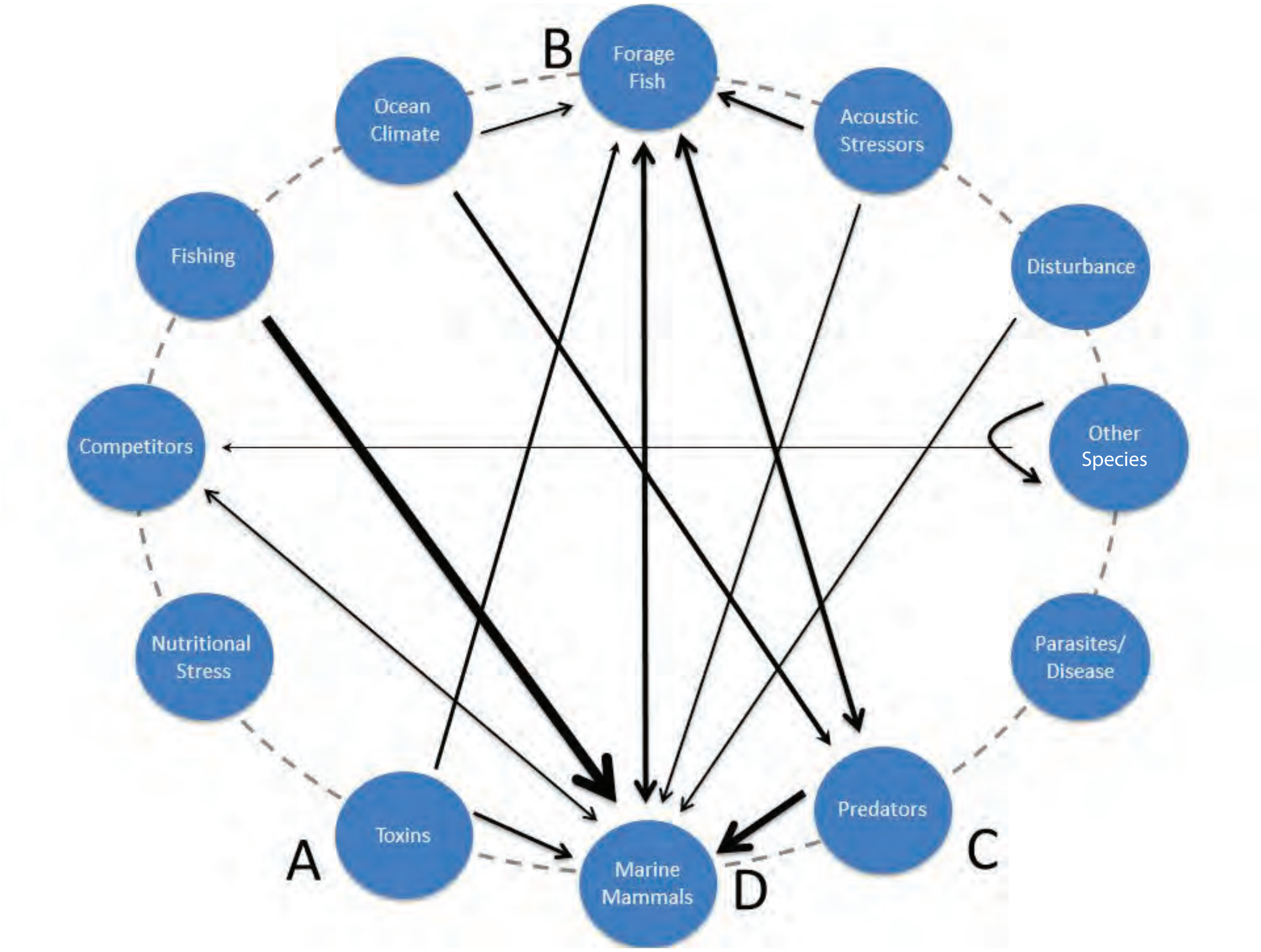

FIGURE S.3 Schematic illustration of an interaction web. Circles around the perimeter of the dashed oval represent species or elements of the abiotic environment (collectively referred to as nodes), and arrows between circles represent species interactions or interactions between species and the abiotic environment. This particular schematic has been stylized to emphasize the nodes of interest and some of their imagined common stressors and interactions. Arrows represent directionality, and line weight represents interaction strength. Note that only a few of the many nodes and their interactions are represented in this schematic. An example of a driver is A (Toxins) operating on B (Forage Fish), a recipient. Forage Fish can also operate as a driver on C (Predators) and vice versa (i.e., both serving as drivers and recipients). Finally, A (Toxins) can operate directly as a driver on D (Marine Mammals) and indirectly as a driver on D through the indirect pathway (A to B to D).

and by testing for whether pathways for adverse health outcomes are shared across stressors. (Chapter 4)

Given that it is problematic to predict when stressors may interact to produce strong effects, there is a critical need for early indicators of risk. However, it is not possible to detect even substantial declines in the size of many marine mammal populations, because precision on population estimates is generally low. Although new survey technologies and analysis methods are improving precision somewhat, it is doubtful that the financial resources and scientific methods are sufficient for adequate population assessments.

Despite the uncertainty, regulators must make decisions on whether and where to allow potentially harmful anthropogenic activities to take place. The concept of adaptive (resource) management offers a framework for making such decisions. In this approach, hypotheses are developed based on current understanding; the optimal action is determined taking into account not just this understanding but also what may be learned as a result of each management action. Adaptive management is also used to identify the optimal data collection strategy to reduce uncertainty.

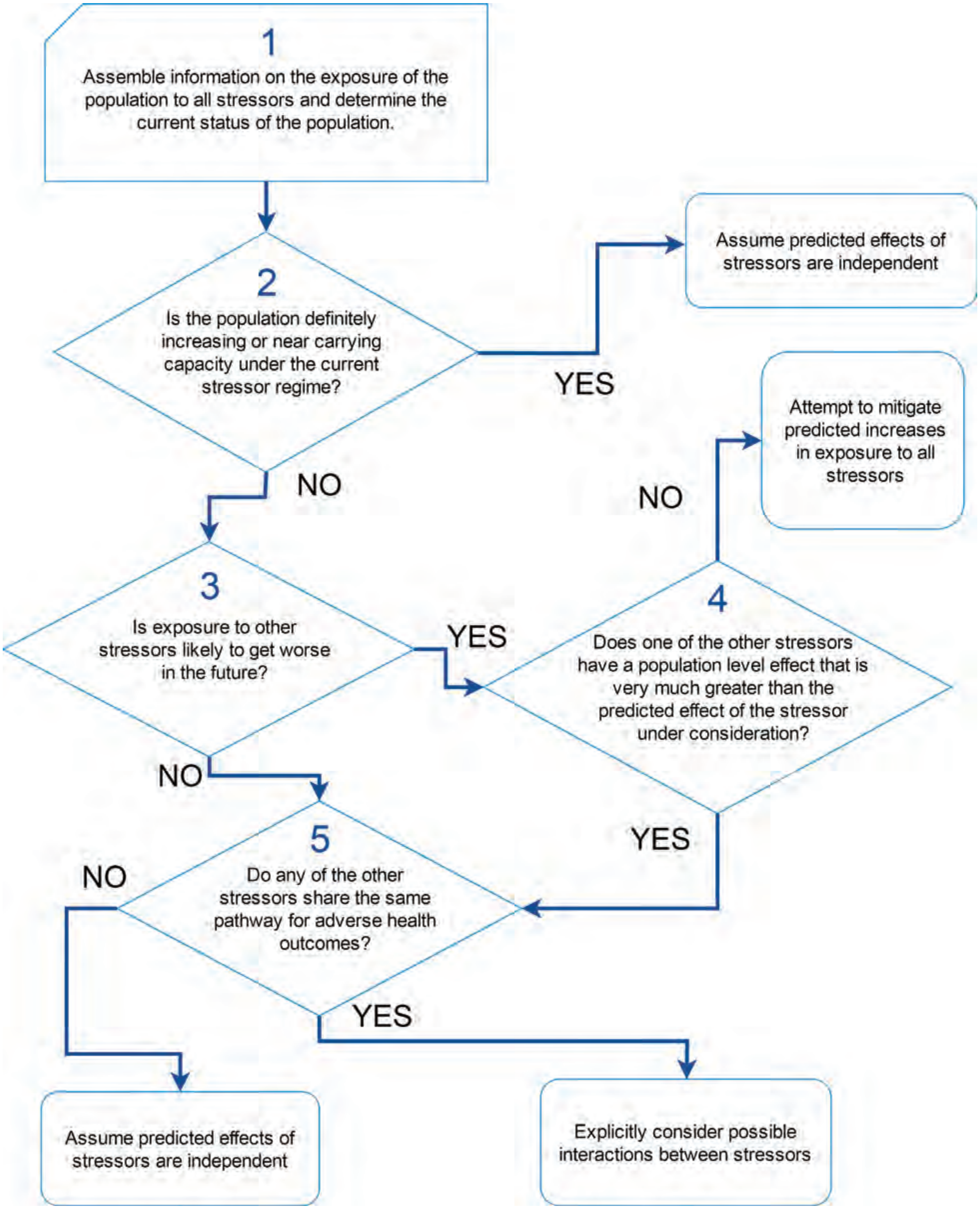

FIGURE S.4 A decision tree for identifying situations where studies of the possible interactions between stressors should be given a high priority when considering the effect of a focal stressor on a population.

Recommendation: Responsible agencies should develop relatively inexpensive surveillance systems that can provide early detection of major changes in population status. (Chapter 7)

Surveillance systems should be developed first for populations that currently lack adequate stock assessments. To be most effective in providing an early warning, the variables monitored will depend on the species and situation, and may change over time with development of new technology and increasing ecological knowledge. Indices of population health, such as mother-to-calf ratios and body condition, are potentially sensitive measures. Abundance indices, such as calibrated acoustic detection rates, may also be appropriate in some circumstances. All measures considered should be evaluated in the context of their ability to inform alternative hypotheses about the mechanisms underlying population changes so that, if a negative change is detected, an early start on evaluating the possible cause could be made. For example, declines in population health indices may indicate increases in exposure to anthropogenic stressors, but they may alternatively be caused by an increase in population size approaching carrying capacity.

Recommendation: Adaptive management should be used to identify which combinations of stressors pose risks to marine mammal populations, and to select which stressors to reduce once a risk is identified. (Chapter 6)

Once a population of marine mammals has been found to be at risk, managers need to identify a stressor or suite of stressors whose reduction can reduce this risk. It may not be possible to reduce some stressors or ecological drivers that contribute to risk. For example, it simply may not be possible to remove persistent toxicants or reverse warming in the ocean due to climate change. This leaves those stressors that in practice can be mitigated within a time period consistent with the population's rate of decline or recovery. Among these remaining stressors, or combination of stressors, it will be important to next identify those whose reduction would be most effective at decreasing the risk. These considerations can be used to establish research priorities for estimating dose–response functions. This approach suggests a new form of effect study—experiments that remove or reduce one or more stressors to study effect of reduction. This experimental design may be more appropriate for adaptive management than the more traditional experiments that add stressors to the current baseline.

The committee recognizes that the state of the science of cumulative effects has low predictive power compared to regulatory demands to assess these effects. The most important goals for managing cumulative effects are (1) identifying when the cumulative effects of stressors risk transitioning a population or ecosystem to an adverse state and (2) identifying practical reductions in stressors to reduce this risk.

1

Introduction

ORIGIN OF THE REPORT AND STATEMENT OF TASK

Four previous reports of the National Research Council (NRC)[1] have documented effects of anthropogenic sound on marine mammals. It is now recognized that intense sounds from human activities such as seismic air guns can have direct physiological effects on marine mammals, and naval sonar triggers behavioral reactions that can lead to death by stranding. However, nonlethal behavioral disturbance is the most common effect of anthropogenic noise on marine mammals. Rather subtle behavioral changes experienced by many marine mammals may have greater population consequences than occasional lethal events. Environmental reviews of human activities that make noise[2] in the ocean routinely assess the number of animals that may be injured or disturbed, and researchers have started to develop methods to estimate effects on populations.

Noise is a stressor for humans and wildlife, and its effects can interact with those of other stressors. Marine mammal populations exist in environments that are being altered simultaneously by various combinations of human activities and their effects, such as pollution and habitat degradation and loss. Natural factors interact in complex ways with effects of human activities to alter climate, the numbers of prey, competitors, pathogens, and predators, potentially contributing to the mix of threats that populations must withstand to remain viable.

Scientists, regulators, and managers have long recognized that the complexity of these interactions must be better understood in order to ensure that marine mammals will continue to be functioning components of their ecosystems. This has led to a strong desire to better understand marine mammal responses to cumulative effects of multiple stressors.

Terminology in the area of cumulative effects in scientific literature has been driven primarily by considerations of environmental chemicals. The U.S. Environmental Protection Agency (EPA, 2007) defines aggregate exposure as the combined exposure of a receptor (individual or population) to a single chemical. The chemical can originate from multiple sources and be present in multiple media, and exposures can occur by different routes and over different time periods. Cumulative risk is defined as the combined risk to a receptor (individual or population) from exposures to multiple agents (here, chemicals) that can come from many sources and exist in different media, and to which multiple exposures can be incurred over time to produce multiple effects. More than one chemical must be involved for the risk to be considered cumulative.

The term *cumulative effect* has been used in marine mammal literature to encompass both aggregate exposure and cumulative risk. For example, noise has been considered to have cumulative effects when an animal is exposed to multiple noise sources, such as shipping plus seismic. To be consistent with the much larger field of environmental chemical exposure, noise should be considered one of a number of stressors experienced by marine mammals. As such the effects of various noises on an individual or a population would be considered components of an overall aggregate exposure to noise. Cumulative effect would derive from the combination of noise and other anthropogenic stressors, such as chemical pollution, marine debris, introduced pathogens, and changes in temperature or pH induced by climate change, and also natural stressors, such as presence of predators, pathogens, parasites, or reduced availability of prey.

The committee defines *aggregate exposure* as the

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[1] Until 2015, reports were published under the authorship of the National Research Council.

[2] Noise refers to sounds that are unwanted or are not useful for a receiver.

combined exposure to one stressor from multiple sources or pathways and *cumulative effect* as the combined effect of exposures to multiple stressors integrated over a defined relevant period: a day, a season, a year, or a lifetime.

When assessing cumulative effects, biologists focus on cumulative effects on an individual animal or population when they are repeatedly exposed to the same or different stressors. By contrast, definitions of "cumulative effects" used in relevant laws and regulations, particularly the National Environmental Policy Act of 1969 (NEPA) and the Endangered Species Act (ESA), focus on the effects of multiple "actions." In addition to NEPA and ESA, there are a number of other acts and implementing regulations dealing with environmental impacts on marine mammals, which are summarized in Appendix B.

Finding 1.1: There is an important difference between the definition of cumulative effects as used by most biologists and cumulative effects as defined under the implementing regulations for the National Environmental Policy Act and the Endangered Species Act. The regulatory definition focuses on the incremental effect of a proposed human action when added to those of other human actions. Most biologists think of effects accumulating when individual animals or populations are repeatedly exposed to the same or different stressors, taking into consideration natural factors that may affect the response to human activities.

NEPA recognized the importance of these interactions by requiring all federal agencies to assess the environmental impacts of their actions. At the heart of NEPA is a requirement that federal agencies "include in every recommendation or report on proposals for legislation and other major Federal actions significantly affecting the quality of the human environment, a detailed statement by the responsible official on—(i) the environmental impact of the proposed action, (ii) any adverse environmental effects which cannot be avoided should the proposal be implemented, (iii) alternatives to the proposed action, (iv) the relationship between local short-term uses of man's environment and the maintenance and enhancement of long-term productivity, and (v) any irreversible and irretrievable commitments of resources which would be involved in the proposed action should it be implemented."[3] The detailed statement called for in NEPA is termed an Environmental Impact Statement (EIS). NEPA regulations require agencies to include in each EIS an evaluation of direct, indirect, and cumulative impacts associated with the action and proposed alternatives. *Cumulative impact* is defined for these purposes as "the impact on the environment which results from the incremental impact of the action when added to the other past, present, and reasonably foreseeable future actions regardless of what agency (federal or non-federal) or person undertakes such other actions." The regulations add that "[c]umulative impacts can result from individually minor but collectively significant actions taking place over a period of time."[4]

Section 7 of the ESA directs federal agencies to carry out programs for the conservation of threatened and endangered species. It further requires federal agencies to ensure that their actions (i.e., all actions authorized, funded, or carried out by the agency) are not likely to jeopardize the existence of a listed species or adversely modify the critical habitat of a listed species. As part of these assurances, Section 7 also requires agencies to consult with the U.S. Fish and Wildlife Service (FWS) or National Marine Fisheries Service (NMFS) (Steiger, 1994) regarding any activities that may affect listed species.[5] "Procedurally, before initiating any action in an area that contains threatened or endangered species, federal agencies must consult with the FWS (for land based species and selected marine mammals) or NMFS (for all other marine species) to determine the likely effects of any proposed action on species and their critical habitat."[6]

The text of the ESA does not directly address cumulative impacts or effects, but the implementing agencies (FWS and NMFS) and the courts have interpreted Section 7 as to require consideration of cumulative effects during the consultation process. The regulations promulgated under the ESA define "cumulative effects" as "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."[7] Guidance produced by the FWS and NMFS regarding Section 7 consultations specifically states that this more narrow definition should not be conflated with the broader definition of "cumulative impacts" used in NEPA and pertains only to ESA Section 7 analyses.[8]

The science is not currently in place to allow quantitative estimates of how different stressors will interact as they impact individuals and populations or what the impact will be of repeated exposure to stressors. For federal agencies that seek to continue to improve their consideration of cumulative effects, such as the U.S. Navy, the U.S. Department of the Interior's Bureau of Ocean Energy Management (BOEM), and the National Oceanic and Atmospheric Administration's

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[3] 42 U.S.C. § 4332(2)(C).

[4] 40 C.F.R. § 1508.7.

[5] 16 U.S.C. § 1536(a). The agency first determines whether their proposed action "may affect" a listed species or its habitat. If the agency determines it may, then formal consultation with either FWS or NOAA Fisheries is automatically required. If the agency determines that the action is not likely to affect a listed species or its habitat and the consulting agency agrees with this assessment, then further formal consultation is not necessary. If, however, the consulting agency does not agree with the assessment, then a formal consultation is required. *Conservation Congress v. USFS*, 720 F.3d 1048 (9th Cir. 2013).

[6] *Conservation Congress v. USFS* 720 F.3d 1048 (9th Cir. 2013) citing *Natural Resources Defense Council v. Houston*, 146 F.3d 1118, 1125 (9th Cir. 1998) and *Forest Guardians v. Johanns*, 450 F.3d 455, 457 n.1.

[7] 50 C.F.R. § 1508.7.

[8] See https://www.fws.gov/ENDANGERED/esa-library/pdf/esa_section7_handbook.pdf.

(NOAA's) NMFS, this presents a challenge. The U.S. Navy, BOEM, and NMFS each either fund and conduct noise-making activities, issue authorizations and permits for such activities, or regulate impacts of sound on most marine mammals. These agencies, along with the U.S. Marine Mammal Commission, funded the present study in order to review current understanding of cumulative effects of anthropogenic stressors, including sound, on marine mammals, to assess current methodologies, and to identify new approaches that may improve the ability to estimate cumulative effects.

REVIEW OF PREVIOUS NRC REPORTS ON MARINE MAMMALS AND SOUND

There has been a consistent expansion of focus in the series of NRC reports on marine mammals and sound from 1994 to 2005. Aside from scientific concern that noise from shipping might reduce the range over which whales may communicate (Payne and Webb, 1971) and studies on the impact of noise from offshore oil industry activities (Malme et al., 1983, 1984), there was little interchange before 1990 between marine mammal biologists and the ocean acoustics community, which understood how well low-frequency sound propagates in the deep ocean. The first NRC report on low-frequency sound and marine mammals (NRC, 1994) was motivated in large measure by a single ocean acoustics experiment designed to monitor changes in ocean temperature by measuring the speed with which a sound travels across ocean basins (Baggeroer and Munk, 1992). Four federal agencies funded a $1.7 million feasibility test for this project, which would involve sending a ship with powerful underwater loudspeakers to a site in the Indian Ocean where a low-frequency sound projected from the ship could be heard in Bermuda and California. When a report in *Science* (Gibbons, 1990) showed how the sound could be detected over much of the global oceans, the executive director of the U.S. Marine Mammal Commission could not understand how this federal action had not required permitting for effects of sound on marine mammals, because it covered such large ranges. His concerns led to the addition of a program to monitor effects on marine mammals, and the transmissions were permitted as marine mammal research (Cohen, 1991). This feasibility test succeeded in precisely timing how long sounds took to travel as far as 16,000 km (Munk et al., 1994). This success led to plans to operate a low-frequency source over a decade or more to measure changes in ocean temperature (in a project called Acoustic Thermometry of Ocean Climate, or ATOC). The long period of operation of such a long-range sound source raised concern about the impact of ATOC on marine mammals. The 1994 NRC report was tasked to review the effects of these kinds of low-frequency sounds on marine mammals and "to consider the trade-offs between the benefits of underwater sound as a research tool and the possibility of its having harmful effects on marine mammals" (NRC, 1994, p. 1). The NRC (1994) report addressed the state of knowledge on the effect of low-frequency sound on marine mammals and found very little relevant data. The 1994 report provided a number of research recommendations to close these data gaps.

The second NRC report, *Marine Mammals and Low-Frequency Sound* (NRC, 2000), was specifically tasked with assessing progress in research on effects of low-frequency sound on seals and cetaceans since 1994, with an evaluation of the marine mammal research program associated with ATOC. Given that the Marine Mammal Protection Act was coming up for reauthorization, NRC (2000) made specific recommendations for changes in the Act, along with recommendations to NOAA for setting priorities for regulating effects of noise, and recommendations for research sponsors. The 2000 report made a suite of recommendations calling for research that could address the uncertainty around the effects of different types and sources of sound on various marine mammal species, both in the context of biological consequences and for monitoring and regulatory purposes (NRC, 2000).

The third NRC report was tasked to evaluate all frequencies and sources of anthropogenic sound that could affect marine mammals, rather than simply low-frequency sound, to identify data gaps in ocean noise databases, and to recommend research to develop a model of ocean noise (NRC, 2003a). Consistent with this charge, the NRC (2003a) expanded the work of prior committees to recommend monitoring noise and marine mammal populations globally. This NRC report (2003a) also recommended that research on effects of sound on marine mammals be structured to test for population-level effects. This latter problem became the primary focus of the fourth NRC report (NRC, 2005), titled *Marine Mammal Populations and Ocean Noise: Determining When Noise Causes Biologically Significant Effects*. In order to begin to address the question of when a behavioral response will become significant to the individual animal, and, more importantly, significant to the population, the NRC (2005) developed a conceptual heuristic[9] model that outlined how behavioral changes could have population consequences. This model, named the Population Consequences of Acoustic Disturbance (PCAD) model, identified a series of stages for relating the effects of acoustic disturbance on the life history of marine mammals, through to the impact on populations. The only stressor this model focused on was sound, and the model recognized that population-level consequences would be likely only when the stressor was repeatedly encountered. Specifically it looked at the aggregate effect of anthropogenic noise as a stressor over a sufficient period—a season or year—that could result in changes in life-history parameters for the exposed animals. These

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[9] A qualitative model informed by expert opinion that links processes and states, in this case the linking of acoustic disturbance through behavior and physiology to its impact on individuals and populations. The heuristic model informs research that can quantify the processes so the qualitative model is turned into a predictive model.

aggregate effects were modeled on the concept of allostatic load/overload (McEwen and Wingfield, 2003).

The model has subsequently been expanded to consider the population consequences of all forms of disturbance (PCoD). New et al. (2014) describe the PCoD model and present an early attempt to quantify fitness effects of behavioral disturbance. The recognition of the importance of identifying intermediate scales between short-term disturbance and population effects was a key element of the 2005 report that is taken up again by this report.

This report develops a metric of health of the individual that can integrate effects which can be related to survival or reproduction over periods of seasons up to the lifetime. The model defines how the distribution of the health of individuals can be used to determine the cumulative risk to the stock, population, or species.

The statement of task for this report is provided in Box 1.1.

BOX 1.1
Statement of Task

The National Academies of Sciences, Engineering, and Medicine's Ocean Studies Board has previously convened four highly successful panels on the subject of biological effects of manmade underwater sound, which produced a progressive series of reports published in 1994, 2000, 2003, and 2005, with the latest report focusing on the potential for biologically significant effects on marine mammal populations. Sound, however, is only one of a variety of potential anthropogenic or natural stressors that marine mammals encounter, and it is often evaluated in isolation without consideration of the effects of other stressors (e.g., fishing, climate change, pollution, etc.), or consideration of how these other stressors may affect an animal's response to sound exposure. The committee will conduct a workshop and review the present scientific understanding of cumulative effects of anthropogenic stressors on marine mammals with a focus on anthropogenic sound. The committee will assess current methodologies used for evaluating cumulative effects and identify new approaches that could improve these assessments. The committee will examine theoretical and field methods used to assess the effect of anthropogenic stressors for

- short or infrequent exposure in the context of other known stressors (i.e., multiple stressors, both natural and anthropogenic) and
- chronic exposure in the context of other known stressors.

The review of methodologies will begin by focusing on ways to quantify exposure-related changes in the behavior, health, or body condition of individual marine mammals and assess the potential to use quantitative indicators of health or body condition to estimate changes in vital rates and, in turn, estimate the potential population-level effects.

REPORT OVERVIEW AND ORGANIZATION

Nine committee members were selected, representing a broad range of expertise (marine mammalogy, ecology, animal behavior, biostatistics, physiology, global change biology, zoology, and bioacoustics). Beginning with its first meeting in June 2015, the committee held four meetings and a workshop. The workshop, held in October 2015, was an information-gathering opportunity designed to survey approaches and methodologies that have been developed to identify and measure animals' exposure to stressors and their responses. The committee was particularly interested in efforts developed for human and terrestrial ecosystems because they wanted to hear how other disciplines addressed these same challenges and questions of assessing cumulative impacts. The workshop discussions also helped the committee members identify innovations (in thinking and application) that they could consider in their review of the current approaches and methods.

In this chapter, the committee begins by defining some of the terminology associated with cumulative effects and the contrasts in their interpretation by biologists and regulators. This is followed by a brief introduction of select U.S. legislation that provides the general legal framework for addressing impacts to marine mammals that the sponsors of this report also use to guide their programmatic activities and responsibilities relevant to marine mammals. The chapter closes with a review of earlier NRC studies that looked at marine mammals and sound.

The effects of sound on wildlife are the focus of Chapter 2 and the committee examines the various sources and the variations in time, frequency, and intensity of sound. Both terrestrial and marine studies are reviewed, and particular attention is given to the perception of or responses to sound by animals. The chapter discusses auditory sensitivities, shifts in hearing (both temporary and permanent), and dose–response relationships in the context of stressors. Characterizing these relationships is an essential step in understanding exposure and outcomes, an approach that is revisited in the remaining chapters in the reviews of other types of stressors and their effects. The chapter includes an explanation of how dose–response functions, properly obtained, can provide much more accurate estimates and variances of marine mammal "take" in association with sound-generating activities.

Chapter 3 transitions away from sound to explore the current state of knowledge regarding the many other types and sources of stressors, with a particular focus on extrinsic stressors (factors in the animal's external environment that create stress). The committee reviewed the effects of extrinsic stressors associated with anthropogenic activities, such as pollutants or ship strikes, and ones that are associated with natural factors. The chapter concludes with a discussion of how the spatial and temporal variation among stressors affects the potential for cumulative effects of individual and combined stressors.

Understanding how the effects of extrinsic stressors might interact to create cumulative effects is the focus of Chapter 4. The committee reviewed studies of interactions of multiple stressors and discussed the challenges of applying the findings from these studies to management of marine mammals and their environment. The chapter examines how multiple stressors are likely to interact, and then identifies approaches for prioritizing stressors for cumulative effects analysis with the use of a decision tree. The committee also explored a set of case studies involving marine mammal population declines that illustrate the difficulty of inferring causes—but also provided the committee an opportunity to investigate what conclusions might have been drawn if the decision tree had been used with these case studies.

Chapter 5 provides a conceptual framework via a new model, titled Population Consequences of Multiple Stressors (PCoMS), developed for assessing the risks associated with aggregate exposures to one kind of stressor, such as sound, and the cumulative exposure associated with sound and other stressors. The PCoMS model documents the pathways from exposure to stressors through their effects on health to their effects on vital rates and population dynamics. A key component of this framework is an assessment of the health of an individual. The chapter discusses a suite of measures that the committee identifies as useful for assessing health in the target populations.

In Chapter 6 the committee broadened its review from cumulative effects of stressors on individuals and populations to consider how interactions among stressors may affect multiple species and entire ecosystems. In doing so, committee members review the components of an interaction web and the various species or abiotic elements that affect the distribution and abundance of species of interest, and specifically how interaction webs can help identify the factors that need to be considered in evaluating cumulative effects on populations and ecosystems.

Chapter 7 acknowledges the challenges of detecting and anticipating the cumulative effects of multiple stressors on marine mammal populations and discusses a suite of population-monitoring parameters that could facilitate the early detection of unexpected population declines and, where possible, the rapid diagnosis of the main factors contributing to them.

In the final chapter of the report (Chapter 8), the committee reviews a broad range of approaches for assessing cumulative impacts that include approaches with limited use for marine mammals as well as those with more utility. The committee identifies the use of comprehensive health assessment as a broadly applicable approach that can serve as a key component of the PCoMS model framework as well as an early warning indicator of population risk prior to population decline.

The tasks asked of this committee span a broad range of scientific disciplines from toxicology to marine ecology. Terms such as *interaction* have different meanings to different specialties, and the dose–response functions discussed in the report span many levels of biological organization from molecules to ecosystems. Nearly every reader may have questions about the usage of some terms. The committee has included a glossary of important terms used throughout this report (Appendix D).

2

Estimating Exposure and Effects of Sound on Wildlife

INTRODUCTION

The world is a cacophony of sounds—from natural sources such as wind-blown vegetation and ocean waves or calling insects, birds, fish, and whales—so all animals have evolved mechanisms to modify their vocalizations to compensate for noise and to focus as listeners on relevant sounds (Tyack and Janik, 2013). However, the increasing levels of anthropogenic noise create acoustic conditions unprecedented in the evolutionary record (Swaddle et al., 2015). Worldwide expansion of human activities and infrastructure is increasing the exposure of terrestrial and marine environments to anthropogenic sound (Hildebrand, 2009; Barber et al., 2010; Shannon et al., 2015). Recent estimates suggest that more than 88% of the contiguous United States experiences elevated sound levels due to anthropogenic activities (Mennitt et al., 2013) and that the propulsion noise from ships elevated ocean sound levels in the 25-50 Hz band by 8-10 decibels (dB) from the mid-1960s to the mid-1990s, which then remained constant or showed a slight decline in the next decade (Andrew et al., 2011).

Most of the human activities that produce noise are common to terrestrial and marine ecosystems. These include transportation, exploration for and extraction of oil and gas, construction, mining, and military operations. Sounds from these sources can influence terrestrial and marine animals in similar ways. Although this report focuses on the cumulative effects of anthropogenic stressors, including sound, on marine mammals, recent terrestrial studies have evaluated consequences of noise exposure in ways that have not been thoroughly investigated in marine mammals, such as declines in foraging efficiency (owls [Mason et al., 2016; Senzaki et al., 2016] and bats [Siemers and Schaub, 2011; Bunkley and Barber, 2015]), heightened vigilance (prairie dogs [Shannon et al., 2014, 2016] and songbirds [Quinn et al., 2006; Ware et al., 2015]), declines in reproductive success (Halfwerk et al., 2011), and altered predator–prey relationships (Francis et al., 2009). Insights from such terrestrial research help point to potential effects that deserve more attention in marine studies, and these studies can serve as guides for future efforts to determine whether noise affects marine mammals in similar ways.

Because research on land and at sea has largely progressed in isolation, we summarize the research status of each ecosystem separately below. Nevertheless, research in these disparate ecosystems provides a general framework for investigating how diverse noise stimuli present a multitude of challenges to wildlife.

When assessing the potential influence of a sound stimulus on an animal, determining whether the stimulus is within the organism's sensory capabilities is critical. Most animals have developed sensory organs that allow them to detect either pressure waves or particle motion in the environment somewhere in the range of frequencies from below 10 Hz to above 180 kHz. They use this sensory input to communicate, orient, avoid predators, detect prey, and monitor their environment. If the stimulus falls outside of an animal's sensory capabilities, i.e., higher or lower in frequency than its sensory organs can detect, the stimulus is likely not to have a direct effect (Francis and Barber, 2013), although indirect consequences of noise exposure are possible (e.g., Francis et al., 2009, 2012a).

There is a diverse array of anthropogenic sound sources, which vary in time, frequency, and intensity. Variation along these axes is not only relevant to the detection capabilities of an organism's sensory system, but is also relevant to how organisms perceive sound stimuli. Sounds that are sudden, unpredictable, and loud often generate startle responses that can be similar to those associated with predation risk (see Figure 2.1). Sounds with these characteristics need not be associated with real threats to elicit strong responses. For example, the acoustic startle response in mammals is stimulated by sounds that increase to 80-90 dB above the threshold of hearing in 15 milliseconds (Fleshler, 1965). Götz and Janik (2011) demonstrated that the startle responses triggered by these stimuli are aversive enough to lead grey seals (*Halichoerus grypus*) to show fear conditioning with strong flight responses. Other sounds that animals interpret as originating from either predators or aggressive conspecifics may evoke disturbance responses similar to those that function to defend against risk of predation (Frid and Dill, 2002) or potential intraspecific confrontation. Beaked whales (*Mesoplodon densirostris*) respond to military sonar through antipredator behavior in a manner similar to, albeit less intense than, their responses to playback of predator calls (killer whales [Tyack et al., 2011]). Military sonar sounds in the 1-10 kHz band are well below the frequencies used in beaked whale vocalizations and those at which they hear best, but these sonar signals share a similar duration and frequency structure with the stereotyped calls of killer whales. The stronger response of killer whales (*Orcinus orca*) than that of sperm whales (*Physeter macrocephalus*) or long-finned pilot whales (*Globicephala melas*) to playbacks of sonar signals (Miller et al., 2012; Harris et al., 2015) suggests that killer whales also perceive the sonar as threatening.

Sounds that are frequent, continuous, or chronic may not be perceived as threatening but nonetheless can affect animals by interfering with their ability to detect acoustic signals or cues, such as calls from conspecifics or sounds made by predators or prey (see Figure 2.1). The more overlap there is in spectral bandwidth between anthropogenic sounds and those used by an organism, the more likely they are to interfere with detecting biologically important signals. Masking of relevant sounds has the potential to reduce an organism's auditory perceptual range, or listening area (Payne and Webb, 1971; Clark et al., 2009; Barber et al., 2010), and can interfere with an organism's abilities to detect, interpret, and respond to cues in their environment. As early as 1971, Payne and Webb (1971) suggested that shipping noise could have reduced by a factor of 6 the range over which one fin whale could hear another vocalizing at 20 Hz. Male fin whales (*Balaenoptera physalus*) repeat series of 20 Hz songs that can be detected at ranges of hundreds of kilometers (Croll et al., 2002). During the 20th century, when shipping noise increased, commercial whaling also reduced fin whale populations to 10% or less of their original numbers (Rocha et al., 2014). If females listen to these 20 Hz songs to find and select a mate, then this reduction in the range could interact with the decrease in abundance of whales to reduce the reproductive rate of this endangered species (Croll et al., 2002).

Anthropogenic sounds can also distract animals (Chan et al., 2010), causing them to divert their attention to a sound stimulus away from other important environmental stimuli, whether acoustic or via another sensory modality. For example, exposure to shipping noise disrupts feeding in shore crabs (*Carcinus maenus*) and causes them to take longer to find shelter after a simulated predatory attack, even if the attack does not involve acoustic cues (Wale et al., 2013). Finally, in addition to the sound characteristics, the behavioral context of the animal is critical to understanding how and why organisms respond to various anthropogenic sounds (Ellison et al., 2011).

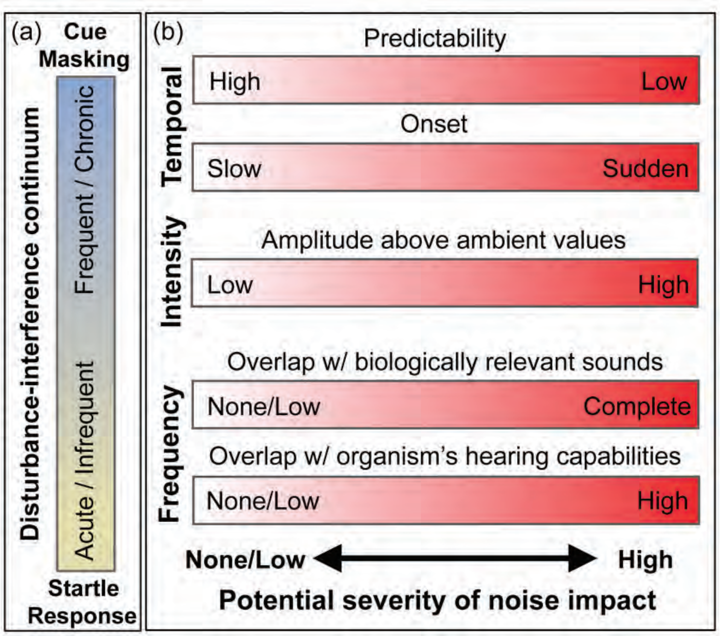

FIGURE 2.1 (a) The disturbance–interference continuum can range from acute or infrequent noise stimuli that will likely trigger startle, flight, or hide responses to frequent or chronic noises that interfere with cue detection. (b) The severity of an impact from a noise stimulus will depend on the temporal, intensity, and frequency features of the stimulus. SOURCE: Francis and Barber (2013).

TERRESTRIAL STUDIES

The most extensive research on the effects of noise has been conducted on humans where noise has been shown to have cardiovascular, endocrinological, neurological, and auditory effects (Basner et al., 2014). Cognition is also impacted; chronic noise at levels typically found in residential areas can impair cognitive processes in children (Lercher et al., 2003). Whether marine mammals and other nonhuman animals experience similar consequences of noise exposure is less well known. Research in the last decade demonstrates many effects of noise for taxonomically diverse wildlife, but many potential consequences have not been adequately investigated.

Researchers have known for decades that acute intense sound events, such as those generated by aircraft overflight, gunshot, or chainsaws, can trigger immediate behavioral responses, such as hiding or fleeing (reviewed by Ortega [2012]). Additionally, early road ecology studies suggested that traffic noise reduces the density of vertebrates, especially birds, near roads (e.g., van der Zande et al., 1980; Reijnen et al., 1995; Kuitunen et al., 1998). However, these early studies were viewed with skepticism because confounding factors also associated with roads (e.g., mortalities from collisions with vehicles, changes in predator densities, and land cover changes) could also explain observed changes. Recent work has bolstered these early studies; research that isolates noise as a single environmental stimulus or introduces noise experimentally demonstrates that noise alone can explain declines in bird abundance and species richness (Bayne et al., 2008; Francis et al., 2009). More recently, experimental approaches that broadcast playbacks of traffic noise (McClure et al., 2013; Shannon et al., 2014) or energy-sector noise (Blickley et al., 2012a) over large areas have supported earlier observational studies and “natural” experiments. For example, at an important migratory bird stopover site McClure et al. (2013) constructed a 0.5 km “phantom road” where they simulated 12 vehicle pass-by events per minute for vehicles traveling ~70 km/h and alternated 4 days of noise “on” and 4 days of noise “off.” Noise “on” periods resulted in a one-quarter decline in bird abundance, and several species avoided areas exposed to the playback entirely. Another study experimentally introduced traffic noise via playback to prairie dog (*Cynomys ludovicianus*) colonies such that received levels at the center of colonies were approximately 52 dbA L_{eq} (re 20 µPa; Shannon et al., 2014).[1] In response to exposure, prairie dogs significantly reduced aboveground activity, and those that remained above ground increased visual vigilance at the expense of active foraging. There was no evidence of habituation to repeated exposure to the stimulus across the 3-month study period. Prairie dogs respond to an approaching human at greater distances in the presence of road noise than during quieter control periods (Shannon et al., 2016).

BOX 2.1
Acoustic Terminology

The decibel (dB) is a logarithmic scale for measuring a quantity with respect to a specified reference level.

The sound pressure level (SPL) in dB is equal to 20 $\log_{10}$ (sound pressure/reference pressure).

In water the reference pressure is 1 μPa and in air it is 20 μPa, where Pa is an abbreviation for a pascal or newton per square meter.

The sound energy level (SEL_{cum}) is the cumulative sound energy level over the time interval of interest. The reference value for dB_{SEL} is 1 μPa^2-s.

SPL_{pk} is the peak SPL encountered over the time interval of interest.

$SPL_{P\text{-}P}$ is the maximum difference between the compression and rarefaction phases associated with an impulsive sound source.

SPL_{RMS} (reported in dB_{RMS}) is the root mean square SPL measured over an appropriate time interval. The value of a SPL_{RMS} for a transient signal is influenced by the time interval over which the SPL_{RMS} is calculated.

dBA is a measure of the SPL with different frequencies weighted by the frequency-dependent sensitivity of human hearing.

L_{eq} is the steady SPL that over a given period of time has the same total energy as the energy in the varying sound of interest. It can be reported as either dB or dBA.

Impulsive noise is defined by short duration, rapid rise, and broad frequency content.

The costs in reduction of habitat are obvious for species that avoid noisy areas entirely or that decline in abundance with noise exposure, but there also may be costs for those individuals that remain in noisy areas. For example, the number of males in courtship displays (leks) of greater sage-grouse (*Centrocercus urophasianus*) declines in response to experimental playback of natural gas compressor noise or energy-sector truck traffic (Blickley et al., 2012a). Individuals that remain in the leks exposed to noise experience elevated stress hormone levels relative to those in leks that were not exposed to playbacks (Blickley et al., 2012b). Experimental playback of traffic noise also increases stress hormones in

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[1] See Box 2.1 for acoustic terminology.

female wood frogs (*Lithobates sylvaticus*) and appears to impair navigation toward chorusing males at breeding ponds (Tennessen et al., 2014). Whether mediated by physiological stress responses or due to other factors, avian reproductive success can decline in response to noise. The most obvious of these declines in success include examples in which male birds occupying noisy territories have lower pairing success than individuals in areas that are less noisy (Habib et al., 2007; Gross et al., 2010). In other cases, birds breeding in noisy areas lay fewer eggs (Halfwerk et al., 2011) or fledge fewer young (Kight et al., 2012). It is unclear whether the lower breeding success is due to the influence of noise on these pairs or if the lower success is due to less fit birds being marginalized to the noisy habitat. If the latter, and if there remain better territories for the more fit pairs, then it likely will not lead to population-level effects.

Even relatively short exposure (i.e., approximately 4 days) to experimentally introduced traffic noise causes declines in a body condition index (i.e., mass-to-wing chord length ratio) among migrating songbirds (Ware et al., 2015). This decline in health appears to be mediated by a foraging–vigilance trade-off; in noisy conditions, birds increase visual vigilance in response to impaired acoustic surveillance capabilities, but decrease time spent actively foraging. Frid and Dill (2002) argue that disturbance generally causes animals to reduce time allocated to other critical activities, such as foraging, which may pose increasing fitness costs as disturbance increases. Noise can also directly impair foraging by masking the acoustic cues used by predators to locate prey, such as in gleaning bats (e.g., Schaub et al., 2008; Siemers and Schaub, 2011). Additional evidence from a comparative study examining responses of 183 bird species suggests that birds with animal-based diets are more sensitive to human-made noise than birds with plant-based diets, perhaps due to an underappreciated use of hearing alongside vision when hunting (Francis, 2015). Regardless of the precise mechanisms responsible for predator sensitivities to noise, decreases in predator abundance, or decreases in predator efficiency, can have broader ecological consequences. For example, declines in common nest predators in areas exposed to energy-sector noise results in higher nesting success among several songbird species that persist in noisy areas (Francis et al., 2009). Similarly, noise-induced declines in the abundance of species that perform key ecological functions, such as the seed-dispersing activities of Woodhouse's scrub-jay (*Aphelocoma woodhouseii*), can trigger the reorganization of foundational species (Francis et al., 2012b; see "Indirect Effects of Sound on Marine Mammals" on p. 31).

MARINE STUDIES

This section provides a selection of studies showing the anatomical, physiological, and behavioral responses of marine mammals to different intensities of sound. It begins with an overview of U.S. regulations that established criteria and thresholds for various levels of acoustic disturbance of marine mammals that correlate with the legal definition of a take.[2]

Criteria, Thresholds, and Takes

While shock waves from underwater explosions have resulted in mechanical trauma in whale ears (Ketten et al., 1993), the most severe acoustic injury associated with intense sound waves is a permanent hearing threshold shift (PTS)—a loss of hearing within a particular frequency range that is not reversible. Sounds not intense or energetic enough to cause PTS can cause a temporary threshold shift (TTS)—reduced hearing sensitivity within a particular frequency range that lasts for a period of minutes to hours, but recovers to its prior level of sensitivity. Sounds at all levels can cause behavioral changes as long as they are audible. Animals can reduce the physiological impact of sound through behaviors in which they move down the sound gradient. They can also respond to noise masking relevant sounds through behavioral changes.

The prohibitions against taking marine mammals under the Marine Mammal Protection Act described in Appendix B focus on two kinds of takes: Level A takes that have the potential to injure an animal, and Level B takes that harass animals by disrupting behavior. In spite of the early focus on the global scales at which shipping noise might mask fish and whale communication, these regulatory definitions led research in the United States to focus on identifying how intense sounds may injure animals or disrupt their behavior. The National Marine Fisheries Service (NMFS) has defined acoustic injury as a PTS. Studies of the toxic effects of chemicals typically determine the dose that kills half of a sample, whereas studies that involve intentional injury or death of marine mammals are rarely permitted. This led to the development of experiments that use TTS as a reversible indicator of risk of injury.

For sound sources, two critical measures are sound pressure level (SPL) measured in dB re 1 µPa, a measure of sound intensity, and sound exposure level (SEL) measured in dB re 1 μPa^2-s, a measure of the energy received due to the aggregate exposure to all sound sources over a defined interval of time. SEL accumulates the energy in short, intense sounds, such as pile driving, with longer, lower-level sounds, such as shipping. One critical decision for SEL calculations is the duration over which energy is accumulated. Several different integration times are important for marine mammals. The mammalian ear integrates sound energy over a period of about 200 milliseconds (msec) (Green, 1985), so 200 msec can be used as a maximum integration time to estimate apparent loudness of a sound. The animals are more likely to react behaviorally to short, intense sounds,

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[2] Defined in the Marine Mammal Protection Act as "harass, hunt, capture, or kill, or attempt to harass, hunt, capture, or kill" (16 U.S.C. § 1362; see also 50 C.F.R. § 216.3), and in the Endangered Species Act as "harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect" (16 U.S.C. § 1532 (19)).

whereas physiological effects are greater for equivalent energy delivered as long, less intense sounds. To estimate effects of noise exposure on the sensitivity of hearing, longer integration times are required. For humans, the 8-hour daily exposure in a workplace is commonly used as an integration time. There is no obvious equivalent for marine mammals in the wild, but the longer SEL accumulates sound energy, the higher the value. Most animals go through daily cycles of behavior, so a 24-hour integration time has been adopted (e.g., Southall et al., 2007; NMFS, 2016a), but the critical point for assessing noise impact on hearing is whether the animal has long enough time at low enough exposure levels for the auditory system to recover from any temporary effects of noise exposure (Ward et al., 1976). Thus, although there is an appropriate energy metric for aggregate exposure to sound sources, it is more effective as a physical measure than as a predictor of aggregate impact on marine mammals. Predicting impacts on hearing requires integrating SEL until the animal has a long enough period of relative quiet to recover.

Southall et al. (2007) conducted a very thorough study of the available science and laid the groundwork for more recent updated approaches to determining onset of TTS and PTS (e.g., Finneran, 2016). They categorized marine mammals into five hearing groups: low-, mid-, and high-frequency cetaceans; pinnipeds in water; and pinnipeds in air. For each hearing group, they established the SPL and the SEL that would result in PTS or behavioral disturbance for three categories of sounds: single pulses, multiple pulses, and nonpulses. NMFS recently published acoustic thresholds for the onset of TTS and PTS (NMFS, 2016a) that aim to be based on the best current available science. These guidelines have separate PTS thresholds for impulsive and nonimpulsive sounds for five categories of marine mammals: low-, mid-, and high-frequency cetaceans; phocids; and otariids.[3] For each marine mammal category two thresholds are given for impulsive sounds: one for peak sound pressure level (SPL_{pk}) and one for cumulative sound exposure level (SEL_{cum}) accumulated over 24 hours; and one threshold is given for nonimpulsive sounds: the cumulative sound exposure level (SEL_{cum}) accumulated over 24 hours. The SPL_{pk} ranges from 202 dB re 1 μPa for high-frequency cetaceans to 232 dB re 1 μPa for otariid pinnipeds in water. The SEL values for impulsive sounds range from 155 dB re 1 μPa^2-s for high-frequency cetaceans to 203 dB re 1 μPa^2-s for otariids, and the threshold values for nonimpulsive sounds range from 173 dB re 1 μPa^2-s for high-frequency cetaceans to 219 dB re 1 μPa^2-s for otariids.

The Level B behavioral harassment criteria used by NMFS for most situations are thresholds of SPL_{RMS}[4] of 160 dB re 1 μPa[5] for impulsive sounds and 120 dB_{RMS} for nonimpulse sounds.[6] NMFS classifies a variety of sonar signals as impulsive for Level B criteria, but as nonimpulsive for Level A criteria (NMFS, 2016a). These thresholds are treated as all-or-nothing thresholds, with all animals exposed above the threshold treated as harassed and no animals below the threshold considered to be harassed. The primary exception involves estimates of "takes" by Navy sonar, which are estimated using a behavioral response function developed by Finneran and Jenkins (2012) to estimate the proportion of animals receiving a given sound level that will show the criterion behavioral response. This response function has a sigmoidal shape in which the probability of response varies more gradually as a function of dosage than in the step function threshold. The Navy has adopted more conservative criteria for behavioral response thresholds for beaked whales (all-or-nothing threshold of 140 dB_{RMS}) and for harbor porpoises (all or nothing threshold of 120 dB_{RMS}) exposed to sonar (Finneran and Jenkins, 2012).

In order to determine received sound levels, the propagation of a sound from a point source can be modeled to determine the spatial distribution of the sound field. The level of exposure can then be determined by combining this with an estimate of the animals' distribution. There is generally much greater uncertainty associated with estimating the distribution of animals than the sound field. The principles of underwater sound propagation are relatively well understood (Keenan, 2000), whereas the information available on the movements and distribution of marine mammal species is highly variable geographically and by species. Spatially explicit marine mammal density estimates have been calculated based on transect-based (typically visual) surveys (Hammond et al., 2002; Redfern et al., 2006; Roberts et al., 2016) and telemetry data (Aarts et al., 2008; Whitehead and Jonsen, 2013), as well as through the use of habitat-based models (Forney, 2000; Redfern et al., 2006). More complex individual-based animal three-dimensional movement models have also been used to estimate the SEL_{cum} for individuals (Frankel et al., 2002; Gisiner et al., 2006; Donovan et al., 2013).

Takes have typically been calculated based on determining the 190 dB_{RMS} or 180 dB_{RMS} (Level A) or the 160 dB_{RMS} or 120 dB_{RMS} (Level B) isopleth[7] and moving that area through space as the source moves. The total area encompassed over the course of 24 hours is multiplied by the density of a given marine mammal species in that general geographical area at the time of year of the activity to produce a single value take estimate for that species for that 24-hour period. However, a hard threshold typically based

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[3] Low-frequency cetaceans are all the baleen whales. High-frequency cetaceans are all porpoises, river dolphins, pygmy and dwarf sperm whales, all dolphins in the genus *Cephalorhynchus*, and two species of *Lagenorhynchus, L. australis* and *L. cruciger*. Mid-frequency cetaceans are all the odontocetes not in the high-frequency group.

[4] RMS is root mean square.

[5] All underwater acoustic intensity dB are re 1 μPa. This reference level will not be repeated for future dB.

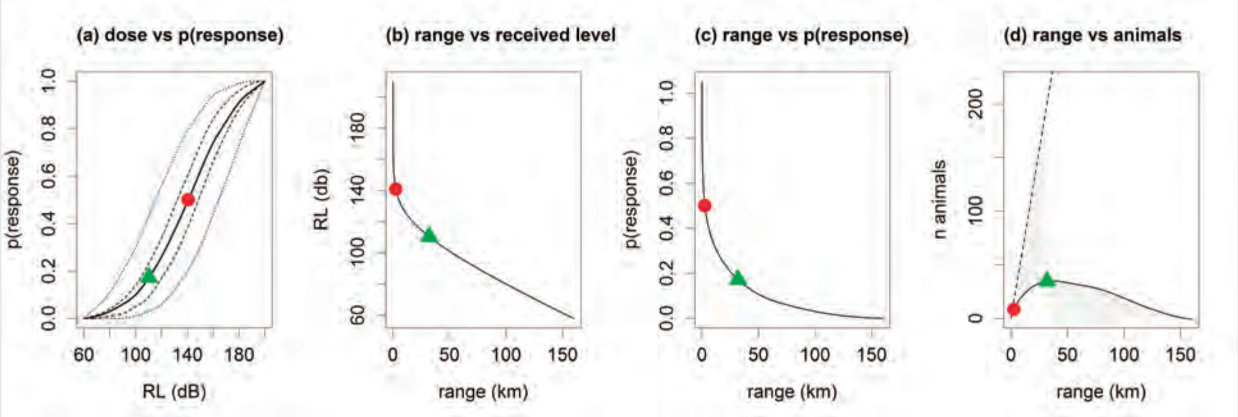
[6] See http://www.westcoast.fisheries.noaa.gov/protected_species/marine_mammals/threshold_guidance.html.

[7] Typically a circle centered at the source with a radius equal to the distance at which the signal falls to the criterion value.

BOX 2.2
Estimating the Number of Behavioral Takes from a Dose–Response Function

Behavioral dose–response functions based on experimental data are now available for a number of marine mammal species (reviewed later in this chapter). One approach for determining the threshold for response is to use the received sound pressure level (RL) at which the probability of response is 0.5, the “RL_{p50}.” For example, this is the origin of the 120 dB_{RMS} Level B harassment criterion used by National Marine Fisheries Service (NMFS) for nonpulse sounds (NRC, 1994, p. 19). There are two problems with this approach. First, using RL_{p50} as a threshold typically results in a substantial underestimate of the number of takes implied by the dose–response function. Second, this procedure ignores uncertainty in the dose–response function, as well as in the source level, propagation model, and density estimate. These issues are illustrated here using the fitted dose–response function from Miller et al. (2014) for killer whales showing onset of avoidance behavior in a controlled exposure experiment that used a scaled mid-frequency sonar source as the stimulus (see Box Figure 1a).

To illustrate the first issue, the average estimated dose–response function is used (solid line in Box Figure 1a); a stationary single-frequency 6 kHz source is assumed, with a source level of 210 dB re 1 µPa at 1 m and a simple propagation model (spherical spreading and frequency-dependent absorption; see Box Figure 1b). The resulting probability of response as a function of range from the source is shown in Box Figure 1c. If the spatial distribution of animals is independent of the source location, then, on average, the number of animals at each range will increase linearly with range (see Box Figure 2). The expected number of animals responding is the number at each range multiplied by the probability of response at that range (see Box Figure 1d), integrated over all ranges. Assuming a density of one animal per km^2 gives an expected take of 3,215 animals. If, instead, a threshold is set at $RL_{p50} = 141$ dB_{RMS} (the red dot on Box Figures 1a-d), this translates to a threshold range of 2.63 km, and an estimated take of $\pi 2.63^2 = 21.8$ animals, more than two orders of magnitude too low.



Box Figure 1 (a) Example dose–response function from Miller et al. (2014): solid line is posterior mean; dashed lines show 50% CI; dotted lines 95% CI. Red dot shows received level corresponding with probability of response of 0.5 (RL_{p50}); green triangle shows effective received level (ERL; see box text). (b) Range versus received level from a simple transmission loss model. (c) Dose–response model reexpressed in terms of range. (d) Expected number of animals as a function of range in 1 km bins (dashed line); expected number of responding animals as a function of range in 1 km bins (solid line). ERL is at the range (green triangle) where as many animals are expected to fail to respond within this range as are expected to respond outside this range (i.e., the two shaded regions have the same area).

If a fixed threshold must be used (e.g., for reporting), the correct take value can be obtained by using the “effective RL” (ERL)—this is the RL corresponding to the range at which the number of animals expected to respond at larger ranges is balanced by the number failing to respond at smaller ranges (analogous to the effective detection radius in Buckland et al. [2001, Ch. 5]). In this example, the ERL is 110 dB_{RMS} corresponding to a range of 32.0 km (green triangle in Box Figures 1a-d).

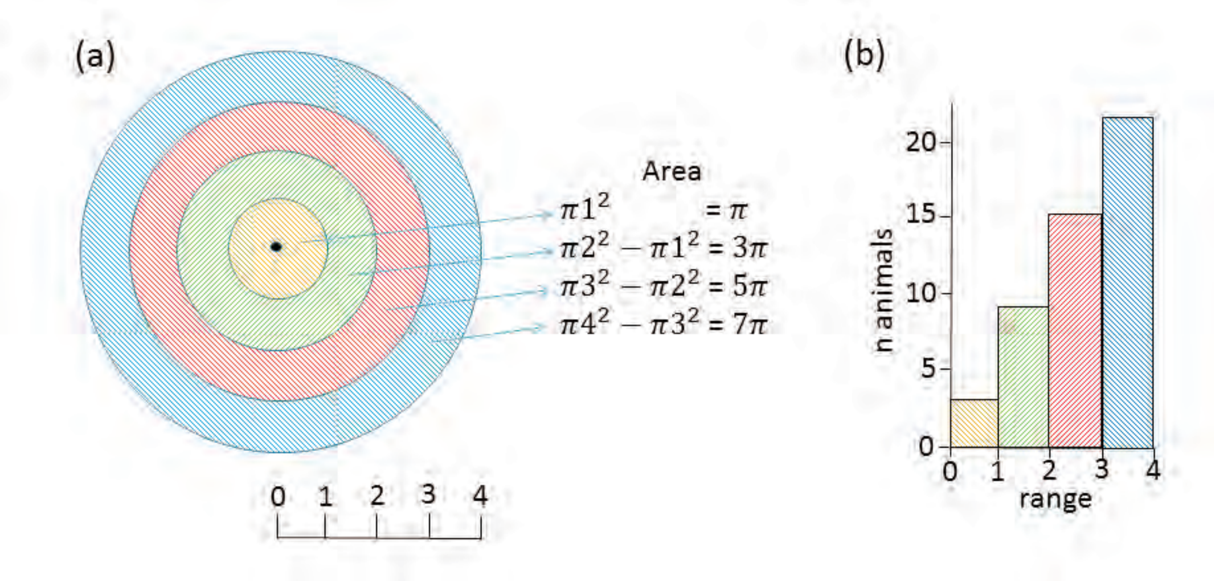
Regarding the second issue, uncertainty on inputs can be translated into uncertainty on take estimates readily through stochastic simulation. Regulators may then choose the level of risk they wish to use in deciding whether to permit an activity (e.g., Taylor et al., 2000). For simple cases, simulation is unnecessary: for example, if it is desired to include only uncertainty in the dose–response function, the above calculations can be repeated using the 2.5% and 97.5% quantiles (dotted lines in Box Figure 1a), yielding a 95% confidence interval (CI) of 313 to 9,910 takes. However, there are often multiple sources of uncertainty and other complications, making simulation the best approach.

To allow the calculations outlined here, researchers should provide sufficient information to allow reconstruction of their dose–response functions, and uncertainty about these functions. For example, Miller et al. (2014) provide a table of quantiles for probability of response over a range of doses. Unfortunately, this is not common practice, and only RL_{p50} values are reported for many studies (see main text). The current NMFS Level B harassment criterion of 120 dB_{RMS} was based on reported levels from the 1980s at which approximately 50% of gray and bowhead whales responded; Malme et al. (1984) reported dose–response functions for gray whales exposed to experimental oil exploration and production-related activity that could be used to calculate the extent to which the 120 dB_{RMS} criterion may underestimate the number of whales taken.

Finding 2.1: Current methods for calculating behavioral take based on animals within a range determined by the 50% probability-of-response threshold lead to potentially significant underestimates of the total number of animals taken. An "effective received level" can be calculated that corrects the take estimate.

Finding 2.2: Take numbers are currently requested and approved based on a point value estimate. Changes in transmission patterns of sound in the ocean, distribution of animals, variable responsiveness of individual animals, and temporal, spatial, and social determinants of response all create uncertainty in the number of animals taken by sound. Thus, any effort to include measures of uncertainty, such as confidence intervals for estimates of predicted take, as required under the Marine Mammal Protection Act, would be more consistent with the state of our knowledge than providing a single number for takes.

Calculations of take are very sensitive to the shape of the dose–response function at low levels of dose, because this corresponds to larger distances, where relatively more animals are exposed. Increased realism can be introduced by accounting for animals' auditory sensitivity, where known (Miller et al., 2014; see next section), and by experimental information about how RL and range interact to affect animals' responses.



Box Figure 2 (a) The areas of rings of fixed width increase linearly with their distance (range) from a central point. (b) If the point is located at random with respect to animals then the number of animals within each ring is, on average, proportional to the area of the ring, and so also increases linearly with range.

on a 50% probability-of-response criterion can significantly underestimate the number of animals taken. Even though the probability of an exposed animal responding is smaller outside of the impact threshold than inside it, the greater number of animals experiencing low exposures may overwhelm this difference in risk and ultimately result in more animals being affected at distances that are greater than the ones currently considered for monitoring and mitigation (see Box 2.2).

Models that estimate the number of "takes" do not describe how this "taking" may affect the population, which requires further understanding how these impacts on individuals affect their survival and reproduction. Changes in these vital rates can then be incorporated into a dynamic population model to estimate population-level impacts (Thompson et al., 2013b; New et al., 2014; King et al., 2015).

Auditory Sensitivity

Studying what sounds cause masking or TTS demands understanding how the sensitivity of hearing varies with frequency, which is achieved by measuring audiograms of different species. It has become apparent from studies on marine mammal hearing that their auditory capabilities differ considerably among species. Underwater audiograms have been determined using either behavioral or physiological methods for 18 species of cetaceans (14 in the mid-frequency hearing group, 4 in the high-frequency hearing group, and none for baleen whales) and 11 species of pinnipeds and other marine carnivores (6 phocids and 5 in the combined otariids, sea otters, and walrus) (Mooney et al., 2012; Finneran, 2016). Behaviorally determined audiograms are available for individuals from four of the five marine mammal groups (mid- and high-frequency cetaceans and phocids and otariids in water). Within each group, the audiograms were combined to arrive at a best-fit composite audiogram for that group as shown in Figure 2.2. No hearing measurements have been made on low-frequency cetaceans. Hence the estimated hearing thresholds were calculated based on data from Cranford and Krysl (2015), Houser et al. (2001), Parks et al. (2007a), and Tubelli et al. (2012) as described by Finneran (2016).

The curves for all hearing groups follow a typical mammalian pattern in which there is a best frequency of hearing. Below the best frequency there is a gradual falloff in hearing sensitivity for low frequencies and above there is a much more rapid falloff in hearing sensitivity for high frequencies. These curves represent the best available peer-reviewed data. It is recognized that the curves are based on small numbers of animals, and only a few species are surrogates for each entire hearing group. No data were available for low-frequency cetaceans, so this estimate is based on correlation and assumptions.

Finding 2.3: A behavioral dose–response relationship can be determined without knowing the subject's audiogram. However, understanding the physiological effects of sound from TTS through PTS requires an audiogram. For baleen whales physiological sound impacts are estimated based on modeling of the skull, estimated historical ocean noise thresholds, and data from other cetacean hearing groups. An audiogram from at least one species of baleen whale would be beneficial in understanding the effects of anthropogenic sound on baleen whales.

Permanent and Temporary Threshold Shift

If sounds are loud enough, they can lead to TTS. As indicated by the name, the hearing threshold returns to baseline in minutes to hours after the cessation of the stimulus, depending on the amount of TTS. The energy in the sound that generates a TTS is expressed as the SEL and measured in dB re 1μPa^2-s. TTS and the growth in TTS with increasing SEL have been measured in four cetacean and three pinniped species. The weighted TTS threshold ranged from 153 dB$_{\mathrm{SEL}}$ for high-frequency (HF) cetaceans to 193 dB$_{\mathrm{SEL}}$ for otariids in water (Finneran, 2016). TTS can reduce an animal's communication space and its abilities to detect predator and prey during the minutes to hours it takes for the threshold to return to its preexposure state. It is arguable whether this temporary reduction in hearing sensitivity represents an injury in itself. Kujawa and Liberman (2006) demonstrated in laboratory mice that noise exposures that cause only TTS may cause pathological changes that render the auditory system more vulnerable to age-related hearing loss. However, TTS is not considered an injury in the U.S. regulatory framework. No experiments have investigated the long-term effects of TTS in marine mammals, or have tried to create a PTS in a marine mammal (but see Kastak et al., 2008). Based on data from terrestrial mammals, the onset of PTS has been set by Southall et al. (2007) at an SEL that would produce 40 dB of TTS. Thresholds for PTS can then be calculated by knowing the threshold for onset of TTS and estimating the growth in TTS with increasing sound levels. For impulsive sounds, TTS in laboratory animals increases with a slope of 2.3 dB of TTS per dB of noise, suggesting a minimum of 15 dB SEL above TTS onset for PTS caused by impulsive sound. Similarly the slope for nonimpulsive sounds, based on human data, is 1.6 dB of TTS per dB of noise or conservatively rounded down to 20 dB SEL above TTS onset for PTS (Southall et al., 2007). The amount of sound energy required to produce injury based on TTS data has been summarized by Southall et al. (2007) and the NMFS (2016a) for each of the marine mammal hearing groups. The HF cetaceans have the lowest estimated PTS threshold, 173 dB$_{\mathrm{SEL}}$ for nonimpulse sounds, but the predicted range of injury is not necessarily much less than for the higher thresholds at lower frequencies, because lower frequencies propagate better than higher frequencies. The sound energy required to cause injury judged by PTS is so great that zones of injury for even intense sound sources such as airguns and naval sonars are estimated at less than 1 km for all but a few cases. For example, a single one-second ping from one of the loudest naval sonars, the 53C, would be above the PTS threshold for HF cetaceans out to a range of 1 km given omnidirectional propagation, while it would be above the PTS threshold for mid-frequency and low-frequency cetaceans for less than 100 m from the source. These ranges suggested monitoring and mitigation measures that focused on detecting animals close to the source ship and suggest that the probability of marine mammals experiencing PTS from anthropogenic activities will likely be sufficiently low as to preclude any population-level effects.

Finding 2.4: Studies of noise levels that cause TTS and the growth in TTS with increasing noise are used to predict the occurrence of permanent hearing loss. Currently data exist for one species of otariid, two species of phocids, two species of mid-frequency (delphinid) cetaceans, and two species of high-frequency (phocoenid) cetaceans. Only a few individuals (one to five) of each species have been tested and within hearing groups there is wide variation in TTS onset and growth with increasing levels of noise. This variation indicates that the physiological effects of sound cannot be generalized based on testing of a few species of marine mammals, and more species need to be studied.

Behavioral Responses

Just about the time that data from TTS studies started to suggest limits on the ranges at which sound could injure marine mammals, evidence began to accumulate that lethal strandings of a poorly known group of whales called beaked whales coincided with naval sonar exercises. Frantzis (1998) described an atypical mass stranding where 12 Cuvier's beaked whales (*Ziphius cavirostris*) stranded over 38 km of a Greek bay over 2 days when a naval sonar was being tested. Issues with mid-frequency sonar came to national attention in the United States following the stranding of 17 cetaceans and the death of 7 during a naval sonar exercise on March 15-16, 2000, in the Northeast and Northwest Providence Channels of the Bahamas Islands. A joint U.S. Navy and U.S. Department of Commerce report (Evans and England, 2001) determined that "the cause of this stranding event was the confluence of the Navy tactical mid-range frequency sonar and the contributory factors . . . a strong surface duct, unusual underwater bathymetry, intensive active use of multiple sonar units over an extended period of time, a constricted channel with limited egress, and the presence of beaked whales that appear to be sensitive to the frequencies produced by these sonars." Usually when whales mass strand, they strand together at the same time. D'Amico et al. (2009) cataloged 12 atypical mass strandings of beaked whales that coincided with naval exercises that may have transmitted sonar. These strandings represent the most obvious and clearly lethal impact of anthropogenic sound on marine mammals.

Cox et al. (2006) reported on a workshop convened by the U.S. Marine Mammal Commission in 2004 to synthesize the current understanding of beaked whale strandings and to recommend research initiatives to determine the most probable causal pathways between transmission of mid-frequency sonar and strandings of beaked whales. The consensus from that meeting, which has not changed to date, was that a behavioral response occurring under a combination of contributory conditions was the progenitor of the strandings and the associated pathologies. Extensive behavioral, physiological, and anatomical research has been conducted over the last decade and a half to better understand not only this extreme example of the effect of anthropogenic sound on marine mammals but that of less dramatic chronic and episodic exposures. Some of the beaked whales that stranded during sonar exercises showed gas and fat emboli apparently caused by a decompression sickness (DCS) (Jepson et al., 2003; Fernández et al., 2005). Fernández et al. (2012) reported on three beaked whales that appear to have died at sea from decompression symptoms and then washed ashore, suggesting that whales do not just die from stranding, but may die directly from DCS at sea. These results have reinvigorated analysis of the diving physiology of deep-diving whales to better understand how they manage N_2 and other gases under hydrostatic pressure (Hooker et al., 2012). Current thinking is that anthropogenic noise can in some situations trigger behavioral reactions that may interfere with the ways whales manage gas under pressure and/or may cause whales to strand and die.

Dose–Response Relationships

This understanding that sound can trigger behavioral responses that may lead to injury or death motivated research to better define the relationship between exposure to sound and behavioral responses that could lead to effects that regulators view as "Level B takes" under the U.S. Marine Mammal Protection Act. Managing the impacts of underwater sound requires an understanding of the effect of this disturbance on individuals and the risk to the population. Dose–response relationships have commonly been used in toxicology to relate the level of exposure to the probability of a particular response or to the elicitation of different responses with differing levels of severity. When we discuss the first case, we will call these dose–p(response) relationships, and when we discuss the latter, we will call these dose–s(response) relationships. Toxicologists typically study genetically inbred laboratory animals under conditions designed to minimize stress, narrow the diversity of subjects, and control all variables except the experimental one to provide the strongest baseline condition for experimental detection of effects of known dosages of a single stressor. Behavioral responses of marine mammals are highly context dependent, being influenced by age (Houser et al., 2013a), sex (Symons et al., 2014), behavioral state (Sivle et al., 2012; Goldbogen et al., 2013), location (Tyack and Clark, 1998),

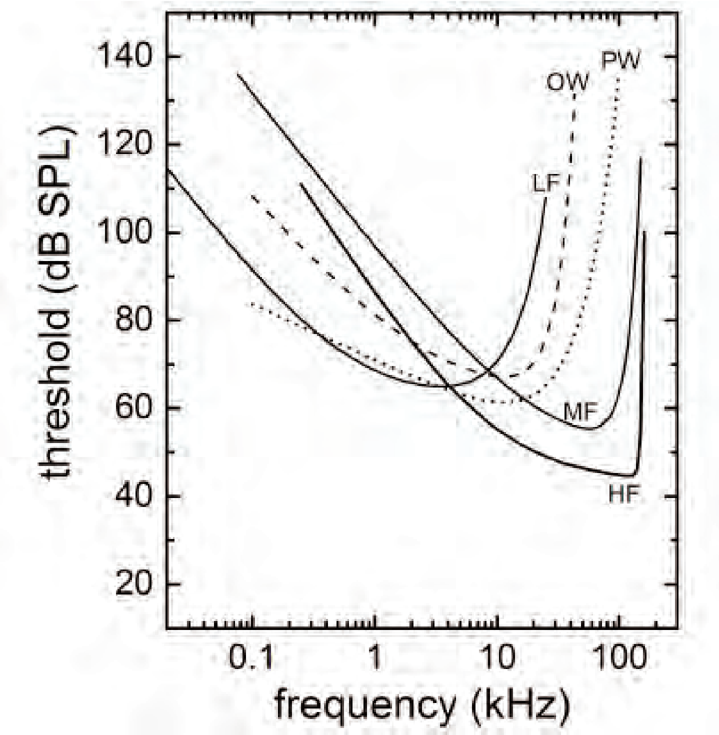


FIGURE 2.2 Composite audiograms obtained through behavioral testing except for LF that was calculated. NOTE: HF = high-frequency cetaceans; LF = low-frequency cetaceans; MF = mid-frequency cetaceans; OW = otariids, walrus, and sea otter in water; PW = phocids in water. Thresholds are expressed in dB_{RMS} re 1 µPa. SOURCE: Adapted from Finneran (2016; peer reviewed for NMFS [2016a]).

prior exposure resulting in habituation (Houser et al., 2013b) or sensitization (Kastelein et al., 2011), and individual sensitivities. Most experimental studies on the effects of an anthropogenic sound stimulus on marine mammals have been conducted with subjects drawn from wild populations. If the subjects are a representative sample of the contexts that affect responses, then the dose–response functions and other behavioral observations should be appropriate for the populations under study. Behavioral dose–response functions for three species were obtained from captive animals, and all TTS research has been done with captive animals.

One approach to estimating dose–response functions assumes a specific functional relationship between exposure and response. Many methods to estimate dose–response functions often assume a sigmoidal shape with a monotonic relationship between exposure and response. Some toxicological dose–response curves do not have this functional form (Calabrese, 2005), and we cannot assume that behavioral responses to sound will have a sigmoidal shape. Most dose–p(response) analyses assume a minimum exposure below which no response is expected and a maximum exposure above which all of the animals are assumed to respond. In the case of behavioral responses to sound, the minimum exposure can be assumed to occur at the limits of detectability as determined by the frequency-dependent audiograms. Ellison et al. (2011) emphasize the importance of context and environment in modulating the behavioral response to a given received level. Context includes current behavioral state and past exposure to the signal, and environment includes all the environmental factors that influence the signal-to-noise ratio and may result in a masked response threshold. DeRuiter et al. (2013) provided evidence that animals are more likely to show a response to a nearby signal at lower intensity than they do to a signal coming from farther away but with a greater received level. For example, tagged Cuvier's beaked whales responded to the simulated sonar at received levels as low as 89 dB re 1 µPa but did not respond to sonar from an active naval ship farther away with a received level up to 106 dB.

Within the U.S. regulatory structure, Level A takes (injury) are equated with exposures resulting in PTS, whereas both TTS and behavioral disruption are regarded as Level B takes. Level B behavioral takes are generally considered to be less severe than Level B physiological takes (TTS). It is likely that, at the maximum exposure for behavioral response, animals may already be experiencing TTS. Note that in the case of the beaked whale strandings, exposures well below those required for PTS did disrupt behavior in a way that led to the death of the animals that stranded, so the logic of this regulatory structure is questionable for some settings.

The importance of understanding how sonar initiates a behavioral response in cetaceans has been the impetus to several studies that have developed empirical dose–p(response) curves linking the probability of a behavioral response to a given sound exposure. Finneran and Jenkins (2012) constructed a behavioral response curve that is used by the U.S. Navy and its regulator to estimate the proportion of animals receiving a given sound level that will show the criterion behavioral response. The Finneran and Jenkins (2012) curve is based on a mathematical formula following Feller (1968) and based on data from Finneran and Schlundt (2004), Fromm (2009), and Nowacek et al. (2004). The threshold response level is set at 120 dB_{RMS} and the level at which the probability of response is 0.5 is at 165 dB_{RMS}, resulting in an asymptotic value of approximately 200 dB_{RMS} for 100% response.

Another approach used to estimate probabilistic dose–p(response) functions assumes that the distribution of the probability of responses as a function of exposure is Gaussian (truncated at a lower and upper SEL) and estimates the mean and variance for this relationship (Antunes et al., 2014; Miller et al., 2014). Hierarchical Bayesian models can be used to estimate dose–p(response) functions, assuming that each individual has a response threshold, and that the distribution of thresholds across the population is (truncated) normal. Observed levels associated with responses are then used to estimate the population mean and variance, which together with the minimum and maximum values can be used to estimate the dose–p(response) function.

Figure 1a in Box 2.2 shows the dose–p(response) function for killer whales exposed to 1-2 kHz and 6-7 kHz sonar, where the 50% response was at 141 ± 15 dB_{RMS} with thresholds ranging from 94 to 164 dB (Miller et al., 2014). Similar dose–p(response) functions have been determined for exposure to sonar for Blainville's beaked whale (RL_{p50} at 150 dB_{RMS}; Moretti et al., 2014), long-finned pilot whales (RL_{p50} at approximately 170 dB_{RMS}; Antunes et al. 2014), a captive harbor porpoise (RL_{p50} at 124-144 dB_{RMS} depending on sonar type; Kastelein et al., 2013), captive bottlenose dolphins (RL_{p50} at 162 dB_{RMS} on first trial and 174 dB_{RMS} by tenth trial; Houser et al., 2013b), and captive California sea lions (RL_{p50} at 147 dB_{RMS} increasing to 158 dB_{RMS} when sensitive juveniles [<2 years] were removed; Houser et al., 2013a). The responses used to establish the response function varied: presence or absence of a foraging dive in a 30-minute period for Blainville's beaked whale where the stimulus was actual naval sonar operations; a change in two-dimensional movement tracks for long-finned pilot whales where the stimulus was simulated sonar in a controlled exposure experiment (CEE); an avoidance reaction as determined by an expert group consensus for killer whales where the stimulus was simulated sonar in a CEE; a sudden change in swimming speed or direction for the captive harbor porpoise where the stimulus was synthesized sonar signals; and primarily based on a statistically significant change in breathing during a 30-second period for captive bottlenose dolphins and California sea lions where the stimulus was simulated sonar. These studies have generally been based on relatively small sample sizes, in some cases a single animal, but have indicated that the responses are dissimilar enough that taxon-specific rather than a generic odontocete exposure–response relationship is necessary for impact assessments (Antunes et al., 2014; Harris et al., 2015). The responses of captive bottlenose dolphins also suggested that they may be capable of habituation to repeated exposures (Houser et al., 2013b), in contrast to California sea lions that did not demonstrate habituation under a similar experimental protocol (Houser et al., 2013a). This does not mean that pinnipeds do not habituate to sounds under other circumstances, but simply that they did not show habituation under this experimental protocol.

The responses used to establish the above-referenced dose–p(response) functions have varied in severity and most of them would be considered minor on the 10-point severity scale presented by Southall et al. (2007). The responses noted above range in severity from 2 (brief or minor changes in respiration rate) for captive bottlenose dolphins and California sea lions, to 3 (minor changes in locomotion speed, direction, and/or dive profile but no avoidance of sound source) for captive harbor porpoises and long-finned pilot whales, to 4 (moderate changes in locomotion speed, direction, and/or dive profile but no avoidance of sound source) for Blainville's beaked whale, to 6 (minor avoidance of sound source) for killer whales. These experiments are designed so as not to harm the subjects. In this sense the experiments have succeeded, but it may take some extrapolation to predict thresholds for more severe responses if those are more relevant for a specific regulatory regime. Miller et al. (2012a) reviewed data from dose–s(response) experiments on killer, long-finned pilot, and sperm whales and reported that there was no consistent relationship between exposure and the severity score assigned to a response. It was noted that just-audible signals could result in responses of severity levels between 0 and 7. This variation highlights how different the responses of different individuals may be to similar acoustic levels of exposure. Ellison et al. (2011) suggest that contextual factors cause variability in responsiveness at low received levels, but annoyance/disturbance responses may be evoked in most animals over a relatively narrow range of high levels of acoustic exposure. This argues against assuming that the distribution of responses is likely to fit a symmetric normal distribution around a mean, but might better be viewed as a hybrid of several distributions driven by different processes.

Harris et al. (2015) demonstrated when combined killer whale, sperm whale, and long-finned pilot whale dose–p(response) data were plotted for three different levels of severity of response, a basically sigmoidal curve was generated for each severity level. For low severity of response, the curve reached 0.5 response probability at 153 dB_{SEL} and asymptoted at 1.0 probability at 167 dB_{SEL}. For medium severity of response, the curve reached 0.5 response probability at 155 dB_{SEL} and reached 1.0 probability at 180 dB_{SEL}. For the highest severity of response, the curve asymptoted at a 0.1 probability of response at 160 dB_{SEL}. The overall population effect will be a function of the probability of a response and the severity of the response. It is not yet possible to determine whether a greater probability of a less severe response or a lower probability of a more severe response will have the greatest population consequences.

Dose–p(response) relationships have not been estimated for the same marine mammal species in both captive and natural settings, but limited data suggest different responsiveness across these contexts, albeit using different criteria for the response. A free-ranging bottlenose dolphin tagged before the start of naval sonar exercises remained in the same general area during the 3 days of exercises and had modeled exposure levels up to 168 dB_{RMS} (Baird et al., 2014). This value is above the RL_{p50} for captive dolphins on the first trial at an exposure SPL of 162 dB_{RMS}. The response of free-ranging harbor porpoises to a commercial two-dimensional seismic airgun survey in the North Sea was determined through passive acoustic tracking. The density of porpoises was unchanged at 10 km at received SPL of 148 dB_{RMS} and reduced by 6% at 5 km at received levels of 155 dB_{RMS} (Thompson et al., 2013a). These levels are well above the RL_{p50} estimated for a captive harbor porpoise exposed to sonar (124-144 dB_{RMS}), although another captive harbor porpoise consistently exhibited an aversive behavioral reaction to seismic airgun sound at SPL above 174 dB_{RMS} (Lucke et al., 2009). Captive studies have provided necessary first-order information on dose–response relationships for species too small or too difficult to tag under current methods, but they are an inadequate proxy for dose–response relationships determined in free-ranging animals because the context is so different, and the suite of behavioral responses available to captive animals is restricted compared to that available to free-ranging animals. This lack of dose–response data is particularly important for small pelagic odontocetes that form the majority of animals predicted to be taken in many environmental assessments (e.g., U.S. Department of the Navy, 2013). The responses observed in captivity are also low on the severity scale and would be unlikely to have population consequences in the wild.

Finding 2.5: The selected response criterion for dose–response studies has typically been a low-severity response, but anomalous high-severity responses have been observed during these studies. Just-audible signals have resulted in responses of severity levels between 0 and 7. The severity levels were established based on assumed effects on individual fitness, and thus severe responses to low sound levels raise concerns regarding population consequences.

Finding 2.6: A primary reason for having no free-ranging dose–response curves for any of the smaller cetaceans is the lack of a suitable data recording package for attachment to these animals. The development of such a data recording package that would combine GPS with a measurement of sound exposure level is essential to estimate the impact of sound on these species that constitute the vast majority of cetaceans exposed to anthropogenic sound.

Many species of marine mammals continue to occupy U.S. naval test and training ranges in Southern California, the Bahamas, and Hawaii (Falcone et al., 2009; McCarthy et al., 2011; and Baird et al., 2014, respectively). These range animals have been observed to respond to sonar activities with changes in diving patterns and movements. For example, Blainville's beaked whales move to the periphery of the U.S. Navy's Atlantic Undersea Test and Evaluation Center (AUTEC) range during training exercises with multiple ships operating sonar. They return to the range within a few days after the training exercises have concluded (McCarthy et al., 2011; Tyack et al., 2011). It is very difficult for observational studies to demonstrate that sonar is the cause of these reactions (see Chapter 6). A combination of controlled experiments to demonstrate causation, with opportunistic observations of actual exercises to study the scale and significance of responses (Tyack et al., 2011), has proven particularly informative. The long-term consequences of the energetic costs of displacement and changes in foraging location and potential changes in foraging resources are not completely known, but a recent study (Claridge, 2013) has shown that the average animal abundance of beaked whales at AUTEC is lower than in an equivalent area at Abaco, an area 170 km away in the Bahamas where sonar exposure is limited. Also the female-to-calf ratio at AUTEC is higher, suggesting lower recruitment. Beaked whales have both capital and income breeding characteristics (Huang et al., 2011). New et al. (2013b) developed an energetic model that considered the impact of displacement from food resources on survival and reproduction of beaked whales. Their results showed that, while adult survival was relatively robust under reduced energy input, minor reduction in energy intake over an extended period could affect lifetime reproductive output.

Killer whales represent an existential threat to marine mammals of several species, so playback of killer whale calls has been used as a positive control in studies of responses to anthropogenic sound. Blainville's beaked whales (Tyack et al., 2011) and gray whales (Malme et al., 1983) show behavioral responses to playbacks of killer whale vocalizations when the signal-to-noise ratio is 0 dB. Some cetaceans also respond to some anthropogenic sounds, such as sonar at levels well below the current criteria for disturbance used in the United States. The 50% probability of a startle response for a captive harbor porpoise to playback of 6-7 kHz up-sweeps mimicking naval sonar signals occurred at SPL received levels of 101 dB_{RMS} (Kastelein et al., 2012). The minimum level for response of Cuvier's beaked whales to playback of sonar signals occurred at SPL received levels of 89-127 dB_{RMS}, although the whales did not respond to sonar from a distant warship at received SPL of 78-106 dB_{RMS} (deRuiter et al., 2013). The above data show that the thresholds defining behavioral harrassment used by NMFS (160 dB_{RMS} impulsive sounds; 120 dB_{RMS} nonimpulsive) need to be updated in light of the new data for sonar. Some harbor porpoises and Cuvier's beaked whales respond at levels well below the 120 and 140 dB_{RMS} response thresholds currently used for these species. Similarly, the 50% probabilities of response are in most cases below the 165 dB_{RMS} previously used in environmental impact assessments for naval activities. As described in Box 2.2, the current method of calculating takes based on response thresholds can lead to an underestimate of the number of animals taken.

Masking

With behavioral responses being observed at dose levels close to the limits of detectability in some cases, and with detectability used to set the minimum exposure at which the dose–response function starts, the acoustic signal-to-noise ratio needs to be considered when it limits detectability through masking. Masking occurs when the level of detectability for one sound is increased in the presence of a second sound by an amount expressed in dB. The mammalian ear has been modeled as a bank of overlapping band-pass filters[8] and only energy in the band-pass filter centered on the sound being detected, the critical band, contributes to the masking of that sound (Fletcher, 1940). While this has been investigated most thoroughly for Gaussian[9] noise, it does not hold true for many natural and anthropogenic noises that have complex spectra and amplitude fluctuations. Through a phenomenon known as comodulation masking release (Trickey et al., 2010), the broader the frequency band of the natural noise is outside the critical band, the more the masking is reduced compared to what it would have been with Gaussian noise in the critical band. Masking has been considered primarily in the case where the second sound represents noise for the species or individual in question. For example, concern has been expressed that shipping noise, which has increased since the advent of motorized vessels, overlaps with the frequency range of important social calls of baleen whales, including blue (Mellinger and Clark, 2003), fin (Watkins et al., 1987), and right (Parks et al., 2007a) whales. The primary concern here has been that elevated ambient noise would reduce the range over which whales could detect calls of conspecifics.

Clark et al. (2009) have proposed analyzing the potential effect of masking through a calculation of the reduction in communication space for several species of baleen whales. They found the most profound reductions due to the modeled passage of two ships within 4 km of a right whale in the Stellwagen Bank National Marine Sanctuary, where the aggregate exposure resulted in an 84% reduction in the communication space for that animal. Hatch et al. (2012) calculated an overall 63% reduction in communication space for right whales in Stellwagen Bank National Marine Sanctuary compared to what they experienced in the mid-20th century, when background levels were estimated to be 10 dB below the lowest 5% of all the background levels currently recorded.

One serious problem with these predictions is that they ignore compensation mechanisms that whales use to maintain the effective range of their communication signals in noise. The natural environment in which animal communication evolved has significant variation in noise, for example from rain (heavy rain causes up to a 40 dB increase) or waves and bubbles caused by wind (8 dB increase between Beaufort 0.5 and 1.0), and most birds and mammals have evolved mechanisms to compensate for this natural variation in noise. One of the most pervasive compensation mechanisms is the Lombard effect, by which animals increase the source level of their calls in increased noise (Brumm and Zollinger, 2011). All birds and mammals tested have shown the Lombard effect, and marine mammals are no exception. Killer whales increased their call amplitude by 1 dB for every dB increase in background noise created by motorized vessels (Holt et al., 2009). Making louder calls in increased noise can have an energetic cost; bottlenose dolphins increase their metabolic rate as the acoustic energy of their vocalizations increases (Holt et al., 2015). In the case of the right whales in Cape Cod Bay, the location modeled by Clark et al. (2009), Parks et al. (2010) showed that individual right whales elevate the source level of their calls as the noise level increases. In addition, as shipping noise chronically increased from the 1960s to the 1990s, right whales have increased the fundamental frequency of their calls by about an octave, outside of the peak frequency of shipping noise (Parks et al., 2007b). These mechanisms are not taken into account in the Clark et al. (2009) model, making it unrealistically extreme in its predictions of reduction of effective space. Other mechanisms by which human engineers compensate for noise include making signals longer and/or more redundant. These mechanisms are also used by marine mammals; humpback whales increased the duration of their songs by 29% in the presence of low-frequency active sonar, and this was produced by increasing the redundancy of the song (Miller et al., 2000).

In addition to potential effects on communication space, shipping can also act as a physiological stressor. Rolland et al. (2012) measured fecal glucocorticoids in North Atlantic right whales in the Bay of Fundy during the summers of 2001-2005. Shipping activity was reduced by 67% and the associated noise levels declined by about 6 dB immediately after the attack on the World Trade Center on September 11, 2001. This reduction in ship movement and noise was associated with a reduction in stress-related glucocorticoids compared to other years and before September 11, 2001. However, this opportunistic study lacked the controls required for standard experimental design.

Impulsive Sources

Impulsive sources affect animals differently than relatively continuous sources. The rise time and peak pressure (measured in kPa) are more important metrics than the root mean square (RMS) value of the received level. Depending on the interpulse interval, the auditory system may have an opportunity to partially recover between pulses. As noted previously, the current NMFS threshold for behavioral response to impulsive sounds is 160 dB_{RMS} and for nonimpulsive sounds it is 120 dB_{RMS}. The primary sources of impulsive sounds that marine mammals experience come from seismic activity associated with oil and gas exploration; pile driving associated with construction of bridges, docks, and wind farms; and some acoustic deterrent devices associated with fishing and aquaculture.

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[8] A band-pass filter allows a range of frequencies to pass with minimum attenuation and strongly attenuates frequencies outside that band. The width of the band-pass is typically given as the frequencies above and below the center frequency at which the attenuation is 3 dB.

[9] Gaussian noise has a normal distribution of instantaneous amplitudes over time.

Seismic Surveys

Responses to seismic surveys have been studied in a variety of marine mammals. The following overview captures most of the salient results but is not a comprehensive literature review. Romano et al. (2004) sampled blood from a captive beluga whale (*Delphinapterus leucas*) and bottlenose dolphin (*Tursiops truncatus*) after exposure to underwater impulsive sounds from a seismic water gun. For the beluga whale, levels of norepinephrine, epinephrine, and dopamine were significantly higher for peak pressure levels of 116 to 198 kPa. For the dolphin, serum levels of aldosterone were significantly elevated and monocytes decreased after exposure to peak pressure levels of 146 to 220 kPa. Miller et al. (2009) conducted controlled approaches of a commercial seismic survey vessel to make pass-bys of sperm whales in the Gulf of Mexico. The whales, which were exposed to received levels varying from 120 to 147 dB_{RMS} at ranges varying from 1.4 to 12.8 km, did not change their direction of travel or behavioral state in response to exposure, but did decrease the energy they put into swimming and showed a trend for reduced foraging. Madsen et al. (2002) studied responses of sperm whales in Norwegian waters to seismic surveys at ranges greater than 20 km and reported no responses at exposure ranging up to 123-130 dB_{RMS}. Avoidance responses have more commonly been reported for baleen whales. Avoidance responses to airgun sounds at received levels of 160-170 dB_{P-P} re 1 μPa have been reported for migrating gray whales (Malme et al., 1983), bowhead whales (Richardson et al., 1986), and migrating humpback whales (McCauley et al., 2000). Fin whales moved away from a 10-day seismic survey in the Mediterranean and were spatially displaced for at least 14 days after the seismic airgun shooting period (Castellote et al., 2012). The survey area affected was estimated to be about 100,000 km^2 (Castellote et al., 2012).

Pile Driving

Pile driving is used in the construction of structures, such as piers and bridges, and the installation of oil and gas platforms and offshore wind turbines. The impact of pile driving for offshore wind turbines has been of particular concern for marine mammals because of the high source level (Madsen et al., 2006). Pile driving produces broadband, multiple pulsed sounds, similar to seismic airgun surveys, with the peak energy below 1 kHz (Bailey et al., 2010). During pile driving, hammer strikes occur about every 1-2 seconds and the piling duration is generally several hours for each pile with the interval between piles varying from minutes to days (Bailey et al., 2010; Dähne et al., 2013). Source levels vary depending on the size of the pile and method of pile driving, but have been estimated to be 226-257 dB_{P-P} re 1 μPa at 1 m based on recorded levels back-calculated to 1 m (OSPAR, 2009; Bailey et al., 2010). Sound levels of 205 dB_{P-P} at 100 m (Bailey et al., 2010) and energy up to 176 dB_{SEL} re 1 μPa^2-s at 720-750 m distance (Brandt et al., 2011; Dähne et al., 2013) have been reported.

In Europe, assessments of the impacts of offshore wind developments on marine mammals have focused on small cetaceans and pinnipeds (Bailey et al., 2014).The response of marine animals to the construction phase, particularly the pile-driving activity, has primarily been studied for the most abundant cetacean species in the North Sea, the harbor porpoise (*Phocoena phocoena*). Harbor porpoises have been reported to exhibit an avoidance response to the impulsive sound of pile driving at distances of 20 km or more and for up to 3 days (Tougaard et al., 2009; Thompson et al., 2010; Brandt et al., 2011). There is currently a lack of data for large whales. Large whales are classified as having low-frequency hearing (see Figure 2.2), which suggests that they may be most sensitive to pile-driving sounds. Offshore wind energy areas have been identified and leased by the Bureau of Ocean Energy Management on the U.S. Outer Continental Shelf where a number of whale species, many of which are listed as endangered species, are known to occur. As offshore wind energy facilities begin to be installed off the U.S. coast, studies on the short- and long-term responses of large whales will be particularly important for determining the potential population-level consequences.

Acoustic Deterrent Devices

Acoustic deterrent devices (ADDs) are intentionally designed to deter wildlife such as marine mammals from depredating resources such as fish in a fish farm. A variety of different ADDs have been developed to deter seals from depredating fish farms (reviewed by Nowacek et al., 2007; Götz and Janik, 2013). Götz and Janik (2013) reviewed mixed evidence on the effectiveness of ADDs in reducing depredation by seals. Activation of ADDs in some settings was associated with increased depredation, perhaps through broadcasting the location of a food source (Geiger and Jeffries, 1987; Jefferson and Curry, 1996). In other settings, ADDs were judged by fish farmers to vary from ineffective to moderate effectiveness in different sites (Quick et al., 2004; Sepulveda and Oliva, 2005). In cases where ADDs were associated with reduced depredation, some showed a decreased effect over time, which could be due to habituation (Groves and Thompson, 1970), tolerance (Bejder et al., 2009), or hearing damage due to exposure to the ADDs (Reeves et al., 1996).

In contrast to the mixed evidence for effectiveness of ADDs on the target pinnipeds, there is strong evidence that operation of ADDs causes some odontocetes to avoid large areas of habitat. Morton and Symonds (2002) studied the presence of killer whales in inshore waters of British Columbia where their distribution had been well studied for more than a decade before four ADDs were installed. Sightings of killer whales were significantly reduced in the roughly 10 km^2 area where the ADDs were installed during the 6-year period of their use, and then recovered to baseline after their use ended. Olesiuk et al. (2002) report a similar sharp decline in sightings of harbor porpoise out to their maximum sighting range of 3.5 km when ADDs were activated for periods of 3 weeks. Brandt et al. (2013) showed a similar decrease in the abundance of porpoises detected out to ranges of 7.5 km from an ADD when it was operating. None of these studies suggest much habitation in the response of odontocetes to ADD signals.

INDIRECT EFFECTS OF SOUND ON MARINE MAMMALS

Marine mammals are among the animals with the most sensitive underwater hearing, but sound may also affect them indirectly through effects on prey, predators, or competitors. Indirect effects of stressors may be more important than direct ones (Ockendon et al., 2014).

Effects on Prey

Some fish are specialized to hear the pressure component of sound. A few species of herring (subfamily Alosinae) can detect the ultrasonic clicks that toothed whales use to find their prey. Wilson et al. (2011) demonstrated that one of these species swims away from these clicks, in a directional antipredator response. Mann et al. (1998) showed that shad respond to echolocation clicks at received levels of 171 dB$_{\text{P-P}}$. This level is high enough that few sources of noise would be likely to mask the clicks, so it is unlikely that elevated noise would make the shad less likely to escape. Most prey of marine mammals detect the particle motion component of sound rather than the pressure component. This mode of hearing limits the ability of animals to hear sounds with wavelengths smaller than roughly their body size, so these animals do not hear well above a few kilohertz. However, some low-frequency sources of anthropogenic sound, such as airguns used in seismic surveys, have been shown to affect the hearing and behavior of fish. McCauley et al. (2003) found that caged fish exposed to repeated passes of a seismic air gun (source level of 222.6 dB$_{\text{P-P}}$ re 1 μPa at 1 m) starting 400-800 m away and passing within 5-15 m of the cage experienced significant hair cell damage that remained unresolved 58 days later. They note that, had the fish not been caged, they would have swum away as they tried to do within the confines of the cage at first hearing of the seismic gun. Engås et al. (1996) report that the catch of cod and haddock was reduced by 50% when airguns began to transmit sound. Reductions in catch were observed 33 km away from the survey and lasted more than 5 days after the airguns stopped operating. The acoustic density of cod and haddock was reduced by 45% during the seismic survey and by 64% post survey. In contrast Løkkeborg et al. (2012) found that gillnet fisheries yields increased during a seismic survey while longline fisheries yields decreased. Acoustic mapping of fish abundance showed only pollock were displaced from the fishing grounds in this study. Løkkeborg et al. (2012) note that the airgun discharge rate was 19 times higher in the Engås et al. (1996) study, and they point out that the lower levels of exposure could explain the lower level of response in their study. If avoidance behavior reduces the prey of marine mammals, it could affect their feeding even if the sound does not affect them directly. However, short-term displacement of prey may have few consequences for marine mammals. Prey often move considerable distances for a variety of reasons, and presumably marine mammals can usually move to relocate them.

There is evidence that continuous noise, similar to the sound of shipping, may increase the mortality of eggs and larvae of a minnow (*Cyprinodon variegatus*; Banner and Hyatt, 1973) and decrease the growth of larvae of the minnow and longnose killifish (*Fundulus similis*). Regnault and Lagardère (1983) showed that exposure to noise 30 dB above ambient increased the metabolic rate of the shrimp *Crangon crangon* in an aquarium, with a significant reduction in growth and reproduction and elevated mortality (Lagardère, 1982). If chronic exposure to noise reduces the abundance of fish and invertebrate prey of marine mammals, this could reduce the quality of their habitats, resulting in site abandonment or survival and reproductive costs for individuals that remain.

Effects on Predators

Sharks and killer whales are some of the primary predators of marine mammals. Sharks do not have particularly sensitive hearing, so effects of noise are likely to be minimal. However, killer whales not only have excellent hearing, but have also been shown to be more responsive to low- and mid-frequency sonar than some other toothed whales, such as sperm and pilot whales (Harris et al., 2015). If killer whales avoid noise sources at greater ranges than potential prey, this could create a zone near the noise source with a lower risk of predation. Noise-mediated predator shelters or shields have been documented in terrestrial systems where songbird nest predators appear to be more sensitive to chronic noise than are their prey (Francis et al., 2009). In the same system, Francis et al. (2012b) found evidence of additional indirect effects with potential long-lasting consequences for the ecosystem. Specifically, the reduced recruitment of piñon pine (*Pinus edulis*), a foundational species, in noisy areas is linked to avoidance of noisy areas by a key seed disperser, the Woodhouse's scrub-jay (*Aphelocoma woodhouseii*), and increased abundance of important seed predators. These studies highlight how noise, like other anthropogenic stressors, can have indirect effects that reverberate throughout communities by interfering with interactions among species. Given the many pathways by which anthropogenic noise could affect marine mammals, a potential benefit from a predator shield must be weighed against potential costs of persisting in noise-exposed zones.

Effects on Conspecifics

Different kinds of noise can have varying effects on social cohesion in different species. Buckstaff (2004) showed that, as a motorboat approaches a group of bottlenose dolphins (*Tursiops truncatus*), the dolphins will increase the rate at which they produce signature whistles, followed by increased social cohesion (Nowacek et al., 2001). When sonar signals trigger a flight reaction, this can interfere with normal social cohesion, leading to separation of members of a group. For example, Miller et al. (2012a) report on a group of killer whales exposed to a playback of mid-frequency sonar sounds. When the received level of these sounds reached 152 dB_{RMS}, a calf that had been in the group was seen to have separated from the group. Miller et al. (2011) notes three unique characteristics of this experiment to this exposure session: it was the only repeated mid-frequency active sonar up-sweep exposure presented to the same group of animals; the experiment was conducted in an unusually narrow fjord roughly 1 km wide; and transmissions were started unusually close to the subjects. The calf rejoined the group after 86 minutes, and remained with the group for many hours after exposure. However, this separation was scored as quite a severe response because it could have had more serious consequences for the calf. High-latitude adult male sperm whales that are usually solitary responded to playback of killer whale vocalizations by clustering together at the surface and producing social alerting sounds (Curé et al., 2013).

RECOMMENDATIONS

Recommendation 2.1: Additional research will be necessary to establish the probabilistic relationships between exposure to sound, contextual factors, and severity of response.

Recommendation 2.2: Uncertainties about animal densities, sound propagation, and effects should be translated into uncertainty on take estimates, for example, through stochastic simulation. Regulators may then choose the level of risk they wish to use in deciding whether to permit an activity.

3

Current Understanding of Stressors

INTRODUCTION

Although increased noise exposure is a concern for marine mammals, other anthropogenic activities also serve as potential stressors that can alter individual behavior and health and contribute to cumulative impacts. In general, a stressor can be defined as any causal factor or stimulus, occurring in either the animal's internal or external environment that challenges the homeostasis of the animal. Marine mammals are exposed to a diverse set of both intrinsic and extrinsic stressors during their lifespan (see Table 3.1).

There are short-term internal stimuli that evoke myriad physiological responses occurring daily to maintain an organism near its homeostatic set points, but these are not considered stressors. However, aspects of the life cycle that result in significant changes to the set points are considered intrinsic stressors, and inherent in the life-history strategies of marine mammals are numerous features that constitute such stress. Many marine mammals are capital breeders that fast during reproduction or periods on shore. These species are intrinsically nutritionally stressed during reproduction and during migration away from foraging habitat. The amphibious lifestyle of pinnipeds requires that even income breeding species undergo food deprivation while on shore for breeding. Extended periods on shore have been associated with increases in stress hormones in numerous species (Champagne et al., 2012). Species that fast as part of their natural life history may exhibit intrinsic stress during or just after reproduction. During pregnancy, even species that do not fast will undergo significant physiological changes, including metabolic, cardiovascular, respiratory, immuno-

TABLE 3.1 Definition and Examples of Types of Stressors

	Definition	Examples
Intrinsic Stressor	An internal factor or stimulus that results in a significant change to an animal's homeostatic set points	Pregnancy, lactation, migration, molting, fasting (e.g., during the breeding season in capital breeders)
Extrinsic Stressor	A factor in an animal's external environment that creates stress in an animal	*Anthropogenic:* Pollutants, ship strike, entanglement, noise, psychological factors (e.g., perceived threat) *Natural, but potentially influenced by anthropogenic activity:* Harmful algal blooms, resource limitation, predator pressure, pathogens, temperature, salinity, naturally occurring chemicals, intra- or interspecific competition
Ecological Driver	A biotic or abiotic feature of the environment that affects multiple components of an ecosystem directly and/or indirectly by changing exposure to a suite of extrinsic stressors	Loss of keystone or foundation species, recurring climate patterns such as El Niño, climate change

logical, and hematological changes, in order to accommodate the growing fetus.

In addition, there are extrinsic stressors that arise from chemical, physical, or biological factors in an animal's external environment. Extrinsic stressors may be specifically associated with anthropogenic activities (e.g., pollutants or ship strike) and include psychological factors that occur when human activities are perceived as a threat, typically a predatory threat (e.g., sonar; Isojunno et al., 2016). Extrinsic stressors may also be prompted by natural factors, although these natural factors are often influenced by anthropogenic activities to some degree (e.g., disease or resource limitation), making it difficult to classify the extrinsic stressor as unequivocally natural. Regardless of whether causal factors are purely natural or not, these stressors have potential to influence an animal's responses to other anthropogenic stressors. In addition, how the animal responds to extrinsic stressors is dependent on its physiological capacity, which is modulated by intrinsic stressors. So long as the extrinsic stressors and intrinsic stressors do not exceed the animal's ability to maintain organismal function (i.e., allostasis; McEwen and Wingfield, 2003), effects on health and reproduction that lead to population impacts are unlikely. Numerous studies have evaluated the impact of the various extrinsic stressors on the individual health, survival, and reproduction of marine mammal species, although these studies have been biased toward pinnipeds (reviewed by Atkinson et al., 2015). At the extreme, extrinsic stressors can result in increased mortality, demographic impacts, and even cohort failures in some marine mammal species. The cumulative effect of whatever combination of these existing intrinsic and extrinsic stressors to which an individual is exposed will influence the impact of any additional anthropogenic stressors on individuals and consequently their population-level effect.

Many extrinsic stressors can be the products of larger phenomena that are identified as ecological drivers. An ecological driver is a biotic or an abiotic feature of the environment that affects multiple components of an ecosystem directly and/or indirectly by changing exposure to a suite of extrinsic stressors. Ecological drivers may operate on multiple species at varying trophic levels and may even affect multiple ecosystems.

POTENTIAL ENVIRONMENTAL (EXTRINSIC) STRESSORS

Human activities can potentially cause mortality, injury, disturbance, and stress to marine mammals. Activities that result in immediate fatalities, such as bycatch, hunting (or other deliberate killing), and collisions with ships, will increase the population mortality rate above that caused by natural factors alone. These lethal stressors directly affect population abundance. In contrast, human activities with nonlethal effects on marine mammals may affect their behavior and physiology and lead to impacts on their health. The cumulative effect of these human activities, along with natural extrinsic stressors, on the health of individual animals may result in changes in their reproduction and survival that then affect population dynamics. In this section the committee reviews and discusses environmental stressors and their associated effects that have been reported for marine mammals. The focus is on those stressors that have been emphasized in the literature, and/or that have strong potential to interact with other stressors due to chronicity of exposure (e.g., persistent chemical contaminants to which many marine mammals are exposed over a lifetime), or the potential for a sublethal but chronic effect (e.g., permanent damage to an organ system). This should not be considered an exhaustive list of all possible environmental stressors that have potential to affect marine mammals. A comprehensive review of all potential stressors is beyond the scope of this report.

Physical Injury

Fishery Interactions

Entanglement in fishing gear represents an important source of injury and mortality in marine mammals. Bycatch mortality is estimated globally to exceed hundreds of thousands of marine mammals each year (Read et al., 2006). Bycatch occurs most frequently in association with gillnet fisheries. There is a strong spatial component to bycatch of marine mammals, with "hotspots" influenced by marine mammal density (Block et al., 2011), fishing intensity (Stewart et al., 2010), or both (Lewison et al., 2014). Spatial overlap between fisheries and marine mammals is often associated with coastal zones, shelf breaks, upwelling regions, and frontal zones (Hyrenbach et al., 2000; Scales et al., 2014). When not immediately fatal, entanglement or ingestion of fishing gear can impede the ability of marine mammals to feed and can cause injuries that eventually lead to infection and death (Wells et al., 2008; Cassoff et al., 2011; Moore and van der Hoop, 2012). Weakened animals may be more susceptible to predation (Moore and Barlow, 2013). There are also costs likely to be associated with nonlethal entanglements in terms of energy and stress (Moore and van der Hoop, 2012). The prevalence of scars on North Atlantic right whales (*Eubalaena glacialis*) associated with entanglements indicates the persistent and repetitive nature of this threat (Knowlton et al., 2012).

Vessel Collision

Collision with ships is a key threat to large whales (Laist et al., 2001; Thomas et al., 2016). Vessel strike also poses a risk to manatees (Runge et al., 2015) and small cetaceans in heavily populated coastal regions (e.g., Wells et al., 2008), and the risk may increase when illegal feeding has conditioned the animals to approach vessels (Donaldson et al., 2010). Several studies have estimated quantitative relationships (i.e., dose–response relationships) between vessel speed and the lethality of collisions for large whales (Vanderlaan and Taggart, 2007; Wiley et al., 2011; Conn and Silber, 2013). Even when it is not lethal, collision with a vessel causes stress and injury, which could make individuals more susceptible to negative sequelae following exposure to subsequent stressors.

Toxic Compounds

Nonbiological Toxins

Chemical contaminants, particularly those that are persistent in the environment, are a concern for marine mammals that often occupy high trophic positions. Persistent organic pollutants (POPs), which include legacy pesticides (e.g., DDT and chlordane), legacy industrial-use chemicals (e.g., polychlorinated biphenyls [PCBs]), and emerging contaminants of concern (e.g., polybrominated diphenyl ethers and perfluorinated compounds) accumulate in fatty tissues of marine organisms and are magnified through the food chain, leading upper trophic predators to be highly exposed. High concentrations of PCBs and DDT have been reported in tissues of marine mammals in most parts of the world, particularly in coastal regions adjacent to heavy coastal development and/or industry (Ross et al., 2000; Houde et al., 2005; Kajiwara et al., 2006; Kucklick et al., 2011). These legacy POPs have been linked to a number of adverse health effects, but primary concerns relate to endocrine disruption, and specifically thyroid hormone disruption (Sormo et al., 2005; Boas et al., 2006; Tabuchi et al., 2006; Schwacke et al., 2012), reproductive impairment or developmental effects (Reijnders, 1986; Ulbrich and Stahlmann, 2004; Hall et al., 2009), and immune dysfunction or disease susceptibility (De Guise et al., 1998; Van Loveren et al., 2000; Jepson et al., 2005). Polybrominated diphenyl ethers (PBDEs), commonly used as flame retardants, are another class of POPs that have spread globally in the environment and have also been reported in a broad array of marine mammal species (Houde et al., 2009; Rotander et al., 2012). The toxicity of PBDEs has not been as thoroughly investigated in comparison to PCBs, but rodent studies have suggested developmental neurotoxicity with learning and memory impairment that can persist into adulthood, and decreased thyroid hormone production similar to the toxic effects of PCBs (Eriksson et al., 2001; Branchi et al., 2003). PBDEs can be biotransformed to hydroxylated brominated diphenyl ethers, which exhibit greater toxicity for some effect end points as compared to their parent compound, and some studies have suggested that biotransformation of naturally occurring compounds in the marine environment may be an even greater source of the hydroxylated analogues as compared to the anthropogenic flame retardants (Wiseman et al., 2011).

POPs bind to fatty tissues and as such are sequestered in the blubber of marine mammals. Concentrations are likely maintained at equilibrium, or increase with age if the exposure continues, until an event (e.g., parturition, lactation, seasonal blubber changes, or loss of prey base) prompts blubber depletion and mobilization of the sequestered contaminants (reviewed by Houde et al., 2005). Once contaminants are mobilized, they may be more likely to reach target organs and initiate mechanistic pathways for adverse health effects. Therefore, POPs have potential to affect an individual over a lifetime, depending on life events and whether or not there is continued exposure. Neonates and dependent calves or pups may be particularly susceptible due to high concentrations of POPs that are offloaded from mother to offspring through milk (Wolkers et al., 2004; Yordy et al., 2010).

Aside from POPs, other organic compounds of concern include polycyclic aromatic hydrocarbons (PAHs). PAHs exist naturally in the environment but can also be from anthropogenic sources. Crude oil, fumes, vehicle exhaust, coal, organic solvents, and wildfires are all potential sources for PAHs. Exposure may be continual, associated with runoff from impervious cover in developed coastal regions, or natural seeps that produce low-level but steady exposure. Acute events such as oil spills may produce pulses of more significant exposure. Depending on the route of exposure (inhalation/aspiration, ingestion, or direct dermal contact), PAHs can produce a broad range of health effects. Lung disease, disruption of the hypothalamic-pituitary-adrenal (HPA) axis, and altered immune response have been reported in marine mammals as well as experimental mammal species following exposure to oil (Mazet et al., 2000; Schwartz et al., 2004; Mohr et al., 2008; Schwacke et al., 2014a) or inhalation of smoke associated with wildfires (Venn-Watson et al., 2013). Although PAHs are more rapidly metabolized and do not accumulate as is the case with POPs, the toxic effects (lung disease and HPA-axis damage) may be long lasting and initiate chronic disease conditions (Smith et al., 2017). Heavy metals, particularly mercury—which has been associated with immunological and neurotoxic effects and can cause permanent damage to the brain (Kakuschke and Prange, 2007; Farina et al. 2011)—have also been widely measured in the tissues of marine mammals (Dietz et al., 1996; Wagemann et al., 1996; Weihe et al., 1996; Seixas et al., 2008). Comparison of mercury tissue concentrations with established toxicological thresholds have indicated that some Arctic marine mammal species are at risk of neurological effects (Dietz et al., 2013), and levels of mercury in Arctic regions have been increasing in recent decades (Dietz et al., 2009; Rigét et al., 2011).

Despite the vast evidence to suggest that marine mammals are exposed to anthropogenic, as well as natural, chemicals capable of producing significant toxic effects, only a few studies have actually examined the impacts on population survival or reproductive rates (e.g., Hall et al., 2006; Lane et al., 2015). Such observational assessments are inherently challenging due to the difficulty in controlling for confounding or interacting variables, as well as the sublethal but chronic nature of chemical contaminant effects, and the difficulty of observing mortality or reproductive end points in long-lived marine mammal species, particularly cetaceans. Even fewer studies have attempted to develop quantitative relationships relating a given dose of a chemical to changes in a vital rate (e.g., reduced fecundity) and have had to rely on data from experiments with other mammalian species (e.g., Schwacke et al., 2002; Hall et al., 2006).

Biological Toxins

Marine algal toxins are produced by unicellular algae that are often present at low concentrations but that may proliferate to form dense concentrations under certain environmental conditions. When high cell concentrations form, the toxins that they produce can harm the health of marine life, and this is referred to as a harmful algal bloom (HAB). Marine mammals can be exposed to HAB toxins directly by inhalation or indirectly through food web transfer (Durbin et al. 2002), and these toxins can cause severe neurotoxic effects (reviewed by Van Dolah, 2005). Mortality and morbidity related to HAB toxins have been increasingly reported over the past several decades, and biotoxicosis has been a primary contributor to large-scale die-offs across marine mammal taxa (Van Dolah, 2005; Simeone et al., 2015). Since 1998, multiple die-offs as well as abortions and premature parturition have been reported for California sea lions (*Zalophus californianus*) in relation to domoic acid, a toxin produced by diatoms of the genus *Pseudonitzschia* (Scholin et al., 2000; Bejarano et al., 2008a). Furthermore, studies have determined that even sea lions that survive can suffer sublethal effects that could influence reproduction and longer-term survival (Gulland et al., 2002; Goldstein et al., 2008, 2009). Impacts of *Pseudonitzschia* blooms on marine mammal populations along the western U.S. coast have not been limited to sea lions; domoic acid has also been linked to mortalities of balaenopterids, delphinids, phocoenids, and mustelids (Van Dolah, 2005). Domoic acid has also been detected in tissues of marine mammals along the southeast U.S. coast (Schwacke et al., 2010; Twiner et al., 2011), but perhaps of greater concern in this area are the brevetoxins produced by Gulf of Mexico red tides. Brevetoxin has been implicated in multiple die-offs involving common bottlenose dolphins (*Tursiops truncatus*), as well as the endangered Florida manatee (*Trichechus manatus latirostris*) (Flewelling et al., 2005; Twiner et al., 2012; Simeone et al., 2015). Other HAB toxins, such as saxitoxin and ciguatera toxins, have been implicated in morbidity or mortality of other marine mammals, including humpback whales (*Megaptera novaeangliae*) and endangered monk seals (*Monachus* sp.) (Reyero et al., 1999; Bottein et al., 2011; summarized by Van Dolah, 2005).

Parasites and Pathogens

Parasites are ubiquitous. Parasites have the ability to cause disease and to function as pathogens. Microparasites, which include viruses, bacteria, fungi, and protozoa, multiply inside the host and are frequently associated with immune responses and development of host immunity in healthy animals. Macroparasites, which include helminths and arthropods, are larger in size and have complex life cycles that frequently involve more than one host for reproduction.

Microparasites can infect respiratory, central nervous, or other organ systems causing morbidity and mortality (e.g., Guzmán-Verri et al., 2012; Van Bressem et al., 2014; Simeone et al., 2015), and in some cases have been associated with epidemics that produce significant mortality. For example, viral pathogens of the genus *Morbillivirus* have been associated with severe respiratory illness and linked to large-scale die-offs of marine mammal populations worldwide (Van Bressem et al., 2014). Endemic microparasites may sporadically infect a smaller number of animals, but contribute to natural mortality as well as to widespread, low-level disease that in some cases may affect reproduction (e.g., *Brucella* sp.; Guzmán-Verri et al., 2012). Similarly, macroparasites may chronically infect marine mammals and contribute to low-level mortality or morbidity that reduces fitness or resilience (Simeone et al., 2015). Perrin and Powers (1980) estimated that 11-14% of natural mortality in spotted dolphins (*Stenella attenuata*) was attributable to the nematode *Crassicauda* sp. based on the prevalence of cranial lesions by age in spotted dolphins incidentally killed in the eastern tropical Pacific tuna fishery. The distribution of parasites and thus the risk of exposure and subsequent infection in marine mammals can be influenced by human activities. For example, domestic or human-managed animal populations and landscape alteration can affect terrestrial parasite distribution, and in coastal areas this can influence the risk for land-to-sea transmission. Such an influence has been supported by studies of *Toxoplasma gondii* transmission from terrestrial animals (feral cats and wildlife) to marine mammals in adjacent coastal waters (VanWormer et al., 2013, 2014).

Resource Limitation

Competition between marine mammals and fisheries has long been recognized (Northridge, 1984), and there is little doubt that this competition can be significant. For example, Punt and Butterworth (1995) estimated that the South African west coast population of Cape fur seals consumed some 600,000 tons of commercially valuable fish, such as Cape hake—in contrast to the average annual landings of 50,000 tons of Cape hake by South African fishing fleets. Conversely, Ford et al. (2010) discovered a strong bottom-up effect on the abundance of fish-eating killer whales in the northeastern Pacific Ocean from the availability of their

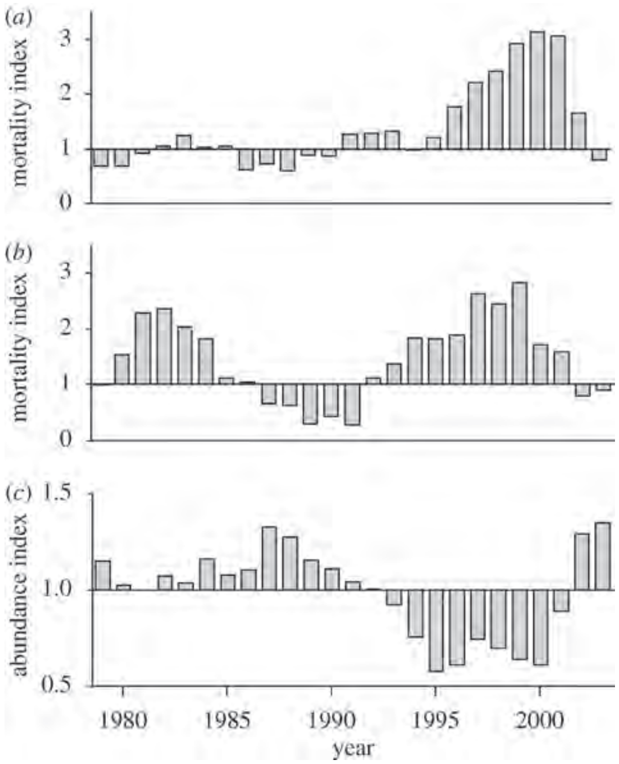

FIGURE 3.1 Mortality of (a) northern and (b) southern resident killer whales negatively covary with (c) abundance of Chinook salmon. (a, b) Values above or below 1 reflect higher or lower mortality rates than expected or (c) higher or lower abundance of Chinook salmon than the average for the time series. SOURCE: Ford et al. (2010).

preferred prey, Chinook salmon (see Figure 3.1), although there is some uncertainty about how this interaction affects population growth (Vélez-Espino et al., 2015).

However, despite this clear connection, the systems involved are complex, and unraveling the nature and extent of the competition between marine mammals and fisheries has been challenging (Matthiopoulos et al., 2008). Fisheries may also result in a variety of indirect effects by changing the ecosystem and decreasing or increasing the abundance of potential marine mammal prey such as forage fish. Analysis challenges stem from complexities in ecosystems, such as spatial heterogeneity and multispecies interactions, which constrain the ability to clearly interpret cause and effect (Harwood, 1992; Matthiopoulos et al., 2008). Other difficulties for quantifying competition emerge from the fact that many marine mammals are generalist predators. The prey consumption of generalist predators varies with the availability of all their preferred prey species (Asseburg et al., 2006; Smout et al., 2014). As a result, more data than are usually available in field studies of marine mammals are required to realistically characterize these interactions. Thus, despite the intuitive connection between fisheries and marine mammals, there is currently no existing demonstration that resource depletion from fisheries has demographic consequences for marine mammals. Other influences of fisheries on marine mammals, such as bycatch, have been well documented.

In addition to food resources, critical marine mammal habitat can be limited by human activities. Critical habitats are areas essential to an animal's survival, such as the islands and protected beaches that grey seals (*Halichoerus grypus*) need for successful breeding (Harwood, 2001). Human disturbance may reduce the ability of seals, such as Hawaiian monk seals (*Monachus schauinslandi*), to use critical breeding beaches (Gerrodette and Gilmartin, 1990). These habitats, and others like the seagrass beds that manatees (*Trichechus manatus*) require for foraging, may also become limited by environmental drivers such as sea level rise (Burns, 1997). While some marine mammals can move to find other habitats, others such as freshwater river dolphins cannot (Harwood, 2001). Ice-associated species that rely on sea ice for pupping, molting, and transportation may be particularly vulnerable to population consequences of reduction of sea ice resulting from climate change (Kovacs and Lydersen, 2008; Kovacs et al., 2011). For example, ringed seals (*Phoca hispida*) show a decrease in body condition, ovulation rates, and recruitment that is correlated with low ice years (Harwood et al., 2000; Ferguson et al., 2005). Likewise, in polar bears (*Ursus maritimus*), decreased ice cover leads to longer periods of fasting, lower reproductive rates, declining body condition and survival, and increased contact with human settlements (Stirling et al., 1999, 2004; Stirling and Parkinson, 2006). At present, few examples exist that demonstrate direct impacts of habitat limitation on marine mammal populations, but as critical habitats become more limited by ecological drivers, this type of stress may become more apparent.

As an adaptive response to reducing intraspecific competition when prey is limited, dietary specialization may occur among individuals (Tinker et al., 2008). This can result in different exposure risks to pathogens within the population. For example, sea otter feeding on abalone, a preferred prey species, had a low risk of infection by *Toxoplasma gondii* and *Sarcocystis neurona* compared to otters feeding on small marine snails, despite foraging in the same habitat (Johnson et al., 2009). Food resource limitation can therefore lead to changes in pathogen exposure and have potential adverse effects on health as a consequence of the interaction between disease and increasing prey limitation.

Perceived Threat

Frid and Dill (2002) made an important contribution to studies of disturbance in wildlife when they pointed out that anthropogenic disturbance stimuli may evoke responses similar to those evoked by predators or other threats, with which a species may have a long evolutionary history. Some species with strong flight responses to threat may be at risk of acute lethal effects of disturbances. Cox et al. (2006) reviewed data on atypical mass strandings of beaked whales that coincided with sonar exercises and concluded that the most likely cause of these strandings involved sonar triggering a behavioral reaction that ultimately led to stranding. If sonar triggers a strong enough avoidance response to send beaked whales from their deep water habitat to water shallow enough to pose a risk of stranding, this suggests that the whales perceive the sonar as a potential threat. As mentioned in Chapter 2, mid-frequency sonar signals share some similarities with calls of killer whales, an important predator, and beaked whale responses to sonar share some similarities to responses to playback of killer whale sounds. These observations are consistent with the hypothesis that beaked whales perceive sonar as a threat, similar to the risk of predation.

Other forms of disturbance that evoke less drastic acute responses may have aggregate effects in wildlife populations. Wildlife tourism, which focuses on experiencing or interacting with wild animals, is a rapidly expanding industry (Newsome et al., 2002; Burgin and Hardiman, 2015). Although effects on marine mammal behavior have been documented, their impact at the population level is not well known (New et al., 2015). It appears that it is not only the sound produced by a whale-watching vessel that elicits a response, but the physical presence of a boat also plays a role in disturbance and the perceived threat risk. Pirotta et al. (2015a) found that the probability that bottlenose dolphins would engage in foraging activity declined by almost half in the presence of boats, but there was no relationship with the sound level. Various other short-term responses of marine mammals to boat traffic and swimmers have been reported. Well-documented examples include avoidance behavior by bottlenose dolphins (*Tursiops truncatus*) of swimmers (Constantine, 2001), and a reduction in resting and surface activity combined with faster swimming among southern right whales (*Eubalaena australis*), also in response to swimmers (Lundquist et al., 2013). Bejder et al. (2006) documented a significant reduction in the abundance of bottlenose dolphins in Shark Bay, Australia, when there were two or more wildlife tour operators compared to control sites with no tourism or when there was only one tour operator. Their findings indicated that the decline was due to a displacement of individuals, potentially those more sensitive, and a long-term shift in habitat use from disturbed sites with high vessel traffic to areas with lower activity. A study of bottlenose dolphins in Fiordland, New Zealand, also found that dolphins avoided areas where there was high tourism traffic (Lusseau et al., 2006; Lusseau and Bejder, 2007). A threshold of 68 minutes between boat interactions was identified below which dolphins switched from a short-term behavioral avoidance strategy to long-term habitat displacement. If this threshold was regularly exceeded, the population was predicted to decline as a result of a reduction in reproductive success, an increase in stillbirths, and decline in calf survival (Lusseau et al., 2006; Lusseau and Bejder, 2007). However, a recent study (Brough et al., 2016) has suggested that some of the decline in reproductive success in this population may be the result of an increase in the discharge of freshwater into the system after 2002. The Lussau and Bejder (2007) results contrast with dolphins in Sarasota Bay, Florida, where the dolphins remain even though a boat passes within 100 m every 6 minutes (Nowacek et al., 2001). One difference between these examples is that most boats in Sarasota Bay may be passing with no activity directed toward the dolphins in contrast with the tourist boat activities in Fiordland.

These studies indicate that population-level effects may be more likely to occur when individuals have small home ranges and high fidelity to sites with a high level of whale watching. In these circumstances a large number of individuals may experience repeated and long-term disturbance. In cases where individual exposure is relatively short, such as for migratory baleen whales, the effects are expected to be less. For example, Christiansen and Lusseau (2015) found that interactions between minke whales and whale-watching boats off Iceland resulted in a 42% decrease in feeding activity and an estimated 64% decrease in net energy intake. However, the aggregate exposure of individuals to whale-watching boats over the course of a summer was low (less than 450 minutes), leading to only a small decrease in female body condition that was unlikely to affect reproductive success (Christiansen and Lusseau, 2015). An examination of calving rates of humpback whales and calf survival off New England also found no evidence for negative effects of exposure to whale watching (Weinrich and Corbelli, 2009). Frameworks using individual-based models are being developed to simulate the potential effects of boat traffic and other human activities on marine mammal populations (New et al., 2013a; Pirotta et al., 2015b).

Ocean Climate and Conditions

Oceanographic and meteorological phenomena can profoundly alter characteristics of the marine environment, which, in turn, affect the distribution and resource acquisition of marine mammals. One of the strongest is the atmospheric forcing of the El Niño–Southern Oscillation (ENSO), which results in major changes in the physical structure and productivity of the North Pacific subtropical gyre (Karl et al., 1995). These changes directly impact low-latitude and coastal upwelling zones that are important habitat for marine mammals and have time-lagged effects at higher latitudes (Brinton et al., 1987). El Niño alters water temperature and structure on large spatial scales and reduces coastal upwelling. These features are important in determining habitat use and movement patterns of marine mammals (Croll et al., 2005; Doniol-Valcroze et al., 2007), altering the range and abundance of some species and concentrating individuals in areas with high productivity (Gardner and Chávez-Rosales, 2000; Benson et al., 2002). These changes in distribution may also influence exposure to other stressors that have geospatial components. Prey limitation associated with El Niño may have severe impacts on coastal and pelagic foraging species, reducing survivorship and reproductive rates and impacting local population dynamics of cetaceans and pinnipeds (Trillmich et al., 1991; Crocker et al., 2006; Leaper et al., 2006).

Multidecadal changes in ocean climate, or regime shifts, also influence sea surface temperature, upwelling, and biological productivity (Croxall et al., 1992; Francis and Hare, 1994). These alterations that persist over longer time scales can amplify effects of ENSO variation. The Pacific Decadal Oscillation (PDO) may influence the periodicity of El Niño events, resulting in stronger cumulative impacts on individuals and populations. Warm water regimes of the PDO are associated with increased nutritional stress in Pacific marine mammals (Le Boeuf and Crocker, 2005). Similarly, a multidecadal oscillation in the climate of the North Atlantic, the North Atlantic Oscillation (NAO), influences the distribution and foraging of numerous marine mammal species and impacts reproductive rates and population dynamics (Fujiwara and Caswell, 2001; Greene and Pershing, 2004; Jiang et al., 2007). Ocean climate is thus a major driver of distribution, abundance, and reproduction of marine mammals with enormous potential to influence the way that individuals and populations respond to extrinsic stressors. However, clear linkages between ocean climate and marine mammal population trends have not been well documented. A study on southern elephant seals spanning five decades also highlighted the importance of considering density effects in combination with environmental conditions to evaluate effects on populations because these factors can interact (de Little et al., 2007).

Besides ocean climate shifts due to ENSO, PDO, or NAO, changes in global and ocean climate that result from anthropogenic climate alteration are likely to have profound impacts on marine mammals (Moore and Huntington, 2008) that will potentially interact with other stressors. Some marine mammals associated with polar ice are already showing shifts in distribution, reduced body condition, and declines in abundance and reproduction in response to declines in sea ice (Kovacs et al., 2011). However, the quality of abundance estimates varies greatly among location and species and in most cases the data currently are not sufficient for analyzing population trends (Laidre et al., 2015). For bowhead whales, the warming Arctic regions have proved beneficial. Their axial-girth-based body condition index (BCI_G) is positively correlated with summer sea ice loss over the past 2.5 decades, and BCI_G is significantly correlated with the duration of the melt season (George et al., 2015). Range expansions of temperate species may alter resource competition in high-latitude habitats. Long-term impacts may include alteration in oceanographic features used in foraging strategies. Changes in prey distribution and abundance may also occur as a result of disruption of ocean currents and increases in the energetic cost of calcification caused by ocean acidification (Doney et al., 2012). Ocean warming has been implicated in reports of rising disease prevalence in marine organisms, including marine mammals (Harvell et al., 2002; Lafferty et al., 2004; Burek et al., 2008; Van Bressem et al., 2009). Emerging evidence from climate change studies (Ockendon et al., 2014) suggests that indirect effects of stressors, through the disruption of interspecific interactions, may be more important than direct ones**.** Apparently caused largely by increased nutritification, dead zones (hypoxic areas) have increased in recent years in many coastal areas, such as the northern Gulf of Mexico (Rabalais et al., 2002; Diaz and Rosenburg, 2008). Although the influences of dead zones on marine mammals have not been well documented, reduced production and prey availability (Grimes, 2001) almost surely are detrimental to these animals.

SPATIAL AND TEMPORAL VARIATION AMONG STRESSORS

The range of extrinsic stressors to which marine mammals can potentially be exposed over a lifetime has been briefly reviewed, but to appreciate the potential for cumulative effects of these combined stressors, the spatial and temporal patterns of exposure should also be considered. The occurrence of individual stressors may show strong spatial variation, and their effects depend on the habitat used by a given marine mammal species. Even ubiquitous stressors, like anthropogenic noise and globally dispersed chemical contaminants, show variation in magnitude across geographic regions. Species that exhibit long-distance movements may be exposed to diverse stressors in disparate ecosystems, and consideration of cumulative effects must include stressors throughout this range. Although highly migratory species may be exposed to a wide range of stressors, the aggregate exposure of individuals may be low (e.g., Christiansen and Lusseau, 2015), affecting the overall impact at a population level. In contrast, species with smaller home ranges may potentially be exposed to fewer stressors, but with greater exposure times to those that occur in the region.

There is also a potential temporal component to variation in vulnerability to stressors related to life-history variation within species. For example, the need of capital breeding species to conserve energy may outweigh short-term costs of local stressors during breeding (Bishop et al., 2015). However, once breeding is completed they may be at an exceptionally low nutritional plane with high allostatic load that reduces their ability to respond to new stressors. Females with calves or pups may also be more sensitive to disturbance and perceived threats (Engelhard et al., 2002; Stamation et al., 2009). During key foraging periods, animals may be less vigilant in responding to threats, which may increase their vulnerability to other stressors such as predators. Some behavioral states also increase vulnerability to stressors. For example, during feeding North Atlantic right whales spend much of their time just below the surface, increasing the risk of vessel collisions (Parks et al., 2012). Stressors that affect prey availability and predation risk on the feeding ground may directly impact animals' body condition, pregnancy rate, and survival (Williams et al., 2013). Because these life-history periods are often associated with specific habitats or spatial use, managers should consider this dimension when assessing the potential impacts of the spatial component of exposure to stressors. From this perspective, chronic stressors that impact individuals across multiple life-history stages are more likely to have deleterious effects than those that impact only one life-history stage. Species or populations that are continually exposed to stressors in a particular location with a given geospatial distribution are also more likely to suffer deleterious effects than species that migrate through that location and are only periodically exposed.

The physiological and behavioral impacts of single and multiple stressors will also vary depending on the frequency of exposure. Ongoing or continuously occurring (i.e., chronic) exposure can be associated with dysregulation of endocrine and homeostatic function and therefore have negative impacts on individual fitness. Chronic activation of generalized stress responses may be an important mechanism through which cumulative impacts arise. Conversely, when exposure to a stressor is acute, occurring for a single discrete period, or intermittent, occurring repeatedly but not necessarily at frequent or regular intervals (e.g., HABs or sonar), animals may accommodate. That is, a physiological response may be invoked but normal function is then restored or a new homeostatic set point is reached. In some cases, the resulting physiological responses may be adaptive and even enhance the ability to respond to future stressors through hormesis[1] (Calebrese et al., 2007). However, even if the exposure is not chronic, an alternative mechanism for cumulative impacts emerges when the adverse effect produced by the stressor persists or is irreversible (i.e., a chronic effect). For example, a permanent threshold shift in auditory sensitivity will impact behavior.

SUMMARY AND CONCLUSIONS

Numerous studies have evaluated the impact of various extrinsic stressors on the individual health, survival, or reproduction of marine mammal species. Stressors such as fishery interaction, vessel strike, HAB toxins, and pathogens can cause acute mortality. Even when there are effects that are nonfatal, they can induce sublethal effects that continue to affect the animal's ability to maintain homeostasis and respond appropriately to other extrinsic or intrinsic stressors. The broad array of chemicals to which many marine mammals are exposed, often chronically over their lifetime, also produce sublethal physiological effects. Such effects have been documented from observational studies of marine mammals and in many cases are supported by findings from experimental studies in other mammalian species. However, linking chemical stressors to decreases in vital rates through observational assessments is inherently challenging due to the chronic nature of many exposures or effects, the complexity involved in controlling for confounding or interacting variables, and the difficulty of observing mortality or reproductive end points in long-lived marine mammal species, particularly cetaceans. These challenges extend to other stressors that induce sublethal effects. Regardless of the stressor, few studies have explicitly defined *quantitative* relationships between varying doses and associated mortality, reproductive, or physiological effects for marine mammals.

Finding 3.1: Numerous studies have demonstrated direct physiological effects from a broad array of extrinsic stressors in marine mammals. However, few studies have explicitly quantified the relationship between varying doses of a given stressor and the level of mortality, reproductive, or physiological effect (i.e., defined a dose–response relationship).

Ecological drivers such as ocean climate shifts act directly or indirectly through prey or other resources to induce stress on marine mammal populations. Similarly, fisheries can directly create competition for resources, or indirectly affect prey availability through ecosystem changes. Wildlife tourism or other forms of disturbance that may be perceived as a threat evoke more acute responses but may have aggregate effects. For these stressors, analysis challenges stem from complexities in ecosystems and/or difficulties in elucidating long-term shifts in behavior or habitat use, constraining the ability to clearly interpret cause and effect at the population level.

The occurrence of some stressors may show strong spatial variation. In addition, an animal's vulnerability to stressors may vary temporally in relation to life history. Therefore, temporal and spatial variation in exposure to stressors must be considered. Ongoing or continuously occurring (i.e., chronic) exposure to a stressor can be associated with dysregulation of endocrine and homeostatic function and therefore may be an important mechanism through which a cumulative effect manifests within individuals. Even if the exposure is not chronic, an alternative mechanism for a cumulative impact emerges when the adverse effect produced by the stressor persists or is irreversible (i.e., a chronic effect).

Finding 3.2: The effects of stressors on marine mammals depend on temporal and spatial overlap in the distribution of stressors and the target organisms. Chronic exposure or a chronic effect resulting from an acute exposure provides mechanisms through which cumulative impacts may arise.

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[1] A phenomenon of dose–response relationships wherein a stressor that produces harmful biological effects at moderate to high doses may produce beneficial effects at low doses.

4

Assessing Interactions Among Stressors

INTRODUCTION

As described in Chapters 2 and 3, marine mammals are exposed to a diverse set of extrinsic stressors during their lifespan. Understanding the way exposure to any one stressor may affect marine mammal populations is challenging; understanding the population-level consequences of exposure to multiple stressors is far more challenging. However, a key to understanding how the effects of extrinsic stressors might integrate to create cumulative effects is determining how specific stressors create responses, and evaluating the potential for interactions between the effects of these responses over the lifespan of an individual. It is important to be clear what is meant by an interaction between stressors. Gennings et al. (2005) reviewed the models that have been used to quantify toxicological interactions and defined an interaction between two chemicals as occurring when the shape of the dose–response relationship for one chemical is affected by the dose of the other chemical. The committee adopted the same definition for interactions between stressors. If the shape of the dose–response relationship of one stressor does not change in the presence of another stressor, then these stressors do not interact, and the responses are said to combine additively.

The impact of multiple extrinsic stressors can be studied at different levels of biological organization from molecular, cellular, or organ responses, to effects on the individual, to higher-order population- and community-level responses (see Figure 4.1). Accommodation, or recovery that restores normal function, may occur at any level of organization (e.g., Nichols et al., 2011). However, when the exposure to a stressor is sufficient, the response at one level will be propagated to the next level. For example, at the molecular level, changes in gene expression, enzymatic reactions, and receptor function may occur in response to a stressor; these in turn may initiate cellular responses such as differentiation, proliferation, or altered hormone synthesis. When sufficient, these cellular responses can produce an injury to an organ or disruption of an endocrine axis that eventually leads to morbidity, mortality, or reproductive failure for the individual. If sufficient individual-level responses occur, there can be impacts on populations and, ultimately, communities and ecosystems. It is at these higher levels of biological organization that responses are of greatest societal relevance and greatest concern for natural resource, coastal, and ocean management.

Although the flow of responses in Figure 4.1 is depicted as moving upward through increasingly higher levels of biological organization, responses may also be introduced at a higher level (e.g., ecosystem or community) and then initiate a cascade of responses within an individual marine mammal. The El Niño–Southern Oscillation would be an example of an ecological driver initiated at the ecosystem level, which can cause prey depletion, prompting a response at the molecular level, which then propagates upward to an individual-level response.

Unfortunately in many cases, responses at the higher levels cannot be detected until the process is so far along that the change may be catastrophic and irreversible. It is therefore important to study effects of stressors at the lower levels of biological organization. However, it is imperative to supplement the information on lower-level responses with an understanding of the linkages and processes by which such responses eventually translate into higher-level impacts. The linkages and associations of responses across different levels of biological organization are considered by ecotoxicologists when describing adverse outcome pathways (AOPs) (Ankley et al., 2010; Connon et al., 2012) and by conservation physiologists when describing biological upscaling (Cooke et al., 2014). Depending on the context, an AOP may be considered to extend from molecular-level responses all the way through to population-, community-, or even ecosystem-level responses. Similarly, the Population Consequences of Disturbance (PCoD) model structure (New et al., 2014), which is used in Chapter 5 as the basis for a model of the population consequences of exposure to multiple stressors, describes a series of compartments and transfer functions that upscale from physiological or behavioral changes to anticipated impacts on population vital rates. The series of transfer functions between compartments from the initial physiological change to the ultimate effect on individual vital rate or population dynamics in the PCoD model is essentially equivalent to an AOP. However, for this report, the committee defines an AOP to span the molecular- to individual-level responses shown in Figure 4.1.

In practice, it is extremely difficult to detect interactions between two stressors by determining the dose–response relationship for one stressor at different dosages of the second stressor. Instead, most research has focused on detecting deviations from additivity, usually by assessing the significance of the interaction term in an analysis of variance (ANOVA) or other linear model analysis of results from a controlled factorial experiment (Folt et al., 1999), or the deviations from a null model of additive effects (e.g., Darling and Côté, 2008). However, as Greenland (2007) notes, "concepts of biologic interaction do not in general correspond to the concept of statistical interaction, because the latter is only the need for a product term in a statistical model."

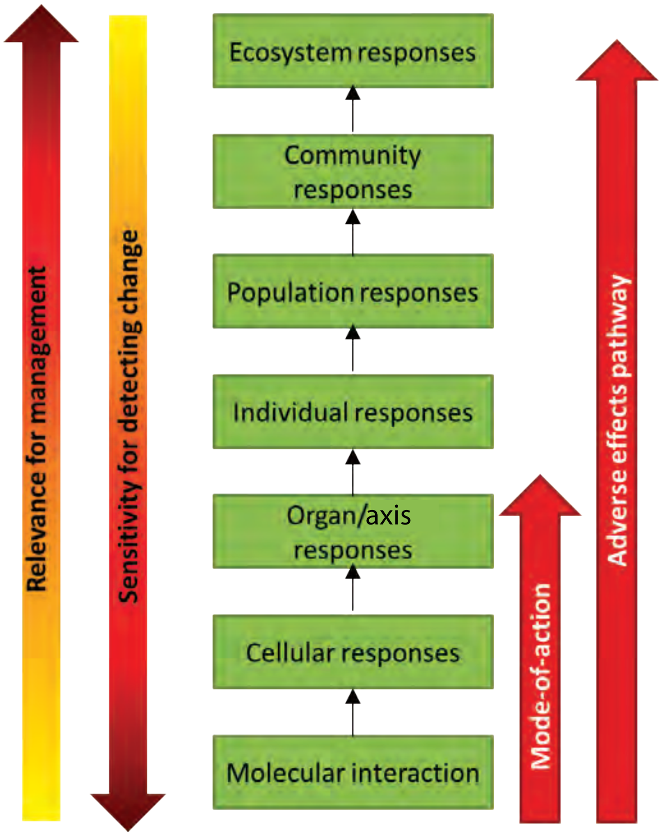

FIGURE 4.1 The hierarchy of responses to a stressor across multiple levels of biological organization.

In the next section, the results of recent meta-analyses of studies of the interactions between stressor effects that have used this statistical approach are reviewed in order to assess the prevalence and nature of interactions between extrinsic stressors in marine and freshwater systems. However, as noted above, these meta-analyses only provide information on whether statistical interactions have been detected: they do not provide quantitative models of the way the stressors actually interact. In subsequent sections the committee describes how interaction effects may be quantified by considering common pathways for adverse health outcomes along which different stressors act, provides some examples of the way in which the extrinsic stressors to which marine mammals are exposed may interact, and explains how stressors might be prioritized for cumulative effects analysis. Finally, that approach is used to look at the potential causes of some unexplained declines in marine mammal populations.

STUDIES OF MULTIPLE STRESSORS: A BRIEF REVIEW

As noted in the previous section, most studies of interactions among multiple stressors test whether the effect of the stressors together is significantly different from the combined effect of each stressor acting independently. The magnitude of effect expected depends on the mathematical operation used to combine the independent effects. For example, stressor effects may be combined additively or multiplicatively depending on the nature of the response being tested. Because a multiplicative combination of stressor effects is additive on the logarithmic scale, both methods of combination are usually referred to as "additive." The test statistics that are most commonly used are Hedges' *d*, which, according to Crain et al. (2008), is "constructed similar to ANOVA where a significant interaction effect signifies deviation from the null model of additivity," and the sum of the natural logarithms of the response ratios (lnRR) for each stressor. For the latter metric, an interaction is identified if the difference between the lnRR when both stressors are present and the sum of the lnRR values for the individual stressors is significantly greater than zero. If the combined effect of two or more stressors is greater than the combination of their individual effects, this is referred to as a synergistic interaction. If it is less than the combination of the individual effects it is referred to as an antagonistic interaction. If there is no significant difference, the cumulative effect is referred to as additive.

The complications that can arise with these simple null models are elegantly summarized by Côté et al. (2016). For example, synergistic interactions are impossible to detect with these methods if the sum of the individual effects is greater than 100% (Folt et al., 1999). These issues can be overcome by using the "multiplicative risk model" as described by Sih et al. (1998). The predicted combined effect using the multiplicative risk model is less than the predicted effect from a simple additive model, and its use as the null model is therefore more likely to result in the detection of synergistic interactions. Further complications occur if the effect of one stressor is so large that it results in the death of most experimental animals before any other stressor can have an effect. This is referred to as "dominance" by Côté et al. (2016). It would be incorrectly identified as an antagonistic interaction using a simple additive model. Additional problems arise if the stressors under consideration have opposite effects. In these cases, the threshold for a synergistic or antagonistic effect is actually smaller than the effect of either of the stressors. Such effects have been referred to as "reversals" (Jackson et al., 2016). Finally, in some cases the combined effect of the two stressors is in the opposite direction to the effects of either of the individual stressors, a phenomenon called "mitigating synergism" by Piggott et al. (2015).

Crain et al. (2008) reviewed 171 studies that used factorial experimental designs to investigate the effects of two or more of 13 stressors on marine and coastal environments. About 90% of the experiments were done in the laboratory and three-quarters of the studies subjected single species rather than entire communities or ecosystems to the stressors. They detected synergistic interactions using Hedges' *d* in 36% of the studies and antagonistic interactions in 38%. When a third stressor was added, the proportion of synergistic pairwise interactions increased from 33% to 66%. Piggott et al. (2015) reanalyzed the same data set as that used by Crain et al. (2008) to take account of comparisons in which the stressors had opposite effects and the potential for mitigating synergisms. They found fewer examples of synergistic interactions (31% versus 36%) and more examples of antagonistic interactions (43% versus 38%).

Harvey et al. (2013) analyzed 623 observations from controlled factorial studies of the cumulative effects of temperature and acidification on calcification, photosynthesis, reproduction, survival, and growth in marine organisms using lnRR as the test statistic. Their analysis found evidence for synergistic interactions between the two stressors for four of the response variables. This was the result of a greater than expected increase in photosynthesis, and a greater than expected reduction in calcification, reproduction, and survival.

Ban et al. (2014) used a parametric bootstrap approach for calculating the standard error of the interaction term in an ANOVA of the results from studies of the effects of multiple stressors on coral reefs. Their aim was to increase the statistical power of more conventional analyses, which can result in failure to detect an interaction when one is, in fact, present. They analyzed the results of 26 fully factorial studies that investigated the cumulative effect of irradiance and temperature on photosynthesis in corals and found that the mean effect size of the combined treatments was statistically indistinguishable from a purely additive model.

Jackson et al. (2016) analyzed values of Hedges' *d* extracted from 286 observations of the responses of freshwater ecosystems to paired stressors in controlled factorial experiments. They found that multiple stressors exerted significant antagonistic effects on animal abundance/biomass, animal condition, animal growth/size, and animal survival.

Przeslawski et al. (2015) analyzed values of Hedges' *d* extracted from the results of 104 factorial experiments that examined the cumulative effects of temperature, salinity, and pH on growth and/or survival of the embryos or larvae of marine organisms using a generalized linear mixed-effects model. They found evidence for synergistic interactions between temperature and pH in 76% of the experiments, and for synergistic interactions between temperature and salinity in 58%.

This review of meta-analyses establishes that the cumulative effects of multiple stressors may be additive, antagonistic, or synergistic in almost every setting tested. The proportion of cases providing evidence for antagonism and synergism varied substantially among studies. As a result, the prevalence of interactions between stressors in nature remains uncertain, especially because the relatively low statistical power of most of the studies (Ban et al., 2014) will have resulted in some interactions going undetected. Nonetheless, the basic conclusion that one can take from all of these studies is that there are few situations where one can confidently assume that the effects of multiple stressors are additive. Although Côté et al. (2016) have pointed out that synergies are not the most prevalent form of interaction reported in the literature, and caution about the risks of managing antagonistic interactions as if they were synergistic, they also found that "physiological response variables have so far not yielded evidence of antagonisms." Because physiological responses are a fundamental component of most of the observed reactions of marine mammals to extrinsic stressors, this suggests that assuming the effects of individual stressors are additive may frequently lead to an underestimation of their cumulative impact.

Finding 4.1: There are few situations where one can assume that the effects of multiple stressors are simply additive, and this assumption may lead to an underestimation or overestimation of their cumulative impact.

Most of the studies of cumulative effects of multiple stressors that contributed to these reviews have used factorial designs. This leads to elegant experiments with simple analyses in situations where the conditions can be replicated and controlled. However, if the factorial design does not actually provide a dose–response relationship for each stressor–effect pair, or for any relevant combinations of stressors, then it is of little use to management. The critical questions for managers who aim to prevent threats are "What stressor effects threaten populations or ecosystems, and what combinations of dosages of stressors elevate the effect enough to pose a risk?" Given that many anthropogenic stressors have negative effects on marine mammals, simply evaluating whether their cumulative effects may be antagonistic, additive, or synergistic does not provide the information needed to decide whether specific dosages of one or more stressors are likely to cause an effect that poses a risk to species of concern. The critical point for managers in the planning phase is to define population-level effects that need to be avoided, and then to evaluate whether the cumulative impact of a planned activity, of other activities, and of the relevant array of natural stressors poses a risk of causing the deleterious effects. After it is discovered that a population or ecosystem is in danger, then the critical issue is to evaluate what changes in stressors will provide the best reduction in risk at the least disruption of other critical human priorities. Both of these problems require assessment of dose–response relationships across the relevant range of dosages and effects. Ideally this assessment should be conducted under realistic field conditions, coupled with quantitative assessments of the interaction between all stressors that may cause the effect of concern.

Finding 4.2: The critical question for managing risk of cumulative effects is "What combinations of dosages of stressors are likely to elevate the effect enough to pose a risk to populations or ecosystems?" Once a population is found to be at risk, then the critical issue is to determine which combination of stressors could be reduced in order to bring the population or ecosystem into a more favorable state.

CUMULATIVE IMPACT SCORES

Halpern et al. (2008) used expert-derived vulnerability weights from Halpern et al. (2007) and a cumulative impact model to identify what they believed to be the greatest threats among 38 different stressors and ecological drivers at large or small spatial scales of marine ecosystems, and to identify the most threatened ecosystems. They used this method to create a global map of human impacts on marine ecosystems, and they argue further that this map can be used to allocate conservation resources for ecosystem-based management. Maxwell et al. (2013) adapted the methods of Halpern et al. (2007, 2008) and used them to estimate cumulative impacts for marine mammals and other marine predators. Here a critical review of this approach is provided.

Halpern et al. (2008) calculated cumulative impact scores I_C for each 1 km^2 of ocean using the following equation:

$$I_C = \sum_{i=1}^{n} \sum_{j=1}^{m} D_i \times E_j \times \boldsymbol{\mu}_{ij},$$

where D_i is the log-transformed and normalized value of the intensity of the driver at location i, E_j is the presence or absence of ecosystem j, and $\boldsymbol{\mu}_{i,j}$ is an impact weighting for each driver–ecosystem pair. Drivers were allowed to have different weights for different ecosystems, but this calculation of cumulative impact assumes the effects of the drivers are additive, with no interaction between them. Maxwell et al. (2013) estimated the cumulative impact of multiple stressors (CUI) using a similar equation:

$$CUI = \sum_{i=1}^{n} \sum_{j=1}^{m} D_i \times \mathrm{S}_j \times \boldsymbol{\mu}_{i,j}$$

where D_i is the normalized and log-transformed value of intensity of an anthropogenic stressor at location i, S_j is the probability distribution of species j being present in a given cell, and $\boldsymbol{\mu}_{i,j}$ is the impact weight, which reflects the potential effect of anthropogenic stressor i on species j. The impact weight for each stressor–species combination is calculated from expert rankings of the importance of a number of different vulnerability measures for that combination.

The determination of impact weights is a critical aspect of this approach. Halpern et al. (2007) used two numerical measures (area and recovery time) of vulnerability, and three ordinal variables (frequency, extent of ecosystem impacted, and resistance of the ecosystem to the threat). Maxwell et al. (2013) used six measures (frequency of impact, whether the impact was direct or indirect, likelihood of mortality, individual recovery time, reproductive impact, and spread of the impact across the population). These rankings are then combined into a single vulnerability score.

This kind of arbitrary tallying of ordinal scores is not uncommon in situations where, for example, a health practitioner wants a simple repeatable way to assess the cumulative risk of a series of factors for a specific adverse outcome. However, the committee thinks that the arbitrary tallying of this kind of scale requires validation. When Halpern et al. (2007) asked the experts to identify the three top threats in the ecosystems, only half of the results of the vulnerability ranking matched the judgment of the experts, indicating either that there was low confidence in the resulting rankings or that the experts suffered from perception bias.

The cumulative impact scores used by Halpern et al. (2008) and Maxwell et al. (2013) assume that cumulative effects are additive across threats within an ecosystem. As discussed above, all the reviews of the effects of multiple stressors found evidence for synergistic and antagonistic interactions, which suggests that this simple additive approach may overestimate some impacts and is likely to underestimate others. The committee recognizes the enormous amount of work that has gone into developing this approach and compiling the databases needed for its application. Determining the spatial overlap between human activities and species or ecosystems is an important first step in identifying locations where interactions between stressors are likely to occur. However, the committee believes that a better quantitative understanding of potential exposure levels, dose–response functions, and linkages to vital rates is required to provide an adequate assessment of cumulative effects in these locations.

PREDICTING HOW MULTIPLE STRESSORS ARE LIKELY TO INTERACT

A consideration of cumulative effects has been often discussed with respect to marine mammals (Wright and Kyhn, 2015), and such effects must be considered in Environmental Assessments and Environmental Impact Studies (40 C.F.R. § 1508.7). However, in spite of the large number of factorial experiments in other taxa, no experiments have examined the cumulative effects of multiple stressors on marine mammals. Quantification of the interactions between these stressors is hindered by a limited understanding of the physiological and behavioral effects of cumulative exposure, and the logistical difficulties of measuring the impacts of this exposure on free-ranging individuals over their lifespans.

Any stressor that induces effects up to at least an individual level (e.g., mortality or reproductive impairment), whether exposure is acute, intermittent, or chronic, has the potential to contribute to a cumulative population-level impact. For example, direct lethal effects may occur as a result of acute exposure to ship strike, intermittent exposure to infectious disease outbreaks or harmful algal blooms, or to the risk of bycatch in fishing gear that is left in the water for long periods (e.g., gillnets). In most cases, the acute effects of each stressor on survival can be evaluated independently and their cumulative effect calculated using a multiplicative risk model that accounts for the fact that an individual can only be killed once.

However, it is more difficult to predict the interactions that may occur among stressors that have a chronic effect on survival and reproduction, and that therefore have the potential to generate unexpected, nonadditive effects for populations and communities. These occur when a stressor affects an individual's homeostatic systems so that it can no longer respond appropriately to its environment, and its vulnerability to other stressors is increased. Interactions may also occur at the population level if the stressor effects result in demographic changes, for example, if mortality is preferentially focused on adult females. They may also occur at a higher level of biological organization (community or ecosystem level) if a tipping point (see Chapter 6) is reached because an ecological driver has, for example, caused a collapse in the prey base. In the rest of this section, approaches that can be used to improve understanding of potential interactions between stressor effects at the individual level are explored. The potential for interactive effects at higher levels is discussed in Chapter 6.

Insight for predicting cumulative effects at the individual level can be gained from the environmental health and ecological risk assessment communities, where scientists are grappling with the complicated issue of cumulative risk assessment for chemical mixtures. There are more than a hundred million chemical substances known to date,[1] and a recent report from the Centers for Disease Control and Prevention provides data for 265 environmental chemicals that are a potential concern for human exposure.[2] People, other terrestrial organisms, and marine organisms are all exposed to this plethora of potentially toxic substances to varying degrees and are most often exposed to mixtures of these chemicals chronically or repeatedly throughout their lives.

A number of different approaches have been proposed for assessing the cumulative risk for multiple chemicals. They often involve identifying a group of chemicals that can be considered collectively (EPA, 2000). One mathematical modeling approach integrates an index for chemicals that co-occur in the environment and have similar structure or mode of action in order to predict a cumulative dose (EPA, 2002; Connon et al., 2012). The index for each chemical can be based on its concentration and toxic potential; therefore, the approach is most applicable for chemicals with a well-characterized mechanism for toxicity, such as the dioxin-like compounds whose toxicity is induced through the aryl hydrocarbon receptor (Van den Berg et al., 2006). Alternative approaches have been suggested that focus on the overall physiological process, rather than mechanisms or modes of action, because there can be a multitude of underlying molecular mechanisms that contribute to a given adverse outcome. This potentially expands the array of chemicals to be considered collectively, because chemicals that have distinct modes of actions may still disrupt the same endocrine pathway or organ system and, ultimately, result in the same disease.

There are clearly limitations to the expansion of these approaches to the multitude of stressors, particularly non-chemical ones, that are of potential concern for marine mammals. However, the paradigm of using co-occurrence, and a common mechanism of action or a common outcome, may be valuable. At the molecular level, it may be possible to predict the effect of stressors that have a similar mode of action using a common dose–response relationship. The cumulative effect of these stressors will only be additive in the unlikely event that the common dose–response function is linear (see Figure 4.2).

One common assumption of ecotoxicologists is that if two or more stressors act through a common mechanism of action, then their doses can be summed to provide a cumulative dose that can then be used with a single dose–response function. Many dose–response functions are sigmoidal in shape or are otherwise nonlinear, and in these cases the sum of two doses may produce a response that is greater or less than the added responses of each stressor alone. A simple

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[1] See http://www.cas.org.

[2] See http://www.cdc.gov/exposurereport.

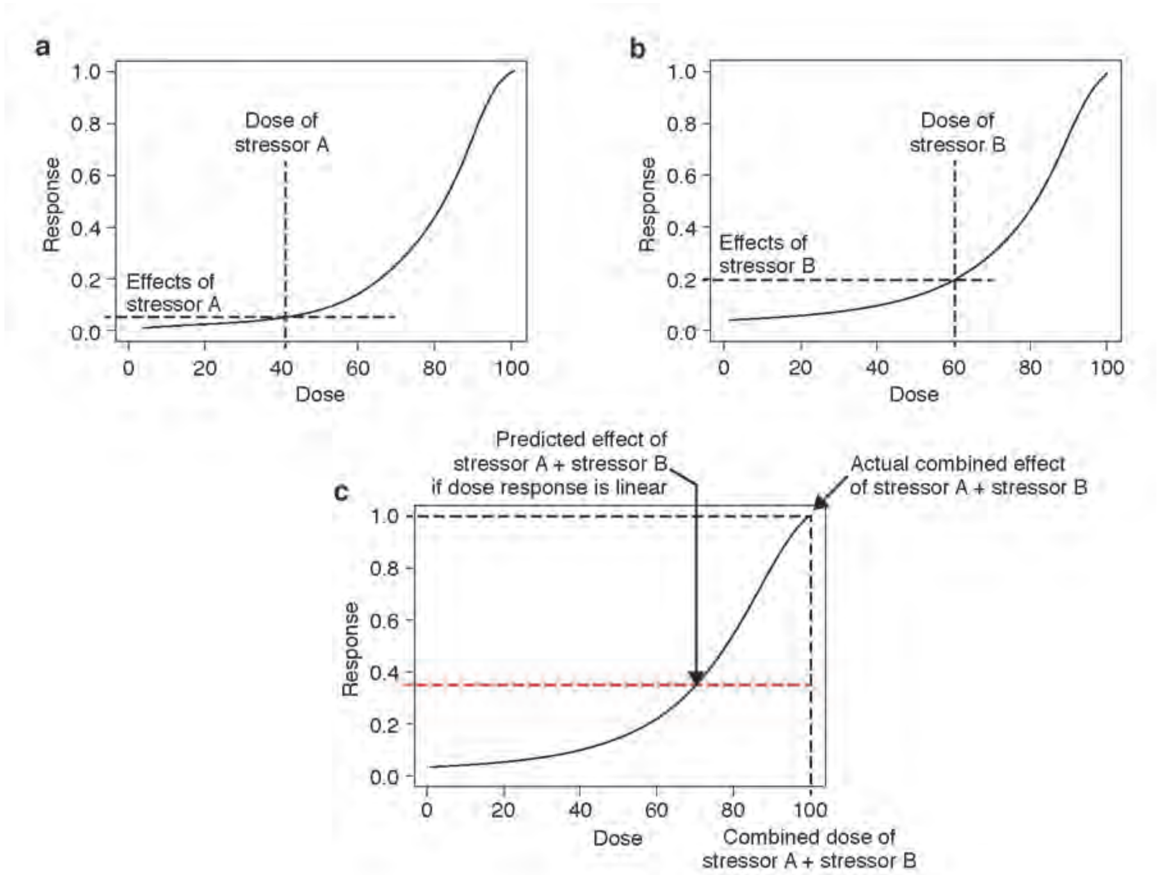

FIGURE 4.2 This figure illustrates how the potential for interaction between two stressors (A and B) that share a common mechanism of action depends on the form of the dose–response relationship. (a) Effect of stressor A alone. (b) Effect of stressor B alone. (c) Effect of a combined dose of stressor A and stressor B, obtained by adding the dose from stressor A to that of stressor B. The effect predicted from the dose–response relationship shared by the two stressors is three times higher than the prediction if their effects are assumed to be additive (red line).

example to illustrate the complexity introduced when a dose–response function is nonlinear is discussed below.

Consider two stressors that act through a common mechanism of action. If one of these stressors is more powerful than the other, then its dosage needs to be adjusted by a metric that corrects for the difference in their relative strengths (e.g., a toxicity factor for chemical stressors). After this correction, the doses of the two stressors can be added to give a combined dosage and compared to a dose–response function (see Figure 4.2). Stressor A has an effect of 0.10 given a dose of 40 units (see Figure 4.2a), and stressor B has an effect of 0.20 given a dose of 60 units (see Figure 4.2b). If responses were additive, then the response to stressors A and B combined is expected to be 0.30. However, due to the sigmoidal shape of the dose–response function, the added doses of the two stressors (100 units) produces an effect of 1.0, more than threefold higher than the sum of the individual responses (see Figure 4.2c). Therefore, although these stressors are considered additive in terms of dosage, they produce a synergistic response. Note that this same phenomenon could also occur with aggregate exposure to a single stressor. Even for this simple situation, a prediction cannot be made of the effects of most stressors because the dose, the relative strengths of the stressors, and the dose–response functions are not known.

Similar interactions may occur at the organ system and individual levels if the stressors act through a common or connected pathway. This may occur if the stressors induce damage or provoke a physiological perturbation within the same organ system or endocrine axis, in which glands signal each other in sequence and/or with feedback loops, such as the hypothalamic-pituitary-adrenal (HPA) axis. In addition, effects via one cellular mechanism or component of an endocrine axis may impact the function of other components through shared signaling pathways. Due to this complexity, the overall physiological process or pathway for an adverse health outcome should be considered. Of primary concern are those pathways that lead to a permanent or at least long-lasting (persistent) adverse health condition, because co-occurrence of the health effects of multiple stressors within an individual is necessary for an interaction to ensue. Alternatively, although the health effect associated with a particular exposure to a stressor could be transient, co-occurrence with other stressor effects is still likely if the exposure to the stressor is chronic.

Finding 4.3: Predicting which combinations of dosages of stressors are likely to elevate cumulative effects enough to pose a risk to populations or ecosystems will be challenging, particularly for stressors that have a chronic effect on survival and reproduction. The paradigm of using co-occurrence and a common pathway for adverse health outcomes, developed by the environmental health and ecological risk assessment communities, could be applicable for addressing this challenge.

Marine mammals are exposed to stressors that have the potential to interact as a result of chronic exposure, or because they may cause permanent or persistent health conditions. The pathways for a persistent health outcome along which each stressor may act are indicated in Table 4.1. Nonbiological toxins are divided into persistent organic pollutants (POPs), inorganic pollutants, and petroleum-associated chemicals and organic solvents, because these most often exert effects through differing pathways. Note that this table is not intended to provide an exhaustive list of all the possible sublethal effects associated with each stressor. Only the principal and previously recognized pathways are indicated, with one or more illustrative references. In addition, only direct pathways are indicated as priorities for consideration. The potential for interaction between pathways should not be disregarded. For example, although the hypothalamic-pituitary-thyroid (HPT) and HPA endocrine pathways are presented separately, effects on one axis may impact the function of the other because of shared molecular substrates, enzymatic reactions, and signaling pathways (Nichols et al., 2011). Ultimately, they may impact other connected pathways, such as the immune or central nervous systems (CNS). There are strong associations in some marine mammals of contaminant burdens with suppression of sex hormones, including testosterone and estradiol. In some cases low levels of sex hormones concomitant with high POP burdens were associated with sterility or reproductive failure (Reijnders, 2003).

POTENTIAL INTERACTIONS AMONG STRESSORS

In this section the committee reviews documented or proposed interactions between stressors, focusing on interactions that occur along the same pathways for persistent health outcomes (see Table 4.1). Most of the interactions we consider are synergistic, not only because ignoring such interactions in an assessment of cumulative impacts increases the risk of underestimating those impacts, but also because Côté et al. (2016) found no evidence for antagonistic interactions involving physiological responses to stressors, such as those mediated by pathways for persistent health outcomes.

Acute Mortality

A number of the stressors listed in Table 4.1 (noise, some organic chemicals and solvents, biotoxins, microparasites, prey limitation, and predation pressure) may have direct, acute effects on survival or reproduction. In some situations where marine mammals are exposed to several of these stressors there may be little opportunity for stressor effects to interact, because individuals are likely to die from the effects of one stressor before they can be affected by any of the others. In these circumstances, as noted earlier in this chapter, treating the effects of each stressor as independent can be justified. However, it should be recognized that historical exposure to other stressors may increase an individual's susceptibility to acute effects from a particular stressor. For example, Hall et al. (2006) showed that previous exposure to polychlorinated biphenyls (PCBs) increased the risk of death from infectious diseases in harbor porpoises. In addition, a multiplicative risk model should be used to account for the fact that individuals are unlikely to die from the effects of more than one acute stressor. Because acute effects are normally evaluated by attributing cause of death to a particular stressor, the simplest approach is to calculate the survival rate of individuals exposed to each stressor. The cumulative effect of all the stressors to which the population is exposed is then calculated by multiplying together the survival rates associated with each stressor.

Although there is little opportunity for interaction among the acute effects of different stressors, chronic effects caused by the same or other stressors can interact with acute effects if they alter individual exposure or susceptibility to the acute stressors. These interactions between acute and chronic stressor effects may be antagonistic. A classic example is the use of active sound emitters ("pingers") to reduce the risks of cetacean bycatch in fishing gear (Dawson et al., 2013). Noise from these emitters displaces marine mammals from the area around the gear to which they are attached, thus reducing their risk of physical injury as a result of entanglement but imposing potential energetic costs.

Hypothalamic-Pituitary-Adrenal Axis

The HPA axis has a central role in coordinating an organism's response to stress, controlling the release of glucocorticoids into circulation and moderating levels through negative feedback (Sapolsky et al., 2000). Glucocorticoid secretion is further modulated by neuronal effects of other brain structures; also gene–environment interactions in response to stressors may have long-term impacts on subsequent secretion (Alexander et al., 2009). Disruption of the HPA axis may therefore interact with the effects of other stressors, particularly if the disruption is the result of chronic

TABLE 4.1 Stressors with Potential for Chronic/Repeated Exposure or Persistent Effects, and Associated Pathways for Adverse Health Outcomes

Stressor	Pathway for Persistent Adverse Health Outcome							
	HPA axis	HPT axis	Nutritional	Immune	Reproductive	Respiratory	Brain/CNS	Auditory (hearing loss)
Noise	**Rolland et al., 2012**		**Isojunno et al., 2016;** Ware et al., 2015	Celi et al., 2015	Halfwerk et al., 2011; Kight et al., 2012			**Finneran, 2016**
Non-biological toxins: potential chronic exposure (POPs, inorganic pollutants) and/or persistent adverse health outcome								
POPs (primarily PCBs)	Possible, but not well described; Harvey, 2016; **Oskam et al., 2004**	Patrick, 2009; Tabuchi et al., 2006; **Schwacke et al., 2012**		Diamanti-Kandarakis et al., 2009; **Ross et al., 1996a, 1996b; Lie et al., 2004, 2005**	**Reijnders, 1986;** Diamanti-Kandarakis et al., 2009		Developmental; Zoeller et al, 2002	Developmental; Crofton et al., 2000; Kenet et al., 2007
Inorganic pollutants				**Kakuschke and Prange, 2007**			Farina et al., 2011	
Petroleum-associated or other organic chemicals or solvents	Mohr et al., 2008, 2010; **Schwacke et al., 2014a**			**Schwartze et al., 2004**	**Lane et al., 2015**	**Schwacke et al., 2014a**		Fuente and McPherson, 2006
Biological toxins: potential recurrent exposure, and potential persistent adverse health outcome								
Biotoxin	Gulland et al., 2012			**Schwacke et al., 2010**	**Goldstein et al., 2009**		**Cook et al., 2015**	
Pathogens: persistent adverse health outcome (microparasites) or potential chronic exposure (macroparasites)								
Micro- or macro-parasites			**Reif et al., 2006**	**Some, e.g., morbillivirus; Van Bressem et al., 2014**	**Some, e.g., Brucella; Meegan et al., 2012**		**Some, e.g., morbillivirus; Van Bressem et al., 2014**	
Other stressors with potential for chronic or repeated exposure								
Prey limitation	**Rosen and Kumagai, 2008; Shero et al., 2015**	Eales, 1988; **Ayres et al., 2012; Gobush et al., 2014**	**Crocker et al., 2006**	**Brock et al., 2013a; Peck et al., 2016**	Meyer-Gutbrod et al., 2015; Ward et al., 2009; Robinson et al., 2012			
Perceived threat	**Spoon and Romano, 2012; Di Poi et al., 2015; Champagne et al., 2012**		**Isojunno et al., 2016**	**Brock et al., 2013b**	**French et al., 2011**			
Predation pressure	Newman et al., 2013; Narayan et al., 2013		Creel et al., 2009		Creel et al., 2007; 2009; Hua et al., 2014; Zanette et al., 2011			
Salinity				**Wilson et al., 1999; Mullin et al., 2015**				

NOTE: Publications highlighted in bold refer to studies involving marine mammals.

exposure to a persistent chemical contaminant, because of the numerous points of regulation and complexity of the involved biochemical pathways. However, an understanding of specific mechanisms for a given set of stressors would be needed to accurately predict the consequences of any resulting interactions.

The analysis provided in Table 4.1 suggests that cumulative risk associated with sound and other stressors will occur primarily through the HPA axis. While there is some evidence that the presence of ships and their accompanying sounds affect the HPA axis (Rolland et al., 2012), no studies have looked at the cumulative risk of sound and other stressors through the HPA axis. The indirect effects of sound through prey limitation and predator response are discussed in Chapter 2.

There is strong evidence that petroleum-associated chemicals can adversely affect the HPA axis, providing a potential pathway for interactions with other stressors. Studies by Mohr et al. (2008, 2010) of mink (*Mustela vison*) as a surrogate for sea otters (*Enhydra lutris*) found that exposure to fuel oil interfered with the HPA pathway, resulting in damage to the adrenal gland and an insufficient stress response when the animals were experimentally stimulated with adrenocorticotropic hormone. Polycyclic aromatic hydrocarbons (PAHs), the predominant class of chemicals in fuel oils that are linked to adverse health effects, are more rapidly metabolized (Mohr et al., 2008, 2010) than POPs. Unless there is continuing exposure to an environmental source, exposure of marine mammals to PAHs is generally more limited than to persistent organochlorines. However, the effects on the HPA pathway as a result of acute exposure from, for example, an oil spill may persist for many years. Nearly half of the live bottlenose dolphins (*Tursiops truncatus*) sampled from a bay within the *Deepwater Horizon* (*DWH*) oil spill footprint approximately 1 year after the massive spill had indications of insufficient production of adrenal hormones (Schwacke et al., 2014b). Adrenal insufficiency can lead to adrenal crisis and death in animals that are challenged with other stressors, such as physical injury, microparasites, or temperature extremes, to which a healthy animal would otherwise adapt. Many of the dead dolphins that were recovered in the 1.5 years post-spill had rare adrenal gland lesions, and Venn-Watson et al. (2015) suggested that a likely cause of death for these dolphins was an adrenal crisis brought on by an interaction between the effects of petroleum-associated chemicals with the HPA axis and thermal stress (a particularly cold winter in the year after the spill) or a pathogen infection. Indications of adrenal insufficiency were found in dolphins from the same bay sampled 3 to 4 years after the *DWH* spill (Smith et al., 2017), suggesting that injuries to the HPA axis may be long lasting.

It has been suggested that some POPs may also disrupt the HPA axis by interfering with glucocorticoid receptors or the synthesis of adrenal steroids (Martineau, 2007; Diamanti-Kandarakis et al., 2009; Harvey, 2016), but studies to support such effects are still lacking. However, there is strong evidence for an HPA axis effect for one POP: the DDT derivative o,p′-DDD, which is a well-known inhibitor of adrenal steroidogenesis and is used in the treatment of hyperadrenocorticism (chronic overproduction of glucocorticoid) in dogs (Klein and Peterson, 2010).

Permanent or persistent adverse health outcomes, including decreased glucocorticoid measures, have also been reported in survivors of toxic algal blooms (Bejarano et al., 2008b; Goldstein et al., 2008; Gulland et al., 2012), and these provide the potential synergistic interactions with other stressors. For example, sea lions exposed to domoic acid, a potent neurotoxin, from algal blooms were found to have low serum cortisol concentrations as compared to unexposed controls (Gulland et al., 2012). This effect was seen in sea lions with indication of recent exposure (domoic acid in urine or feces sample), as well as in sea lions that were assumed to have been previously exposed (undetectable domoic acid in urine or feces sample). It is unclear whether the low cortisol concentrations were due to binding of domoic acid to glutamate receptors in the endocrine glands, adrenal gland exhaustion, or other disruption of the HPA axis (see Gulland et al. [2012] for discussion). Regardless, the low cortisol suggests that these individuals were more vulnerable to the effects of other stressors (e.g., petroleum-associated chemicals, noise, and perceived threat) that affect the HPA pathway.

Hypothalamic-Pituitary-Thyroid Axis

The effects of prey limitation may interact with the effect of POPs via the HPT axis. The interference of POPs with the HPT pathway has been well established in terrestrial animals (Patrick, 2009), and there is evidence that similar HPT disruption occurs in marine mammals (Tabuchi et al., 2006; Schwacke et al., 2012). HPT disruption can produce adverse effects during critical stages of development and growth (see Zoeller et al. [2002] and Diamanti-Kandarakis et al. [2009] for review). There is strong evidence for the relationship of POP burdens to suppression of thyroid hormones in diverse species of marine mammals, including pinnipeds, cetaceans, and polar bears (Jenssen, 2006). These effects could potentially act synergistically with the effects of prey limitation, in times of nutritional stress or when animals are faced with other environmental challenges. Ford et al. (2010) suggest high POP concentrations in Pacific killer whales (Ross et al., 2000) may have acted synergistically with the effects of prey limitation, resulting in increased mortality during times of low prey abundance. Reduced prey availability would have resulted in the depletion of fat stores and could have led to mobilization of POPs sequestered in the blubber. The increase in circulating POPs could have interfered with metabolic processes. It could also have further increased suppression of immune responses that were already being modulated by the nutritional stress, resulting in increased disease susceptibility.

Immune Pathway

Numerous researchers have suggested a potential for synergistic interactions between the effects of chemical contaminants and microparasites through the immune pathway. This is based on the well-known immunosuppressive effects of many POPs. Evidence for a greater incidence of infections in relation to POP exposure has been demonstrated in human studies (reviewed by Carpenter [2006] and Gascon et al. [2013]), and effects on immunity have been demonstrated in marine mammals using indices of immune function and/or in vitro experiments using marine mammal leukocytes (Ross et al., 1995, 1996a; De Guise et al., 1998). Exposure to POPs has been considered as a potential exacerbating factor for a number of viral epidemics, including the morbillivirus epidemics of striped dolphins in the Mediterranean in the early 1990s (Aguilar and Borrell, 1994) and common bottlenose dolphins along the Atlantic coast in the late 1980s (Kuehl et al., 1991). However, the cross-sectional nature of the studies (i.e., POP concentrations were measured simultaneously with the mortality outcome) has made it difficult to demonstrate a causal link between these stressors in wild populations because disease-related weight loss may have resulted in an increased concentration of lipophilic POPs in the remaining blubber layer (Hall et al., 1992). In order to overcome this problem, Hall et al. (2006) adopted a case-control design to analyze data from a long-term study of harbor porpoises stranded around the United Kingdom. They found an increased risk of mortality from infectious disease in animals with high tissue concentrations of POPs.

Other potential synergistic interactions mediated by the immune pathway involve petroleum-associated chemicals and microparasites. Persistent adverse health outcomes involving this pathway were reported in bottlenose dolphins following the *DWH* oil spill (Schwacke et al., 2014a, 2014b; Lane et al., 2015; Venn-Watson et al., 2015). The reported immune perturbations were compatible with an increased susceptibility to intracellular bacterial infections (e.g., brucellosis) that can cause reproductive failure (S. De Guise, personal communication), and in the years immediately following the spill, a higher than expected prevalence of primary bacterial pneumonia was noted in recovered dolphin carcasses (Venn-Watson et al., 2015).

The chronic effects of one pathogen may result in a synergistic interaction with the effects of other pathogens via the immune pathway. For example, morbillivirus infection may result in residual immune system perturbations. It has been shown to erase immunological memory in laboratory animals, leading to a persistent increased susceptibility to other infectious agents (de Vries et al., 2012). Impairment of cell-mediated adaptive immunity and partially upregulated humoral immune response has been reported in bottlenose dolphins with morbillivirus-positive antibody titers (Bossart et al., 2011). These perturbations could impact an animal's ability to mount an appropriate immune response when challenged. Furthermore, opportunistic secondary infections leading to mortality following the acute phase of morbillivirus infection have been reported following a number of cetacean morbillivirus outbreaks (see Van Bressem et al. [2014] for review).

Brain/CNS Pathway

Maternal exposure to POPs, and specifically PCBs, has been linked to adverse developmental effects in human offspring, including neurological effects and reduced cognitive function (e.g., Jacobson and Jacobson, 1996; Stewart et al., 2003, 2008; reviewed by Boucher et al., 2009). Such effects would produce less fit offspring, and if similar effects occur for wild marine mammals this could clearly lead to decreased survival in the earliest life stages, if individuals are exposed to other stressors that require increased foraging proficiency or rapid avoidance responses (e.g., prey limitation, perceived threat, and noise). In addition, a recent study by Cook et al. (2015) provides evidence that hippocampal lesions caused by sublethal exposure to domoic acid linked to toxic algal blooms affect spatial memory, which potentially could impair an animal's ability to navigate and forage. Such effects would be permanent for the individual and would likely interact with the effects of other stressors, such as prey limitation.

Animals that survive morbillivirus infection may be plagued with persistent chronic CNS infection. Chronic encephalitis was identified as a common cause of death in stranded striped dolphins (*Stenella coeruleoalba*) for years following a morbillivirus outbreak in the Mediterranean (Soto et al., 2011) and has also been identified in other cetacean species following morbillivirus outbreaks after the outbreak had subsided (Uchida et al., 1999; Yang et al., 2006). These chronic CNS infections could affect behavioral and physiological responses to other stressors, such as noise, particularly for deep-diving cetaceans. However, the estimated prevalence of CNS infection even following the substantial Mediterranean dolphin morbillivirus epidemic was relatively low (1-3 per 1,000 cases of infected individuals) (Soto et al., 2011) and therefore may not be a significant factor for population-level effects.

Auditory Pathway

One of the documented developmental effects of POP exposure is hearing loss, potentially mediated at least in part through the HPT axis; it involves loss of outer hair cells (Crofton et al., 2000; Lilienthal et al., 2011) and distorted development of the primary auditory cortex (Kenet et al., 2007). Such permanent conditions could result in an interaction between POP exposure and the effects of other stressors, such as prey availability and predation pressure, mediated by the auditory pathway.

Organic solvents may also induce permanent hearing loss by damaging the outer hair cells or through effects on central auditory pathways. Studies of other mammal species (primarily rats and humans) demonstrate that the hearing frequencies affected by solvents are different from those affected by noise (reviewed by Fuente and McPherson, 2006). Furthermore, studies in rats have reported synergistic effects between some solvents and noise, demonstrating that simultaneous exposure to both produces a more severe hearing loss than the summed hearing loss produced by exposure to either agent alone (Lataye and Campo, 1997; Brandt-Lassen et al., 2000; Lataye et al., 2000; Mäkitie et al., 2003). The timing of exposure may be important as studies have also shown that the interactive effect between toluene and noise exposure was only synergistic if the exposures occurred simultaneously, or if the toluene immediately preceded the noise exposure. When the noise exposure was prior to the toluene exposure, the effects of the two stressors were independent (Johnson et al., 1990).

Interactions Across Pathways

All of the actual or potential interactions between stressor effects we have described above occur when the effects of different stressors act along the same pathway for persistent health outcomes. However, interactions may also occur across such pathways.

For example, interactions between the immune and reproductive pathways have been documented when prey is limited. The substantial metabolic cost of mounting an immune response has been well documented in diverse taxa, including mammals, birds, reptiles, and insects (Lochmiller and Deerenberg, 2000). Responses to moderate infections can lead to energetic costs as high as 55% increases in metabolic rate and 150-200% increases in the rates of glucose production. If prey is limited, animals can make allocation trade-offs between competing physiological processes. Ecological immunology theory predicts allocation trade-offs between reproductive effort and immune responses under conditions of energy limitation (Graham et al., 2011). When energy is limited, low-intensity infections may be allowed to persist if the energetic costs outweigh the benefits of clearing the infection (Sheldon and Verhulst, 1996; Martin et al., 2011). Individuals may prioritize innate immune responses over more expensive adaptive immune responses, despite greater potential for oxidative damage and autoimmunity (Downs and Dochterman, 2014).

During reproduction, nutrient limitation can force individuals to reduce their energy allocation to immune response so that they can support current reproductive effort in a way that may affect their future reproductive potential (Sheldon and Verhulst, 1996; Svensson et al., 1998). Thus, nutrient limitation may lead to impaired immune response especially during periods of reproduction. Because reproduction is associated with increased potential for pathogen exposure from conspecifics (e.g., during colonial breeding), energetic impacts on immune response can influence the survival costs of reproduction in marine mammals (Peck et al., 2016).

There is also potential for interactions between the HPA and immune pathways as a result of exposure to a range of stressors. Chronic elevation of stress hormones is known to downregulate immune response in wildlife systems (Sheldon and Verhulst, 1996; Råberg et al., 1998) through several pathways, including altering antibody responses (Fowles et al., 1993) and inhibiting lymphocyte proliferation (Rollins-Smith and Blair, 1993). Effects of glucocorticoid stress hormones are hypothesized to be an important mechanism underlying trade-offs between energy expenditure and immune response and may help to reduce the response to injury or infection during nutrient limitation (Sternberg et al., 1992; DeRijk et al., 1997).

There have been numerous efforts to examine the effect of stress hormones on immune responses in wildlife (Ricklefs and Wikelski, 2002; Acevedo-Whitehouse and Duffus, 2009; Peck et al., 2016). The few studies in marine mammals suggest that stress modulation of immune function in marine mammals is complex. Body reserves, foraging success, and the degree of plasticity in immune response may impact disease risk synergistically, through a trade-off between immunity and starvation resistance (Brock et al., 2013a; Peck et al., 2016). Immune investment may be directly impacted by anthropogenic disturbance. Brock et al. (2013b) revealed negative associations between body condition and immune response but only in a population exposed to anthropogenic disturbance. These findings implied energetic costs to disturbance that influenced energy allocation toward fighting infection. Finally, individual components of the immune response may be impacted differentially by elevations in stress hormones and variation in body reserves in ways that differ from biomedical model species (Peck et al., 2016).

PRIORITIZING STRESSORS FOR CUMULATIVE EFFECTS ANALYSIS

As noted above, there is only limited understanding of how exposure to individual stressors may affect demographic rates or population dynamics in marine mammals. Yet most marine mammal populations are actually exposed to multiple stressors, and the committee's review of studies of multiple stressors indicates that they are as likely to interact synergistically or antagonistically as they are to act in a simple additive way. It is necessary to find a way to understand the nature of these interactions, while recognizing that experimental investigations of the combined effect of multiple stressors on marine mammals are unlikely to be feasible or ethical. Figure 4.3 is a decision tree that can be used to identify situations in which studies of the interactions between stressors

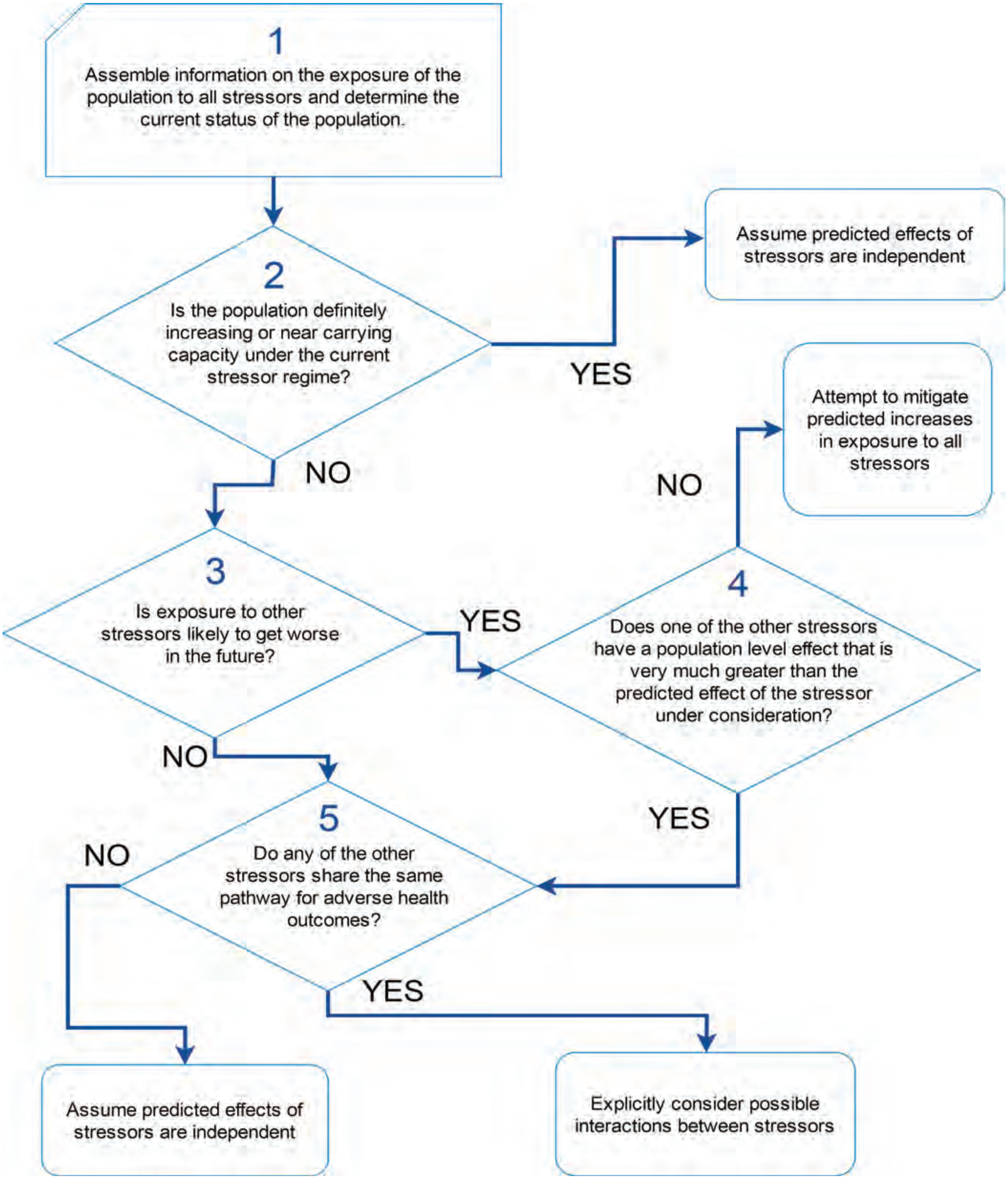

FIGURE 4.3 A decision tree for identifying situations where studies of the possible interactions between stressors should be given a high priority when considering the effect of a focal stressor on a population. See text for a detailed description of the decision-making process.

should be given high priority. It is based on the assumption that interactions are most likely to occur among stressors that share a common pathway for a persistent health outcome (Côté et al., 2016).

Step 1 in the decision process is to determine the spatial and temporal overlap between each stressor and the population of interest. Geospatial approaches, such as those described by Halpern et al. (2007) and Maxwell et al. (2013), can be used to determine this overlap, although, as noted above, these approaches do not provide a rigorous assessment of cumulative impacts. However, several issues make the estimation of exposure to multiple stressors more complicated than first meets the eye. For example, many marine mammal populations are migratory and they will therefore experience considerable temporal variation in their exposure to particular stressors. Thus, the actual duration of exposure to a stressor that is present in a particular area is limited by the amount of time the population actually spends in that area. Quantifying temporal variation in stressor presence is also important for resident populations, because the presence of a stressor may not coincide with sensitive life-history stages. In addition, prior exposure to pathogens or toxins may increase an individual's sensitivity to additional stressors that are encountered in different locations or long after the initial exposure to the pathogen or toxin. Step 2 is to determine the current status of the population of interest (i.e., is it increasing, neither obviously decreasing nor increasing, or decreasing). Chapter 7 describes the methods that can be used to ascertain population status. If a population is definitely increasing, or if it is close to carrying capacity, it should be reasonably resilient (Taylor and DeMaster, 1993) to additional mortality caused by interactive effects between stressors. Large adverse population-level effects of these interactions are likely to be detected before the population has declined to levels of concern. In these circumstances, studies of possible synergies between stressors would not be a high priority.

Steps 3 and 4 allow the identification of situations in which the population is decreasing and the population's exposure to stressors is expected to increase over time. If one of the existing stressors to which the population is exposed is known to have a dominant effect (Step 4), possible interactive effects should be considered for stressors that share the same pathways for adverse health outcomes as the dominant stressor. If there is no dominant stressor, efforts will likely be required to mitigate any potential increases in stressor exposure, even if there is no evidence of interaction between the stressors.

In Step 5 the other stressors to which the population is currently exposed should be reviewed to see if they share the same pathway for adverse health outcomes. If they do, then the possibility that these stressors may interact synergistically should be investigated.

When considering the way the effects of multiple stressors may be analyzed, it is important to take account of the lessons that have been learned from epidemiological studies, where confounding variables are known to give rise to spurious associations between exposure variables and effects of interest. This is particularly likely to be the case when the effects of one stressor operate along the same causal pathway as other variables. This situation may result in colinearity between stressor variables in linear models, or it may mask the indirect effects of stressors through other variables when fixed effects are assessed in an ANOVA. In these cases, analyses that are based on structural equation modeling or some other latent state modeling may better account for the causal pathways by which stressors impact physiology, behavior, health, or vital rates.

Recommendation 4.1: Situations where studies of cumulative effects should be prioritized can be identified using tools such as the decision tree developed by the committee and testing for whether pathways for adverse health outcomes are shared across stressors.

CASE STUDIES: DIFFICULTIES IN INFERRING CAUSES OF DECLINES

In this section, three case studies of marine mammal populations that have either suffered a precipitous, unexplained decline, or have failed to recover following the removal of a dominant stressor are considered. This is not a critique of the work that has been done to investigate these declines, nor is it an attempt to suggest how these populations should be managed to promote their recovery. Rather, the committee's aim is to describe how the potential causes of the decline were initially identified, and to investigate what conclusions might have been drawn if the decision tree shown in Figure 4.3 had been used as part of this process.

Cook Inlet Beluga

The Cook Inlet (CI) beluga whale (*Delphinapterus leucas*) population, which is separated by the Alaska Peninsula from other beluga populations in Alaskan waters, declined from around 1,300 whales in 1979 to 367 in 1999 (Hobbs et al., 2000; see Figure 4.4). Alaskan Native subsistence harvest between 1993 and 1998 ranged from 21 in 1994 to 123 in 1996. The most reliable data come from 1995-1997, when an average of 87 whales were taken per year (Angliss and Lodge, 2002). Including this subsistence take in models of the population's dynamics indicated that it was sufficient to account for most of the observed decline over this period. Alaskan Natives imposed a voluntary moratorium in 1999, and in 2000 the National Marine Fisheries Service (NMFS) declared the population depleted under the Marine Mammal Protection Act (65 Fed. Reg. 34590). The expectation was that with greatly reduced subsistence take the population would grow between 2% and 6% annually. Since 1999 the total subsistence harvest has been five whales, with none

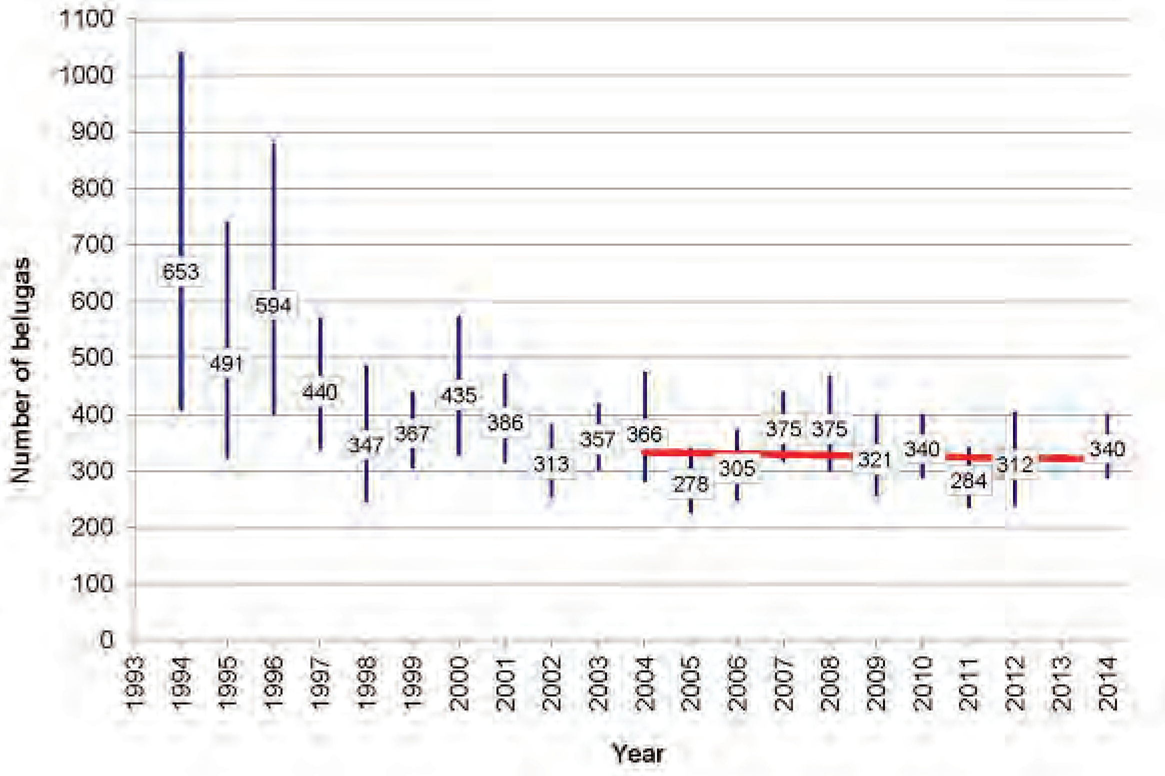

FIGURE 4.4 Figure 13 from NMFS (2015) showing abundance estimates for Cook Inlet beluga whales between 1994 and 2014. Vertical bars indicate the 95% confidence interval for each estimate. The trend from 1999 (when the hunt was managed) to 2014 was −1.3% per year (standard error [SE] = 0.7%).

taken after 2005 (NMFS, 2015). Nonetheless, the population has shown no sign of recovery (see Figure 4.4). The most recent estimate of population size is 340 in 2014 (Shelden et al., 2015). Based on aerial surveys and satellite telemetry data, the core summer distribution of the population has contracted from more than 7,000 km^2 in 1978-1979 to 2,800 km^2 in 1998-2008 (Rugh et al., 2010). As a result, most of the population is concentrated in upper Cook Inlet, during the summer months. This is close to the port of Anchorage, where the population is most likely to be exposed to disturbance from human activities (NMFS, 2015). Why there has been this change of distribution is not known, although several possible reasons have been suggested (Moore et al., 2000; Shelden et al., 2003; Goetz et al., 2007).

In 2010, the NMFS established a Cook Inlet Beluga Recovery Team (CIBRT). The CIBRT drew up a list of threats which they believed "might significantly impact CI recovery" (NMFS, 2015) and used their "best professional judgment" to identify the most important threats. These threats were then ranked on the basis of their extent, frequency, trend, probability of occurrence, and potential magnitude. The 10 threats of greatest concern are listed below, with an indication (in parentheses) of which of the stressors listed in Chapter 3 might be associated with each threat:

1. catastrophic events, such as an oil spill
2. cumulative and synergistic effects of multiple stressors (primarily between noise, nonbiological toxins, and perceived threats)
3. noise (noise, perceived threat)
4. disease agents (pathogens) and harmful algal blooms (biotoxins)
5. habitat loss or degradation (habitat limitation)
6. reductions in prey (prey limitation)
7. subsistence hunting (acute physical injury)
8. unauthorized take (acute physical injury)
9. pollution (nonbiological toxins)
10. predation (acute physical injury, perceived threat)

Threats 1-3 were categorized as of "high relative concern," threats 4-7 as "medium" concern, and threats 8-10 as "low" concern. The only threats for which data on beluga morbidity and mortality exist were placed in the low- and medium-concern categories. The justification for this placement is that CI belugas generally have lower contaminant loads than belugas studied elsewhere, that killer whales (*Orcinus orca*) were suspected in the deaths of only three CI beluga whales in the past 17 years and that mammal-eating killer whales have not been observed in the population's core summer range, and that the subsistence hunt is suspended until at least 2018 and would be reinstated at a low level only if it did not place the recovery of the population in jeopardy.

The draft recovery plan concluded that "disease as a factor in the deaths of CI belugas appears to be low, and there is little evidence to suggest diseases of concern are present in other mammals in the area." It is therefore slightly surprising that disease was considered to be a threat of medium concern. However, this categorization may be because of the potential role of diseases in catastrophic events. In contrast, the draft recovery plan recognizes that "the trend of habitat loss or degradation . . . is . . . increasing over time," but habitat degradation was only categorized as a medium concern "due to limited understanding of how . . . habitat may be altered . . . and its resilience to perturbation." Prey limitation was also categorized as being of medium concern because "the magnitude of the impact of a reduction in prey on . . . belugas is unknown, as is the trend."

Catastrophic events are known to strongly influence extinction risk for small populations (Morris and Doak, 2002, p. 21). Such events are particularly likely to occur when a large proportion of the population is concentrated in a small area at certain times of the year. This is one of the consequences of the contraction in the summer range of CI belugas and, as a result, many animals could be exposed to episodic stressors such as spills of petroleum-associated chemicals and solvents and outbreaks of infectious disease.

There have been no documented direct or indirect effects of noise on CI belugas, and the categorization of noise as a threat of high relative concern appears to be primarily based on "evidence from other odontocete species . . . to conclude that a high potential exists for negative impacts (of noise)." As noted in Chapter 2, evidence of the effects of noise on marine mammal populations is largely circumstantial or conjectural.

When the decision tree from Figure 4.3 is applied to the CI beluga population, one can see that the population is declining, existing stressor levels are likely to get worse in the future, there is no dominant stressor, and there are a number of stressors (noise, nonbiological toxins, microparasites, and prey limitation) that share potential pathways for adverse effects. This leads to the conclusion that efforts will be required to mitigate any potential increases in stressor exposure, even if there is no evidence of interaction between the stressors.

In summary, the initial decline of the CI beluga population can be largely explained by excessive harvesting, but the reasons why the population has failed to recover remain unknown. However, interactions between some of the many stressors to which the population is exposed may be involved in this failure. The recovery plan is primarily concerned with mitigating the threats of high and medium relative concern; this is also the recommendation that emerges from application of the decision tree in Figure 4.3. The population monitoring planned as part of the recovery plan will focus on photo-identification studies which, as we note in Chapter 7, have the potential to provide relatively precise information on many of the demographic characteristics of the population.

Collapse of Pinniped and Sea Otter Populations in the Northern North Pacific Ocean and Southern Bering Sea

Once abundant populations of harbor seals (*Phoca vitulina*), Steller sea lions (*Eumetopias jubata*), and sea otters (*Enhydra lutris*) have collapsed over large areas of the Gulf of Alaska, Aleutian archipelago, and southern Bering Sea during the past four or five decades (Doroff et al., 2003; NRC, 2003b; Small et al., 2008). Despite high levels of public interest in these species and legal mandates to define and assess their various stocks under the U.S. Marine Mammal Protection Act, considerable uncertainty and scientific debate remain over the patterns, causes, and consequences of these declines.

Although there is no question that these three species have declined, data on the timing and magnitude of their declines varies in quality among the species. This is largely a consequence of when the surveys were done relative to the periods of decline. For harbor seals and Steller sea lions, rigorous monitoring programs were not initiated until the 1990s after the declines had begun (NRC, 2003b; Small et al., 2008). This shortcoming is most acute for harbor seals, which were effectively unmonitored in southwestern Alaska until after the decline had run its course. Monitoring data for Steller sea lions are better in that more systematic surveys were initiated in the 1970s while the decline was ongoing (NRC, 2003b). However, few data exist from before the decline or during its early stages, thus creating uncertainty over the onset and magnitude of the decline. This shortcoming is most severe in the central and western Aleutian Islands.

While the monitoring data range from problematic to less than ideal for pinnipeds and sea otters, they are essentially nonexistent for regional stocks of small cetaceans except for killer whales. Two species are common in this area (harbor porpoise [*Phocoena phocoena*] and Dall's porpoise [*Phocoena dalli*]), and there are a variety of rarer species (e.g., Cuvier's beaked whale [*Ziphius cavirostris*], Baird's beaked whale [*Berardius bairdii*], Stejneger's beaked whale [*Mesoplodon stejnegeri*], beluga [*Delphinapterus leucas*]; possibly striped dolphin [*Stenella coeruleoalba*], Pacific white-sided dolphin [*Lagenorhynchus obliquidens*], Risso's dolphin [*Grampus griseus*], false killer whale [*Pseudorca crassidens*]; and conceivably one or more as-yet-to-be-described species). Part of the difficulty for monitoring these cetacean species is that they spend their entire lives in a vast oceanic environment that is difficult to access and to survey.

Except for sea otters, both the causes and consequences of the marine mammal population declines are poorly known. In the sea otter's case, the weight of available evidence points to killer whale predation as the likely cause (Estes et al., 1998; USFWS, 2013). Ecological consequences of the sea otter collapse, which also have been reasonably well documented, include a widespread ecosystem phase shift (e.g., Selkoe et al., 2015) from a kelp-dominated to a deforested, sea urchin–dominated coastal sea floor (Estes et al., 1998) and various knock-on influences of this "trophic cascade" to other species and ecological processes (Estes et al., 2009a).

In the case of pinnipeds, there are at least four reasons for the general lack of causal understanding. A primary reason, in contrast with the sea otter decline, is that none of the systems were observed closely or carefully while the declines were in the process of occurring. Other than the declines themselves, few data exist on co-occurring patterns of changes in the abundance and distribution of other species. A second reason arises from a generally poor understanding of food web structure and dynamic process that led to spatiotemporal variation in prey in the open sea. In contrast with the sea otter's food web, which is easy to observe and measure and can be studied experimentally, water column and oceanic food webs that sustain pinnipeds are difficult to observe and even more difficult to study experimentally. A third reason for the lack of understanding of the pinniped declines arises from the mobile nature of their predators and prey, which, when coupled with convective influences of ocean currents, produces an ecosystem in which meaningful measurements of the distribution and abundance of species must be done at large spatial scales. Finally, until the early 2000s, the pinniped declines were believed to have resulted from bottom-up forcing—detrimental impacts on survival or reproduction resulting from changes in the abundance or quality of food, which in turn were mostly thought to have resulted from changes in physical oceanography or competition with fisheries. This belief in nutritional limitation has been, and continues to be, embraced by many people in the local research and management communities, despite a general lack of evidence (NRC, 2003b). While the pervasiveness of bottom-up forcing processes in driving the sea lion declines has been questioned (Springer et al., 2003), there has been no concurrence and considerable debate over both the cause of the sea lion decline and the failure of the species to recover following various conservation and management actions (DeMaster et al., 2006; Trites et al., 2007; Wade et al., 2007, 2009; Springer et al., 2008; Estes et al., 2009b; and many others). These differing views are evident in the remarkably different perspectives and conclusions in two separate overview reports—one by the National Research Council (NRC, 2003b) and the other by the NMFS (NMFS, 2008).

This particular case study of the causal factors for the declines in sea otters and pinnipeds illustrates how the nature of evidence, together with differences in belief and scientific philosophy (i.e., one's foundational bases for making inferences), can prevent consensus on the potential roles of even simple direct effects in marine mammal population declines. It is possible, if not likely, that sea otter and pinniped declines are the consequence of multiple stressors. However, so long as such strong debate surrounds the potential importance of the single stressors, progress in assessing the impacts of multiple stressors on marine mammals will remain an elusive goal.

Because of the lack of suitable data, it is difficult to apply the decision tree in Figure 4.3 to this case study. The two principal stressors for all species that have definitely declined appear to be food limitation, predation pressure, and (possibly) perceived threat. These do not share potential pathways for adverse effects.

Collapse of U.K. Harbor Seal Populations

U.K. populations of harbor seals are monitored on a 5-year cycle using aerial surveys of haul-out concentrations conducted during the summer molt. These surveys provided evidence of declines of around 40% between 2001 and 2006 in a number of Scottish populations (Lonergan et al., 2007). The declines have continued, with an estimated decline of 65% since 2001 in Orkney (Hanson et al., 2013), and 90% since 2002 in the Firth of Tay (Hanson et al., 2015). However, the pattern of decline has not been consistent. For example, counts in the Moray Firth declined by 50% between 1993 and 2005 (Thompson et al., 2007), probably because of the effects of deliberate killing (Matthiopoulos et al., 2014); although levels of deliberate killing have been reduced, the population has continued to fluctuate in size. Populations on the west coast of Scotland and in the southern North Sea populations have shown no obvious long-term declines (see Figure 4.5).

A workshop held in 2012 identified a long list of potential causes for these declines that included almost all of the stressors listed in Chapter 3. However, by the time a second workshop was held in 2014, this list had been narrowed down to three "key potential drivers" (Hall et al., 2015): physical injury (spiral lesions; Bexton et al., 2012), prey limitation, and biotoxins. The spiral lesions, originally attributed to collisions with ducted propellers, are now believed to be the result of predatory attacks by male grey seals (van Neer et al., 2014; Thompson et al., 2015). Deaths from these injuries may be sufficient to explain the precipitous decline of the small Firth of Tay population (Hanson et al., 2015), but it is not clear whether they can explain the decline in the much larger Orkney population. Although there is evidence that harbor seals around the United Kingdom are regularly

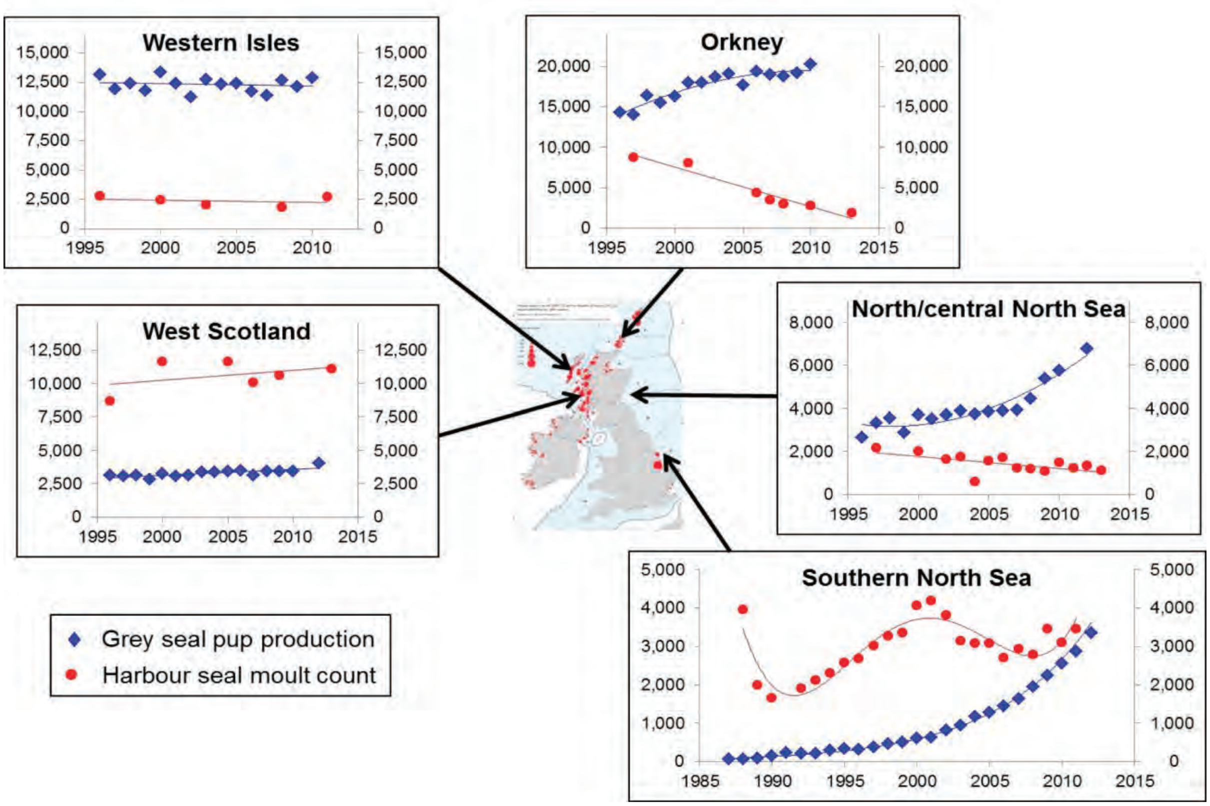

FIGURE 4.5 Changes in harbor seal molt counts and grey seal pup counts for the United Kingdom over the period 1996-2013. SOURCE: Taken from Figure 1 of Hall et al. (2015).

exposed to biotoxins, no deaths have actually been attributed to this cause (Jensen et al., 2015).

Application of the decision tree from Figure 4.3 indicates that the affected populations are not increasing or near carrying capacity, that some stressor levels are likely to increase (grey seal numbers, and therefore grey seal predation, are increasing, as is the incidence of toxic algal blooms in Scottish waters [Hall and Frame, 2010]), and that some of the stressors (prey limitation and biotoxins) share two pathways for adverse outcomes. There has been some preliminary work to investigate possible interactions between these stressors. Caillat and Smout (2015) modified the state-space population model developed by Matthiopoulos et al. (2014) for the Moray Firth population to include the potential effects of prey availability, grey seal numbers, and exposure to biotoxins. They used a series of logistic equations to model the potential effects of all these stressors on fecundity and pup survival. Although the logistic equation does not explicitly include an interaction term, the predicted effects of the different stressors are not additive. In fact, Caillat and Smout (2015) found that only grey seal numbers had a significant effect on pup survival, and the only stressor affecting fecundity was prey limitation. This suggests that each of these stressors had a dominant effect on one demographic rate, and that there was no interaction between their effects. This analysis was only possible because detailed information on changes in demographic rates over time were available from photo-identification studies of the Moray Firth population (Cordes, 2011).

5

Modeling the Population Consequences of Exposure to Multiple Stressors

INTRODUCTION

A conceptual model of the Population Consequences of Acoustic Disturbance (PCAD) was first developed by the National Research Council (NRC) (2005). A working group established by the U.S. Office of Naval Research in 2009 has formalized this model structure and extended it to cover all forms of disturbance. This Population Consequences of Disturbance (PCoD) model is described by New et al. (2014). It consists of a series of transfer functions that describe how

- exposure to stressors (such as noise) affects individual behavior,
- the resulting changes in behavior can affect health (defined as all internal factors that affect fitness or homeostasis),
- variations in health may affect individual vital rates (the probability of survival, giving birth, or growth/attaining sexual maturity for an individual), and
- data on the variation in the level of exposure to the stressor experienced by different individuals can be used to scale up the anticipated changes in vital rates so that they can be used to predict population-level effects.

As noted in Chapter 4, these transfer functions and their associated causal flows correspond to the first five levels of biological organization in the hierarchy of responses to a stressor illustrated in Figure 4.1. Approaches for assessing the effects of stressors on the two higher levels of biological organization (communities and ecosystems) are described in Chapter 6.

Full PCoD models have been developed for a number of marine mammal populations (Lusseau et al., 2012; Nabe-Nielsen et al., 2014; New et al., 2014; King et al., 2015). Ideally, the predictions of these models should be fitted to appropriate time series of empirical data obtained over a range of levels of disturbance, and the results of the fitting process used to improve the parameter estimates and quantify the uncertainty associated with the model predictions. Approaches such as Bayesian hidden-process modeling (Newman et al., 2006) may be appropriate for this purpose. However, in no case has this been possible, and such models should be considered “exploratory.” Exploratory models are most useful for comparing the possible consequences of different scenarios and for identifying priority areas for research. It is particularly important that the uncertainties associated with their underlying parameter values are documented, and that the effects of these uncertainties on their predictions are quantified.

New et al. (2014) used the PCoD model structure to investigate the potential effects of lost foraging dives on the health (measured by total lipid mass; see Schick et al., 2013) of adult female southern elephant seals, and the implications of variation in health for pup survival and population dynamics. They used information obtained from data loggers that were attached to animals immediately before they embarked on their ~240-day post-molt foraging trips. The data loggers allowed a reconstruction of their surface transit time and their foraging dive time. During portions of some foraging dives, elephant seals drift, and the rate of vertical movement during the drift is related to the ratio of lipid to lean body mass. The data logger information was calibrated against actual lipid gain during the foraging trip using measurements of body composition collected before and after the foraging trip. The results of other studies were used to link maternal mass to pup mass at weaning (Arnbom et al., 1993) and pup mass at weaning to pup survival (McMahon et al., 2000, 2003). The model was then used to determine the effect of foraging dive disturbance on pup survival. It was assumed that there were no foraging dives for the duration of the disturbance, and surface transit time was set to the observed maximum for that individual. If animals were disturbed for 50% of their time at sea in 1 year, the predicted decline in population size was small (<1%). However, if this level of disturbance persisted for an extended period (for example, as a result of variations in the extent of the Antarctic ice sheet caused by climate change), the predicted effects were much greater (a 10% decline in abundance over 30 years). This analysis was only possible because detailed longitudinal data on the movements, health, and reproductive success of a large number of adult female seals were available. Such extensive data sets require decades of intensive research and are only available for a few marine mammal populations.

Researchers have adopted a range of techniques to build PCoD models in situations where empirical data are more limited. Nabe-Nielsen et al. (2014) used an individual-based model of the movements of harbor porpoises to estimate the potential effects of responses to the noise associated with wind turbine operation and shipping on their energy reserves. They then used a hypothetical relationship between energy reserves and survival to calculate population-level consequences. Villegas-Amtman et al. (2015) used a similar approach to predict the potential effects of reduced energy intake on reproductive success and survival for gray whales.

If empirical data are sufficient to estimate a relation between behavioral change and health, but not between health and vital rates, it may be possible to use a surrogate measure for the relevant vital rate. Christiansen and Lusseau (2015) used a bioenergetic model and empirical information on the behavioral response of adult female minke whales (*Balaenoptera acutorostrata*) to whale-watching boats on their summer feeding grounds in Iceland to estimate the effects of these responses on the whales' health (as measured by their blubber volume). They calculated how different rates of encounter with whale-watching boats would affect an individual whale's health at the end of the summer, and then used an empirically derived relation between female blubber volume and fetal length (Christiansen et al., 2014) as a surrogate for the relationship between health and the probability of giving birth. Although interactions with whale-watching boats resulted in a 40% reduction in feeding activity, the predicted reduction in a female's body condition over the course of the summer was very small (0.049%), because encounters with boats were rare. This reduction in body condition was not predicted to affect fetal survival. However, even if Christiansen and Lusseau (2015) had detected a significant effect on fetal survival, they would have been unable to forecast the population-level effects of exposure to whale-watching boats because the proportion of the North Atlantic minke whale population that feeds in Icelandic waters and the percentage that has actually encountered boats is not known.

In situations where even surrogate measures are unavailable, expert elicitation (Sutherland and Burgman, 2015) can be used to parameterize some of the transfer functions of the PCoD model. Expert elicitation is a formal process in which a number of experts on a particular topic are asked to predict what may happen in a particular situation. The process is used in conservation science when data are lacking but there is an urgent need for management decisions (Runge et al., 2011; Martin et al., 2012). It is designed to mitigate the well-documented problems that arise when expert judgments are canvassed in an unstructured way. These include anchoring, availability bias, confirmation bias, and overconfidence (Cooke, 1991). These predictions are combined into calibrated, quantitative statements, with associated uncertainty, which can be incorporated into mathematical models (Martin et al., 2012). King et al. (2015) used this approach to parameterize relationships between the number of days on which harbor porpoises were disturbed by noise associated with the construction of offshore wind farms and their survival and reproductive success. These relationships were then used to predict the potential population consequences of different scenarios for the construction of multiple wind farms. Lusseau et al. (2012) used a similar approach to predict the potential aggregate effect of noise associated with wind farm construction, tour boat operation, and harbor expansion on the bottlenose dolphin population in the Moray Firth, Scotland.

In the remainder of this chapter, how the PCoD framework can be expanded to assess the potential population-level effects of exposure to multiple stressors is considered.

DEFINING INDIVIDUAL HEALTH

Evaluation of the potential demographic impacts on marine mammal populations of cumulative exposure to multiple stressors requires the biological upscaling (Cooke et al., 2014) of many levels of organization, including the behavioral responses of individuals, and the effects of these responses on population dynamics, biogeography, and community ecology (see Figure 4.1 in Chapter 4). In this chapter, we consider upscaling to the level of population dynamics. One important factor that links individual behavioral and physiological responses to population dynamics is the "health" of individuals. In 1948, the World Health Organization (WHO) defined health as "a state of complete physical, mental and social well-being and not merely the absence of disease or infirmity." Similarly, definitions of "disease" in wildlife are broader than just infection by pathogens. They include the potential for cumulative impacts on health from nutrition, exposure to toxic chemicals, and climate (Wobeser, 1981). The WHO definition has been debated and criticized over the years (Jadad and O'Grady, 2008; Huber et al., 2011; Stephen, 2014), and recently it has been proposed that health be considered as "the ability to adapt and self-manage" (Huber et al., 2011), implying that a healthy organism is capable of maintaining physiological homeostasis under changing conditions. For wildlife, such definitions are effectively proxies for fitness, emphasizing the potential effects of health on lifetime reproductive success. The committee therefore adopts "the ability to adapt and self-manage" as the definition of health.

Given this background, an assessment of an individual's health provides a useful integration of the way physiological and behavioral responses to multiple stressors may affect that individual's fitness. Potential health indices include body condition, hematological and serum biochemical parameters, steroid hormone levels, and markers of immune function and oxidative stress. This approach offers some potential advantages over empirical attempts to correlate variations in demographic rates with exposure to different stressors, because it can provide an assessment of the potential for reduced survival and reproductive output *prior to* an actual alteration in these rates. In addition, the application of health-based approaches to modeling the cumulative effects of exposure to multiple stressors may increase understanding of the mechanisms by which these stressors affect fitness.

A CONCEPTUAL MODEL FOR THE CUMULATIVE EFFECTS OF MULTIPLE STRESSORS

In this section, an expanded version of the PCoD model shown in Figure 6 of New et al. (2014) is described that can also be used to understand how specific stressors affect individual animals, how these effects can accumulate as a result of exposure to multiple stressors, and how these cumulative effects may translate into population-level consequences. This model, identified as Population Consequences of Multiple Stressors (PCoMS), provides a framework around which quantitative, predictive models for particular situations can be constructed. Figure 5.1 shows the structure of this framework for a single individual exposed to one stressor. It differs from the original PCoD model in the following ways:

- It can be used to describe the effects of any dosage scenario for any stressor, not just those that cause disturbance.
- The individual-based nature of the model is made explicit.
- It includes the direct, acute effects of predation and anthropogenic causes of mortality, such as bycatch, collisions, and deliberate killing.
- Following the model outlined by McEwan (1998, Figure 1), the initial effect of any stressor is assumed to be on an individual's physiology. The resulting physiological changes may or may not be translated into behavioral responses, depending on the context (Killen et al., 2013).
- The direct link between the behavioral change and health compartments in the PCoD model has been removed because, in practice, behavior can only affect health indirectly through its effects on physiology.

The model assumes that an individual's response to any stressor is always mediated, at least initially, by a physiological response because the initial interaction with that stressor will always be through the nervous system. This reflects one of the fundamental aspects of the allostatic

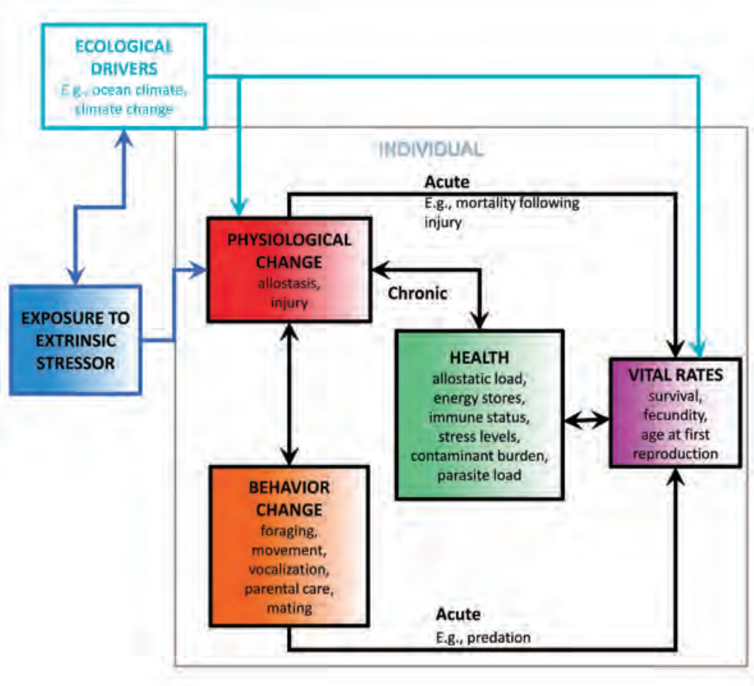

FIGURE 5.1 The Population Consequences of Multiple Stressors (PCoMS) framework for a single individual exposed to one stressor. Each compartment in the framework represents one or more quantities (variables) that evolve over time. Compartments are connected by arrows that represent causal flows ("transfer functions" in the terminology of NRC [2005]). For each individual, changes in physiology may result in changes in behavior (such as movement away from a sound source and cessation of feeding), which may in turn affect physiology.

load concept (McEwan, 1998): whether or not an animal exhibits a behavioral response to a stressor will depend on its internal state and a suite of intrinsic stressors. Consider a foraging individual's response to an approaching vessel. If it perceives the vessel, and its allostatic load is tolerable, it will probably take evasive action (a behavioral response mediated by a physiological response). However, if its body condition is poor, it may choose to keep feeding and may fail to evade the vessel.

Changes in behavior or physiology in response to a stressor may have a direct, acute effect on the vital rates of an individual. For example, an individual may move into an area with a high risk of predation as a result of avoidance behavior, or it may be at increased risk of mortality due to decompression sickness if it changes its diving behavior. For many marine mammal populations, the direct effects of acute stressors, such as bycatch and predation, may be more important than indirect effects. Because these acute effects operate on a short time scale, their cumulative effects are likely to be additive, as discussed in Chapter 4, so they can be modeled in a relatively straightforward way within the PCoMS framework. In this chapter, the focus is on the chronic effects of multiple stressors on health, primarily modeled using the concept of potential allostatic load (McEwen and Wingfield, 2003) that involves the adverse outcome pathways along which nonadditive effects are most likely to occur.

Allostatic load represents the consequences of the individual's efforts to maintain homeostasis. Examples include reduced immune status, increased long-term levels of stress hormones, and reduced body condition relative to normal levels. The allostatic load associated with exposure to a particular stressor is only "potential" because that exposure will not necessarily have an immediate effect. However, it may have an effect on allostatic load at some later date, possibly because of the interaction with other stressors. A high allostatic load will have implications for all of an individual's vital rates. For example, an adult female may choose to forgo breeding in order to reduce her potential for allostatic overload.

In some cases it may be sensible to combine compartments in the PCoMS model (i.e., hypothesize a transfer function that "jumps over" an intermediate compartment) if there is insufficient information to treat them separately. For example, explicitly modeling the physiological processes that occur between exposure to a stressor and a behavioral response is unlikely to be necessary.

The framework can readily be expanded to illustrate the effects of multiple stressors on a single individual (see Figure 5.2). Exposure to a particular dosage scenario for each stressor results in a unique set of physiological and behavioral responses (represented by the stack of responses in Figure 5.2, each layer corresponding to the responses to a different stressor), which may interact with the responses to other stressors. The consequences of the responses to many of these stressors are integrated through their combined effect on an individual's potential allostatic load. Although it is currently impossible to measure allostatic load directly, it may be possible to use proxy measures of health (as described below) as appropriate response variable in studies of cumulative impacts.

The effects of multiple stressors may interact internally to affect allostatic load. For example, contaminants sequestered in the blubber layer may be mobilized during lactation or as a result of elevated stress levels or reduced energy

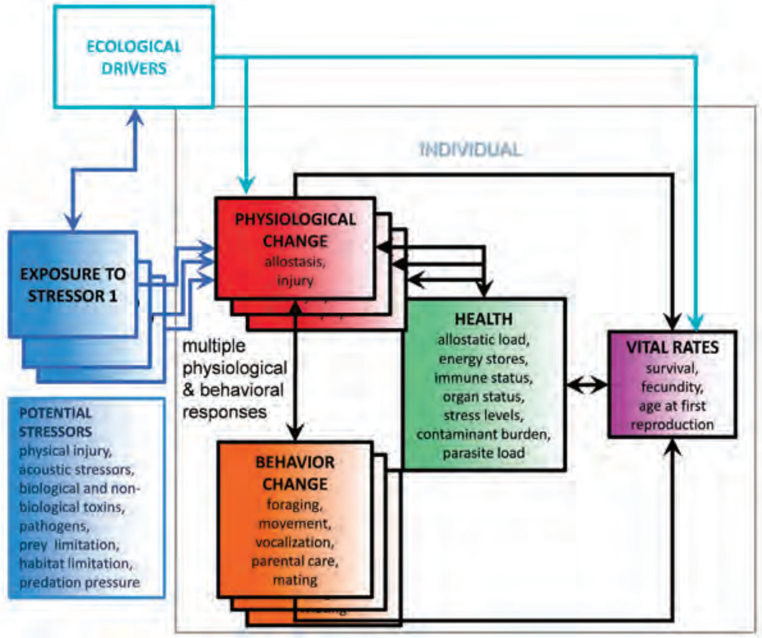

FIGURE 5.2 An expanded version of the framework shown in Figure 5.1 that includes the effect of multiple stressors on a single individual.

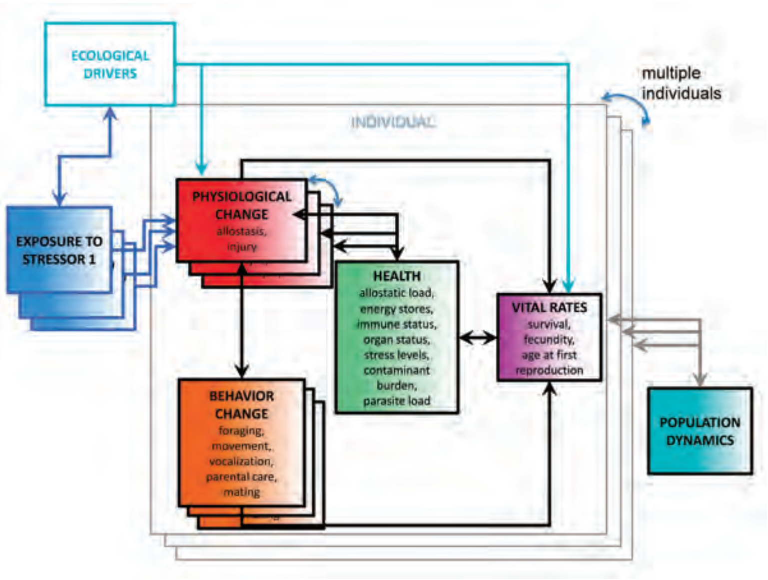

FIGURE 5.3 An expanded version of the framework shown in Figure 5.2 that includes multiple individuals and population-level consequences.

intake that are caused by other stressors. They may then interact with an individual's immune function and affect its response if it is challenged with a novel pathogen. Similarly, the effect of macroparasite burden and dormant pathogens on health may be amplified if immune status is compromised.

The framework can be expanded to the population level if estimates of the potential exposure of each individual in the population to the suite of stressors under consideration, and the effects of this exposure on physiology and behavior, are available (see Figure 5.3). This will require information on the distribution in time and space of the marine mammal species and the stressors, which can be assembled using approaches similar to those used by Maxwell et al. (2013). However, this will also require definition of appropriate dose–response relationships for each stressor, as well as a functional representation of interactions. The committee does not underestimate the difficulties that will be involved in obtaining the information needed to parameterize these functions for even a small number of stressors. The final step is to integrate the effect of these exposures on individual vital rates across the entire population in order to estimate their population-level consequences.

As noted above, the PCoMS framework treats mortality from predation and anthropogenic activities (such as bycatch, deliberate killing, and fatal ship strikes) as acute effects of exposure to the agents of this mortality (predators, fishing gear, hunting pressure, and vessel traffic). It can also be used to model the effects of natural and anthropogenic ecological drivers. For example, as noted in Chapter 3, changes in ocean climate can have profound effects on some marine mammal populations as a result of the redistribution of prey species. In the PCoMS framework this would be modeled as a change in exposure to a prey limitation stressor. Similarly, the effects of climate change are likely to lead to shifts in the distribution of vessel traffic, which can be modeled as changes in exposure to the risk of physical injury, toxic compounds, pathogens, and acoustic stressors. The effects of ice reduction on pagophilic species can be modeled as a habitat limitation stressor. Exposure to this stressor will result in behavioral changes, which could have acute effects (if seal species that normally breed on ice switch to breeding on land, and are therefore at greater risk of predation) or chronic effects (via the health compartment) as a result of the increased travel costs.

The PCoMS framework is similar to the framework developed by Rider et al. (2012) for assessing the role of nonchemical stressors in modulating the human risk factors associated with chemical exposure. However, Rider et al. (2012) place greater emphasis on how to predict the distribution of stressor doses across a population, and they do not consider the consequences of those doses for population dynamics.

The committee stresses that the PCoMS framework, like the original PCAD framework developed in NRC (2005), is only conceptual: it serves primarily to identify what the committee believes are the most important components of any comprehensive model of cumulative effects. The framework needs to be fleshed out with mathematical functions that describe the relationships between the different compartments, and integrated across all the individuals in the population that are exposed to the stressors under consideration. Determining appropriate forms for these functions and then parameterizing these functions will be extremely challenging. In many cases, it may be possible to ignore some of these relationships because they are not relevant to the population under consideration, but such decisions need to be carefully evaluated and fully justified. In situations where one stressor is considered to be dominant (i.e., its effects are so large that the effects of all other stressors to which the population is exposed are negligible by comparison), use of a simplified version of the framework that considers only the dominant stressor is appropriate.

Recommendation 5.1: Future research initiatives should include efforts to develop case studies that apply the PCoMS framework to actual marine mammal populations. These studies will need to estimate exposure to multiple stressors, predict changes in behavior and physiology from those stressors, assess health, and measure vital rates in order to parameterize the functional relationships between these components of the framework. Where possible, the data on changes in demography, population size, and the health of individuals collected in these studies should be used to improve estimates of the parameters of the PCoMS model and reduce uncertainty.

APPLYING THE PCOMS FRAMEWORK TO NORTH ATLANTIC RIGHT WHALES

North Atlantic right whales have been protected since the 1930s and intensively studied since the early 1980s (Kraus and Rolland, 2007), yet their population numbers remain perilously low (Kraus et al., 2005). They are exposed to a wide range of stressors on their summer feeding grounds and over their lengthy migration pathways. These include physical injury as a result of entanglement in fishing gear, collisions with shipping, strong interannual variation in prey availability, and exposure to shipping noise (Clark et al., 2009). The North Atlantic Right Whale Catalogue,[1] curated by the New England Aquarium, contain records of the life histories of many right whale individuals, as well as more than 700,000 photos and drawings. These records can be used to provide information on variations in the health (Pettis et al., 2004) and location of these individuals over time. Values for a set of visual health parameters are added to the catalog each time a whale is photographed. Schick et al. (2013) used these data to estimate the movements and overall health status of these individuals over time and to relate survival to health status. Rolland et al. (2016) used the same health information and model structure to link the health status of females in one year to their calving success in the subsequent year. Successful females were, on average, significantly healthier than unsuccessful ones. There was a dramatic decline in health status and calving success from 1998 to 2000 that coincided with reduced prey availability.

These relationships could be used as the transfer functions linking the health and individual vital rates compartments in a PCoMS framework that described the cumulative effects of physical injury (resulting from entanglement and collisions) and variations in prey availability on this population. Additional information in the North Atlantic Right Whale Catalogue could be used to parameterize a transfer function that would describe the changes in health that occur as a result of different levels of exposure to entanglement over the course of an entire year.

QUANTIFYING EXPOSURE-RELATED CHANGES IN PHYSIOLOGY AND ASSOCIATED CHANGES IN BEHAVIOR

Physiology

As noted above, there will be an immediate physiological response to exposure to a stressor mediated by the central nervous system. These kinds of short-term physiological responses to a stressor have evolved to reduce the risk that the animal's health is compromised. Thus, one of the critical aspects of using physiological measures to assess aggregate and cumulative impacts is the ability to detect physiological changes that actually compromise health. In many cases, the generalized endocrine response to stress can provide relevant information, if there is appropriate contextual information to differentiate between normal adaptive variation and increased allostatic load. Hematological and serum biochemical parameters can be measured from blood to help identify a wide range of disease conditions such as inflammation, liver dysfunction, or anemia. Markers of immune status can provide critical information on the health of an individual, but it may be difficult to differentiate suppression of immune function from absence of exposure to pathogens. The effects of many stressors may be integrated through their impacts on oxidative stress (OS). For example, exposure to organic and inorganic contaminants is associated with dramatic increases in OS and oxidative damage (Ercal et al., 2001; Valavanidis et al., 2006). Exposure to polychlorinated biphenyls is associated with increased OS and oxidative damage to DNA, lipids, and proteins (Stohs, 1990; Oakley et al., 1996). OS also plays an important role in the pathogenesis of viral and bacterial infections (Schwarz, 1996). Chronic activation of the hypothalamic-pituitary-adrenal (HPA) axis and the release of glucocorticoids also enhance OS (Costantini et al., 2008, 2011; Stier et al., 2009; Cote et al., 2010). Such antioxidant responses are energetically expensive and may limit investment in important life-history components (Costantini et al., 2008; Dowling and Simmons, 2009; Monaghan et al., 2009; Metcalfe and Alonso-Alvarez, 2010; Isaksson et al., 2011). Thus, evidence of oxidative damage may provide a valuable marker of the cumulative effect of multiple stressors in marine mammals.

Uses of single physiological markers have yielded strong but inconsistent links to individual and population fitness. For example, a meta-analysis (Bonier et al., 2009) found negative associations between glucocorticoid concentrations and fitness in 51% of published studies. Together, suites of physiological measures that include body condition, hematological and serum biochemical parameters, stress hormones, reproductive hormones, immune markers,

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[1] See http://rwcatalog.neaq.org.

and OS markers provide the most comprehensive measures of individual health. Changes in global gene expression in tissue samples may allow development of biomarkers that integrate these parameters.

Deep-diving marine mammals are exposed to high hydrostatic pressures and must support the metabolic costs of each dive using the oxygen they bring with them on the dive. If exposure to sound or other stressors changes dive behavior, this could have energetic costs and impose risks from effects of pressure. Marine mammals that dive to 500 m or more are exposed to hydrostatic pressures of 50 atmospheres (atm) or more. This would cause high-pressure nervous syndrome in most mammals tested and it is not known how marine mammals avoid this problem (Kooyman and Ponganis, 1998). More is known about how they avoid problems such as toxicity of oxygen at high pressures. When an air-breathing mammal fills its lungs at 1 atm of pressure and then dives, the volume of air reduces under pressure following Boyle's law. The parts of the lung where gas is exchanged with the blood are the most compliant, so they contract before stiffer tissues such as the bronchi and trachea (Fahlman et al., 2009). This limits the risk that breath-hold divers are exposed to P_{O_2} high enough to be toxic.

The shallower the depth at which diffusion stops because of alveolar collapse, the lower the P_{O_2} to which breath-hold divers are exposed. Estimating the depth of alveolar collapse is thus an important parameter for determining change in physiology that may be stimulated by exposure to sounds that affect dive behavior. Measurement of arterial P_{O_2} (McDonald and Ponganis, 2012) or arterial P_{N_2} (Falke et al., 1985) in free-diving pinnipeds has proven a powerful method to estimate depth of lung collapse. The P_{N_2} measurements were made possible by a portable blood sampling device that could be attached to freely diving seals.

The amount of oxygen available in the lungs is limited so that many marine mammal species store most of the oxygen they take on a dive in blood and muscle. The length of time a mammal can dive is limited by the oxygen available and tolerance of tissue for anaerobic metabolism, which can be detected by the presence of lactate in the blood. Thus, diving behavior represents a complex interaction of physiological adaptation and the requirements of foraging and social behaviors. Alterations in behavior in response to disturbance have the potential to create health impacts when they exceed the constraints imposed by physiology. The aerobic dive limit (ADL) has been defined as the dive duration after which there is an increase of lactate in the blood (Kooyman, 1985). Many studies have estimated the ADL by estimating the O_2 store and metabolic rate, but both of these may be modulated by dive behavior, and the estimate is sensitive to assumptions about how low a P_{O_2} an animal can tolerate. Meir et al. (2009) measured arterial and venous P_{O_2} in freely diving elephant seals and found they tolerate unusually low P_{O_2} in their tissues, allowing them to prolong their dives. More measurements of post-dive lactate would improve understanding of ADL, and more measurements of arterial and venous P_{O_2} would help to understand the physiological mechanisms affecting ADL.

Another important exposure-related change in physiology involves the regulation of N_2 and managing risk of decompression. Recent evidence that exposure to sonar can cause decompression sickness (DCS) in deep-diving whales has reinvigorated analysis of risk of DCS in marine mammals (Hooker et al., 2012). When a mammal dives with lungs full, as the hydrostatic pressure increases, N_2 diffuses into the blood and tissues, elevating their P_{N_2}. As the lungs collapse under pressure, this diffusion reduces and ceases. However, as the animal ascends, with reducing hydrostatic pressure, there is a decompression, with risk that bubbles may form if tissues or blood are supersaturated with respect to the ambient hydrostatic pressure. There is evidence that chronic exposure to small bubbles may damage the bones of deep-diving sperm whales (Moore and Early, 2004) and explosive DCS has been reported for beaked whales exposed to naval sonar (Fernández et al., 2005). Models of diving physiology have been used to predict risk of gas bubbles based on the dive profiles of tagged deep-diving marine mammals (Fahlman et al., 2014), and these models help us to understand how reactions to anthropogenic noise might disrupt the mechanisms used by these animals to manage gases under hydrostatic pressure, leading to risk of DCS. Marine mammals are breath-hold divers, so rapid ascent from a single dive poses a low risk of DCS. Furthermore, once an animal dives below the depth of alveolar collapse in the lungs, there is no gas exchange. Therefore, one risk factor for DCS is time spent above the depth of alveolar collapse, but deep enough for hydrostatic pressure to increase the nitrogen tension in tissues. Another risk factor for DCS involves long-duration dives at great depth, as these may cause redistribution of dissolved gases from tissues that take up and release gas quickly (e.g., muscle) to tissues that take up and release gas more slowly (e.g., adipose tissue) (Fahlman et al., 2014).

Behavior

The most comprehensive information on quantifying exposure-related changes in marine mammal behavior as a function of measured levels of exposure to a stressor come from studies of the behavioral responses of an increasing number of species to sounds produced by military sonars, or devices that mimic these sounds. Harris and Thomas (2015) have provided a review of these studies. Behavioral response studies are experiments designed to test the causal link between sound exposure and behavioral responses. One challenge for these studies with marine mammals is the difficulties in quantifying sound exposure at the animal and in obtaining continuous unbiased measures of behavioral responses. Johnson and Tyack (2003) describe a sound and movement recording tag that functions as an acoustic dosimeter and as a sensitive recorder of behavioral responses. These tags have been used in experiments that record baseline behavior, then record exposure and response to controlled playback of sonar and other sounds. Use of a dose escalation design makes it possible to estimate the lowest exposure that elicits each response. Statistical methods for identifying significant changes in behavior are described by Miller et al. (2012a). Miller et al. (2014) used this approach to define the probabilistic dose–response function illustrated in Figure 1a in Box 2.2.

One common response to anthropogenic sound is a marked reduction in marine mammal vocalizations. This may be the result of animals leaving the vicinity of the sound source or ceasing vocalization. Passive acoustic monitoring can be used to derive a relationship between received sound levels and this response. For example, Moretti et al. (2014) used data from an array of hydrophones on a Navy range to derive a relationship between acoustic detections of Blainville's beaked whales and calculated exposure level of sonar. Thompson et al. (2013b) deployed their own array of acoustic sensors to relate the detection rate of harbor porpoise clicks to distance from a seismic survey.

Controlled experiments and opportunistic monitoring of behavioral responses to anthropogenic noises can often complement one another. Controlled experiments can be critical for demonstrating that a sound causes a response, and for defining how animals respond to the sound. These results, which are often derived from a small sample of short-term experiments, can be used to design a monitoring scheme for the actual activities that produce the sounds. The Moretti et al. (2014) study showed responses to actual sonar exercises that were similar to those predicted from the experiments. Thompson et al. (2013b) were not only able to show the spatial scale of responses to seismic surveys, but were also able to demonstrate how that response reduced over the duration of the survey.

QUANTIFYING EXPOSURE-RELATED CHANGES IN INDIVIDUAL HEALTH

Measures of Body Condition That Are Useful for Assessing Health

Body condition is one of the few proxies for allostatic load that can be measured using conventional methods. Classic methods to measure energy stores involve separating skin, blubber, and other tissues, weighing them, and estimating their caloric values. Noninvasive measures such as ultrasound can also be used to measure blubber layers. The total amount of water in the body (total body water or TBW) can be estimated by diluting a known volume of isotopically labeled water, and total body lipid (TBL) can then be estimated by known relationships between TBW and TBL. Less specific morphometric measurements such as length, weight, and girth are also often used to estimate body condition. These measurements do not require dead animals, but they often require handling live animals. Biuw et al. (2003) used the dilution technique to validate a method for estimating body condition on tagged elephant seals while they were at sea. They used the rate of vertical change in depth of these animals while they were drifting passively through the water column to estimate their buoyancy. The lean tissue of marine mammals is denser than seawater, but lipid stores are less dense, so that the buoyancy of an animal is largely a function of the ratio of lean to lipid tissues (Crocker et al., 1997). Schick et al. (2013) used information of this kind to estimate variations in the health of individual elephant seals over time. These health estimates were then incorporated into the PCoD model developed by New et al. (2014). Monitoring buoyancy appears to be a useful method for quantifying changes in body condition in a number of species. For example, Gordine et al. (2015) describe a filtering method that can reliably detect buoyancy changes in the dive records of drift diving species using the highly summarized data that are normally collected by most of the tags fitted to marine mammals. Aoki et al. (2011) demonstrated that estimates of the body density of elephant seals fitted with tags that could record depth, swim speed, and temperature at 1 second intervals, and three-dimensional accelerations (for detecting pitch and hind flipper movements), were within 1% of the equivalent estimates from isotope dilution from the same individuals. In addition to these detailed studies of buoyancy, information on changes in body condition may be obtained from time series of aerial photogrammetry of the same individual collected using unmanned vehicles (e.g., Durban et al., 2015).

Measures of Organ Status That Are Useful for Assessing Health

Hematology and serum chemistry parameters are routinely used in human health care to assess physiological state and are generally organized into panels that represent specific pathological processes or organ systems. In circumstances where blood samples can be collected from marine mammals these measures can provide information on basic metabolic status, kidney function, inflammation, liver disease, or thyroid disorders.

Measures of Immune Status That Are Useful for Assessing Health

Wild populations and individuals are constantly challenged by pathogens. The immune responses to these pathogens influence the demographic parameters of populations (Daszak et al., 2000; Morens et al., 2004). Immune responses are energetically expensive, and the ability to mount them may be influenced by nutritional state, stress hormones, and toxics exposure (Hammond et al., 2005; Peck et al., 2016). The primary difficulty of assessing immune response is interpreting variation in markers without information on the exposure of individuals to pathogens. To date, studies on immune function in marine mammals suggest that they share all of the primary immune components identified in biomedical studies. However, it is likely that there are modifications to marine mammal immune function that serve to preserve response under the diverse environmental conditions experienced, including high pressure, cold temperatures, and extreme hypoxemia, conditions that are immunosuppressive in many human studies (Shepard and Shek, 1998; Brenner et al., 1999).

A variety of approaches have been developed to assess immune competency from cross-sectional samples. Functional immune assays have been developed for both pinniped and cetacean species that quantify the proliferative response of lymphocytes (e.g., Levin et al., 2005; Mori et al., 2006; Schwacke et al., 2012). Cytokines regulate the development of humoral and cellular immune responses. For species where blood or tissue sampling is feasible, a suite of markers are available to measure individual innate and adaptive immune responses, including circulating levels of cytokines, acute phase proteins, and immunoglobulins. Microarrays and RNA sequencing allow examination of cytokine expression in tissue. Multiplex cytokine arrays have been optimized for individual marine mammal species (Mancia et al., 2007; Vechhione et al., 2008; Eberle et al., 2013). DNA sequences for cytokines for many species have been published and can be used to develop quantitative assays (King et al., 1996; Inoue et al., 1999). Commercial assay antibodies have also been validated for use in numerous marine mammal species (e.g., Peck et al., 2016). Innate immune function can be assessed with serum from any species through simple complement killing assays, such as hemolytic complement (CH50) and bacteria killing assays. As measures of adaptive immune response, total immunoglobulin levels have been measured using species-specific and commercial antibodies (King et al., 1998; Peck et al., 2016), and pathogen-specific immunoglobulins have been measured to document exposure to a wide variety of diseases using direct agglutination assays, immunohistochemical staining, and commercial enzyme-linked immunosorbent assays. Together these measures represent a formidable arsenal of tools that could, in principle, be used to assess individual and population innate and adaptive immune function. However, collecting the appropriate samples for analysis will be challenging, particularly because large cross-sectional data sets on immune markers in populations are needed to differentiate robust and appropriate immune responses that occur as part of life-history variation from exaggerated or suppressed immune responses in individuals that indicate impaired health. The association between immunosuppression and increased infections is well documented in humans (Luebke et al., 2004), but the form of that relationship varies with life stage and the level of immune suppression. Given the well-documented exposure to pathogens and parasites in wild marine mammals, it is likely that immunosuppression will lead to an increase in rates of infection.

Measures of Stress That Are Useful for Assessing Health

One approach to measuring the cumulative physiological impact of multiple stressors on marine mammals is through the measurement of stress hormones. Physiological stress can be defined as a complex physiological response to aversive environmental stimuli that challenge fluctuating homeostatic set points. The mammalian neuroendocrine stress response is driven largely by activation of the HPA axis, which results in the release of glucocorticoids into circulation (Sapolsky et al., 2000). Glucocorticoids bind to tissue receptors and alter expression of genes affecting a diverse array of physiological processes, including metabolism. Meta-analysis has shown that anthropogenic disturbances are associated with elevation of glucocorticoids in wildlife regardless of the kind of disturbance (Dantzer et al., 2014), although the fitness impacts of these elevations are less clear. While acute stress responses are usually adaptive, and may even increase subsequent fitness through the process of hormesis (Boonstra, 2005), biomedical studies have suggested that chronic activation of stress responses can have negative effects on survival and reproduction, mainly through suppression of immune and gonad function. Thus, chronic activation of the HPA axis may be an important mechanism by which cumulative exposure to diverse stressors leads to physiological and demographic impacts. Chronic stress resulting from persistent or cumulative exposure to stressors may lead to dysregulation of the HPA axis. This dysregulation is thought to result from loss of negative feedback, when chronic elevation of glucocorticoids decreases the number of glucocorticoid receptors in areas of the brain that regulate activation of the response (Dickens et al., 2009).

Several conceptual models have been developed to represent the physiological impacts of chronic stress, including allostatic overload (McEwan and Wingfield, 2003) and homeostatic overload (Romero et al., 2009). Individuals undergoing chronic stress responses would be expected to exhibit higher baseline levels of circulating glucocorticoids, enhanced glucocorticoid responses to environmental stressors, and increased time for glucocorticoid levels to return to baseline following a stressor (Dickens and Romero, 2013). In biomedical studies, chronic elevation of glucocorticoids directly suppresses immune and gonad function (Sapolsky et al., 2000), although these relationships are less well established in wildlife species than in humans. Because the detrimental physiological effects of chronic stress are thought to result from a larger cumulative exposure to glucocorticoids and because conserved glucocorticoid stress responses can result from a wide variety of stressors, measurement of glucocorticoids represents a potentially important proxy for cumulative stress and health in marine mammals.

Unfortunately, measurement of the magnitude of stress responses and the status of negative feedback regulation is not possible for most marine mammal species, because it requires repetitive blood samples or experimental manipulations (adrenocorticotropic hormone or dexamethasone injection). Baseline (i.e., not altered by sampling) glucocorticoid concentrations can be measured in rapidly acquired blood samples, although this kind of sampling is not feasible for most species of marine mammals. For pinniped species that haul out on land, studies have suggested that chemical immobilization may ameliorate the stress response to handling, allowing measurement of baseline levels in some species (Champagne et al., 2012). Extensive work is under way to develop and validate techniques for measurement of glucocorticoids in other sample matrices that are appropriate for use in free-ranging cetaceans, including fecal samples, blow, blubber, and skin (reviewed by Hunt et al., 2013), sometimes called “integrated measures.” Measures from these matrices may be superior to blood samples in allowing identification of chronic elevation in baseline glucocorticoids. Fecal measures are the least invasive and may be more sensitive to anthropogenic disturbances (Dantzer et al., 2014) but are sometimes difficult to link to targeted individuals. Blubber samples acquired by biopsy dart have perhaps the greatest potential as a matrix for measurement of glucocorticoids in large whales. Highly fat-soluble glucocorticoid hormones dissolve in perfused blubber. Blubber samples can be targeted to specific individuals and taken prior to any alteration in glucocorticoids from sampling. In addition to measurement of glucocorticoids, blubber samples can also be analyzed for reproductive hormones, fatty acids, and contaminants, allowing increased understanding of potential integration among stressors. One key limitation in the current utility of measuring blubber glucocorticoids is understanding how blubber concentrations respond to acute and baseline changes in plasma (i.e., turnover and lag times). This issue can potentially be addressed through controlled experiments in tractable species that allow manipulation of cortisol levels and repetitive sampling. It is also important to understand how blubber cortisol levels may be influenced by important life-history events like fasting or reproduction. This need can be addressed through large sample size, cross-sectional, or longitudinal studies that measure glucocorticoids across multiple matrices. Finally, there is great potential for development of gene expression markers in marine mammal blubber that differentiate between acute and chronic elevation in glucocorticoids (Khudyakov et al., 2015).

Recent developments in the technologies available for long-term time series of stress and reproductive hormones, as well as potential exposure to contaminants, have the potential to provide unique insights into the historical variation in stress responses and reproduction. Earplugs from several species of large cetaceans provide time series of hormone and contaminant data over the lifetime of the individual, as long as 65 years in currently analyzed samples (Trumble et al., 2013). These profiles potentially reveal the timing of pregnancies and lactation, baseline stress hormones, and exposure to several important classes of contaminants. Similarly, baleen samples can provide individual time series of stress and reproductive hormones lasting up to 20-25 years (Hunt et al., 2014).

Interpretation of the potential relationship between glucocorticoid levels and individual fitness requires extensive contextual data. Currently there are few large cross-sectional data sets of stress hormones from marine mammals that can be used to quantify natural variation in glucocorticoids with age, gender, season, and/or reproductive status. However, such data are critical for assessing anthropogenic impacts on stress hormone levels and their potential for health and reproductive effects as well as for determining key periods where sampling is likely to be most informative about health. A primary research need is to collect glucocorticoid measurements across life-history stages in species of interest. These data will not only provide a basis for identifying unusual glucocorticoid levels in individuals or populations but will also enhance understanding of how natural variation in glucocorticoids may regulate the allocation of energy resources between immune response and reproduction, and how intrinsic factors might modify responses to anthropogenic stressors. For example, a large literature in seabirds has focused on the roles that natural variation in glucocorticoids plays in regulating breeding decisions (e.g., Kitaysky et al., 2007), carry-over effects between stress responses at various life-history stages (e.g., Schultner et al., 2014), and the interaction of glucocorticoid stress responses with exposure to toxins (e.g., Nordstad et al., 2012; Tartu et al., 2015). Currently, no parallel literature exists for marine mammals. Understanding the adaptive uses of stress responses in marine mammal systems is critical to assessing how cumulative stress impacts might integrate and when they are most likely to have demographic consequences.

6

Interactions Among Stressors and Challenges to Understanding Their Cumulative Effects

INTRODUCTION

The assessment of aggregate and cumulative effects from stressors (anthropogenic or natural) on any particular species or stock of marine mammal involves two fundamental elements: conceptualizing the process by which the potential stressors might influence the mammal population, and designing and implementing approaches to test specific hypotheses for relationships among stressors and demographic responses. Both of these needs present particular challenges in the case of marine mammals. Chapter 6 explores these challenges in further detail.

CONCEPTUALIZING PROCESS

Understanding the impacts of a potential stressor on any species in nature is always best served by first establishing a conceptual model that defines the pathways and processes by which that impact might occur. This general approach further involves defining the relationship between dosage of the stressor and response of the individual marine mammal, the population, or the associated ecosystem. Multiple potential stressors add to the challenge of understanding impacts. One commonly used approach to this difficulty that has been used in biomedical research involves estimating whether the impacts of two or more stressors occur via common pathways. Sharing common modes of action is thought to increase the likelihood of interaction (see Table 4.1). However, demonstrating or even predicting how the diverse set of stressors considered in this report may interact to influence marine mammals will be no mean feat. In this chapter the problem is treated in a manner that is broadly conceptual. The discussion begins by introducing the "interaction web" as a way of envisioning how the distribution and abundance of marine mammals will be influenced by stressors of any sort. Next is a discussion of functional relationships between stressor level and marine mammal response. In the third short section of this chapter, "ecological surprises" are introduced and discussed as the likely manifestation of what science does not yet understand about the way interaction webs are assembled and how they function. The section on ecological surprises is followed by an exploration of how the understanding of stressor–response relationships for marine mammals might be improved through a discussion of the principles of experimental design and scientific inference. The chapter concludes with a section on adaptive management: how best to use the insights derived from the various studies of marine mammals, stressors, and responses for their conservation and management.

THE INTERACTION WEB

Although various approaches have been taken to define the network of interactions among species and between species and their abiotic environments, in this report the idea of an *interaction web*, as defined by Dunne et al. (2002) is used. The older, more well-known, and more widely used notion of a *food web* (the network of trophic interactions among species [Pimm, 1979]) is embedded in the interaction web concept. The conception of the interaction web is based on a single broad premise—that the distribution and abundance of species in any ecosystem is dictated by interactions among species and between these species and their abiotic environment. In the case of food webs, abiotic factors are not considered, and species interactions are restricted to those involving consumers and their prey. The interaction web broadens the concept of interactions to include abiotic and biotic ecological drivers that have effects on populations that are broadly similar to the effects of stressors on individuals. Stated in the specific context of this report, a stressor stimulates the

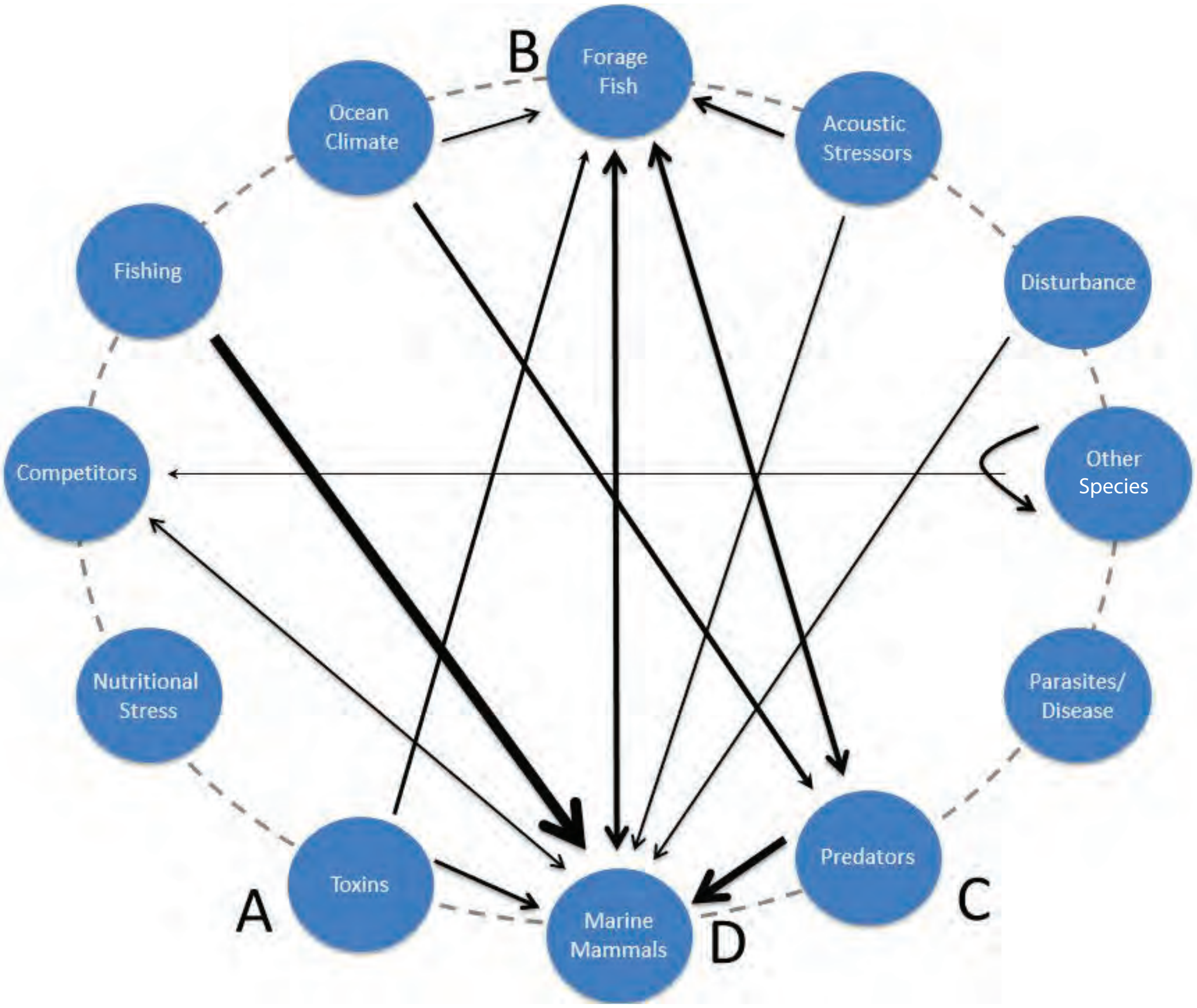

FIGURE 6.1 Schematic illustration of an interaction web. Circles around the perimeter of the dashed oval represent species or elements of the abiotic environment (collectively referred to as nodes), and arrows between circles represent species interactions or interactions between species and the abiotic environment. This particular schematic has been stylized to emphasize the nodes of interest and some of their imagined common stressors and interactions. Arrows represent directionality and line weight represents interaction strength. Note that only a few of the many nodes and their interactions are represented in this schematic. An example of a driver is A (Toxins) operating on B (Forage Fish), a recipient. Forage Fish can also operate as a driver on C (Predators) and vice versa (i.e., both serving as drivers and recipients). Finally, A (Toxins) can operate directly as a driver on D (Marine Mammals) and indirectly as a driver on D through the indirect pathway (A to B to D).

physiological response in an individual, and an ecological driver is a species or abiotic element of the environment that has an influence on a population. The key feature of ecological drivers is that they are biotic or abiotic features of the environment that affect individual animals indirectly by changing exposure to a whole suite of extrinsic stressors.

Interaction webs can be characterized in various ways. In this report it is done visually—as an oval with species and abiotic environmental elements arrayed around the perimeter (referred to subsequently as nodes) and *direct interactions* among species and/or elements of the abiotic environment (referred to subsequently as *linkages*) as the interconnecting lines (see Figure 6.1). The distribution and abundance of species in nature are largely dictated by these linkages, which are further defined by three properties: directionality, sign, and strength. For any two nodes A and B, A may influence B while B has little or no influence on A (in which case A is said to be the *driver* and B is said to be the *recipient*); or two nodes B and D may influence one another (in which case both B and D are drivers and recipients). Interactive effects might be positive (e.g., the influence of a prey species on its consumer) or negative (e.g., the influence of consumer on its prey). Anthropogenic stressors may be negative drivers, in the sense that at the levels occurring in nature they exert a negative influence on the distribution and/or abundance of a marine mammal species, population, or stock. In this context it is important to recognize that stressors at the individual level may have little or no influence, or in some cases even a positive influence, on the species or stock of interest. *Interaction strength*, defined as the magnitude of the direct effect of one node on another node, is visually characterized by line weight (see Figure 6.1).

Interaction web nodes can also affect one another via one or more intervening nodes, in which case their interplay is defined as an *indirect effect*. For example, node A might affect node D both directly and even more strongly through an indirect effect on node D via node B. Indirect effects are often imagined to be weaker than direct effects because the likelihood of a weak link occurring in the interaction chain increases with chain length, and the strength of any indirect effects will be limited by the weakest link in the chain. However, indirect effects can be as strong as or stronger than direct effects, and, in all but the simplest ecosystems, the number of potential indirect effects greatly exceeds the number of potential direct effects (Estes et al., 2013a). The net effects of anthropogenic drivers on marine mammal populations might thus be composed of either direct or indirect effects, or, most likely, both types of effects.

Interaction webs, by their fundamental nature, are exceedingly complex. Endeavors to quantify or otherwise analyze interaction web behavior have employed two broadly similar approaches, use of the community matrix (May, 1972; Yodzis, 1988) and network analysis (Proulx et al., 2005). Although these general methods of analysis will not be discussed further in this report, they may be used for further understanding the influence of anthropogenic stressors on marine mammals and their associated ecosystems.

Finding 6.1: Interaction webs characterize the numerous pathways in which all species within an ecosystem interact with one another and the various elements of their physical environment. This approach can be used to conceptualize the myriad ways extrinsic stressors may influence marine mammals.

Finding 6.2: Any two species may link together in the interaction web via direct or indirect interactions. Direct interactions are those in which there are no intervening species, whereas indirect interactions are those in which there are one or more intervening species. Indirect effects can link species with stressors via long interaction chains that may involve both bottom-up and top-down forcing processes.

RELATIONSHIP BETWEEN STRESSOR LEVEL AND INTERACTION WEB RESPONSE

The effects of a stressor on a population or ecosystem depend on the functional relationship between stressor level and an individual's response through changes in vital rates, the proportion of the population that is exposed to the stressor, and, for those exposed individuals, the level of exposure that each individual experiences.

A critical question here is: How sensitive are the predictions of population- and ecosystem-level effects from stressors to the form of the mathematical function that describes these relationships? If for example this function is linear (see Figure 6.2a), then some change in stressor level is predicted to lead to a constant proportional change in the system in which it acts, whatever the specific value of the stressor. Using this simple function, the magnitude of stressor impact can be estimated from the slope of the stressor–response function and the magnitude of change in the stressor, and even very low doses will have some effect. If the stressor has a point source, large numbers of individuals may be exposed to these very low stressor levels (see Box 2.2), and this could have important population-level effects. If, however, a sigmoidal function of the form shown in Figure 6.2b is assumed, very low doses are predicted to have little or no effect, and the population-level effects associated with the linear function would be ignored. In contrast, if the true function is in fact sigmoidal but linearity is assumed, unanticipated strong effects from small increases in stressor level may occur.

There are many reasons why a nonlinear function is more likely to be appropriate. Some of the more obvious reasons at both the individual and population levels are summarized below:

- For toxicants whose effect depends on binding with a receptor, the well-developed theoretical understanding of receptor-ligand kinetics predicts a nonlinear function.
- The physiological mechanisms that animals use to maintain homeostasis in the face of stressors often mean that adverse effects may not be visible until these systems break down, after which an adverse effect can suddenly appear. This nonlinear pattern can lead to sharp thresholds for effects.
- Any pattern of threshold variation (i.e., any particular density function) among individuals in response to a stressor within a population is likely to lead to a nonlinear cumulative distribution function.
- For a noise effects example, animals are not expected to respond to sounds at levels below their hearing threshold, and responsiveness may not increase above a certain high intensity of sound.

The preceding discussion is not meant to imply that these functional relationships must be understood before stressor effects can be documented. Such functional relationships will likely remain unknown in many cases. Even under this more limiting circumstance, stressor impacts might still be detected.

As explained further in Chapter 5, the Population Consequences of Acoustic Disturbance model (NRC, 2005) aimed

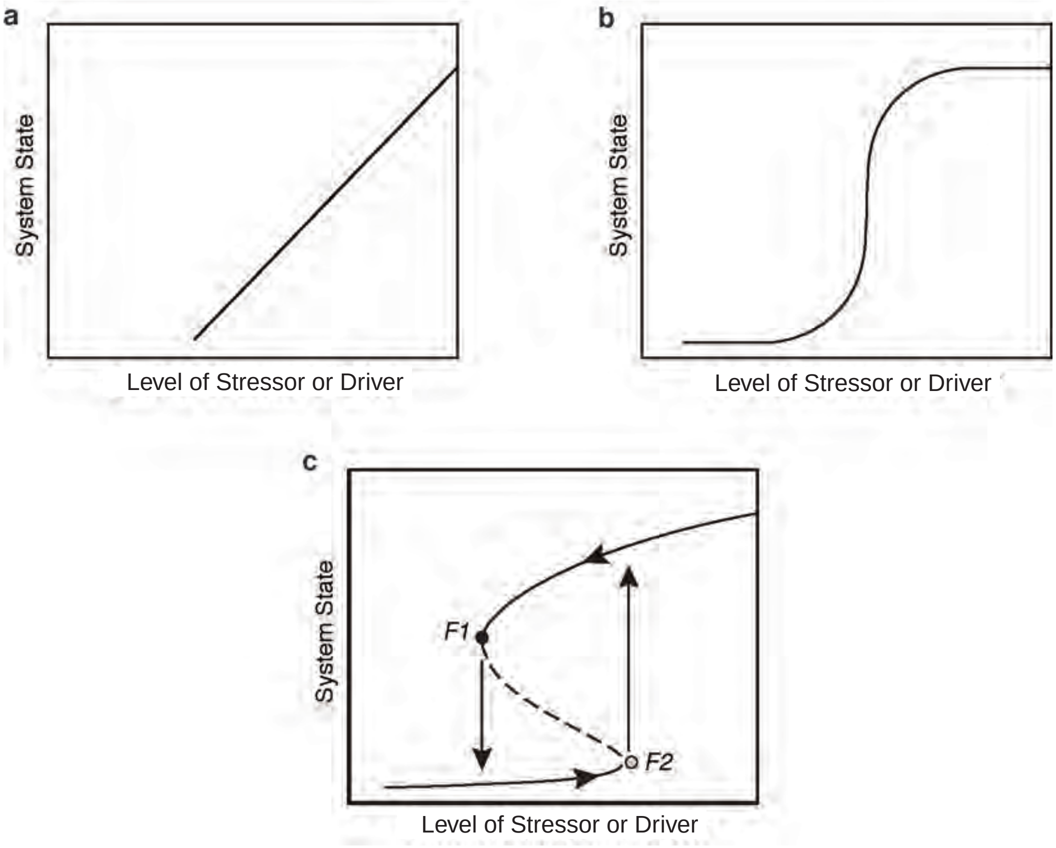

FIGURE 6.2 State-space graphs capture the functional relationships (all direct and indirect interactions) between a stressor and its effect on the state of a system. This relationship may be (a) linear or (b, c) nonlinear. The abrupt transitions depicted in (b) and (c) are often referred to as *phase shifts* or *regime shift*s. When the stressor or driver level at which a phase shift occurs is different when the stressor or driver level is increasing and when it is decreasing (c), the system is said to exhibit *hysteresis*. F1 and F2 are referred to as *tipping points* or *breakpoints*. Figures 6.2b and 6.2c adapted from Scheffer et al., 2009.

to break the causal chain from exposure to the stressor of noise to population effects into a series of sequential functional relationships. Chapter 5 describes recent applications of this model that use measures of body condition to integrate effects of stressors, from which the influences on reproduction and survival are predicted. There is evidence for nonlinear relationships between body condition, which integrates effects of many stressors, and reproduction, and this in turn varies among marine mammal species. Analysis of data from several species of pinnipeds showed that maternal state variables explained twice the variation in natality rates in capital breeders compared with income breeders (55% compared to 25%) and that the relationships between maternal state variables and pregnancy were distinctly nonlinear in capital breeders (Boyd, 2000). Thus, even if disturbance of feeding had a linear effect on body condition, the combined effect of disturbance on condition and then condition on pregnancy would be nonlinear, and the form of this function would likely vary between capital and income breeders.

Hunsicker et al. (2016) reviewed 736 relationships between driver levels and ecosystem responses in marine pelagic ecosystems. They report that nonlinear responses are more common than linear ones. Strongly nonlinear relationships were particularly common among climate and trophodynamic variables but also were associated with anthropogenic drivers, such as overfishing and pollution. The results of their meta-analysis of ecological studies led Hunsicker et al. (2016) to suggest that "in the absence of evidence for a linear relationship, it is safer to assume a relationship is non-linear."

The shape of the functional relationship between a stressor or driver and its effect on an individual, population, or ecosystem has significant implications for management. If managers can assume that gradual changes in intensity of the stressor or driver lead to roughly linear changes in recipients, as in Figure 6.2a, then they can aim to monitor the effects over time to make sure these effects are not becoming adverse. If the slope of this linear relationship is known at low driver levels, this relationship can be extrapolated to predict effects at higher driver levels. By contrast, if the functional relationship is as in Figures 6.2b and 6.2c, then no effect may be seen over a considerable range of driver levels, but beyond this range effects may escalate rapidly with only a small increase in the driver. Functional relationships of this nature lead to what are called *phase shifts* or *regime shifts* (Conversi et al., 2015), defined as abrupt and sometimes catastrophic responses by a system to small changes in driver intensity. The net effects of anthropogenic stressors on marine mammal populations and their associated ecosystems might thus be small and imperceptible until some critical level is reached, at which point the effect is strong. Selkoe et al. (2015) argue that this situation is common enough that resource managers should, "[i]n the absence of evidence to the contrary, assume nonlinearity."

In some situations, the functional relationship between the level of a stressor or driver and the state of a system may vary depending on the directionality of change in stressor or driver level (see Figure 6.2c). This phenomenon is called hysteresis. For example, an individual marine mammal that has been exposed to a sound may habituate or become sensitized, changing its responsiveness to later exposures. Similarly, the initial response of an individual to increasing numbers of a pathogen following infection will differ from the response as the body reduces the number of pathogens. In this case, the state of the organism has changed from when the infection starts to when its immune system is causing the infection to decrease. At the population level, if abundance is reduced to a very low level by a driver, the population may not recover following driver relaxation because of such factors as demographic stochasticity or inverse density dependence (Allee effect; Stephens et al., 1999). For populations governed by the generalized logistic growth equation, the rate of decline following overshoot beyond carrying capacity will be more rapid than the rate of recovery from a similarly sized reduction in abundance below carrying capacity (Gotelli, 2008, p. 30). In multispecies systems (i.e., biological communities), a driver-induced reduction in one species might alter species interactions such that the driver relaxation is not followed by a similar pattern of recovery. A critical point about hysteresis for this report is that managers should not assume the response of a system will follow the same path when the level of a stressor is reduced as it did on increase of the stressor.

Ecosystems can shift among different basins of attraction (Scheffer et al., 2001)—different configurations to the distribution and abundance of species, in which movement from one basin to another requires a strong perturbation. This situation can be likened to the behavior of a ball over a three-dimensional surface of ridges and valleys, in which the valleys are basins of attraction and the ridges are tipping points (also known as breakpoints). Perturbations (changes in driver level) that are sufficient to push the ball over a ridge and into another valley result in regime shifts. The consequences of this process for the functional relationship between driver level and system state is illustrated graphically by Figure 6.2c. When driver level changes from just below F2 to just above F2 (a tipping point), the system jumps from one state to another (a regime shift). Once a regime shift has occurred, driver level must be reduced to below F1 for the system to return to the initial state. These breakpoints or tipping points can be thought of as unstable equilibria between alternative stable states (May, 1976). The first explorations of ecological tipping points and regime shifts were based on theoretical analyses (e.g., Lewontin, 1969; May, 1976). A large and growing body of empirical study confirms the existence of these state shifts and regime shifts in nature (Sutherland, 1974; Scheffer, 2009), including the shift from coral-dominated systems to macroalgae-dominated systems in the Caribbean (Hughes, 1994; Knowlton, 2004), changes in fishery yield (Steele, 2004; Vert-pre et al., 2013), shifts between kelp forests and sea urchin barrens (Steneck et al., 2002); and changes at larger system-wide scales (Beaugrand, 2004; Hare and Mantua, 2000; Möllman et al., 2009). Empirical evidence for hysteresis, although more limited, does exist (see Figure 6.3).

The general situation in which the state or condition of an individual, population, or ecosystem is largely unresponsive over one range of stressor or driver levels but responds strongly at other levels presents a substantial challenge to management. Under this circumstance, managers must know the range of stressor levels over which the desired state is maintained, thereby allowing them to set a threshold below which the risk of transition to the adverse state is suitably low. The actual forms of the functional relationship between stressor levels and their effects on marine mammal physiological systems, individual condition and life-history metrics, or the distribution and abundance of populations are largely undocumented. To the extent possible, the choice of such functional relationships should be based on data and/or theory, not on scientific preconceptions.

ECOLOGICAL SURPRISES

The preceding sections of this chapter establish two key points: (1) that interaction webs are highly complex structural entities, given the great diversity of species and the even greater diversity of ways these species can interact with one another and their physical environment, and (2) that functional relationships among species and between species and their physical environments are commonly nonlinear. Given these two key points, the responses of natural systems to stressors are expected to be difficult to predict and thus often characterized by what have been referred to as ecological surprises. In a paper based on analyses of various case studies and a survey of established field ecologists, Doak et al. (2008) concluded that major surprises (defined as "a substantial change in the abundance of one or more species resulting from a previously unknown or unanticipated process of any kind," p. 593) should be expected in any effort to understand and predict ecological dynamics (Peetchey et

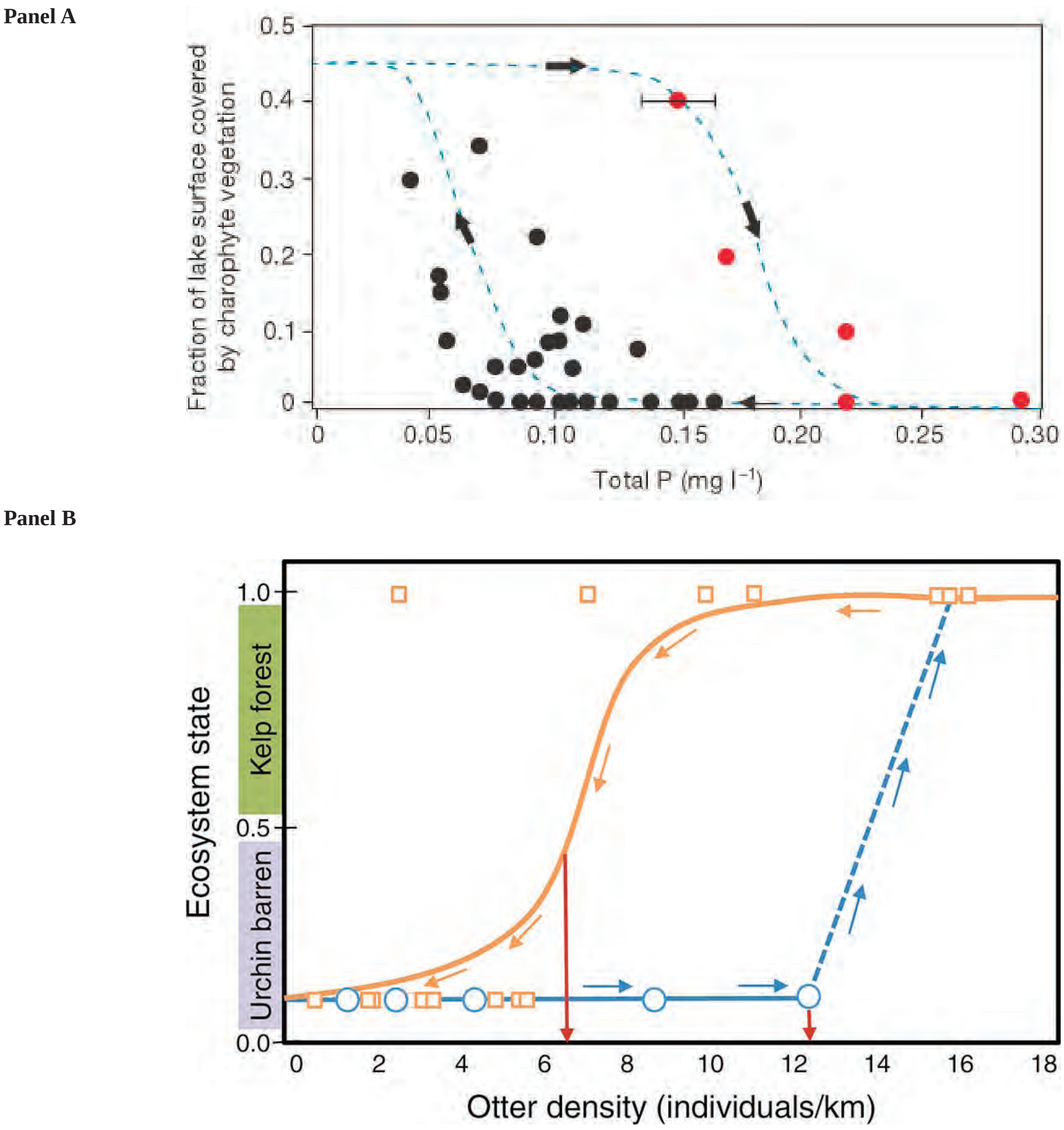

FIGURE 6.3 Two empirical examples of hysteresis: Panel A shows the differing response of charophyte vegetation in a shallow European lake to an increase (red dots) followed by a decline (black dots) in phosphorous concentration. SOURCE: Scheffer et al. (2001). Panel B demonstrates the differing sea otter densities (red arrows) required to precipitate a phase shift between kelp- and urchin-dominated phase states, depending on whether the otter population was growing or declining. SOURCE: Selkoe et al. (2015).

al., 2015). Key attributes of ecological surprises (Doak et al., 2008) include the following:

- Surprises are both dramatic and widespread in scientific studies of all kinds.
- Ecological surprises are especially common and underreported.
    - Ninety percent of well-established field ecologists who responded to a questionnaire in which they were asked if they had ever been surprised (as defined above) answered in the affirmative.
    - Eighty-eight percent of those who responded in the affirmative believed that they understood the reasons for having been surprised after the

    fact, thus suggesting that the causes were easy to understand but previously unanticipated.
  - Many of these examples remained unpublished because the individual investigators thought they were either uninteresting (scientifically) or unpublishable.
- Efforts to improve predictability and quantify uncertainty in ecological models are unlikely to reduce the frequency of ecological surprises because these modeling efforts necessarily are built around things that are known as opposed to things that are unknown.
- Sooner or later, most natural resource management strategies will not work as planned, thus reinforcing the need for management plans that are precautionary.

In keeping with this general view of nature, studies of marine mammals have resulted in numerous surprises. For example, while most populations and species of great whales recovered following protection from exploitation during the whaling era, some (like southern blue whales) have not recovered for reasons that remain unknown (Branch et al., 2007). In Chapter 4, several case studies of population decline were explored where it has been difficult to infer causes, including beluga whales in Cook Inlet, Alaska, pinnipeds and sea otters in the Northern Pacific and Southern Bering Sea, and harbor seals in the United Kingdom. Other examples of surprises involving marine mammals could be described and cited. However, the committee is not aware of any cases where these surprises were subsequently attributed to cumulative impacts or the interaction among multiple stressors. This does not imply that such cumulative or interactive effects are unimportant in causing ecological surprises, but rather that they are not well understood.

To reiterate, the basic reasons for these various surprises are (a) insufficient understanding of interaction web structure, especially with regard to the various important pathways that lead from potentially diverse drivers to marine mammals; (b) complex functional relationships in the interactions among species and between species and the abiotic environment; and (c) overly simplistic views of interaction web structure and process.

Finding 6.3: The functional relationships between interacting species are often nonlinear and characterized by hysteresis. These complex functional relationships, coupled with immensely complex interaction web topologies, often result in unanticipated outcomes, sometimes referred to as ecological surprises.

DESIGNING APPROACHES TO UNDERSTANDING STRESSOR IMPACTS AND THE PRINCIPLES OF SCIENTIFIC INFERENCE

Empirically based scientific inquiry in ecology involves two main elements: a search for pattern (which is commonly based on one's view of interaction web structure and dynamics, as discussed above), and distinguishing between causation and correlation. Empirically based patterns nearly always derive from observation of variation in space or time. These two elements of scientific inquiry are in turn often challenged by two essential inadequacies: (1) inherent difficulties in observing patterns associated with purported or hypothesized causal agents (in the context of this report, stressors and drivers) and (2) the inability to distinguish between causation and correlation with a high level of confidence. These shortcomings are best overcome through the experimental method, wherein the influence of some purported causal agent or agents (e.g., anthropogenic stressors or drivers) is assessed by observing differences between experimental units (e.g., behavior or physiological parameters in the case of stressors; individuals or populations in the case of driver effects on the distribution and abundance of species) that have been treated with the purported causal agent (i.e., by adding or removing the imagined stressor or driver) and those that have not (controls).

The three basic principles of experimental design are *randomization*, *replication*, and *local control*, which exist because experimental units always contain some level of intrinsic variation, independent of that which might be caused by their experimental treatments. For example, no two individuals are exactly the same. One needs to be able to detect and measure experimental treatment effects through this intrinsic variation in experimental units. Randomization (the random matching of experimental treatments to experimental units) is done in order to ensure that intrinsic variation among the experimental units is as likely as possible to be spread evenly between treatments. Replication provides a measure of experimental error, defined as the difference among identically treated experimental units, and causes the average value of the intrinsic variation among identically treated experimental units to converge on zero with increased replicate number. Local control is accomplished by choosing and arranging the experimental units and then assigning treatments to these experimental units so as to reduce experimental error.

Scientific experiments that are conducted in accordance with these design principles have three important properties. First, they minimize the likelihood of mistaking correlation for causation. Second, they provide an inferential template for the assessment of multiple agents of causality and the interactions among these agents. Third, they often permit increased inferential efficiency through the processes of blocking, stratification, and the analysis of covariance, all of which help reduce experimental error. These broad principles are discussed and explained in greater detail in any introductory text on experimental design (e.g., Fisher, 1937; Montgomery, 1997).

As observed in Chapter 4, the predominant approach to studying interactions between stressors uses experiments with a simple factorial design. Although this approach is both powerful and broadly applicable, it has drawbacks and limitations for answering the many questions about nature that scientists have been unable to address experimentally. This is the current state of affairs for the committee's charge in this report, which is to evaluate the cumulative influences of anthropogenic stressors on marine mammals. As noted in Chapter 3, the lack of strong evidence for an influence of fisheries on marine mammals through competition for prey or other indirect interaction web effects is due to the failure to be able to assess these effects experimentally. Instead, the conclusions are more often based on observations of individuals and populations of marine mammals between otherwise similar areas with and without fisheries effects. Other approaches have been used in an effort to make these assessments (most commonly correlative analyses or inferences based on modeling approaches), but in many cases the signal is weak, and in most cases the distinction between causation and correlation is equivocal. For example, despite the great biomass of fish removed from the North Pacific Ocean/southern Bering Sea ground fisheries, it has proven both difficult and contentious to establish whether or not these potential prey removals have contributed to the declines of fur seals, harbor seals, Steller sea lions, and sea otters in southwest Alaska (NRC, 2003b). Moreover, pinniped populations in the northwestern Atlantic Ocean have generally increased, despite the collapsed ground fisheries (Estes et al., 2013b). Similar obstacles apply in the assessment of noise on marine mammals, although in this latter case experimental or quasi-experimental approaches are less problematic because noise is more manageably controlled than fisheries in space and time. However, the assessment of noise effects in combination with other potential stressors on marine mammals is exceedingly challenging because not only is it difficult or impossible to experimentally assess most singular (main) effects, doing so in sufficiently orthogonal combinations to be able to sort out the interactive effects is vastly more challenging. This is the fundamental nature of the problem at hand.

Understanding the influence of anthropogenic or natural stressors on marine mammals can only be rigorously assessed through observations of the manner in which individuals and populations respond to changed intensities of these stressors in their surrounding environments. Such information can be obtained in two general ways—through purposeful experimentation and through correlative studies from regions in which data from marine mammals are available in areas where the purported or hypothesized stressor has also varied. The strength of the experimental method is that, when properly done, the likelihood of misinterpreting results because of potentially confounding factors is eliminated or greatly diminished. As explained previously, the difficulty with experimental approaches for marine mammals is that they are difficult or even impossible to implement at appropriate scales of space and time for a host of fairly obvious reasons, including logistical limitations and legal, social, and economic constraints. Many of the experimental approaches that have been implemented lack sufficient samples to have the necessary statistical power or precision to detect effects. With proper planning, correlative studies are easier to conduct, but these are also usually plagued with uncertainties over whether the purported or hypothesized stressor is the cause of any marine mammal response in the face of other potential confounding variables. This fundamental limitation to correlative analyses will be greatly magnified in efforts to assess the potential influences of multiple stressors or the aggregate influences of single stressors on marine mammals.

The strength of inferences from nonexperimental information can often be improved through various analytical approaches. One of these is a weight-of-evidence analysis in which the array of relevant information is contrasted against the expectations of alternative competing hypotheses. Using this approach, it is sometimes possible to determine the most likely of two or more alternative hypotheses, or to exclude one or more of these hypotheses based on internal inconsistencies with available data. More recently, Sugihara et al. (2012) proposed a general method for distinguishing causality from correlation based on nonlinear state-space reconstruction of time-series data.

Finding 6.4: Controlled experiments are the most rigorous way of testing for the influences of potential stressors on any species. For marine mammals, such experimental approaches are often not possible, in which case inferences must be based on quasi-experiments. Although quasi-experimental data are subject to confounding and thus multiple interpretations, reasonably strong inferences are often possible from time-series analyses and weight-of-evidence approaches.

ADAPTIVE MANAGEMENT

As described above, classical factorial experiments are impractical as a vehicle for evaluating potential cumulative influences of stressors on marine mammal populations, while observational (correlative) studies are more practical to undertake but are likely to result in ambiguous inferences. Despite this, regulators must make decisions on whether and where to allow potentially harmful anthropogenic activities to take place. The concept of adaptive (resource) management offers a framework for making such decisions in the situation where there is some scientific understanding of the link between management action and outcome, and where repeated decisions must be made over time (such as issuing annual permits for activities, or setting harvest limits). Key texts describing the concept include Walters (1986) and Williams (2011a, 2011b). A brief overview is provided here.

Adaptive management involves first setting a conservation objective and then formulating multiple hypotheses about the population response to the different management options, together with an assessment of the probability of each hypothesis being correct. The optimal decision is determined (see later for how "optimal" is defined), and this action taken. The population response is monitored, and the new information gained is used to update the probabilities for each hypothesis, whereupon the process is repeated. A key concept is that "we learn more about the system as we go along" and hence can adapt management decisions in the light of the improved information. There are broadly two approaches of adaptive management, depending on how "optimal" is defined: in passive adaptive management, the optimal decision is the one most likely to bring scientists closest to the conservation objective given the current state of knowledge; in active adaptive management, determining the optimal decision also involves accounting for the learning that is anticipated to occur as a result of each possible decision. (See Williams [2011b] for a more nuanced discussion of the various closely related definitions that have been used.) Hence, in active adaptive management, it is sometimes considered optimal to take management decisions that result in moving away from the conservation objective in the short term if this means one learns more about the biological system and so can make better conservation decisions in the future. Classical experiments may be contemplated, where different management actions are assigned at random to spatially replicated regions (if possible). Active adaptive management is therefore riskier, in that it relies more on having an accurate assessment of the consequences of selected actions (in terms of how much each possible action will help us distinguish between the multiple alternative hypotheses).

Although adaptive management ideas are much discussed, they are relatively little used in practice. A recent literature review by Westgate et al. (2013) identified 1,336 articles published between 1978 and 2011 using the term "adaptive management." Of these only 61 (<5%) explicitly claimed to enact the methods, and only 13 projects were found that the review authors felt met the criteria for actually using adaptive management. There are multiple possible reasons for this lack of usage. First, the method requires the formulation of multiple competing hypotheses, typically expressed as alternative quantitative conceptual models of the system, and it may be that there is simply not enough knowledge about most systems to do this adequately. Second, the realistic rate of learning may be too slow to be useful. This may be because there is strong natural variability (e.g., from ecological drivers such as El Niño in the Pacific or the North Atlantic Oscillation) that nearly masks any signal coming from alternative management actions; because possible management options do not generate a strong signal (e.g., if they can only be applied to a small component of the population); because any signal may take a long time to be manifest (as will be the case for long-lived, slow-reproducing animals like most marine mammals); because standard experimental practices like replication and blocking are not possible; or because the monitoring of outcomes that are feasible is too imprecise to be useful. Third, although adaptive management is designed to cope with uncertainty about which hypothesis is correct, and with observation error in the outcome measurements (both "known unknowns" [Logan, 2009]), it is not robust to the kinds of ecological surprises that were discussed earlier in this chapter (the "unknown unknowns"); hence, focusing only on measuring the best metrics for distinguishing between alternative hypotheses risks missing other important conservation issues. The topic of monitoring is explored in the next chapter. Finally, implementing adaptive management is complex, typically requiring a team with skills in theoretical ecology, applied conservation, statistics and modeling, and, potentially, social sciences if the human aspect of management decisions is to be considered. Resources and commitment over the long term are required, and these are rarely available.

Despite these issues, there does not appear to be a superior alternative to adaptive management as a rational and structured system for making optimal conservation decisions. Trial and error, or "reactive management" (Sutherland, 2006), is clearly inferior. For this reason, the application of adaptive management principles to the management of cumulative effects is encouraged wherever this is possible.

Recommendation 6.1: Adaptive management should be used to identify which combinations of stressors pose risks to marine mammal populations, and to select which stressors to reduce once a risk is identified. In this approach, hypotheses are developed which guide management actions and data collection to assess the strength and impact of individual stressors and their cumulative effects.

CONCLUSIONS

In addition to direct mortality from entanglement in fishing gear, ship strikes, and purposeful killing, marine mammals are exposed to a broad range of potential anthropogenic stressors, including but not necessarily limited to noise, prey depletion by fisheries, disease, pollutants and toxins, and a broad (but still largely unknown) array of indirect effects of these various stressors on the associated ecosystems. In particular cases, each of these direct effects is known or suspected to have negative impacts on marine mammal individuals and populations. A separate literature from experimental studies (see Chapter 4) has demonstrated the cumulative or synergistic influences of stressors on a wide range of aquatic plant and animal species. Therefore, cumulative influences of anthropogenic stressors on marine mammals are nearly a certainty.

The challenge is in conceiving of and especially then demonstrating these effects on marine mammals. The important outstanding questions are these: For which particular stressors under what specific conditions and for which marine mammal species will cumulative effects occur, and what are the functions that relate stressor dosage to the linked effect? Answering these questions in a scientifically rigorous manner is beset by three significant challenges. The first challenge is to properly characterize a topology of influence by stressors on marine mammals. Simple direct effects of singular stressors on marine mammals are relatively easy to imagine, but the potential influences of multiple stressors, acting through both direct and indirect interaction web pathways, will be substantially more difficult. The second key challenge will be in designing studies in which the interactive influences of multiple stressors on marine mammals can be evaluated. Experimental designs that are capable of demonstrating interactive effects while controlling for confounding influences are nearly impossible to carry out without purposely manipulating the purported drivers in an orthogonal manner. A final challenge is in the detection of any real impact from stressors on a marine mammal stock at the individual and especially the population level. Rigorous demonstration of population change has proven to be exceedingly difficult for most marine mammal species. Thus, even when the process by which multiple stressors might influence a marine mammal is well conceived and a study can be properly designed to put the resulting hypothesis to a test, the ability to document an effect on the marine mammal species, population, or stock of interest will often be limiting.

7

Early Warning Signs of Risk to Populations

INTRODUCTION

The previous chapters have attempted to establish that scientists may anticipate the nature of some interacting effects, but in most situations they are not currently able to forecast the cumulative effects of all stressors with any accuracy. Therefore, there is a pressing need for early detection of unexpected population declines and, where possible, rapid diagnosis of the main factors contributing to them. This requires some form of population monitoring. The parameters monitored must be informative about the status of the population; it is also helpful if they are informative about the contributing factors for any decline in status, although that could become part of a secondary, more intensive, data-gathering effort that is instigated if the first stage of monitoring indicates a problem. (An alternative view is given in the following paragraph.) Detecting a deleterious situation involves testing for long-term declines in status over time (trend analysis; see, e.g., Thomas et al., 2004), or a recent sudden drop (sequential surveillance; see, e.g., Anderson and Thompson, 2004; Frisén, 2009). Alternatively a comparison could be made with reference to populations thought to be in good status, although such comparisons need to consider natural variability. The parameters monitored must also be measured with sufficient accuracy and precision that there is a good chance a deleterious change of magnitude large enough to cause concern will be detected (i.e., good statistical power, if a statistical hypothesis test is the detection mechanism).

The above approach has been criticized as being inefficient and ineffective by Nichols and Williams (2006), who refer to it as "surveillance monitoring." They argue that a focus on detecting declines, often using statistical hypothesis testing, is unlikely to lead to optimal conservation decisions and introduces unnecessary time lags, and that identifying the causes of declines is less important than identifying the most effective remedy (although recognizing the cause can often help identify possible solutions). Instead, they advocate embedding monitoring within a larger framework of conservation-oriented science or management, where monitoring is used to enable discrimination between multiple competing hypotheses about the biological system being monitored and hence facilitate better management decisions. Monitoring therefore becomes an integral part of an adaptive management framework, as defined in the previous chapter. This also implies that monitoring programs will change what is measured as the scientific hypotheses under consideration are updated—a paradigm called "adaptive monitoring" by Lindenmayer and Likens (2009).

The committee believes that there is merit in both of these frameworks. Adaptive management, and hence adaptive monitoring, potentially can be effective in situations where there is enough knowledge of the system to formulate working hypotheses about the link between each potential management action and the outcome, to evaluate the *a priori* probability of each hypothesis, and where learning through focused monitoring will be useful. However, there are at least two reasons not to rely exclusively on such adaptive monitoring. First, there are many cases where the above criteria will not be met and adaptive management will not be helpful. Second, as described in Chapter 6, there is a strong potential for "ecological surprises," for example, unexpected declines in species that had not previously been considered to be of conservation concern. Hence, a dual approach is advocated, where the principles of adaptive management and adaptive monitoring are applied where possible, but where, in addition, a "light touch" surveillance program is undertaken in order that very large changes in conservation status of species are not missed until it is too late to do anything about them. It is recognized that such a surveillance program will have low power, but its aim is to detect only large changes in status. The chance of detecting a change in status will be improved if a sensitive indicator can be found that is also relatively inexpensive to monitor.

The committee has previously recommended the use of adaptive management (Recommendation 6.1) to focus data collection and guide management actions. The following recommendation concerns a "light touch" surveillance program.

Recommendation 7.1: Responsible agencies should develop relatively inexpensive surveillance systems that can provide early detection of major changes in population status and health. Surveillance systems should be developed first for populations that currently lack adequate stock assessments.

In the following sections, the population parameters that might best be measured in either of the above frameworks are discussed. One form of ecological surprise described earlier is that of an ecological tipping point. In the last section, suggestions from the literature on the early detection of a species or system approaching a tipping point are described.

MONITORING POPULATION SIZE

Population size is the most basic measure of population state. However, for most marine mammal species, monitoring total population size (or density) over time or space is not a sensitive way to obtain early warning of problems (for surveillance monitoring) or distinguish between different possible management actions (for adaptive monitoring). One issue is that it is often difficult to define what constitutes a biologically appropriate unit of assessment because many local populations are not genetically or demographically isolated. Another is that most marine mammal species are long lived and slow to reproduce, so any negative impact that causes reproductive failure or juvenile mortality, or any beneficial management action, will take a very long time to cause a significant population trend. However, the main issue is that population (or stock) size is a parameter that is notoriously difficult to measure precisely, particularly for marine mammals that often range over a large area and are invisible when underwater. Visual methods requiring human observers remain the most commonly used for marine mammals, particularly cetaceans—either shipboard or aerial line transect surveys or photographic capture–recapture (Buckland and York, 2009). For colonial pinnipeds, colony counts are sometimes used, with a correction factor (derived from animal-borne tags) for those at sea (Buckland and York, 2009); for some pinnipeds such as grey seals, pup production at breeding colonies is estimated and a population dynamics model is used to scale up to total population size (e.g., Thomas et al., 2005). For animals that are widely dispersed, it tends to be the spatial variation that causes low precision; for rare or hard-to-see animals it is the low sample size; for colony counts it is estimating the scaling factor. The result is that the ability to detect all but the most drastic population trends is often limited. For example, Taylor et al. (2007) reviewed the precision of abundance estimates for 127 stocks under U.S. management and concluded that, overall, 70% were not precise enough to detect a precipitous decline of 50% over 15 years of monitoring. Jewell et al. (2012) examined the utility of combining results from multiple abundance surveys worldwide: for the best-fitting model, the smallest population decline detectable with high (>0.8) power was more than 50% for 5 out of the 11 taxonomic and geographic groupings used.

Despite this pessimistic message, more precise monitoring is possible for some stocks, particularly those that live in restricted areas relatively close to shore (e.g., southern resident killer whales) or all pass close to shore at some point in their life cycle (e.g., gray whales). New technology may also play a part in enabling more precise population estimation—for example, potentially replacing visual surveys with remote aerial vehicle surveys using high-definition cameras or video recorders (Buckland et al., 2012) or passive acoustic surveys from fixed or floating sensors, or remote underwater vehicles (Marques et al., 2013). Many of these techniques are still under active development; for passive acoustic methods a critical limitation is knowledge of the acoustic biology of the target species required to convert call density into animal density and abundance. New statistical methods that make better use of existing or emerging data streams also offer the potential for better precision—for example, the recent ability to extend capture–recapture analysis to utilize information about the location of the captures (Borchers, 2012; Royle et al., 2013; Pirotta et al., 2015c). Taylor et al. (2007) discuss some other potential routes to increased precision. However, it is important to emphasize that, at the current time, estimation of population size remains a very imprecise science for almost all marine mammal stocks.

One possibility sometimes suggested for obtaining more precise estimates of population status is to measure indices of population size, such as uncalibrated acoustic detections and sightings from shore-watch schemes or from platforms of opportunity. However, straightforward interpretation of changes in the index as changes in population numbers requires that the relationship between the two is linear and has constant variance over the range of both indices, or that the shape of the relationship and variance is known (Williams et al., 2001, Section 12.7). In practice, the relationship is rarely linear (indeed it may not even be monotonic) or with constant variance. Nevertheless, carefully chosen indices may still be effective as early warning metrics, for example, if they are sensitive to changes in population size or disturbance for the species of interest and are relatively inexpensive to deploy at the population scale. Passive acoustic detections may be a good candidate in this regard, in that large amounts of data can be collected at moderate expense (for vocal species); however, its efficacy has yet to be demonstrated.

In determining the cause of population declines, it is often insightful to focus on the components of the population likely to be affected first. This is discussed in the next section.

MONITORING DEMOGRAPHIC PARAMETERS

Population dynamics are governed by four fundamental demographic parameters: survival, fecundity, immigration, and emigration. One or more of these must decline (or increase in the case of emigration) for population declines to occur. Hence, measuring these parameters may make for a more sensitive monitoring system than waiting for a detectable change in population size. However, it is typically infeasible to monitor all of these parameters with good precision, so one will typically need to prioritize. To do so, one needs to consider which of these parameters is expected to be most strongly affected by cumulative impacts of stressors, the influence changes in these parameters have on population size, and the feasibility of accurately measuring the parameter.

Many marine mammals are relatively long lived and reproduce infrequently but over multiple occasions. Under these circumstances, ecological theory leads us to predict that reproductive-age adult females should evolve strategies that enable them to delay breeding or abandon investment in young when conditions are harsh in order to prioritize their own survival and hence maximize their future reproductive output when conditions may be better. Therefore, there is an expectation that adult female survival will remain high and relatively constant in fluctuating environments, while fecundity and calf or pup survival should fluctuate with the conditions. A similar phenomenon occurs as populations approach carrying capacity and, based partly on empirical observations, Eberhardt (2002 and references therein) proposed the following sequence of changes as conditions worsen:

- increase in mortality rate of immatures
- increase in age of first reproduction
- reduction in reproductive rate of adult females
- increase in mortality rate of adults

The committee's opinion is that there is no strong theoretical reason to suggest that pup or calf mortality should always increase before fecundity-related parameters decrease; this may depend on the cost of pregnancy and gestation, and whether the species is adapted to uncertainty in the ability to provision young. For species where these costs are low, and that are adapted to uncertain provisioning conditions, adult females may tend to continue to produce pups or calves but then not be able to successfully rear them. Hence, from an early warning perspective, fecundity (including age at first breeding) and calf or pup survival are all parameters to target.

To determine influence on population size, it is useful to consider the findings of matrix population modeling (Caswell, 2001), in particular from sensitivity analysis, which quantifies how much population growth will be affected by identically sized changes in each demographic parameter in the model. Exact results depend on the model, but in general, population growth is most sensitive to changes in adult survival, with changes of the same magnitude in fecundity and pup or calf survival having much less effect (Eberhardt, 2002).

Putting these last two threads together it is expected that birth rates and/or pup or calf survival are likely to be first affected by cumulative stressors, but that they will have the least effect on population growth rate. This provides a strong justification for monitoring these parameters as part of an early warning system, where they may show a strong signal of population stress before the population trajectory is strongly affected. However, it is important to recognize that natural population processes such as density dependence will also result in low birth rates and/or with pup or calf survival, and hence measurements need to be put into the context of natural population dynamics. Also, as stated earlier, these demographic parameters are expected to show the highest levels of natural variation, so picking out a declining trend among strong interannual variation may be difficult.

The last consideration is the feasibility of accurately monitoring the parameters. Many demographic parameters can be estimated from an intensive capture–recapture survey; typically for marine mammals this involves photographic identification, although genetic identification from biopsies or fecal samples (or even potentially blow samples) is possible. Each of these methods is labor intensive, and only feasible in situations where animals are accessible and a reasonable recapture rate is likely. In planning a study, the expected precision can readily be evaluated using a straightforward simulation approach (Devineau et al., 2006).

Age-specific mortality can also be derived from analysis of age structure of a population, assuming a stable age structure (as in when the population is growing exponentially, or has reached carrying capacity); this is the basis of life-table analysis. One example of this is Moore and Read (2008), who used the age structure of harbor porpoise deaths from all mortality sources and the age structure of deaths from fisheries bycatch to estimate the effect of bycatch on vital rates and the likelihood of population decline. The use of strandings is, however, problematic due to the length of time required to obtain a sufficient number of carcasses for age structure analysis, and the fact that it can only be used on inshore populations in areas where stranded carcasses are reported and can be investigated. For this reason it cannot be recommended as a general monitoring method.

Fecundity (or at least pregnancy) can also potentially be estimated from hormone analysis (e.g., Kellar et al., 2006;

Hunt et al., 2014) and from looking at pregnancy rates (and possibly pregnancy history) of stranded or sampled animals. However, high pregnancy rates alone may not mean good population status: if calf or pup survival is low then females do not need to devote energy to provisioning their young and hence may recover and breed again more quickly—thus elevating pregnancy rates. Hence pup or calf survival should also be measured.

Overall, although birth rates and pup or calf survival seem at first glance to be the best parameters to monitor for early warnings, it will be important to undertake some form of power or precision analysis to determine whether a signal of the expected magnitude can be detected given expected levels of interannual variation and measurement error.

Another generally applicable approach is to focus on indices of demography that can readily be measured in the field. One prominent example is the ratio of adults to juveniles in a sightings survey (or, relatedly, the proportion of mother–calf pairs in populations where this is an appropriate metric). Calves or pups are typically readily distinguishable from adults; it may also be possible to distinguish juveniles and record similar metrics on them. In conclusion, collection and analysis of stage-structured population data may provide a useful early warning of poor population status.

MONITORING POPULATION HEALTH

Chapter 5 provided a definition of individual health, as well as reviewing some of the various indices used to assess individual health. However, it is important to distinguish between assessing the health of an individual versus assessing the health of a population, the latter being focused on the measurement of the distribution of health outcomes in a population or a subset of a population, *as well as* the determinants or factors that influence those outcomes (Ryser-Degiorgis, 2013). The term “health outcomes” is used rather than the more narrow term “health status” because the latter refers to health at a single point in time rather than over a period of months or even years that it may take for a disease to develop (and demographic consequences to become manifest) (Kindig and Stoddart, 2003). As a field of research, population health focuses on multiple potential contributing factors for health outcomes; it considers the complex interactions among factors, the biological mechanisms underlying a given health outcome, and the influence of different factors over time and throughout an organism’s life cycle (Kindig and Stoddart, 2003; Ryser-Degiorgis, 2013). In this respect, population health studies not only address the detection of changes in health outcomes, but also simultaneously address the potential causal factors.

The concept of population health involves different criteria from population status. The National Marine Fisheries Service (NMFS) assesses the status of a marine mammal population or “stock” by assessing its range, minimum population estimate, current population trends and productivity rates, human-caused mortality, and other factors that may cause a decline or impede recovery (NMFS, 2004). Populations that are large and near carrying capacity will usually have a good population status but could have a lower level of population health. A population that is at or nearing carrying capacity may exhibit a high prevalence of disease (e.g., malnutrition or infectious disease), and the population’s size in relation to its expected carrying capacity should be considered as a potential driver when poor population health is observed. In this context, population health (i.e., the distribution of health outcomes in a population or a subset of a population) may produce a false-positive indication of population decline. While this chance of false positives for populations for which status is completely unknown decreases specificity, population health will in most cases provide greater sensitivity and is a more tractable approach as compared to monitoring population status, which requires precise estimation of population size and current productivity rate in relation to an expected productivity rate. Carrying capacity is generally not known and is difficult to estimate. However, the objective of monitoring as outlined in this chapter is early detection of population declines. If poor population health is observed, continued monitoring over time would allow the hypothesis of carrying capacity being the underlying driver to be confirmed or rejected.

Population health monitoring can take two primary forms: passive health surveillance (also referred to as scanning surveillance) and targeted health surveillance. Passive health surveillance focuses on in-depth investigation of disease incidence and for wild marine mammals is generally conducted using carcasses or tissues collected from stranded animals. In the United States, under the 1992 Amendments to the Marine Mammal Protection Act, the Marine Mammal Health and Stranding Response Program (MMHSRP) was formalized to coordinate efforts to investigate marine mammal strandings.[1] The intent of the program is to improve the knowledge of rates and causes of mortality and morbidity to gain a better understanding of population threats and stressors, and to detect emerging or unusual events. Since 1991, 62 marine mammal unusual mortality events (UMEs) have been recognized in the United States,[2] and in those where causes have been attributed (only 56%), these have included biological toxins, infections, human interactions, oil spills, and changes in oceanographic conditions (Gulland and Hall, 2007). An additional important component of the MMHSRP is biomonitoring, i.e., sampling, archiving, and analysis of tissues to allow for examination of geographic and temporal patterns in exposure to chemical contaminants, biological toxins, and/or pathogens (e.g., Fire et al., 2009; Twiner et al., 2012; Simeone et al., 2015). A real-time, nationally centralized system for reporting marine mammal health data has been proposed (Simeone et al., 2015) and would

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[1] See http://www.nmfs.noaa.gov/pr/health/MMHSRP.html.

[2] See http://www.nmfs.noaa.gov/pr/health/mmume/events.html.

greatly facilitate the conduct of epidemiological analyses to more rapidly detect and identify contributing factors for UMEs, as well as to explore more subtle changes in population health over space and/or time in relation to one or more stressors. Standardization of databases for marine mammal health within and across nations could facilitate more global analyses. However, with the exception of nearshore species, the utility of passive surveillance for marine mammal populations will still be limited due to the extremely low probability of recovering carcasses (Williams et al., 2011; Barbieri et al., 2013; Carretta et al., 2015).

Recommendation 7.2: A real-time, nationally centralized system for reporting marine mammal health data should be established.

In contrast, *targeted health surveillance* is carried out proactively, focusing on live animals that in some cases are apparently healthy, and relying primarily on cross-sectional study designs that require only a single sampling occasion (Ryser-Degiorgis, 2013). Targeted health surveillance in the form of capture–release health assessment has been successfully conducted for a number of species along the U.S. coast (e.g., Wells et al., 2004; Aguirre et al., 2007; Greig et al., 2010). Physical examination, diagnostic ultrasound, and blood sampling for hematology, serum biochemistry, and hormone analysis can be conducted and synthesized to determine the prevalence of specific disease conditions (Schwacke et al., 2014a), and serology (to determine antibody prevalence) can help to evaluate prior pathogen exposure, or lack thereof, assisting in the development of management plans (M. Barbieri, personal communication). Portable auditory evoked potential systems also allow for hearing tests (Finneran and Houser, 2007) to be performed, which are particularly relevant for understanding hearing loss among various populations. Unfortunately, capture–release studies can only be conducted on relatively small, tractable marine mammal species, and to date have focused on the nearshore where individuals can be temporarily caught and restrained on land (e.g., seals and polar bears; Stirling et al., 1989; Polischuk et al., 2001) or in shallow waters (e.g., small delphinids, and manatees; Bonde et al., 2012). However, methods could and should be developed to extend such sampling to other coastal, continental shelf, and/or oceanic species, although an extension of these types of approaches to large cetaceans will be complicated by the logistical challenges of capturing and restraining them. Nevertheless, remote sampling techniques are rapidly advancing and can be applied to large cetaceans. Hunt et al. (2013) review currently available techniques for obtaining physiological information on large whales that include remote collection of respiratory ("blow") samples, skin/blubber samples, and fecal samples. Perhaps most promising is the collection of blow, as techniques for analysis of metabolites, hormones, and pathogens have been demonstrated using cetacean respiratory samples (Acevedo-Whitehouse et al., 2009; Hunt et al., 2013; Aksenov et al., 2014; Cumeras et al., 2014), and recent developments in human breath analysis indicate promise for eventually obtaining a broad array of physiologically relevant indicators of health (reviewed by Hunt et al., 2013). However, collection methods are still being refined and will require extensive validation as well as collection of baseline samples to understand the inherent variability for the suite of measures across species, life-history stages, and varying environmental conditions. Likewise, "-omics" approaches (primarily proteomics and transcriptomics) are being pursued using sampling matrices that can be remotely collected (blow, skin/blubber; reviewed by Hunt et al., 2013), but characterization of expression profiles is still in its infancy, and identifying patterns that provide meaningful information on health state is complicated by lack of information on cetacean genomes (Hunt et al., 2013), variation among life-history stages, genetic stock, and varying environmental conditions (e.g., Van Dolah et al., 2015), and the fact that some remotely collected samples (i.e., skin/blubber) simply may not be appropriate matrices for detecting expressional changes associated with many health conditions.

Targeted surveillance could also be supported through photographic studies. Photographic monitoring has been used to identify emerging zoonotic disease (Rotstein et al., 2009) and support epidemiological investigations of skin disease in both terrestrial (e.g., Oleaga et al., 2011) and marine mammals (e.g., Hart et al., 2012; Van Bressem et al., 2015). Visual health assessment based on body and skin condition, and the presence of cyamids and rake marks, has been applied for right whales (*Eubaleana glacialis*), and an index of health based on these criteria has been developed that is predictive of survival and reproduction (Schick et al., 2013). In addition, Fearnbach et al. (2015) have applied photogrammetry to assess body condition based on proportional head width in endangered Southern Resident killer whales (*Orcinus orca*). Furthermore, recent development of techniques to obtain photographs using unmanned aircraft systems (Durban et al., 2015) will greatly facilitate photographic monitoring to measure body condition and/or assess parasites, skin disease, or other externally visible indicators of compromised health.

These novel health assessment methods are primarily designed to be applied to individuals, but because population health emerges from the health status of a population's members, appropriate sampling at the individual level can lead to inferences about population status. In this vein, body condition, as measured by a visual health assessment or photogrammetry (see above paragraph), could represent a first-pass metric for overall population health. Sampling would need to include a sufficiently large number of animals to assess the health of groups critical to population growth, such as a large cross-sectional sample of adult females across a variety of life-history stages or of juveniles. A broad measure of health, such as body condition, would not necessarily be sensitive to quick changes because fat reserves may not be affected until the late stage of a disease; however, because most pathways of declining health eventually affect body condition, it could capture the consequences of a variety of potential stressors.

One important caveat here, just as with measuring demographic parameters, is that care needs to be taken not to misinterpret poor health caused by natural demographic processes, such as reaching carrying capacity, with poor health that is of concern; in other words, measurements need to be put in the context of expectation given the population status.

EARLY WARNING OF TIPPING POINTS

As described in Chapter 6, the existence of multiple stable states and tipping points in natural ecosystems is now beyond reasonable doubt. However, the real challenge for managers and scientists alike is the ability to anticipate and predict regime shifts, especially as the impacts of anthropogenic stressors and drivers on ecosystem function and processes appear to be increasing. The potential for predicting regime shifts in marine environments and their management depends on the characteristics of the regime shifts: their drivers, scale, and potential for management action.

Recent theoretical findings (Drake and Griffen, 2010; Dai et al., 2012; Dakos et al., 2015) suggest that ecosystems tend to recover more slowly from small perturbations if they are in the vicinity of tipping points. This phenomenon is referred to as “critical slowing down,” and its temporal and spatial indicators may under some conditions provide early warning signals of a system approaching a tipping point where it could easily pass through a critical transition into an alternate state (Dakos et al., 2015). However, applying these theoretical insights to the management of marine mammal populations is limited by a lack of critical ecological data in many species: without these data it is challenging to characterize baseline variability in populations and resources well enough to detect changes that might indicate a potential tipping point. There is also the important consideration that many population parameters for marine mammals are measured with such low precision that detecting any signal among the noise may be nearly impossible.

Levin and Möllmann (2015) argue that “accounting for marine regime shifts in management clearly requires integrative, cross-sectoral ecosystem-based management (EBM) approaches.” EBM is widely used for ocean management worldwide and is well suited for dealing with regime shifts, as it considers the multiple interacting drivers and ecosystem linkages that generate ecosystem shifts. They make a case for the use of Integrated Ecosystem Assessment (IEA) (Levin et al., 2009), an EBM framework used by a number of management agencies in the United States.[3] IEAs are becoming more common, but they are still new enough in their development to allow the inclusion of regime shift concepts in an emerging EBM framework. IEAs could provide a transparent means of characterizing the status of ecosystem components, “prioritizing potential risks and evaluating alternative management strategies against a backdrop of actual environmental conditions.” To be useful, IEAs will need to identify ecosystem attributes and anthropogenic stressors; “develop and test indicators and reference levels that reflect key ecosystem attributes and the drivers; explore the susceptibility of an indicator to natural or human threats as well as the ability of the indicator to return to its previous state after being perturbed; evaluate the potential different management strategies to influence the status of key ecosystem components and the pressures that affect these ecosystem components”; and consider the precision with which the indicator can be measured, relative to the expected strength of the signal generated.

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[3] See http://www.noaa.gov/iea.

8

Approaches to Assess Cumulative Impacts

INTRODUCTION

The previous chapters of this report have reviewed a variety of "approaches to assess cumulative effects of multiple stressors on marine mammal populations that, in turn, have direct and indirect effects on vital rates and population health" as stipulated in the statement of task (see Chapter 1). There are very few situations where one can link exposure to stressors directly to effects on marine mammal populations. Several approaches are discussed, beginning with those of limited use for marine mammals and then moving on to those with greater utility for this task.

APPROACHES WITH LIMITED APPLICATION FOR EVALUATING CUMULATIVE EFFECTS IN MARINE MAMMALS

Factorial Experiments

The primary experimental method used to evaluate cumulative effects of stressors involves factorial experiments that manipulate two or more stressors in animals that can be held in controlled settings. As discussed in Chapter 4, many stressors are likely to interact, and their effects should only be assumed to be additive if there are sound biological (as opposed to purely statistical) reasons for this assumption. The committee's review of meta-analyses of these experiments concluded that there are no obvious generalities that could help us to predict the effects of interactions between stressors on marine mammals in the wild. There are so many stressors affecting marine mammals and the ecosystems upon which they depend that the traditional approach of starting with impacts of individual stressors and then studying interactions when small sets of stressors are added together is not practical. Halpern et al. (2007) found that all of the marine ecosystems they surveyed were threatened by at least nine stressors, leading to hundreds of potential interactions that would need to be studied. This is not practical for marine mammals.

Alternative Model Species

The difficulties of studying cumulative effects in protected, large, long-lived animals such as marine mammals has led some to argue for consideration of other easier-to-study taxa as surrogate model species (Caro and O'Doherty, 1999). However, as Chapter 3 discusses, terrestrial mammals may differ enough in responses to stressors that they may not be good model systems for marine mammals. For example, investigations in pinnipeds have shown that increased oxidative stress during fasting and diving is ameliorated by oxidant-induced hermetic responses that increase antioxidant capacity more than would be predicted using studies from terrestrial mammals (reviewed by Vázquez-Medina et al., 2012). There also are serious questions about extrapolating information about interactions between marine stressors from nonmammalian marine model species to apply to marine mammals. As homeotherms, the response of marine mammals to temperature is very different from that of animals whose temperature matches the ambient. As animals that breathe air, marine mammals are much less sensitive to water-borne compounds than animals that extract oxygen from water. In this report the committee urges caution when extrapolating from non–marine mammal species in assessing cumulative effects of stressors on marine mammals.

Laboratory Studies

There are significant logistical and ethical problems with experiments that intentionally expose marine mammals in the laboratory to stressors such as pathogens. However, studies have been conducted on stressors such as sound, toxins, and temperature. Chapter 2 reviews studies on effects of sound on marine mammals. De Swart et al. (1996) and Ross et al. (1996b) fed harbor seals with herring from either relatively uncontaminated areas of the Atlantic Ocean or from the contaminated Baltic Sea. Baltic herring was immunotoxic to the seals, potentially reducing their resistance and increasing risk from infectious diseases. Yeates and Houser (2008) determined how low the temperature of air or water had to go before the metabolic rate of their bottlenose dolphin subjects became elevated. Water temperature had a stronger effect than air temperature, and little synergy was observed between the two. These studies of physiological responses to stressors illustrate that laboratory studies can demonstrate causal relationships between stressors and effects.

There may be further scope for laboratory research on effects of stressors on marine mammals, but there is a major advantage for research on wild animals. Marine mammals are exposed to such broad and poorly quantified arrays of stressors that it would be difficult to attempt to reproduce these combinations of stressors in the laboratory. By contrast, if one wants to study the effect of adding one stressor, such as sound, to a population influenced by many stressors, then one can select subjects from the wild population that are exposed to the current combination of stressors. Exposure to intrinsic stressors will vary with life history, and exposure to extrinsic stressors will vary in time and space. If the goal is to study animals whose allostatic load is high, this suggests selecting times when both intrinsic and extrinsic stressors lead to the energy demand exceeding supply (McEwan and Wingfield, 2003). This goal suggests an alternative to fully sampling the range of exposures in the wild. However, studies that involve adding one stressor to a wide sample of subjects in the wild actually do evaluate the cumulative effects of all the stressors to which the subjects are exposed. One cannot count on the same being true for studies of animals that are maintained in laboratory environments where animals are well fed and free from predation and many other stressors. These considerations suggest that wild marine mammals may be more appropriate subjects for studies of cumulative effects than captive animals.

SAMPLING STRATEGIES THAT DEPEND ON RANGING PATTERNS

The opportunities and obstacles for making critical measurements depend on the ranging patterns of the species under study. There are four main patterns for marine mammals that are relevant for sampling strategies for assessing cumulative effects of stressors in marine mammals.

Accessible Resident Populations

Species with home ranges that are small and near shore can be studied in a cost-effective manner by biologists using small vessels to sight individuals that can be identified by markings. These kinds of studies have proven valuable for tracking birth, growth, and death of nearly every individual in a population (e.g., Brault and Caswell, 1993). The overall exposure of the population can be measured on a seasonal or annual basis for a range of stressors based on environmental sampling. Comprehensive health assessments also are able to measure the dosage of individuals for some stressors, along with data on responses to stressors. These studies have been conducted with several populations of bottlenose dolphins that live in coastal waters of the southeastern United States, providing demographic data that can be compared across sites. Comprehensive health assessments involving suites of biomedical sampling (Wells et al., 2004) have also taken place at several of these sites, providing critical data for evaluating the dosage and effects of stressors that impact only one or a few of the sites. For example, Schwacke et al. (2014b) compared results from dolphins oiled after the *Deepwater Horizon* event to those from a population in Sarasota Bay, Florida, far from the oiling, and Venn-Watson et al. (2015) compared oiled dolphins to those that had stranded in other areas. For populations with limited home ranges, these concurrent studies in several populations provide a powerful tool for studying effects of stressors whose exposure varies across the locations.

Some species associated with deep oceanic areas have small enough home ranges for observational methods to provide important longitudinal data in areas where deep water is close to shore. For example, some beaked whale species are thought to have limited home ranges near seamounts or undersea canyons. Claridge (2013) was able to obtain important life-history data from populations of Blainville's beaked whale (*Mesoplodon densirostris*) in Bahamian waters. Similar data have been obtained for pilot whales in the Strait of Gibraltar where a small population of pilot whales resides (Verborgh et al., 2009). These situations may give a biased view, however. For example, pilot whales in most other study sites range so widely that there are relatively low rates of resighting individuals in one location.

Species with Predictable Locations for Birth on Land

Pinnipeds that come ashore in between foraging trips at sea and that give birth on land offer special opportunities for study. Long-term studies of identified individuals in this case can more easily involve sampling, weighing, and tagging than studies for species where animals do not come ashore. The foraging trips may take days to months—durations that are well within the scope of established tag attachments. Some of these species are suitable for the analysis of body condition through measuring buoyancy during drift dives. New et al.

(2014) showed how data on weight and survival of mothers and pups could be coupled with tag data measuring how foraging affects body condition. These data can be incorporated into the kind of model developed in Chapter 5 to relate how variation in stressors leads to variation in reproduction and calf survival. The main obstacles to studying interactions between stressors in these species involve development of more studies of identified individuals, and development of ways to measure exposure and response to stressors. These species are among the most promising for development of studies using the model from Chapter 5.

Species That Are Accessible at Some Points Within Large Home Ranges or During Annual Migrations

Some migratory species of cetacean congregate near shore for enough of their annual cycle to be studied by shore-based researchers. When accessible, these populations can be studied by observing individual animals that have distinctive marks. For species with several such sites, comparing sightings can allow movements to be tracked, but this is biased by the observation sites and is likely to lead to an incomplete view of the population range. For example, the population of right whales in the Northwest Atlantic is well studied from sightings during the summer foraging season, enough to estimate risk of extinction (Caswell et al., 1999). A subset of the population migrates to coastal waters off the southeastern United States, but little is known about where the other segment winters. Similarly, long-term observations of a small population of killer whales that are routinely sighted in Puget Sound, Washington, has provided solid evidence of a decline, enough to list the population as endangered (Ford, 2013). However, this population ranges as far as California during the winter, and little is known about their exposure or response to stressors during this part of the year. In these cases, focused tagging efforts may be needed to supplement local field studies. Obtaining measurements and attaching tags to these animals will be more challenging than working with animals that haul out on land. In addition many of these migrations occur on an annual basis, requiring longer tag attachment times than for most species that give birth on land, to cover the time at sea away from the nearshore site. Many species that have large home ranges or migrate annually have been tagged with satellite tags, but this is expensive, so the sample size is low. Few tags are available with longevity sufficient to cover an entire migration period, but the success rate and length of attachment duration are increasing as the technology evolves (Mate et al., 2007).

Open Ocean Species

Species that are widely distributed in the open ocean are the most challenging for studies of cumulative effects. It is difficult to develop longitudinal studies that involve resighting individuals over such large areas, and it is more difficult to sample or tag animals on the high seas than on land or in shallow coastal waters. Some solutions have been developed for these problems. Remote tagging and biopsy methods have been developed, but these are more limited than those available onshore or where one can handle the animals. Further development of sampling and tag attachments will be required to apply the approaches recommended in this report for open ocean species. Researchers studying the stress to pelagic dolphins of encirclement in tuna nets used the encirclement itself to enable handling, sampling, and tagging dolphins in a floating restraint system (Scott and Chivers, 2009), but this is unlikely to be possible for larger whales. Smith et al. (1999) report on a systematic and standardized effort to photo-identify and biopsy sample humpback whales throughout the North Atlantic. Similar scales of effort would likely be required for sampling exposure and response to stressors for populations of marine mammals that span ocean basin scales. The methods recommended in this report for studying cumulative effects will need considerable development to be applicable for these species.

Combining the difficulty of studying these four groups of marine mammals with the vulnerability of their populations suggests a broad set of priorities. The marine mammal species most at risk of extinction over the past few decades have not been the migratory large whale species, but rather populations of river dolphins, such as the baiji or Chinese river dolphin (*Lipotes vexillifer*) (Turvey et al., 2007). A range of anthropogenic stressors have been implicated in the decline and extinction of the baiji, with physical injury as a result of interactions with fishing gear being the most important. The limited home ranges of the resident species make them more vulnerable to localized concentrations of stressors. By contrast, the harder-to-study migratory and open ocean large whale species may be less vulnerable. Even though most of these species were exploited during the era of commercial whaling, some populations are large and/or recovering (Whitehead, 2002; Thomas et al., 2016), and the scale of their distribution and movements may render them less vulnerable to local exposure to stressors. This combination of difficulty of study and lower vulnerability may lower the priority for this group for studies of cumulative effects. However, some migratory baleen whale populations, such as the right whales of the western North Atlantic, are exposed to many stressors and have a small and declining population (Kraus and Rolland, 2007). Their coastal distribution puts them at higher risk and makes them easier to study, promoting their priority.

APPROACHES TO ASSESS COMPONENTS OF THE PCOMS FRAMEWORK

Chapter 5 presented a framework for analyzing cumulative effects of stressors on marine mammals. Here we describe approaches to assess cumulative effects organized by the different components of this framework. This section focuses on methods to estimate critical parameters in the context of studying relationships between exposure to stressors and (1) behavioral or physiological responses, (2) health, or (3) vital rates.

Measuring Exposure to Stressors

Lioy and Rappaport (2011) identified two different ways by which biomedical researchers could estimate exposure to chemical stressors that influence human health: a geographical approach and a subject-oriented approach. The geographical approach focuses on different external sources of exposure to a contaminant, which must be summed up to estimate aggregate exposure. Identifying external sources can help prioritize ways to reduce exposure. However, it can involve massive effort and can miss internal sources of chemical stressors, which may be very important for health (Rappaport, 2011). A subject-oriented approach samples directly from the subjects to measure contaminants or their biomarkers. This subject-oriented approach suggests the utility of sampling blood or other tissues in order to estimate the dosage of stressors at the animal to evaluate their impact on health and vital rates (Rappaport, 2011). Placing the sampler on the subject frees the study from needing to track the changing location of the subject, and to associate exposure with time spent in each location. The pros and cons of geographical and subject-oriented approaches to measuring stressors in marine mammals are similar to those identified by Rappaport (2011) for humans.

Spatial and Temporal Distribution of Stressors in the Environment

The geographical approach to identify potential risks from the complex combination of stressors in the world's oceans requires mapping the distribution of the species of concern along with mapping stressors in space and time. An assumption of this geographical approach is that stressors must overlap with the species to exert a cumulative effect. For example, risk of physical injury from fishing or shipping can be estimated by the flux of categories of ships or the density of fishing gear that pose different threats of injury (e.g., fast versus slow ships, gillnets versus other nets). Similarly if predators, competitors, or anthropogenic sources need to be relatively nearby to be perceived as a threat, then data on the distribution of these stressors may provide a useful estimate of exposure. However, mapping noise from acoustic stressors cannot always be derived from information about the location of intense sources alone. Underwater sound can propagate so well that the same sound produced in the Indian Ocean can be detected off California and off Bermuda but at different levels (Munk et al., 1994). The best way to estimate exposure to one or several intense acoustic stressors is to combine acoustic propagation modeling with measurements of levels of sound produced at known ranges and of the transmission loss in the environment. Acoustic propagation models can use source and transmission loss data to predict the sound field around these sources and to guide selection of recording sites to best ground-truth predictions. In cases where sources cannot be so readily identified or measured, ambient noise can be monitored directly. Increasing numbers of acoustic observing systems are coming online globally (Miksis-Olds and Nichols, 2016), providing useful data on integrated exposure to noise from all acoustic stressors.

Similarly, the risks from biological or nonbiological toxins cannot always be derived simply from mapping occurrence of sources of toxins or concentrations in the environment. The processes by which toxins are released, transported, and distributed from sources through environmental media and potentially through the food web to marine mammals are complex and will depend on a number of variables related to the toxin, the habitat, and the species of marine mammal. In some cases, it is possible to examine environmental samples from water, sediment, or prey to predict exposure for marine mammals, but, for toxins that can be detected directly in marine mammal tissues or fluids, direct collection and measurement in marine mammal samples is a preferred approach for characterizing dosage. As discussed in Chapter 3, persistent organic pollutants (POPs), many inorganic contaminants, and harmful algal bloom toxins have been routinely measured from a variety of remotely collected tissue samples. Metabolomic analyses of respiratory samples and proteomic and transcriptomic analysis of tissue samples hold promise for the development of biomarkers that indicate cumulative dosages of many toxins. Respiratory samples also hold promise for detection of markers indicative of pathogenic infections. Similar to toxins, exposure to pathogens can often be better characterized by direct sampling of the animal as the presence of a pathogen in the environment does not necessarily translate to an exposure risk. The actual exposure the animal experiences will depend on a variety of factors, including the presence of transmission vectors, or social structure and aggregation (e.g., colonial breeding) that affect contact rates with infected conspecifics. However, while direct measurement from actual tissues from marine mammals is a preferred approach to measure dosage for toxins, this approach requires extensive sampling effort and analyses that are often very costly. In this regard, it would be beneficial for researchers from multiple disciplines and agencies to collaborate and leverage efforts across projects to collect and analyze samples, building a baseline of data that allows examination of geographic trends for multiple stressors.

Prey limitation is a key factor influencing body condition and, as Chapter 6 emphasizes, is a critical part of the interaction web for marine mammals. Marine mammals are well adapted to use sensory cues from echolocation, vibrissae, and more standard mammalian senses to detect, select, and capture prey. Human methods using ship-based echosounders and nets to map prey are crude by comparison and cannot yield a complete view of availability of preferred prey for marine mammals. However, Friedlaender et al. (2016) have shown that inclusion of prey density and distribution can explain variation in dive behavior of foraging blue whales in a way that greatly increases the power to detect responses to other stressors, such as anthropogenic sound. Further development of methods to measure prey fields may improve these estimates. However, there are considerable obstacles to measuring prey fields in a way that accurately estimates prey limitation for marine mammals. Well-funded long-term censuses of commercially important fish have not solved the challenge of mapping their distribution, even for informing the management of those commercial stocks. There are very few stock assessments of species that are important prey for marine mammals but not important for human fisheries. In addition, measuring prey fields may not provide a complete estimate for the stressor of prey limitation. For example, if prey change their behavior or localized distribution so they are less accessible, then a foraging marine mammal may experience prey limitation even when the prey are present in the area. Here also, the specifics of how, when, and where marine mammals forage may be needed to assess the level of stress from prey limitation. Exposure to prey limitation as a stressor may be estimated by such measures of prey availabilty, although such data are often limited and difficult to interpret for generalist predators. All of these considerations emphasize the importance of developing measures of foraging success of individual marine mammals over time.

Predation pressure is a stressor that can be an important driver, but measurement of predation risk is difficult for marine mammals. Two important predators of marine mammals are sharks, such as great white sharks (*Carcharodon carcharias*) and the killer whale (*Orcinus orca*) (Jefferson et al., 1991). When killer whales are hunting small marine mammals in coastal waters, kills can often be observed visually for an estimation of predation pressure (Baird and Dill, 1995). Baird and Dill (1996) were able to follow killer whales and observe predation events to estimate rates of predation from the predator's perspective. However, these observations are not the same as estimating the risk of predation from the point of view of marine mammals targeted by the predator. Springer et al. (2008) discussed reasons why killer whale predation on large whales may be underestimated by visual observation. Some preliminary work has demonstrated the ability of tags to detect predation events on tagged pinnipeds. Horning and Mellish (2014) analyzed data from 36 Steller sea lions tagged with life-history tags (Horning and Hill, 2005) and were able to conclude that 15 of these sea lions had been killed by a predator. This tagging work identified a new unsuspected shark predator of these sea lions, but this approach is not appropriate for all species, and its cost limits the sample size, making it unlikely to provide robust estimates of predation risk even for species where it can be used. When predation events cannot be studied directly, another method for estimating the risk of predation is to measure when predators interact with prey. Some investigators use scars from shark or killer whale attacks as indicators of predation pressure (Heithaus, 2001), but this is problematic as the scarred individuals are the ones that got away. Accurate estimation of predation pressure for marine mammals remains a significant challenge.

Animal-Oriented Approaches to Measuring Extrinsic and Intrinsic Stressors

Mapping of stressors allows one to estimate exposure at specific locations. However, many marine mammals range over wide areas. If their path is not known, stressor maps may not suffice to estimate exposure. And, as discussed above, broad geographical overlap is not enough to predict exposure for stressors that concentrate in a narrow part of the geographical area, in particular substrates such as sediment, or in prey that must be ingested. As Chapter 3 notes, in these circumstances, the preferred approach is often to sample tissue from a marine mammal to characterize its dosage of chemical stressors. Tissues can currently be sampled from animals that are held for health assessment, but capabilities for sampling critical tissues such as blood are limited for many marine mammal species. New methods will need to be developed for this subject-oriented approach to reach its full potential for marine mammals.

Passive and active personal dosimeters have become established as useful methods for measuring the dosage of stressors. Here the stressor is either absorbed into a passive matrix (O'Connell et al., 2014) or measured by an active device on the animal or human (Boziari et al., 2010). Dosimeter tags have been developed to measure the dosage of some stressors on marine mammals. Acoustic sensors have been placed on marine mammal tags to quantify the dosage of sound at the animal (Johnson and Tyack, 2003). Optical sensors have also been deployed on tags on marine mammals, both to form images of prey (Hooker et al., 2002) and to measure bioluminescence from potential prey (Vacquié-Garcia et al., 2012). A variety of sensors have been used to detect attempts to capture prey (Plötz et al., 2001; Miller et al., 2004a) or the ingestion of prey (Austin et al., 2006), which may provide direct measures of foraging rates.

Managing Information on Stressors and Ecological Drivers

The obstacles described above for measuring prey limitation and predation pressure highlight the difficulties of assessing single components of interaction webs. The movement toward Integrated Ecosystem Assessments may support broader studies of interaction webs that focus on all human and natural nodes (Samhouri et al., 2014) and that prioritize focal ecosystem components (Levin et al., 2014). However, it will require substantial investments from funders in order to improve the estimates and accuracy of the various exposures to drivers and their effects.

As discussed in Chapter 7, long-term monitoring across broad spatial and temporal scales (including both passive and active surveillance) could help improve understanding of the geographic and temporal patterns of stressors as well as associated adverse effects, and also could help in detecting emerging health issues in marine mammals that are potentially indicative of a population at risk. In addition, understanding patterns of dosage and exposure for multiple stressors could help to inform future study designs to elucidate potential cumulative effects. This information will be most powerful if it is made widely available to scientists and managers through a centralized data management system that can interface with other databases that allows integration of marine mammal health data with ecosystem and oceanographic data.

Such a data management system, the Marine Mammal Health Monitoring and Analysis Platform (MM Health MAP), has been proposed and is in the early developmental stages (Simeone et al., 2015), being led by the U.S. National Marine Fisheries Service's (NMFS's) Marine Mammal Health and Stranding Response Program (MMHSRP) and the U.S. Marine Mammal Commission. The goal of the MM Health MAP is to support mandates under Title IV of the U.S. Marine Mammal Protection Act (MMPA) to gather data on marine mammal health trends and correlate these with biological, physical, and chemical variables.[1] However, the successful development and implementation of the MM Health MAP will depend on support not only from the NMFS but also from other federal managers, as well as cooperation and collaboration across the marine mammal research community. These efforts require willingness of, and financial support for, independent research groups to make data available. Other management and funding agencies should also encourage data management policies that lead to broader analyses and synthesis of information, including incorporation of data and model products into such databases. Similar levels of cooperation between the research community and public-sector agencies involved in tracking emerging diseases and specifically zoonotics have been observed (IOM and NRC, 2009). One such example is the PREDICT program within the U.S. Agency for International Development's Emerging Pandemic Threats Program. The PREDICT program is one of the world's most comprehensive zoonotic disease surveillance and capacity development programs; they have developed training for staff and low-cost detection tools for new viruses from targeted virus families in 32 laboratories in 20 developing nations. Such efforts, supported by modern data management practices and information sharing, have helped characterize human and ecological drivers of disease spillover from animals to people, and strengthened models for predicting disease emergence in wildlife (Jonna Mazet, personal communication).

To ensure comparability of the marine mammal health and stressor exposure data across studies and over space and time, such a system would require standardized information and proper quality assurance plans for the various analytical results. One of the components of the MMHSRP, which was established under the 1992 amendments to the MMPA, has been to coordinate analytical quality assurance of data from chemical analyses of marine mammal tissues. The quality assurance program for analysis of POPs, fatty acids, and trace elements in marine mammal tissues has been implemented through the National Institute of Standards and Technology and includes interlaboratory comparison exercises, as well as the development of control materials and standard reference materials for marine mammal tissues. Similar quality assurance measures would need to be identified and, if not in existence, would need to be established for other types of health data (e.g., stress hormones) in order to ensure accuracy and interpretability of results across laboratories. Such efforts would broaden understanding of stressor exposure across regions, provide necessary information to managers to assist in evaluating potential stressor mitigation strategies, and inform researchers interested in hypothesis generation for future analytical studies.

Finding 8.1: Improving the estimates of the exposure to and dosage of stressors, and their effects, will require better data availability, standardization, and management. The merger of both stressor and ecological driver-related data through a centralized database would facilitate integration and analyses.

Measuring Change in Behavior and Physiology

Most studies on the effects of sound on marine mammals focus on end points related to disturbance, such as behavioral changes. Where concern has focused on acute effects, such as strandings of beaked whales in response to sonar, it can be very useful to document levels of sound below which no short-term response occurs that poses a risk of stranding. Fernández et al. (2005, 2012) argue that exposure to sonar may also pose a risk of decompression sickness (DCS). Analyses of dive profiles using physiological models of gas dynamics during diving have been used to estimate the risk of physiological changes that could lead to DCS (Kvadsheim et al., 2012). Diving responses of beaked whales to actual sonar exercises have not been quantified, but they have been measured for experiments that used controlled exposures of sonar to tagged beaked and other whales. The behavioral responses to sonar observed in these experiments led to modeled end-dive N_2 tensions thought not to pose a significant risk of DCS. However, sonar exercises involve more intense and prolonged exposure than occurred during these experiments, which were designed to minimize risk of

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[1] See http://www.nmfs.noaa.gov/pr/health/MMHSRP.html.

injury. Therefore, while the exposure levels linked to these experiments do not pose a significant risk of DCS, the study cannot rule out that behavioral and physiological responses to actual sonar exercises could cause DCS. Testing for DCS in animals that strand coincident with sonar exercises may benefit from careful measurement of the distribution, volume, and gas composition of bubbles, as this may help discriminate between decompression and decomposition in stranded marine mammals (Bernaldo de Quiros et al., 2012).

For many other responses, there is a critical need to develop methods to evaluate the effects of chronic exposure. Analysis of health in terms of energy stores is a promising way to do this, as it can integrate with energetic models of survival and reproduction (e.g., New et al., 2013b). Further development of methods to estimate the energetic consequences of changes in foraging behavior and the physiology of metabolism will strengthen the promising approaches of Biuw et al. (2003) and New et al. (2014). For example, Wilson et al. (2006, 2008) advocate use of accelerometry to estimate metabolic rates of tagged subjects, and Fahlman et al. (2016) and Roos et al. (2016) describe improvements in methods that use respiration to estimate the metabolic rate of cetaceans.

Another important approach for measuring physiological changes resulting from exposure to stressors involves measuring glucocorticoid stress hormones. A few studies have measured changes in stress hormone levels of marine mammals exposed to sound (Romano et al., 2004; Rolland et al., 2012). Methods are being developed to sample stress hormones from a variety of tissues, such as blubber biopsy, feces, and blows. These methods are critical for practical sampling of animals in the wild, and data from these tissues need to be calibrated against data from blood, which is the standard.

The Functions Relating Exposure to Stressors to Behavioral or Physiological Responses

Short-term tags are well suited to experiments studying responses to acute exposure to intense sounds, and these experiments can produce probabilistic dose–response functions (e.g., Figure 1a in Box 2.2). Once these responses are characterized, monitoring programs can be developed to evaluate responses to longer-term and larger-scale exposures (e.g., Moretti et al., 2014). However, few of these studies have estimated exposure to other stressors that might influence cumulative effects. To evaluate cumulative effects of other stressors in addition to noise, these studies would need to include measurements of exposure to other stressors and responses to them.

The levels of exposure for an individual marine mammal to stressors such as noise, prey limitation, perceived threats, and disease may vary considerably as the animal moves over time periods of minutes to days. The biological responses to a sound stimulus are likely to vary as a function of behavioral states, such as traveling or foraging, and of physiological states, such as oxygen reserves or acute disease infection, that may vary on scales of seconds to days or more. These time scales require behavioral and physiological measurements along with estimates of stressor exposure that are local to the animal. These kinds of data on behavioral and physiological states have been used in experiments to evaluate the effect of behavioral context and the responses of marine mammals to acoustic stimuli (e.g., Goldbogen et al., 2013); this approach may offer some promise for studying cumulative effects involving other stressors.

There is also a data gap for studying effects of chronic exposure to sound. Short-term experiments can expose the same subjects several times to the same or different acoustic stimuli (Antunes et al., 2014; Miller et al., 2014). These experiments enable testing whether responses differ for the first exposure versus later ones, which is a first step in studying responses to repeated sounds. Some studies have taken advantage of unplanned events to study the impact of reductions in chronic noise on marine mammals. For example, Rolland et al. (2012) happened to be studying stress hormones in right whales before and after the terrorist attacks on the World Trade Center and Pentagon on September 11, 2001. Noise levels and the occurrence of ships passing near the whales were greatly reduced due to a pause in commercial shipping after these events; during this period of low noise and ship activity, the levels of stress hormones were lower than those recorded before September 11, 2001, or for the same period in other years. However, this opportunistic study lacks the controls required for a standard experimental design. New designs for experiments and opportunistic studies will be required to document the effects of planned changes in chronic noise and disturbance associated with ship passage induced by changes in shipping lanes or in shipping technology.

Use of Health Indices to Detect and Manage Species at Risk

Chapter 5 developed the Population Consequences of Multiple Stressors (PCoMS) framework that uses health parameters to help integrate effects of multiple stressors over longer time periods than those captured by individual physiological or behavioral responses to acute stressor exposures. Measuring these health parameters can improve the ability to model the linkages between stressor dosage or exposure and long-term effects on populations. Changes in health integrate short-term changes in exposure to multiple stressors, providing a longer-term measure that can more readily be linked to changes in vital rates. Because changes in health can be measured more rapidly than changes in vital rates, health may help provide an early warning indicator for individual animals. If enough individuals in a population are sampled for health, as Chapter 7 discusses, this information on population health may provide an early warning indicator for populations at risk.

Comprehensive Health Evaluation

Comprehensive health assessments are of particular value because they provide information on multiple aspects of an animal's condition and are therefore more likely to detect a compromised health state. In addition, health assessments that utilize an array of indicators can help to identify specific causal factors for compromised health and can inform management decisions about which steps to take to reduce risks. Comprehensive health assessments have been developed for pinnipeds and some cetacean species, such as bottlenose dolphins (*Tursiops truncatus*). In pinnipeds, contaminant burdens measured in tissues, and pathogen exposures sampled from nasal and rectal swabs, can be included in physiology workups for tag deployments and recoveries that also include body condition, stress hormones, and immune markers (e.g., Goldstein et al., 2013; Peterson et al., 2015; Peck et al., 2016). For example, recent work using nasal swabs showed that tagged elephant seals were exposed to the H1N1 virus between instrument deployments and recoveries in 2010 (Goldstein et al., 2013). Comprehensive health assessments have also been conducted for coastal populations of bottlenose dolphins in several sites in the southeastern United States (Wells et al., 2004; Fair et al., 2006; Schwacke et al., 2010). In some cases, these studies have identified adverse health effects in association with stressor exposure. For example, a high prevalence of anemia, low thyroid hormone levels, and immune suppression were associated with polychlorinated biphenyl exposure in bottlenose dolphins inhabiting an estuary near a hazardous waste site in Brunswick, Georgia (Schwacke et al., 2012). Most of these studies rely on sampling of blood but may also include sampling of other tissues or body fluids, and ultrasound examination of organs. Baseline data from these kinds of assessments are critical for studying stressor dosage and responses to stressors.

Understanding the health status of a population aids in the identification of threats that can be effectively mitigated to support recovery, whether or not they have been major contributing factors for the population's decline. For example, health studies of highly endangered Hawaiian monk seals found that the species was immunologically naïve to morbillivirus, which posed a significant epidemic threat, and furthermore that the lack of genetic diversity could potentially limit the ability of the species to respond to other newly introduced diseases such as toxoplasmosis, West Nile virus, and influenza (NMFS, 2016b). In response, NMFS identified an action to "Detect and prevent catastrophic disease outbreak and disease-related mortality" as a priority in the 5-year action plan for recovery of this species that was on the brink of extinction. A disease outbreak preparedness plan, including the development of a morbillivirus vaccination program, has now been implemented as part of ongoing health research activities.

Assessing Health in Populations That Cannot Be Handled

Current methods and technologies limit comprehensive health assessments to a few species that can be temporarily captured, restrained, and evaluated. This limitation has led to the development of less comprehensive health assessments for other species, often including two types of readily accessible indicators of health: body condition and stress hormones. As these measures can be obtained using visually observed indicators for body condition, or remote sampling for stress hormones, they can be collected for many marine mammal species.

Body Condition

As discussed in Chapter 5, body condition is an indicator of health and allostatic or homeostatic load that can be measured directly for species that can be handled. Methods are more limited for species that cannot be handled. These include visual observations of condition and use of tags to estimate changes in buoyancy of wild marine mammals. Pettis et al. (2004) estimated body condition by scoring the concavity of an area just behind the blowhole that accumulates fat and that is visible in some photographs taken to identify individual whales. C.A. Miller et al. (2012) used aerial photographs taken directly over a right whale to more precisely measure the body shape and quantify the condition of right whales. Unmanned aerial or underwater vehicles may offer more cost-effective ways to obtain such images optimized for measuring features of interest. The tagging method for estimating body condition involves measuring the vertical acceleration of diving animals during drifting periods of the dive. Drift dives, however, do not occur in all species. More detailed research on the forces acting on swimming marine mammals may allow estimation of the static buoyancy force and percentage of lipid in animals that are not passively drifting, but are gliding during ascent and descent phases of normal dives (Miller et al., 2004b; Watanabe et al., 2006; Aoki et al., 2011). This may broaden the number of species that can be studied using this method.

Stress

As discussed in Chapter 4, chronic activation of the hypothalamic-pituitary-adrenal axis may be an important mechanism by which cumulative effects of different stressors exert effects on health and vital rates. Glucocorticoid (GC) stress hormones have usually been measured from blood samples, but an array of other matrices for stress hormones, including blubber, feces and exhaled blow, and baleen and earplugs in baleen whales are also being studied for analysis of stress. These other matrices provide longer-term measures of GC levels than blood and may be more useful for investigating long-term stress dosage and effects. Feces and exhaled blow can be collected noninvasively for some species, and blubber can be sampled by biopsy darting in almost all marine mammal species. The promise of these new matrices cannot be fulfilled without cross-sectional and/or longitudinal studies that help to establish distributions for expected values across different species, age classes, sexes, and reproductive states. Pregnancy changes corticosterone levels in blubber, so such samples also need to measure progesterone to control for this effect.

Remote Assessment of Health

Pettis et al. (2004) conducted an early effort to develop a scale for assessing the health of individual right whales in the western North Atlantic. They took advantage of an extensive photo-identification catalog to score body condition, skin condition, presence of "rake marks," and cyamids near the blowhole. This assessment scheme was limited to features that were visible from photographs used to identify individual whales. The development of indices that include information from biopsies, blow, and feces will enrich the power of health assessments that are limited to remote sampling.

Health studies that include assessment of body condition as well as collection of contaminant and health biomarkers have been identified as a priority action for the recovery of highly endangered Southern Resident killer whales (NMFS, 2016c). The goal of these health studies is to compare the health of Southern Residents with other killer whale populations to identify potential sources of decreased survival and/or reproduction. High concentrations of emerging contaminants, and specifically flame-retardant chemicals, have been reported in these apex predators (Rayne et al., 2004). Therefore, the health studies are particularly focused on identifying sources for the emerging contaminants and understanding potential associated health effects in order to guide water quality recommendations and reduce contaminant inputs into Southern Resident killer whale habitat.

Finding 8.2: Assessment of health is central to the PCoMS model proposed in this report. Comprehensive health assessments of a cross section of a marine mammal population can also help managers decide when the population is at risk and help them decide which management actions can most effectively support recovery.

Stressor Exposure: Health Response Function

The PCoMS model presented in Chapter 5 has the capability to analyze the short-term links between a health effect and the combination of stressors to which an animal has been exposed. As a sample of wild animals moves through their habitat and/or experiences seasonal changes, they are likely to be exposed to a wide distribution of the stressors that are present in their environment at that time. If the dosage or exposure to the stressors and the effects of each combination of stressors can be measured, then, as Chapter 6 notes, this approach offers the potential for a much larger sample of dose–response measurements than can be tested in experiments, perhaps improving the ability to identify which combinations of stressors have an observable effect on health.

The desired characteristics of the health variables introduced in Chapter 5 are that they can be measured in wild marine mammals, they integrate effects of repeated exposures to multiple stressors, they change over shorter time scales than vital rates, and yet they can influence the vital rates of each individual. The committee has argued that free-ranging marine mammals are influenced by so many stressors, each of whose effects may vary depending on life-history stage of the animal, and that the number of combinations of stressors is too large for experimental studies of how all combinations interact. The committee's proposed PCoMS framework uses a small number of health variables to integrate the effects from multiple stressors and to improve current understanding of the mechanisms by which combinations of stressors affect vital rates.

Exposure to many of the stressors discussed here varies on an hourly to weekly basis, and even exposure to toxic compounds that have stable concentrations in one area will vary as marine mammals move from area to area. Marine mammals are long lived and give birth at most once per year. This means that studies linking exposure to stressors with reproductive success cannot sample effects more frequently than yearly. By contrast, some of the health variables proposed here have much finer time resolution—more appropriate for linking to stressor exposures. For example, Biuw et al. (2003) state that for estimating body condition from buoyancy in drift dives "biologically realistic changes in drift rate (are) expected to be detectable over a period of 5-6 days."

If changes in health and exposure to stressors can be sampled over shorter time periods than vital rates, then longitudinal studies may be able to repeatedly measure stressor–health combinations many times within a breeding cycle. Longitudinal studies are particularly well suited for situations where tags can be attached for significant parts of the annual cycle and can sample the health variables of interest. Tags can currently sample body condition in the few species with drift dives but are not able directly to sample the other health variables discussed here. Development of long-term tags that can sample such variables could support this approach for studying cumulative effects. Initial scoping for development would be useful, but breakthroughs are not expected in the next 5-10 years. For these other variables and for species where it is not possible to use tags to measure body condition, it may be more productive to conduct cross-sectional studies where exposure to stressors and the health variables are measured in a large number of individuals within a population. Rather than measuring changes in health as the pattern of exposure to stressors changes, this approach would sample each individual at a single time point, linking the stressor and health values observed at that time. This approach assumes that the values of stressors observed are close to those that led to the health value measured at the same time. The cross-sectional approach may be less able to detect adverse outcome pathways that involve sequential exposures to stressors over longer time periods.

These kinds of longitudinal and cross-sectional studies are relatively well established for coastal populations of marine mammals in which individuals are small enough to be handled and where relatively comprehensive health assessments have been established. Remote biopsy methods have been developed, but the data obtained by this method are more limited than those available from onshore populations or when one can handle the animals. However, there are precedents for large-scale efforts to sample large, highly mobile whale species. For example, Smith et al. (1999) report on a systematic and standardized effort to photo-identify and biopsy sample humpback whales throughout the North Atlantic. They report that "during 666 days at sea aboard 28 vessels, 4,207 tail fluke photographs and 2,326 skin biopsies were collected." Their assessment was that "an oceanwide approach to population assessment of baleen whales is practicable."

One of the goals of the statement of task for this committee is to identify how exposure to nonacoustic stressors may affect a marine mammal's response to an acoustic stressor. In this context, evaluation of the health status of potential subjects for response studies may help to identify those individuals that may be particularly sensitive or vulnerable to an acoustic stressor. A basic element of the allostasis model is that animals already carrying a large allostatic load may be driven into allostatic overload by a relatively small additional exposure to a stressor. This would suggest that subjects already in adverse health status may be the most vulnerable to even small doses of another stressor. Note, however, that this does not mean that the subject will be the most sensitive in the sense of most likely to show a behavioral response at low exposure levels (Gill et al., 2001). For example, Beale and Monaghan (2004) have shown that birds under nutritional stress may be less likely to stop feeding and move away from a threat than birds of better body condition that may more easily be able to afford the lost foraging opportunities. This emphasizes the importance of measuring the response to stressor in terms of changes in health as well as observing behavioral reactions.

Health Response: Vital Rates Function

The functional relationship between health and vital rates is an important link in the PCoMS model. Parameterizing this relationship will require measuring health and vital rates in the same individuals and populations. Several different methods are used or have been proposed for studying vital rates.

Mark–Recapture Methods

As Chapter 7 notes, vital rates have been estimated for wild marine mammal populations where the same individuals can reliably be resighted. Many demographic parameters can be estimated from focused mark–recapture surveys of animals that can reliably be sighted nearly every year and for which it can be determined whether adult females have given birth. Birth rates and survival of the young are highlighted in Chapter 7 as early demographic indicators of problems; these are most easily studied in species that give birth on land where it can be observed or where young animals are easily distinguished. Several new methods may be appropriate for species where this is not possible, and these will be discussed next.

Matrices That Store Information on Age-Specific Reproduction and Age at Death

One common method for determining the age of mammals involves counting growth layers in tissues such as teeth, baleen, or wax laid down in the ear canal of baleen whales (called the ear plug). Growth layers in teeth have been used to determine the age of dolphins (Hohn et al., 1989), polar bears (Calvert and Ramsey, 1998), and pinnipeds (Scheffer, 1950). Not only can these tissues be used to age marine mammals, but recent work has shown that ear plugs and baleen can provide time records of reproductive and stress hormones as well as contaminants over the lifespan in the case of the ear plug (Trumble et al., 2013) and over several years in the case of baleen (Hunt et al., 2014). Baleen and earplugs are laid down in layers that differ during different parts of the annual cycle, such as feeding, migration, and breeding, making it possible to track each year of life of the animal. Both of these tissues are relevant only for baleen whales—more work on tissues such as teeth that lay down layers throughout the lifespan would help broaden this approach to other marine mammals. In many organisms that lay down these kinds of layers, characteristics of the layer may also indicate the nutritional state of the organism at the time of deposition (Fritts, 2012), potentially providing information on changes in condition.

Life History Tags

Problems with estimating age-specific mortality, and especially causes of mortality in open ocean species, led Horning and Hill (2005) to develop an electronic tag that is implanted internally, recording life-history data through the life of a marine mammal, and that releases and transmits data upon expulsion from the dead animal. Insertion of a tag into the peritoneal cavity requires surgery, but Horning et al. (2008) report that 4 California sea lions (*Zalophus* *californianus*) and 15 juvenile Steller sea lions (*Eumetopias jubatus*) recovered well under veterinary care after the tag insertion. The sea lions were then released into the wild and tracked with satellite tags. The behavior of sea lions with implanted tags was monitored for up to half a year and was similar to that of sea lions tagged only with satellite tags. Distinct signatures of temperature and light identify when an animal has been killed by a predator (Horning and Mellish, 2014). Temperature data from 15 of the 36 sea lions tagged by Horning and Mellish (2014) indicated that they had been killed by predators. These sea lions were followed for a total of 111 years, so 15 deaths indicate a relatively high predation rate.

The costs and risks of surgical insertion of the life-history tag limit the sample sizes achievable for this kind of tagging, and it may not be appropriate for many marine mammal species. Surgical implantation raises ethical and animal welfare concerns that would require evidence of a clear benefit to these populations that would be sufficient to outweigh the welfare cost. However, this research showed that tags can be developed to record data from within an animal until it dies. This mode of tagging suggests a new approach for active personal dosimeters. The dosimeters described above are designed to measure the dosages of stressors to which an animal is exposed. The potential of a tag that can sample the internal milieu of a marine mammal throughout the lifespan would be greatly expanded if, as with earplugs, it could also sample life-history events, stressor dosage, and response to a variety of stressors. Passive personal dosimeters have been designed with materials optimized for absorbing and storing chemical compounds of interest (Paulik et al., 2016). Tags placed inside the body are best located to measure physiological parameters, such as hormones, and dosages of stressors, such as contaminant loads. For species that do not have tissues from which age-specific samples can be recovered, such as the earplug, there may be benefit in designing passive samplers that can sample compounds of interest at known times throughout the lifespan. Some compounds and other stressors, such as sound, can be detected actively by sensors on an electronic tag, but development of active sensing in lifetime tags will face considerable obstacles in terms of power requirements and space limitations.

Stressor Exposure: Vital Rates Function

Modeling each component of the PCoMS model is very challenging, but it is necessary in most cases, because a direct link cannot be made between stressor exposure and vital rates. However, in cases where a direct link can be made, it may be possible to bypass all the intermediate modeling stages. Such studies have been attempted for several seabird species whose demography and movements have been well documented. Some studies have used the approach taken by Forcada et al. (2006) to compare annual variation in demographic parameters to natural variation in more than one stressor on a year-by-year basis. For example, Rolland et al. (2009) used 26 years of demographic data from a study of black-browed albatross on Kerguelen Island to study the impact of fishing bycatch under various climate conditions. Levels of ocean warming expected for the next century were predicted to enhance the growth of this population, potentially compensating for controlled increases in fishing effort. This analysis was useful to inform management of fisheries in the presence of climate change. However, the authors did not explicitly model potential interactions between stressors.

Few studies on marine mammal populations have used methods similar to those just discussed for seabirds and summarized by Barbraud et al. (2012). However, the demographic parameters for populations of pinnipeds that breed on land could be studied using similar methods. Similar analyses should be possible for species such as resident coastal cetacean populations with long-term studies of identified individuals whose tissue can be sampled and whose vital rates are estimated (Bowen et al., 2010). Exposure to environmental stressors such as ocean temperature and interactions with fisheries can be characterized for marine mammals using spatiotemporal sampling of parameters such as effort statistics similar to those used in the seabird studies. For example, Caillat and Smout (2015) studied the potential effects of prey availability, grey seal numbers, and exposure to biotoxins on the fecundity and pup survival of harbor seals off the east coast of the United Kingdom. They found that a single (but different) dominant stressor explained the observed variations in each demographic rate. It may be possible to identify interactions between these stressors in other populations that have undergone more dramatic changes in abundance.

The potential for tissues such as baleen whale earplugs or manmade sampling devices to provide a lifetime record of age-specific fertility, age at death, and exposure to some stressors suggests the potential for a new approach to studying the relationship between exposure to stressors and vital rates in marine mammal populations. Given the low probability that long-term studies of vital rates and spatiotemporal mapping of exposure to stressors will provide sufficient data over long enough time intervals for marine mammal populations, we recommend research on natural matrices that may provide a lifetime record of stressors and effects. The development of tags to accomplish the same goal for species without such natural matrices faces significant obstacles but is worth scoping as a potential opportunity for the long term.

Finding 8.3: Natural and artificial matrices have potential as tools for documenting dosage of chemical stressors and changes in hormone levels over long enough time periods to test the relationship between stressor dosage and response in terms of health or vital rates. Natural matrices that are laid down in semiannual layers from birth to death are particularly promising.

Measuring the Lifetime Exposure of an Organism to Stressors

Wild (2005) argued for the importance of tracking exposure of stressors throughout the lifespan. He developed the concept of an "exposome"—defined as the lifetime exposure of an organism to stressors from the prenatal period to death. It is clearly a great challenge to measure the exposome, but a series of papers have emphasized the importance of gathering exposure data on stressors, in both the internal and the external environments, throughout the lifetime (e.g., Lioy and Rappaport, 2011). Rappaport (2011) suggests an approach to measuring the exposome by repeated sampling of blood at critical times of life, with each sample analyzed for "important classes of toxic chemicals, notably, reactive electrophiles, metals, metabolic products, hormone-like substances, and persistent organic compounds." He argues that as the extent of this sampling increases, economies of scale should create positive feedback for growth of exposome sampling. A similar sampling scheme for accessible marine mammal populations using cross-sectional studies supplemented by individuals sampled throughout their lifespan could help to define combinations of stressors that cause adverse cumulative effects. Longitudinal, spatially comprehensive collection of data on exposure to and effects of multiple stressors could be excessively costly. However, ongoing research studies being funded and/or conducted by multiple federal agencies (e.g., National Oceanic and Atmospheric Administration, U.S. Navy, Bureau of Ocean Energy Management, and the U.S. Geological Survey) and independent researchers could be leveraged and expanded to simultaneously collect samples and conduct analysis to assess exposure to and effects of multiple stressors. The value of a centralized database would be increased with additional information from active surveillance (see Chapter 7).

Health: Vital Rates Function

Most of the health indices discussed in this report can be measured directly for species that can be handled for sampling. The committee has suggested several other approaches for tagging or sampling other matrices in the wild that can be used to assess health. Vital rates can also be estimated directly for species where individuals can regularly be resighted and where birth of the young can be detected reliably. For other species, the committee suggests some new approaches that also include tagging animals with artificial matrices or sampling natural matrices that lay down tissue in layers that can be used for aging and that can store hormones.

The best example of estimating the function relating health to vital rates comes from New et al. (2014), who took advantage of studies of elephant seals on beaches where lipid and lean mass could be measured from pregnant females as they left and returned from foraging trips. Their pups were weighed soon after birth and after weaning. These measurements allowed New et al. (2014) to estimate the energy transferred from mother to pup, and to relate pup natal mass to survival. The relationships between the health variable of body condition, expressed as maternal lipid mass, to the pup's weaning mass, and between the pup's weaning mass and the vital rate of pup survival enabled the evaluation of the relationship between health and vital rates for this species.

The committee found no examples of similar studies relating health to vital rates in other marine mammals but does suggest some new approaches that may enable such studies. A major problem for these studies is the long time period required to measure vital rates. The discovery that baleen whale earplugs provide a lifetime record of reproductive hormones for each year of life may enable studies of the vital rate of reproduction, and the age at death can be measured from the earplug, providing age-specific mortality. The earplug has been shown to store the health variables of contaminants and stress hormones, and some tissues that are laid down in layers also provide indications of body condition. If large enough samples of earplugs can be recovered and analyzed for health and vital rates, this could enable a new way to evaluate the relationship between these critical parameters. This is the only shortcut found by the committee for retrospective studies of health and vital rates where one can use tissue from dead animals to study these relationships from birth to death. This possibility is promising enough to justify exploration of other matrices, such as teeth and baleen, that may provide similar timelines of health and vital rates.

RECOMMENDATIONS

Recommendation 8.1: Future research initiatives should support evaluation of the range of emerging technologies for sampling and assessing individual health in marine mammals, and identification of a suite of health indices that can be measured for diverse taxa and that best serves to predict future changes in vital rates. Potentially relevant measures include hormones, immune function, body condition, oxidative damage, and indicators of organ status, as well as contaminant burden and parasite load. New technology for remotely obtaining respiratory, blood, and other tissue samples and for remote assessment (e.g., visual assessment of body condition) should also be pursued.

Establishing baseline values of these parameters and their associations in species will provide critical information for assessing individual and population health. Assessment of health is not only central to the PCoMS model proposed in this report, but comprehensive assessments of stressor exposure and health of a population of marine mammals can also help managers decide when the population is at risk, and help them decide which management actions can most effectively support recovery. Long-term studies of known individuals will be important in this regard. Cross-sectional sampling and repeated sampling from the same individuals of blood or other tissues during critical life-history phases can help to document dosages and health effects of stressors.

Recommendation 8.2: Agencies charged with monitoring and managing the effects of human activities on marine mammals should identify baselines and document exposures to stressors for high-priority populations. High-priority populations should be selected to include those likely to experience extremes (both high and low) of stressor exposure in order to increase the probability of detecting relationships. This will require stable, long-term funding to maintain a record of exposures and responses that could inform future management decisions. Information on baselines and contextual variables is critically important to interpreting responses.

Recommendation 8.3: Standards for measurement of stressors should be developed along with national or international databases on exposure of marine mammals to high-priority stressors and associated health measures that are accessible to the research community.

Recommendation 8.4: Techniques should be developed that will allow historical trajectories of stress responses to be constructed based on the chemical composition of the large number of baleen whale earplugs and baleen samples in museums or similar matrices in other species. Artificial matrices should be studied for their potential to absorb materials (hormones or chemical stressors) and thereby provide a record of exposures and responses to stressors.

There are opportunities to explore the potential for natural or artificial matrices (that store chemical stressors and hormones over long enough time periods) to test the relationship between exposure to the stressors and response in terms of health or vital rates.

Such techniques with museum samples could provide critical information on the relationships between contaminants, stress, and reproductive intervals. Natural matrices that are laid down in semiannual layers from birth to death are particularly promising.

References

Aarts, G., M. MacKenzie, B. McConnell, M. Fedak, and J. Matthiopoulos. 2008. Estimating space-use and habitat preference from wildlife telemetry data. *Ecography* 31:140-160.

Acevedo-Whitehouse, K., and A.L.J. Duffus. 2009. Effects of environmental change on wildlife health. *Philosophical Transactions of the Royal Society B: Biological Sciences* 364:3429-3438. doi: 10.1098/rstb.2009.0128.

Acevedo-Whitehouse, K., A. Rocha-Gosselin, and D. Gendron. 2009. A novel non-invasive tool for disease surveillance of free-ranging whales and its relevance to conservation programs. *Animal Conservation* 13(2):217-225.

Aguilar, A., and A. Borrell. 1994. Abnormally high polychlorinated biphenyl levels in striped dolphins (*Stenella coeruleoalba*) affected by the 1990-92 Mediterranean epizootic. *Science of the Total Environment* 154:237-247.

Aguirre, A.A., T.J. Keefe, J.S. Reif, L. Kashinsky, P.K. Yochem, J.T. Saliki, J.L. Stott, T. Goldstein, J.P. Dubey, R. Braun, and G. Antonelis. 2007. Infectious disease monitoring of the endangered Hawaiian monk seal. *Journal of Wildlife Diseases* 43(2):229-241.

Aksenov, A.A., L. Yeates, A. Pasamontes, C. Siebe, Y. Zrodnikov, J. Simmons, M.M. McCartney, J.-P. Deplanque, R.S. Wells, and C.E. Davis. 2014. Metabolite content profiling of bottlenose dolphin exhaled breath. *Analytical Chemistry* 86:10616-10624.

Alexander, N., Y. Kuepper, A. Schmitz, R. Osinsky, E. Kozyra, and J. Hennig. 2009. Gene–environment interactions predict cortisol responses after acute stress: Implications for the etiology of depression. *Psychoneuroendocrinology* 34(9):1294-1303.

Anderson, M.J., and A.A. Thompson. 2004. Multivariate control charts for ecological and environmental monitoring. *Ecological Applications* 14:1921-1935.

Andrew, R.K., B.M. Howe, and J.A. Mercer. 2011. Long-time trends in ship traffic noise for four sites off the North American West Coast. *Journal of the Acoustical Society of America* 129(2):642-651.

Angliss, R.P., and K.L. Lodge. 2002. Alaska Marine Mammal Stock Assessments, 2002. U.S. Department of Commerce, *NOAA Technical Memorandum*. NM FSAFSC-133, 224 pp.

Ankley, G.T., R.S. Bennett, R.J. Erickson, D.J. Hoff, M.W. Hornung, R.D. Johnson, D.R. Mount, J.W. Nichols, C.L. Russom, P.K. Schmieder, and J.A. Serrano. 2010. Adverse outcome pathways: A conceptual framework to support ecotoxicology research and risk assessment. *Environmental Toxicology and Chemistry* 29(3):730-741.

Antunes, R., P.H. Kvadsheim, F.P.A. Lam, P.L. Tyack, L. Thomas, P.J. Wensveen, and P.J.O. Miller. 2014. High thresholds for avoidance of sonar by free-ranging long-finned pilot whales (*Globicephala melas*). *Marine Pollution Bulletin* 83:165-180.

Aoki, K., Y.Y. Watanabe, D.E. Crocker, P.W. Robinson, M. Biuw, D.P. Costa, N. Miyazaki, M. Fedak, and P. Miller. 2011. Northern elephant seals adjust gliding and stroking patterns with changes in buoyancy: Validation of at-sea metrics of body density. *Journal of Experimental Biology* 214(17):2973-2987.

Arnbom, T., M.A. Fedak, I.L. Boyd, and B.J. McConnell. 1993. Variation in weaning mass of pups in relation to maternal mass, postweaning fast duration, and weaned pup behavior in southern elephant seals at South Georgia. *Canadian Journal of Zoology* 71:1772-1781.

Asseburg, C., J. Harwood, J. Matthiopoulos, and S. Smout. 2006. The functional response of generalist predators, and their implications for the monitoring of marine ecosystems. Pp. 262-274 in *Top Predators in Marine Ecosystems: Their Role in Monitoring and Management*, I.L. Boyd, S. Wanless, and K. Camphuysen, eds. Cambridge: Cambridge University Press.

Atkinson, S., D. Crocker, D. Houser, and K. Mashburn. 2015. Stress physiology in marine mammals: How well do they fit the terrestrial model? *Journal of Comparative Physiology B* 185(5):463-486.

Austin, D., W.D. Bowen, J.I. McMillan, and D.J. Boness. 2006. Stomach temperature telemetry reveals temporal patterns of foraging success in a free-ranging marine mammal. *Journal of Animal Ecology* 75(2):408-420.

Ayres, K.L., R.K. Booth, J.A. Hempelmann, K.L. Koski, C.K. Emmons, R.W. Baird, K. Balcomb-Bartok, M.B. Hanson, M.J. Ford, and S.K. Wasser. 2012. Distinguishing the impacts of inadequate prey and vessel traffic on an endangered killer whale (*Orcinus orca*) population. *PLoS ONE* 7:e36842.

Baggeroer, A., and W. Munk. 1992. The Heard Island feasibility test. *Physics Today* 45:22-30.

Bailey, H., B. Senior, D. Simmons, J. Rusin, G. Picken, and P.M. Thompson. 2010. Assessing underwater noise levels during pile-driving at an offshore windfarm and its potential impact on marine mammals. *Marine Pollution Bulletin* 60:888-897.

Bailey, H., K.L. Brookes, and P.M. Thompson. 2014. Assessing environmental impacts of offshore wind farms: Lessons learned and recommendations for the future. *Aquatic Biosystems* 10:8.

Baird, R.W., and L.M. Dill. 1995. Occurrence and behaviour of transient killer whales: Seasonal and pod-specific variability, foraging behaviour, and prey handling. *Canadian Journal of Zoology* 73(7):1300-1311.

Baird, R.W., and L.M. Dill. 1996. Ecological and social determinants of group size in transient killer whales. *Behavioral Ecology* 7(4):408-416.

Baird, R.W., S.W. Martin, D.L. Webster, and B.L. Southall. 2014. *Assessment of Modeled Received Sound Pressure Levels and Movements of Satellite-Tagged Odontocetes Exposed to Mid-Frequency Active Sonar at the Pacific Missile Range Facility: February 2011 Through February 2013*. Prepared for U.S. Pacific Fleet, submitted to NAVFAC PAC by HDR Environmental, Operations and Construction, Inc.

Ban, S.S., N.A.J. Graham, and S.R. Connolly. 2014. Evidence for multiple stressor interactions and effects on coral reefs. *Global Change Biology* 20:681-697.

Banner, A., and M. Hyatt. 1973. Effects of noise on eggs and larvae of two estuarine fishes. *Transactions of the American Fisheries Society* 102(1):134-136.

Barber, J.R., K.R. Crooks, and K.M. Fristrup. 2010. The costs of chronic noise exposure for terrestrial organisms. *Trends in Ecology & Evolution* 25:80-189.

Barbieri, M.M., S. Raverty, M.B. Hanson, S. Venn-Watson, J.K.B. Ford, and J.K. Gaydos. 2013. Spatial and temporal analysis of killer whale (*Orcinus orca*) strandings in the North Pacific Ocean and the benefits of a coordinated stranding response protocol. *Marine Mammal Science* 29:E448-E462.

Barbraud, C., V. Rolland, S. Jenouvrier, M. Nevoux, K. Delord, and H. Weimerskirch. 2012. Effects of climate change and fisheries bycatch on Southern Ocean seabirds: A review. *Marine Ecology Progress Series* 454:285-307.

Basner, M., W. Babisch, A. Davis, M. Brink, C. Clark, S. Janssen, and S. Stansfeld. 2014. Auditory and non-auditory effects of noise on health. *The Lancet* 383:1325-1332.

Bayne, E.M., L. Habib, and S. Boutin. 2008. Impacts of chronic anthropogenic noise from energy-sector activity on abundance of songbirds in the boreal forest. *Conservation Biology* 22:1186-1193.

Beale, C.M., and P. Monaghan. 2004. Behavioural responses to human disturbance: A matter of choice. *Animal Behaviour* 68:1065-1069.

Beaugrand, G. 2004. The North Sea regime shift: Evidence, causes, mechanisms and consequences. *Progress in Oceanography* 60(2):245-262.

Bejarano, A., F. Gulland, T. Goldstein, J.S. Leger, M. Hunter, L. Schwacke, F. VanDolah, and T. Rowles. 2008a. Demographics and spatio-temporal signature of the biotoxin domoic acid in California sea lion (*Zalophus californianus*) stranding records. *Marine Mammal Science* 24:899-912.

Bejarano, A.C., F.M. VanDola, F.M. Gulland, T.K. Rowles, and L.H. Schwacke. 2008b. Production and toxicity of the marine biotoxin domoic acid and its effects on wildlife: A review. *Human and Ecological Risk Assessment* 14(3):544-567.

Bejder, L., A. Samuels, H. Whitehead, and N. Gales. 2006. Interpreting short-term behavioural responses to disturbance within a longitudinal perspective. *Animal Behaviour* 72(5):1149-1158.

Bejder, L., A. Samuels, H. Whitehead, H. Finn, and S. Allen. 2009. Impact assessment research: Use and misuse of habituation, sensitisation and tolerance in describing wildlife responses to anthropogenic stimuli. *Marine Ecology Progress Series* 395:177-185.

Benson, S.R., D.A. Croll, B.B. Marinovic, F.P. Chavez, and J.T. Harvey. 2002. Changes in the cetacean assemblage of a coastal upwelling ecosystem during El Niño 1997–98 and La Niña 1999. *Progress in Oceanography* 54(1):279-291.

Bernaldo de Quiros, Y., O. González-Diaz, M. Arbelo, E. Sierra, S. Sacchini, and A. Fernández. 2012. Decompression vs. decomposition: Distribution, amount, and gas composition of bubbles in stranded marine mammals. *Frontiers in Physiology* 3:177.

Bexton, S., D. Thompson, A. Brownlow, J. Barley, R. Milne, and C. Bidewell. 2012. Unusual mortality of pinnipeds in the United Kingdom associated with helical (corkscrew) injuries of anthropogenic origin. *Aquatic Mammals* 38:229-240.

Bishop, A., P. Pomeroy, and S.D. Twiss. 2015. Breeding male grey seals exhibit similar activity budgets across varying exposures to human activity. *Marine Ecology Progress Series* 527:247-259.

Biuw, M., B. McConnell, C.J.A. Bradshaw, H. Burton, and M.A. Fedak. 2003. Blubber and buoyancy: Monitoring the body condition of free-ranging seals using simple dive characteristics. *Journal of Experimental Biology* 206:3405-3423.

Blickley, J.L., D. Blackwood, and G.L. Patricelli. 2012a. Experimental evidence for the effects of chronic anthropogenic noise on abundance of Greater Sage-Grouse at leks. *Conservation Biology* 26:461-471.

Blickley, J.L., K.R. Word, A.H. Krakauer, J.L. Phillips, S.N. Sells, C.C. Taff, J.C. Wingfield, and G.L. Patricelli. 2012b. Experimental chronic noise is related to elevated fecal corticosteroid metabolites in lekking male Greater Sage-Grouse (*Centrocercus urophasianus*). *PLoS ONE* 7:e50462.

Block, B.A., I.D. Jonsen, S.J. Jorgensen, A.J. Winship, S.A. Shaffer, S.J. Bograd, E.L. Hazen, D.G. Foley, G.A. Breed, A.-L. Harrison, J.E. Ganong, A. Swithenbank, M. Castleton, H. Dewar, B.E. Mate, G.L. Shillinger, K.M. Schaefer, S.R. Benson, M.J. Weise, R.W. Henry, and D.P. Costa. 2011. Tracking apex marine predator movements in a dynamic ocean. *Nature* 475(7354):86-90.

Boas, M., U. Feldt-Rasmussen, N.E. Skakkebaek, and K.M. Main. 2006. Environmental chemicals and thyroid function. *European Journal of Endocrinology* 154:599-611.

Bonde, R.K., A. Garrett, M. Belanger, N. Askin, L. Tan, and C. Wittnich. 2012. Biomedical health assessments of the Florida manatee in Crystal River—providing opportunities for training during the capture, handling, and processing of this unique aquatic mammal. *Journal of Marine Animals and Their Ecology 5*(2):17-28.

Bonier, F., P.R. Martin, I.T. Moore, and J.C. Wingfield. 2009. Do baseline glucocorticoids predict fitness? *Trends in Ecology & Evolution* 24(11):634-642.

Boonstra, R. 2005. Equipped for life: The adaptive role of the stress axis in male mammals. *Journal of Mammalogy* 86(2):236-247.

Borchers, D.L. 2012. A non-technical overview of spatially explicit capture–recapture models. *Journal of Ornithology* 152:435-444.

Bossart, G.D., T.A. Romano, M.M. Peden-Adams, A. Schaefer, S. McCulloch, J.D. Goldstein, C.D. Rice, J.T. Saliki, P.A. Fair, and J.S. Reif. 2011. Clinicoimmunopathologic findings in Atlantic bottlenose dolphins *Tursiops truncatus* with positive cetacean morbillivirus antibody titers. *Diseases of Aquatic Organisms* 97(2):103-112.

Bottein, M.Y., L. Kashinsky, Z. Wang, C. Littnan, and J.S. Ramsdell. 2011. Identification of ciguatoxins in Hawaiian monk seals *Monachus schauinslandi* from the Northwestern and Main Hawaiian Islands. *Environmental Science & Technology* 45:5403-5409.

Boucher, O., G. Muckle, and C.H. Bastien. 2009. Prenatal exposure to polychlorinated biphenyls: A neuropsychologic analysis. *Environmental Health Perspectives* 117:7-16.

Bowen, W.D., J.D. Baker, D. Siniff, I.L. Boyd, R. Wells, J.K.B. Ford, S.D Kraus, J.A. Estes, and I. Stirling. 2010. Long-term studies. Pp. 283-305 in *Marine Mammal Ecology and Conservation: A Handbook of Techniques*, I. Boyd, D. Bowen, and S. Iverson, eds. Oxford: Oxford University Press.

Boyd, I.L. 2000. State-dependent fertility in pinnipeds: Contrasting capital and income breeders. *Functional Ecology* 14(5):623-630.

Boziari, A., C. Koukorava, E. Carinou, C.J. Hourdakis, and V. Kamenopoulou. 2010. The use of active personal dosemeters as a personal monitoring device: Comparison with TL dosimetry. *Radiation Protection Dosimetry* 144:173-176.

Bräger, S., K. Brensing, R. Caddell, K.C. Detloff, S. Dolman, P. Evans, V. Frank, J. Haelters, R. Kless, K. Lucke, L. Nunny, G. Pavan, M. Simmonds, and H. Westerberg. 2009. Final Report of the ASCOBANS Intersessional Working Group on the Assessment of Acoustic Disturbance, August 11, 2009. Available at http://www.ascobans.org/sites/default/files/basic_page_documents/AC17_4-08_ReportWGAcousticDisturbance.pdf.

Branch, T.A., K.M. Stafford, D.M. Palacios, C. Allison, J.L. Bannister, C.L.K. Burton, E. Cabrera, C.A. Carlson, B. Galletti Vernazzani, P.C. Gill, R. Hucke-Gaete, K.C.S. Jenner, M.-N.M. Jenner, K. Matsuoka, Y.A. Mikhalev, T. Miyashita, M.G. Morrice, S. Nishiwaki, V.J. Sturrock, D. Tormosov, R.C. Anderson, A.N. Baker, P.B. Best, P. Borsa, R.L. Brownell, Jr., S. Childerhouse, K.P. Findlay, T. Gerrodette, A.D. Ilangakoon, M. Joergensen, B. Kahn, D.K. Ljungblad, B. Maughan, R.D. McCauley, S. McKay, T.F. Norris, Oman Whale and Dolphin Research Group, R.M. Warneke, S. Rankin, F. Samaran, D. Thiele, and K. Van Waerebeek. 2007. Past and present distribution densities and movements of blue whales in the Southern Hemisphere and Northern Indian Ocean. *Mammal Review* 37:116-175.

Branchi, I., F. Capone, E. Alleva, and L.G. Costa. 2003. Polybrominated diphenyl ethers: Neurobehavioral effects following developmental exposure. *Neurotoxicology* 24(3):449-462.

Brandt, M.J., A. Diederichs, K. Betke, and G. Nehls. 2011. Responses of harbor porpoises to pile driving at the Horns Rev II offshore wind farm in the Danish North Sea. *Marine Ecology Progress Series* 421:205-216.

Brandt, M.J., C. Hoschle, A. Diederichs, K. Betke, R. Matuschek, and G. Nehls. 2013. Seal scarers as a tool to deter harbour porpoises from offshore construction sites. *Marine Ecology Progress Series* 475:291-302.

Brandt-Lassen, R., S.P. Lund, and G.B. Jepsen. 2000. Rats exposed to toluene and noise may develop loss of auditory sensitivity due to synergistic interaction. *Noise Health* 3:33-44.

Brault, S., and H. Caswell. 1993. Pod-specific demography of killer whales (*Orcinus orca*). *Ecology* 74:1444-1454.

Brenner, I., J. Castellani, C. Gabaree, A. Young, J. Zamecnik, R. Shephard, and P. Shek. 1999. Immune changes in humans during cold exposure: Effects of prior heating and exercise. *Journal of Applied Physiology* 87(2):699-710.

Brinton, E., V.J. Loeb, M.C. Macaulay, and E. Shulenberger. 1987. Variability of *Euphausia superba* populations near Elephant Island and the South Shetlands: 1981 vs. 1984. *Polar Biology* 7(6):345-362.

Brock, P.M., A.J. Hall, S.J. Goodman, M. Cruz, and K. Acevedo-Whitehouse. 2013a. Immune activity, body condition and human-associated environmental impacts in a wild marine mammal. *PloS ONE* 8(6):e67132.

Brock, P.M., A.J. Hall, S.J. Goodman, M. Cruz, and K. Acevedo-Whitehouse. 2013b. Applying the tools of ecological immunology to conservation: A test case in the Galapagos sea lion. *Animal Conservation* 16(1):19-31.

Brough, T.E., S. Henderson, M. Guerra, and S.M. Dawson. 2016. Factors influencing heterogeneity in female reproductive success in a critically endangered population of bottlenose dolphins. *Endangered Species Research* 29:255-270.

Brumm, H., and S.A. Zollinger. 2011. The evolution of the Lombard effect: 100 years of psychoacoustic research. *Behaviour* 148:1173-1198.

Buckland, S.T., and A.E. York. 2009. Abundance estimation. Pp. 1-5 in *Encyclopedia of Marine Mammals*, W.F. Perrin, B. Wursig and J.G.M. Thewissen, eds. San Diego, CA: Academic Press.

Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L. and L. Thomas. 2001. *Introduction to Distance Sampling: Estimating Abundance of Biological Populations.* Oxford: Oxford University Press.

Buckland, S.T., M.L. Burt, E. Rexstad, M. Mellor, A. Williams, and R. Woodward. 2012. Aerial surveys of seabirds: The advent of digital methods. *Journal of Applied Ecology* 49:960-967.

Buckstaff, K.C. 2004. Effects of watercraft noise on the acoustic behavior of bottlenose dolphins (*Tursiops truncatus*) in Sarasota Bay, Florida. *Marine Mammal Science* 20:709-725. doi: 10.1111/j.17487692.2004.tb01189.x.

Bunkley, J.P., and J.R. Barber. 2015. Noise reduces foraging efficiency in pallid bats (*Antrozous pallidus*). *Ethology* 121:1116-1121.

Burek, K.A., F. Gulland, and T.M. O'Hara. 2008. Effects of climate change on Arctic marine mammal health. *Ecological Applications* 18(sp2):S126-S134.

Burgin, S., and N. Hardiman. 2015. Effects of non-consumptive wildlife-oriented tourism on marine species and prospects for their sustainable management. *Journal of Environmental Management* 151:210-220.

Burns, W.C. 1997. The International Whaling Commission and future of cetaceans: Problems and prospects. *Colorado Journal of International Environmental Law and Policy* 8(1):31-88.

Caillat, M., and S. Smout. 2015. *Harbour Seal Decline: Population Modeling*. Sea Mammal Research Unit, University of St. Andrews, Report to Scottish Government, no. CSD 4, St. Andrews, 29 pp. Available at http://www.smru.standrews.ac.uk/documents/scotgov/CSD4_population_model_VF2.pdf.

Calabrese, E.J. 2005. Paradigm lost, paradigm found: The re-emergence of hormesis as a fundamental dose response model in the toxicological sciences. *Environmental Pollution* 138(3):378-411.

Calabrese, E.J., K.A. Bachmann, A.J. Bailer, P.M. Bolger, J. Borak, L. Cai, N. Cedergreen, M.G. Cherian, C.C. Chiueh, T.W. Clarkson, and R.R. Cook. 2007. Biological stress response terminology: Integrating the concepts of adaptive response and preconditioning stress within a hormetic dose–response framework. *Toxicology and Applied Pharmacology* 222(1):122-128.

Calvert, W., and M.A. Ramsay. 1998. Evaluation of age determination of polar bears by counts of cementum growth layer groups. *Ursus* 449-453.

Caro, T.M., and G. O'Doherty. 1999. On the use of surrogate species in conservation biology. *Conservation Biology* 13(4):805-814.

Carpenter, D.O. 2006. Polychlorinated biphenyls (PCBs): Routes of exposure and effects on human health. *Reviews on Environmental Health* 21(1):1-23.

Carretta, J.V., K. Danil, S.J. Chivers, D.W. Weller, D.S. Janiger, M. Berman-Kowalewski, K.M. Hernandez, J.T. Harvey, R.C. Dunkin, D.R. Casper, S. Stoudt, M. Flannery, K. Wilkinson, J. Huggins, and D.M. Lambourn. 2015. Recovery rates of bottlenose dolphin (*Tursiops truncatus*) carcasses estimated from stranding and survival rate data. *Marine Mammal Science* 32:349-362.

Cassoff, R.M., K.M. Moore, W.A. McLellan, S.G. Barco, D.S. Rotstein, and M.J. Moore. 2011. Lethal entanglement in baleen whales. *Diseases of Aquatic Organisms* 96:175-185.

Castellote, M., C.W. Clark, and M.O. Lammers. 2012. Acoustic and behavioural changes by fin whales (*Balaenoptera physalus*) in response to shipping and airgun noise. *Biological Conservation* 147(1):115-122.

Caswell, H. 2001. *Matrix Population Models,* 2nd Ed. Sunderland, MA: Sinauer Associates.

Caswell, H., M. Fujiwara, and S. Brault. 1999. Declining survival probability threatens the North Atlantic right whale. *Proceedings of the National Academy of Sciences of the United States of America* 96(6):3308-3313.

Celi, M., F. Filiciotto, M. Vazzana, V. Arizza, V. Maccarrone, M. Ceraulo, S. Mazzola, and G. Buscaino. 2015. Shipping noise affecting immune responses of European spiny lobster (*Palinurus elephas*). *Canadian Journal of Zoology* 93(2):113-121.

Champagne, C., D. Crocker, M. Fowler, and D. Houser. 2012. Fasting physiology of the pinnipeds: The challenges of fasting while maintaining high energy expenditure and nutrient delivery for lactation. Pp. 309-336 in *Comparative Physiology of Fasting, Starvation, and Food Limitation,* M. D. McCue, ed. Berlin: Springer.

Chan, A.A., Y.H. Chan, W.D. Stahlman, D. Garlick, C.D. Fast, D.T. Blumstein, and A.P. Blaisdell. 2010. Increased amplitude and duration of acoustic stimuli enhance distraction. *Animal Behaviour* 80:1075-1079.

Christiansen, F., and D. Lusseau. 2015. Linking behavior to vital rates to measure the effects of non-lethal disturbance on wildlife. *Conservation Letters* 8(6):424-431.

Christiansen, F., G.A. Vikingsson, M.H. Rasmussen, and D. Lusseau. 2014. Female body condition affects foetal growth in a capital breeding mysticete. *Functional Ecology* 28:579-588.

Claridge, D.E. 2013. Population Ecology of Blainville's Beaked Whales (*Mesoplodon densirostris*). Doctoral dissertation, University of St. Andrews.

Clark, C.W., W.T. Ellison, B.L. Southall, L.T. Hatch, S.M. Van Parijs, A. Frankel, and D. Ponirakis. 2009. Acoustic masking in marine ecosystems: Intuitions, analysis, and implications. *Marine Ecology: Progress Series* 395:201-222.

Cohen, J. 1991. Was underwater "shot" harmful to the whales? *Science* 252(5008):912.

Conn, P.B., and G.K. Silber. 2013. Vessel speed restrictions reduce risk of collision-related mortality for North Atlantic right whales. *Ecosphere* 4(4):43.

Connon, R.E., J. Geist, and I. Werner. 2012. Effect-based tools for monitoring and predicting the ecotoxicological effects of chemicals in the aquatic environment. *Sensors* 12(9):12741-12771.

Constantine, R. 2001. Increased avoidance of swimmers by wild bottlenose dolphins (*Tursiops truncatus*) due to long-term exposure to swim-with-dolphin tourism. *Marine Mammal Science* 17(4):689-702.

Conversi, A., V. Dakos, A. Gårdmark, S. Ling, C. Folke, P.J. Mumby, C. Greene, M. Edwards, T. Blenckner, M. Casini, A. Pershing, and C. Möllmann. 2015. A holistic view of marine regime shifts. *Philosophical Transactions of the Royal Society B* 370:20130279. doi: 10.1098/rstb.2013.0279.

Cook, P.F., C. Reichmuth, A.A. Rouse, L.A. Libby, S.E. Dennison, O.T. Carmichael, K.T. Kruse-Elliott, J. Bloom, B. Singh, V.A. Fravel, L. Barbosa, J.J. Stuppino, W.G. Van Bonn, F.M.D. Gulland, and C. Ranganath. 2015. Algal toxin impairs sea lion memory and hippocampal connectivity, with implications for strandings. *Science* 350(6267):1545-1547.

Cooke, R.M. 1991. *Experts in Uncertainty: Opinion and Subjective Probability in Science.* New York: Oxford University Press.

Cooke, S.J., S.S. Killen, J.D. Metcalfe, D.J. McKenzie, D. Mouillot, C. Jørgensen, and M.A. Peck. 2014. Conservation physiology across scales: Insights from the marine realm. *Conservation Physiology* 2(1):cou024.

Cordes, L. 2011. Demography and Breeding Phenology of a Marine Top Predator. PhD thesis, University of Aberdeen.

Costantini, D., A. Fanfani, and G. Dell'Omo. 2008. Effects of corticosteroids on oxidative damage and circulating carotenoids in captive adult kestrels (*Falco tinnunculus*). *Journal of Comparative Physiology B* 178(7):829-835.

Costantini, D., V. Marasco, and A.P. Møller. 2011. A meta-analysis of glucocorticoids as modulators of oxidative stress in vertebrates. *Journal of Comparative Physiology B* 181(4):447-456.

Côté, I.M., E.S. Darling, and C.J. Brown. 2016. Interactions among ecosystem stressors and their importance in conservation. *Proceedings of the Royal Society B: Biological Sciences* 283:2015-2592.

Cote, J., S. Meylan, J. Clobert, and Y. Voituron. 2010. Carotenoid-based coloration, oxidative stress and corticosterone in common lizards. *Journal of Experimental Biology* 213(12):2116-2124.

Cox, T.M., T.J. Ragen, A.J. Read, E. Vos, R.W. Baird, K. Balcomb, J. Barlow, J. Caldwell, T. Cranford, L. Crum, A. D'Amico, G. D'Spain, A. Fednandez, J. Finneran, R. Gentry, W. Gerth, F. Gulland, J. Hildebrand, D. Houser, T. Hullar, P.D. Jepson, D. Ketten, C.D. MacLeod, P. Miller, S. Moore, D.C. Mountain, D. Palka, P. Ponganis, S. Rommel, T. Rowles, B. Taylor, P. Tyack, D. Wartzok, R. Gisiner, J. Mead, and L. Benner. 2006. Understanding the impacts of anthropogenic sound on beaked whales. *Journal of Cetacean Research Management* 7:177-187.

Crain, C.M., K. Kroeker, and B.S. Halpern. 2008. Interactive and cumulative effects of multiple human stressors in marine systems. *Ecology Letters* 11:1304-1315.

Cranford, T.W., and P. Krysl. 2015. Fin whale sound reception mechanisms: Skull vibration enables low-frequency hearing. *PLoS ONE* 10:1-17.

Creel, S., D. Christianson, S. Liley, and J.A. Winnie. 2007. Predation risk affects reproductive physiology and demography of elk. *Science* 315:960.

Creel, S., J.A. Winnie, and D. Christianson. 2009. Glucocorticoid stress hormones and the effect of predation risk on elk reproduction. *Proceedings of the National Academy of Sciences of the United States of America* 106:12388-12393.

Crocker, D.E., B.J. Le Boeuf, and D.P. Costa. 1997. Drift diving in female northern elephant seals: Implications for food processing. *Canadian Journal of Zoology* 75:27-39.

Crocker, D.E., D.P. Costa, B.J. Le Boeuf, P.M. Webb, and D.S. Houser. 2006. Impact of El Niño on the foraging behavior of female northern elephant seals. *Marine Ecology Progress Series* 309:1-10.

Crofton, K.M., D.-L. Ding, R. Padich, M. Taylor, and D. Henderson. 2000. Hearing loss following exposure during development to polychlorinated biphenyls: A cochlear site of action. *Hearing Research* 144(1-2):196-204.

Croll, D.A., C.W. Clark, A. Acevedo, B. Tershy, S. Flores, J. Gedamke, and J. Urban. 2002. Only male fin whales sing loud songs. *Nature* 417:809.

Croll, D.A., B. Marinovic, S. Benson, F.P. Chavez, N. Black, R. Ternullo, and B.R. Tershy. 2005. From wind to whales: Trophic links in a coastal upwelling system. *Marine Ecology: Progress Series* 289:117-130.

Croxall, J.P., T. Callaghan, R. Cervellati, and D.W.H. Walton. 1992. Southern Ocean environmental changes: Effects on seabird, seal and whale populations [and discussion]. *Philosophical Transactions of the Royal Society of London B: Biological Sciences* 338(1285):319-328.

Cumeras, R.H.K., W. Cheung, F. Gulland, D. Goley, and C.E. Davis. 2014. Chemical analysis of whale breath volatiles: A case study for non-invasive field health diagnostics of marine mammals. *Metabolites* 4(3):790-806.

Curé, C., R. Antunes, A.C. Alves, F. Visser, P.H. Kvadsheim, and P.J. Miller. 2013. Responses of male sperm whales (*Physeter macrocephalus*) to killer whale sounds: Implications for anti-predator strategies. *Scientific Reports* 3:1579.

Dähne, M., A. Gilles, K. Lucke, V. Peschko, S. Adler, K. Krügel, J. Sundermeyer, and U. Siebert. 2013. Effects of pile-driving on harbour porpoises (*Phocoena phocoena*) at the first offshore wind farm in Germany. *Environmental Research Letters* 8:025002.

Dai, L., D. Vorselen, K.S. Korolev, and J. Gore. 2012. Generic indicators for loss of resilience before a tipping point leading to population collapse. *Science* 336(6085):1175-1177.

Dakos, V., S.R. Carpenter, E.H. van Nes, and M. Scheffer. 2015. Resilience indicators: Prospects and limitations for early warnings of regime shifts. *Philosophical Transactions of the Royal Society B* 370:1659. doi: 10.1098/rstb.2013.0263.

D'Amico, A.D., R. Gisiner, D.R. Ketten, J.A. Hammock, C. Johnson, P. Tyack, and J. Mead. 2009. Beaked whale strandings and naval exercises. *Aquatic Mammals* 35(4):452-472.

Dantzer, B., Q.E. Fletcher, R. Boonstra, and M.J. Sheriff. 2014. Measures of physiological stress: A transparent or opaque window into the status, management and conservation of species? *Conservation Physiology* 2(1):cou023.

Darling, E.S., and E.M. Côté. 2008. Quantifying the evidence for ecological synergies. *Ecology Letters* 11:1278-1286.

Daszak, P., A.A. Cunningham, and A.D. Hyatt. 2000. Emerging infectious diseases of wildlife—threats to biodiversity and human health. *Science* 287(5452):443-449.

Dawson, S.M., S. Northridge, D. Waples, and A.J. Read. 2013. To ping or not to ping: The use of active acoustic devices in mitigating interactions between small cetaceans and gillnet fisheries. *Endangered Species Research* 19:201-221.

De Guise, S., D. Martineau, P. Beland, and M. Fournier. 1998. Effects of in vitro exposure of beluga whale leukocytes to selected organochlorines. *Journal of Toxicology and Environmental Health, Part A: Current Issues* 55:479-493.

de Little, S.C., C.J.A. Bradshaw, C.R. McMahon, and M.A. Hindell. 2007. Complex interplay between intrinsic and extrinsic drivers of long-term survival trends in southern elephant seals. *BMC Ecology* 7:3. doi: 10.1186/1472-6785-7-3.

De Swart, R.L., P.S. Ross, J.G. Vos, and A.D. Osterhaus. 1996. Impaired immunity in harbour seals (*Phoca vitulina*) exposed to bioaccumulated environmental contaminants: Review of a long-term feeding study. *Environmental Health Perspectives* 104(Suppl. 4):823-828.

De Vries, R.D., S. McQuaid, G. Van Amerongen, S. Yüksel, R.J. Verburgh, A.D. Osterhaus, W.P. Duprex, and R.L. De Swart. 2012. Measles immune suppression: Lessons from the macaque model. *PLoS Pathogens* 8(8):e1002885.

DeMaster, D.P., A.W. Trites, P. Clapham, S. Mizrock, P. Wade, R.J. Small, and J. Ver Hoef. 2006. The sequential megafauna collapse hypothesis: Testing with existing data. *Progress in Oceanography* 68:329-342.

DeRijk, R.H., E.M. Sternberg, and E.R. De Kloet. 1997. Glucocorticoid receptor function in health and disease. *Current Opinion in Endocrinology, Diabetes and Obesity* 4(3)185-193.

DeRuiter, S.L., B.L. Southall, J. Calambokidis, W.M.X. Zimmer, D. Sadykova, E.A. Falcone, A.S. Friedlaender, J.E. Joseph, D. Moretti, G.S. Schorr, L. Thomas, and P.L. Tyack. 2013. First direct measurements of behavioral responses by Cuvier's beaked whales to mid-frequency active sonar. *Biology Letters* 9. doi: 10.1098/rsbl.2013.0223.

Devineau, O., R. Choquet, and J.-D. Lebreton. 2006. Planning capture-recapture studies: Straightforward precision, bias and power calculations. *Wildlife Society Bulletin* 34:1028-1035.

Di Poi, C., S. Atkinson, A. Hoover-Miller, and G. Blundell. 2015. Maternal buffering of stress response in free-ranging Pacific harbor seal pups in Alaska. *Marine Mammal Science* 31:1098-1117.

Diamanti-Kandarakis, E., J.P. Bourguignon, L.C. Giudice, R. Hauser, G.S. Prins, A.M. Soto, R.T. Zoeller, and A.C. Gore. 2009. Endocrine-disrupting chemicals: An Endocrine Society scientific statement. *Endocrine Reviews* 30:293-342.

Diaz, R.J., and R. Rosenberg. 2008. Spreading dead zones and consequences for marine ecosystems. *Science* 321(5891):926-929.

Dickens, M.J., and L.M. Romero. 2013. A consensus endocrine profile for chronically stressed wild animals does not exist. *General and Comparative Endocrinology* 191:177-189.

Dickens, M.J., D.J. Delehanty, and L.M. Romero. 2009. Stress and translocation: Alterations in the stress physiology of translocated birds. *Proceedings of the Royal Society B: Biological Sciences* 276:1664. doi: 10.1098/rspb.2008.1778.

Dietz, R., F. Riget, and P. Johansen. 1996. Lead, cadmium, mercury and selenium in Greenland marine animals. *Science of the Total Environment* 186(1):67-93.

Dietz, R., P.M. Outridge, and K.A. Hobson. 2009. Anthropogenic contributions to mercury levels in present-day Arctic animals—A review. *Science of the Total Environment* 407(24):6120-613.

Dietz, R., C. Sonne, N. Basu, B. Braune,T. O'Hara, R.J. Letcher, T. Scheuhammer, M. Andersen, C. Andreasen, D. Andriashek, and G. Asmund. 2013. What are the toxicological effects of mercury in Arctic biota? *Science of the Total Environment* 443:775-790.

Doak, D.F., J.A. Estes, B.S. Halpern, U. Jacob, D.R. Lindberg, J. Lovvorn, D.H. Monson, M.T. Tinker, T.M. Williams, J.T. Wootton, I. Carroll, M. Emmerson, F. Micheli, and M. Novak. 2008. Understanding and predicting ecological dynamics: Are major surprises inevitable? *Ecology* 89(4):952-961.

Donaldson, R., H. Finn, and M. Calver. 2010. Illegal feeding increases risk of boat-strike and entanglement in bottlenose dolphins in Perth, Western Australia. *Pacific Conservation Biology* 16(3):157-161.

Doney, S.C., M. Ruckelshaus, J.E. Duffy, J.P. Barry, F. Chan, C.A. English, H.M. Galindo, J.M. Grebmeier, A.B. Hollowed, N. Knowlton, and J. Polovina. 2012. Climate change impacts on marine ecosystems. *Marine Science* 4:11-37.

Doniol-Valcroze, T., D. Berteaux, P. Larouche, and R. Sears. 2007. Influence of thermal fronts on habitat selection by four rorqual whale species in the Gulf of St. Lawrence. *Marine Ecology: Progress Series* 335:207-216.

Donovan, C.R., C. Harris, J. Harwood, and L. Milazzo. 2013. A simulation-based method for quantifying and mitigating the effects of anthropogenic sound on marine mammals. *Proceedings of Meetings on Acoustics* 17:070043.

Doroff, A.M., J.A. Estes, M.T. Tinker, D.M. Burn, and T.J. Evans. 2003. Sea otter population declines in the Aleutian archipelago. *Journal of Mammalogy* 84(1):55-64.

Dowling, D.K. and L.W. Simmons. 2009. Reactive oxygen species as universal constraints in life-history evolution. *Proceedings of the Royal Society B: Biological Sciences* 276(1663):1737-1745.

Downs, C.J., and N.A. Dochtermann. 2014. Testing hypotheses in ecoimmunology using mixed models: Disentangling hierarchical correlations. *Integrative and Comparative Biology* 54(3):407-418.

Drake, J.M., and B.D. Griffen. 2010. Early warning signals of extinction in deteriorating environments. *Nature* 467(7314):456-459.

Dunne, J.A., R.J. Williams, and N.D. Martinez. 2002. Food-web structure and network theory: The role of connectance and size. *Proceedings of the National Academy of Sciences of the United States of America* 99:12917-12922.

Durban, J., H. Fearnbach, L. Barret-Lennard, W. Perryman, and D. LeRoi. 2015. Photogrammetry of killer whales using a small hexacopter launched at sea. *21st Biennial Conference on the Biology of Marine Mammals*. San Francisco, CA, December 2015.

Durbin, E., G. Teegarden, R. Campbell, A. Cembella, M.F. Baumgartner, and B.R. Mate. 2002. North Atlantic right whales, *Eubalaena glacialis*, exposed to paralytic shellfish poisoning (PSP) toxins via a zooplankton vector, *Calanus finmarchicus*. *Harmful Algae* 1:243-251.

Eales, J.G. 1988. The influence of nutritional state on thyroid function in various vertebrates. *American Zoologist* 28:351-362.

Eberhardt, L.L. 2002. A paradigm for population analysis of long-lived vertebrates. *Ecology* 83:2841-2854.

Eberle, K.C., T.E. Waters, E.D. Jensen, S.K. Venn-Watson, and R.E. Sacco. 2013. Development and application of specific cytokine assays in tissue samples from a bottlenose dolphin with hyperinsulinemia. *Frontiers in Endocrinology* 4:134.

Ellison, W.T., B.L. Southall, C.W. Clark, and A.S. Frankel. 2011. A new context-based approach to assess marine mammal behavioral responses to anthropogenic sounds. *Conservation Biology* 26:21-28.

Engås, A., S. Løkkeborg, E. Ona, and A.V. Soldal. 1996. Effects of seismic shooting on local abundance and catch rates of cod (*Gadus morhua*) and haddock (*Melanogrammus aeglefinus*). *Canadian Journal of Fisheries and Aquatic Sciences* 53(10):2238-2249.

Engelhard, G.H., A.N.J. Baarspul, M. Broekman, J.C.S. Creuwels, and P.J.H. Reijnders. 2002. Human disturbance, nursing behaviour, and lactational pup growth in a declining southern elephant seal (*Mirounga leonina*) population. *Canadian Journal of Zoology* 80(11):876-1886.

EPA (U.S. Environmental Protection Agency). 2000. *Supplementary Guidance for Conducting Risk Assessment of Chemical Mixtures*. EPA/630/R-00/002, August 2000.

EPA. 2002. *Guidance on Cumulative Risk Assessment of Pesticide Chemicals That Have a Common Mechanism of Toxicity*. Office of Pesticide Programs, January 14, 2002.

EPA. 2007. *Concepts, Methods and Data Sources for Cumulative Health Risk Assessment of Multiple Chemicals, Exposures and Effects: A Resource Document.* EPA/600/R-06/013F. National Center for Environmental Assessment, Office of Research and Development, U.S. Environmental Protection Agency, Cincinnati, OH. http://ofmpub.epa.gov/eims/eimscomm.getfile?p_download_id=474337.

Erbe, C. 2013. International regulation of underwater noise. *Acoustics Australia* 41:12-19.

Ercal, N., H. Gurer-Orhan, and N. Aykin-Burns. 2001. Toxic metals and oxidative stress part I: Mechanisms involved in metal-induced oxidative damage. *Current Topics in Medicinal Chemistry* 1(6):529-539.

Eriksson, P., E. Jakobsson, and A. Fredriksson. 2001. Brominated flame retardants: A novel class of developmental neurotoxicants in our environment? *Environmental Health Perspectives* 109(9):903-908.

Estes, J.A., M. Tinker, T.M. Williams, and D.F. Doak. 1998. Killer whale predation on sea otters linking oceanic and nearshore ecosystems. *Science* 282(5388):473-476.

Estes, J.A., D.F. Doak, A.M. Springer, and T.M. Williams. 2009a. Causes and consequences of marine mammal population declines in southwest Alaska: A food-web perspective. *Philosophical Transactions of the Royal Society of London, Series B: Biological Sciences* 364(May):1647-1658.

Estes, J.A., D.F. Doak, A.M. Springer, T.M. Williams, and G.B. van Vliet. 2009b. Trend data do support the sequential nature of pinniped and sea otter declines in the North Pacific, but does it really matter? *Marine Mammal Science* 25:748-754.

Estes, J.A., J.S. Brashares, and M.E. Power. 2013a. Predicting and detecting reciprocity between indirect ecological interactions and evolution. *The American Naturalist* 181(Suppl.):S76-S99.

Estes, J.A., R.S. Steneck, and D.R. Lindberg. 2013b. Exploring the consequences of species interactions through the assembly and disassembly of food webs: A Pacific/Atlantic comparison. *Bulletin of Marine Science* 89:11-29.

European Union. 2008. Directive 2008/56/Ec of European Parliament and of The Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive). *Official Journal of the European Union, L: Legislation* 164:19-40.

Evans, D.L., and G.R. England. 2001. *Joint Interim Report Bahamas Marine Mammal Stranding Event of 14-16 March 2000.* Washington, DC: U.S. Department of Commerce and U.S. Navy. Available at http://www.nmfs.noaa.gov/pr/pdfs/health/stranding_bahamas2000.pdf.

Fahlman, A., S.K. Hooker, A. Olszowka, B.L. Bostrom, and D.R. Jones. 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.

Fahlman, A., P.L. Tyack, P.J. Miller, and P.H. Kvadsheim. 2014. How man-made interference might cause gas bubble emboli in deep diving whales. *Frontiers in Physiology* 5:13.

Fahlman, A., J. van der Hoop, M.J. Moore, G. Levine, J. Rocho-Levine, and M. Brodsky. 2016. Estimating energetics in cetaceans from respiratory frequency: Why we need to understand physiology. *Biology Open* 5(4):436-442. doi: 10.1242/bio.017251.

Fair, P.A., T.C. Hulsey, R.A. Varela, and J.D. Goldstein. 2006. Hematology, serum chemistry, and cytology findings from apparently healthy Atlantic bottlenose dolphins (*Tursiops truncatus*) inhabiting the estuarine waters of Charleston, South Carolina. *Aquatic Mammals* 32(2):182.

Falcone, E.A., G.S. Schorr, A.B. Douglas, J. Calambokidis, E. Henderson M.F. McKenna, J. Hildebrand, and D. Moretti. 2009. Sighting characteristics and photo-identification of Cuvier's beaked whales (*Ziphius cavirostris*) near San Clemente Island, California: A key area for beaked whales and the military? *Marine Biology* 156:2631-2640.

Falke, K.J., R.D. Hill, J. Qvist, R.C. Schneider, M. Guppy, G.C. Liggins, P.W. Hochachka, R.E. Elliott, and W.M. Zapol. 1985. Seal lungs collapse during free diving: Evidence from arterial nitrogen tensions. *Science* 229(4713):556-558. doi: 10.1126/science.4023700.

Farina, M., J.B. Rocha, and M. Aschner. 2011. Mechanisms of methylmercury induced neurotoxicity: evidence from experimental studies. *Life Sciences* 89(15-16):553-563. doi: 10.1016/j.lfs.2011.05.019.Epub 2011 Jun 13.

Fearnbach, H., J. Durban, D. Ellifrit, K. Balcomb, and L. Barre. 2015. Aerial photogrammetry reveals declining body condition and impaired reproduction in endangered Southern Resident killer whales. *21st Biennial Conference on the Biology of Marine Mammals, San Francisco, CA, December 2015.*

Feller, W. 1968. *Introduction to Probability Theory and Its Application.* New York: John Wiley & Sons.

Ferguson, S.H., I. Stirling, and P. McLoughlin. 2005. Climate change and ringed seal (*Phoca hispida*) recruitment in western Hudson Bay. *Marine Mammal Science* 21(1):121-135.

Fernández, A., J.F. Edwards, F. Rodriguez, A.E. De Los Monteros, P. Herraez, P. Castro, J.R. Jaber, V. Martin, and M. Arbelo. 2005. Gas and fat embolic syndrome involving a mass stranding of beaked whales (family Ziphiidae) exposed to anthropogenic sonar signals. *Veterinary Pathology* 42(4):446-457.

Fernández, A., E. Sierra, V. Martín, M. Méndez, S. Sacchini, Y. Bernaldo de Quiros, M.Andrada, M. Rivero, O. Quesada, M. Tejedor, and M. Arbelo. 2012. Last "atypical" beaked whales mass stranding in the Canary Islands (July, 2004). *Journal of Marine Science: Research and Development* 2:107.

Finneran, J.J. 2016. Auditory weighting functions and TTS/PTS exposure functions for marine mammals exposed to underwater noise. Pp. 38-110 in National Marine Fisheries Service. Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing: Underwater Acoustic Thresholds for Onset of Permanent and Temporary Threshold Shifts. U.S. Department of Commerce, NOAA. *NOAA Technical Memorandum.* NMFS-OPR-55.

Finneran, J.J., and D.S. Houser. 2007. Bottlenose dolphin (*Tursiops truncatus*) steady-state evoked responses to multiple simultaneous sinusoidal amplitude modulated tones. *Journal of the Acoustical Society of America* 121:1775-1782.

Finneran, J.J., and A.K. Jenkins. 2012. *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis.* Technical Report. Available at http://www.dtic.mil/dtic/tr/fulltext/u2/a561707.pdf.

Finneran, J.J., and C.E. Schlundt. 2004. *Effects of Intense Pure Tones on the Behavior of Trained Odontocetes*. Technical Report 1913. SPAWAR Systems Center, San Diego, CA. Available at https://ntrl.ntis.gov/NTRL/dashboard/searchResults/titleDetail/ADA424426.xhtml.

Fire, S.E., Z. Wang, T.A. Leighfield, S.L. Morton, W.E. McFee, W.A. McLellan, R.W. Litaker, P.A. Tester, A.A. Hohn, G. Lovewell, C. Harms, D.S. Rotstein, S.G. Barco, A. Costidis, B. Sheppard, G.D. Bossart, M. Stolen, W.N. Durden, and F.M. Van Dolah. 2009. Domoic acid exposure in pygmy and dwarf sperm whales (*Kogia spp.*) from southeastern and mid-Atlantic U.S. waters. *Harmful Algae* 8(5):658.

Fisher, R.A. 1937. *The Design of Experiments,* 9th Ed. London: Macmillan.

Fleshler, M. 1965. Adequate acoustic stimulus for startle reaction in the cat. *Journal of Comparative Physiological Psychology* 60:200-207.

Fletcher, H. 1940. Auditory patterns. *Reviews of Modern Physics* 12:47-65.

Flewelling, L.J., J.P. Naar, J.P. Abbott, D.G. Baden, N.B. Barros, G.D. Bossart, M.Y.D. Bottein, D.G. Hammond, E.M. Haubold, C.A. Heil, M.S. Henry, H.M. Jacocks, T.A. Leighfield, R.H. Pierce, T.D. Pitchford, S.A. Rommel, P.S. Scott, K.A. Steidinger, E.W. Truby, F.M. Van Dolah, and J.H. Landsberg. 2005. Red tides and marine mammal mortalities. *Nature* 435:755-756.

Folt, C.L., C.Y. Chen, M.V. Moore, and J. Burnaford. 1999. Synergism and antagonism among multiple stressors. *Limnology and Oceanography* 44(3):865-877.

Forcada, J., P.N. Trathan, K. Reid, E.J. Murphy, and J.P. Croxall. 2006. Contrasting population changes in sympatric penguin species in association with climate warming. *Global Change Biology* 12(3):411-423.

Ford, J.K.B., G.M. Ellis, P.F. Olesiuk, and K.C. Balcomb. 2010. Linking killer whale survival and prey abundance: Food limitation in the oceans' apex predator? *Biology Letters* 6:139-142.

Ford, M.J. 2013. *Status Review Update of Southern Resident Killer Whales.* Conservation Biology Division Northwest Fisheries Science Center, National Marine Fisheries Service.

Forney, K.A. 2000. Environmental models of cetacean abundance: Reducing uncertainty in population trends. *Conservation Biology* 14:1271-1286.

Fowles, J.R., A. Fairbrother, M. Fix, S. Schiller, and N. Kerkvliet. 1993. Glucocorticoid effects on natural and humoral immunity in mallards. *Developmental & Comparative Immunology* 17(2):165-177.

Francis, C.D. 2015. Vocal traits and diet explain avian sensitivities to anthropogenic noise. *Global Change Biology* 21:1809-1820.

Francis, C.D., and J.R. Barber. 2013. A framework for understanding noise impacts on wildlife: An urgent conservation priority. *Frontiers in Ecology and the Environment* 11:305-313.

Francis, C.D., C.P. Ortega, and A. Cruz. 2009. Noise pollution changes avian communities and species interactions. *Current Biology* 19:1415-1419.

Francis, C.D., C.P. Ortega, R.I. Kennedy, and P.J. Nylander. 2012a. Are nest predators absent from noisy areas or unable to locate nests? *Ornithological Monographs* 74:101-110.

Francis, C.D., N.J. Kleist, C.P. Ortega, and A. Cruz. 2012b. Noise pollution alters ecological services: Enhanced pollination and disrupted seed dispersal. *Proceedings of the Royal Society B: Biological Sciences* 279:2727-2735.

Francis, R.C., and S.R. Hare. 1994. Decadal-scale regime shifts in the large marine ecosystems of the north-east Pacific: A case for historical science. *Fisheries Oceanography* 3(4):279-291.

Frankel, A.S., W.T. Ellison, and J. Buchanan. 2002. Application of the Acoustic Integration Model (AIM) to predict and minimize environmental impacts. *Oceans'02 MTS/IEEE* 3:1438-1443.

Frantzis, A. 1998. Does acoustic testing strand whales? *Nature* 392(6671):29.

French, S.S., M. González-Suárez, J.K. Young, S. Durham, and L.R. Gerber. 2011. Human disturbance influences reproductive success and growth rate in California sea lions (*Zalophus californianus*). *PLoS ONE* 6(3):e17686.

Frid, A., and L.M. Dill. 2002. Human-caused disturbance stimuli as a form of predation risk. *Conservation Ecology* 6(1):11.

Friedlaender, A.S., E.L. Hazen, J.A. Goldbogen, A.K. Stimpert, J. Calambokidis, and B.L. Southall. 2016. Prey-mediated behavioral responses of feeding blue whales in controlled sound exposure experiments. *Ecological Applications* 26(4):1075-1085.

Frisén, M. 2009. Optimal sequential surveillance for finance, public health and other areas. *Sequential Analysis: Design Methods and Applications* 28:310-337.

Fritts, H.C. 2012. *Tree Rings and Climate*. New York: Elsevier.

Fromm, D.M. 2009. Reconstruction of acoustic exposure on orcas in Haro Strait. *NRL Review* 2009:127-129.

Fuente, A., and B. McPherson. 2006. Organic solvents and hearing loss: The challenge for audiology. *International Journal of Audiology* 45:367-381.

Fujiwara, M., and H. Caswell. 2001. Demography of the endangered North Atlantic right whale. *Nature* 414(6863):537-541.

Gardner, S.C. and S. Chávez-Rosales. 2000. Changes in the relative abundance and distribution of gray whales (*Eschrichtius robustus*) in Magdalena Bay, Mexico during an El Niño event. *Marine Mammal Science* 16(4):728-738.

Gascon, M., E. Morales, J. Sunyer, and M. Vrijheid. 2013. Effects of persistent organic pollutants on the developing respiratory and immune systems: A systematic review. *Environmental International* 52:51-56.

Geiger, A.C., and C.J. Jeffries. 1987. Evaluation of seal harassment techniques to protect gill netted salmon. *Acoustical Deterrents in Marine Mammal Conflicts with Fisheries* 37−55. Workshop held February 17-18, 1986, Newport, Oregon.

Gennings, C., W.H. Carter, R.A. Carchman, L.K. Teuschler, J.F. Simmons, and E.W. Carney. 2005. A unifying concept for assessing toxicological interactions: Changes in slope. *Toxicological Sciences* 88(2):287-297.

George, J.C., M.L. Druckenmiller, K.L. Laidre, R. Suydam, and B. Person. 2015. Bowhead whale body condition and links to summer sea ice and upwelling in the Beaufort Sea. *Progress in Oceanography* 136:250-262.

Gerrodette, T. and W.G. Gilmartin. 1990. Demographic consequences of changed pupping and hauling sites of the Hawaiian monk seal. *Conservation Biology* 4(4):423-430.

Gibbons, A. 1990. What's the sound of one ocean warming? *Science* 248(4951):33-34.

Gill, J.A., K. Norris, and W.J. Sutherland. 2001. Why behavioural responses may not reflect the population consequences of human disturbance. *Biological Conservation* 97:265-268.

Gillespie, A. 2010. Noise pollution, the oceans, and the limits of international law. *Yearbook of International Environmental Law* 21:114-139.

Gisiner, R., S. Harper, E. Livingston, and J. Simmen. 2006. Effects of Sound on the Marine Environment (ESME): An underwater noise risk model. *IEEE Journal of Oceanic Engineering* 138(4):1067-1081.

Gobush, K.S., R.K. Booth, and S.K. Wasser. 2014. Validation and application of noninvasive glucocorticoid and thyroid hormone measures in free-ranging Hawaiian monk seals. *General and Comparative Endocrinology* 195:174-182.

Goetz, K.T., D.J. Rugh, A.J. Read, and R.C. Hobbs. 2007. Habitat use in a marine ecosystem: Beluga whales *Delphinapterus leucas* in Cook Inlet, Alaska. *Marine Ecology Progress Series* 330:247-256.

Goldbogen, J.A., B.L. Southall, S.L. DeRuiter, J. Calambokidis, A.S. Friedlaender, E.L. Hazen, E.A. Falcone, G.S. Schorr, A. Douglas, D.J. Moretti, C. Kyburg, M.F. McKenna, and P.L. Tyack. 2013. Blue whales respond to simulated mid-frequency military sonar. *Proceedings of the Royal Society B: Biological Sciences* 280(1765):20130657.

Goldstein, T., J.A. Mazet, T.S. Zabka, G. Langlois, K.M. Colegrove, M. Silver, S. Bargu, F. Van Dolah, T. Leighfield, P.A. Conrad, J. Barakos, D.C. Williams, S. Dennison, M. Haulena, and F.M. Gulland. 2008. Novel symptomatology and changing epidemiology of domoic acid toxicosis in California sea lions (*Zalophus californianus*): An increasing risk to marine mammal health. *Proceedings of the Royal Society B: Biological Sciences* 275:267-276.

Goldstein, T., T.S. Zabka, R.L. Delong, E.A. Wheeler, G. Ylitalo, S. Bargu, M. Silver, T. Leighfield, F. Van Dolah, G. Langlois, I. Sidor, J.L. Dunn, and F.M. Gulland. 2009. The role of domoic acid in abortion and premature parturition of California sea lions (*Zalophus californianus*) on San Miguel Island, California. *Journal of Wildlife Diseases* 45:91-108.

Goldstein, T., I. Mena, S.J. Anthony, R. Medina, P.W. Robinson, D.J. Greig, D.P. Costa, W.I. Lipkin, A. Garcia-Sastre, and W.M. Boyce. 2013. Pandemic H1N1 influenza isolated from free-ranging northern elephant seals in 2010 off the central California coast. *PLoS ONE* 8(5):e62259.

Gordine, S.A., Fedak, M., and L. Boehme. 2015. Fishing for drifts: detecting buoyancy changes of a top marine predator using a step-wise filtering method. *Journal of Experimental Biology* 218:3816-3824.

Gotelli, N.J. 2008. *A Primer of Ecology*, 4th Ed. Sunderland, MA: Sinauer Associates.

Götz, T., and V.M. Janik. 2011. Repeated elicitation of the acoustic startle reflex leads to sensitisation in subsequent avoidance behaviour and induces fear conditioning. *BMC Neuroscience* 12:30.

Götz, T., and V.M. Janik. 2013. Acoustic deterrent devices to prevent pinniped depredation: Efficiency, conservation concerns and possible solutions. *Marine Ecology: Progress Series* 492:285-302.

Graham, A.L., D.M. Shuker, L.C. Pollitt, S.K. Auld, A.J. Wilson, and T.J. Little. 2011. Fitness consequences of immune responses: Strengthening the empirical framework for ecoimmunology. *Functional Ecology* 25(1):5-17.

Green, D.M. 1985. Temporal factors in psychoacoustics. Pp. 122-140 in *Time Resolution in Auditory Systems*. Berlin: Springer.

Greene, C.H., and A.J. Pershing. 2004. Climate and the conservation biology of North Atlantic right whales: The right whale at the wrong time? *Frontiers in Ecology and the Environment* 2:29-34.

Greenland, S. 2007. Interactions in epidemiology: Relevance, identification, and estimation. *Epidemiology* 20:14-17.

Greig, D.J., F.M. Gulland, C.A. Rios, and A.J. Hall. 2010. Hematology and serum chemistry in stranded and wild-caught harbor seals in central California: Reference intervals, predictors of survival, and parameters affecting blood variables. *Journal of Wildlife Diseases* 46(4):1172-1184.

Grimes, C.B. 2001. Fishery production and the Mississippi River discharge. *Fisheries* 26(8):17-26.

Gross, K., G. Pasinelli, and H.P. Kunc. 2010. Behavioral plasticity allows short-term adjustment to a novel environment. *American Naturalist* 176(4):456-464.

Groves, P.M., and R.F. Thompson. 1970. Habituation: A dual-process theory. *Psychological Review* 77(5):419-450.

Gulland, F.M.D., and A.J. Hall. 2007. Is marine mammal health deteriorating? Trends in the global reporting of marine mammal disease. *EcoHealth* 4:135-150.

Gulland, F.M.D., M. Haulena, D. Fauquier, G. Langlois, M.E. Lander, T. Zabka, and R. Duerr. 2002. Domoic acid toxicity in Californian sea lions (*Zalophus californianus*): Clinical signs, treatment and survival. *Veterinary Record* 150:475-480.

Gulland, F.M.D., A.J. Hall, D.J. Greig, E.R. Frame, K.M. Colegrove, R.K. Booth, S K. Wasser, and J.C. Scott-Moncrieff. 2012. Evaluation of circulating eosinophil count and adrenal gland function in California sea lions naturally exposed to domoic acid. *Journal of the American Veterinary Medical Association* 241:943-949.

Guzmán-Verri, C., R. González-Barrientos, G. Hernández-Mora, J.-A. Morales, E. Baquero-Calvo, E. Chavez-Olarte, and E. Moreno. 2012. *Brucella ceti* and brucellosis in cetaceans. *Frontiers in Cellular and Infection Microbiology* 2:1-22. doi: 10.3389/fcimb.2012.00003.

Habib, L., E.M. Bayne, and S. Boutin. 2007. Chronic industrial noise affects pairing success and age structure of ovenbirds *Seiurus aurocapilla*. *Journal of Applied Ecology* 44(1):176-184.

Halfwerk, W., L.J.M. Holleman, C.M. Lessells, and H. Slabbekoorn. 2011. Negative impact of traffic noise on avian reproductive success. *Journal of Applied Ecology* 48:210-219.

Hall, A., C. Duck, P. Hammond, G. Hastie, E. Jones, B. McConnell, C. Morris, J. Onoufriou, P. Pomeroy, D. Thompson, D. Russell, S. Smout, L. Wilson, and P. Thompson. 2015. Harbour Seal Decline Workshop II. Sea Mammal Research Unit, University of St. Andrews, Report to Scottish Government, no CSD6, St. Andrews, 12 pp. Available at http://www.smru.st-andrews.ac.uk/documents/scotgov/CSD6_harbour_seal_workshop_II_VF1.pdf.

Hall, A.J., and E. Frame. 2010. Evidence of domoic acid exposure in harbour seals from Scotland: A potential factor in the decline in abundance? *Harmful Algae* 9:489-493.

Hall, A.J., R.J. Law, D.E. Wells, J. Harwood, H.M. Ross, S. Kennedy, C.R. Allchin, L.A. Campbell, and P.P. Pomeroy. 1992. Organochlorine levels in common seals (*Phoca vitulina*) which were victims and survivors of the 1988 Phocine Distemper Epizootic. *Science of the Total Environment* 115:145-162.

Hall, A.J., K. Hugunin, R. Deaville, R.J. Law, C.R. Allchin, and P.D. Jepson. 2006. The risk of infection from polychlorinated biphenyl exposure in the harbor porpoise (*Phocoena phocoena*): A case-control approach. *Environmental Health Perspectives* 114(5):704-711.

Hall, A.J., G.O. Thomas, and B.J. McConnell. 2009. Exposure to persistent organic pollutants and first-year survival probability in gray seal pups. *Environmental Science and Technology* 43:6364-6369.

Halpern, B.S., K.A. Selkoe, F. Micheli, and C.V. Kappel. 2007. Evaluating and ranking the vulnerability of global marine ecosystems to anthropogenic threats. *Conservation Biology* 21:1301-1315.

Halpern, B.S., S. Walbridge, K.A. Selkoe, C.V. Kappel, F. Micheli, C. D'Agrosa, J.F. Bruno, K.S. Casey, C. Ebert, H.E. Fox, R. Fujita, D. Heinemann, H.S. Lenihan, E.M.P. Madin, M.T. Perry, E.R. Selig, M. Spalding, R. Steneck, and R. Watson. 2008. A global map of human impact on marine ecosystems. *Science* 319:948-952.

Hammond, J.A., A.J. Hall, and E.A. Dyrynda. 2005. Comparison of polychlorinated biphenyl (PCB) induced effects on innate immune functions in harbour and grey seals. *Aquatic Toxicology* 74(2):126-138.

Hammond, P.S., P. Berggren, H. Benke, D.L. Borchers, A. Collet, M.P. Heide-Jørgensen, S. Heimlich, A.R. Hiby, M.F. Leopold, and N. Øien. 2002. Abundance of harbour porpoises and other cetaceans in the North Sea and adjacent waters. *Journal of Applied Ecology* 39:361-376.

Hanson, N.N., D. Thompson, C.D. Duck, S. Moss, and M. Lonergan. 2013. Pup mortality in a rapidly declining harbour seal (*Phoca vitulina*) population. *PLoS ONE* 8(11):e80727.

Hanson, N.N., D. Thompson, C.D. Duck, J. Baxter, and M.E. Lonergan. 2015. Harbour seal (*Phoca vitulina*) abundance within the Firth of Tay and Eden Estuary, Scotland: Recent trends and extrapolation to extinction. *Aquatic Conservation: Marine and Freshwater Ecosystems*. doi: 10.1002/aqc.2609.

Hare, S.R., and N.J. Mantua. 2000. Empirical evidence for North Pacific regime shifts in 1977 and 1989. *Progress in Oceanography* 47(2):103-145.

Harris, C.M., and L. Thomas. 2015. *Status and Future of Research on the Behavioral Responses of Marine Mammals to U.S. Navy Sonar.* CREEM Technical Report 2015-3, University of St. Andrews. Available at http://hdl.handle.net/10023/7741.

Harris, C.M., D. Sadykova, S.L. DeRuiter, P.L. Tyack, P.J.O. Miller, P.H. Kvadsheim, F.P.A. Lam, and L. Thomas. 2015. Dose response severity functions for acoustic disturbance in cetaceans using recurrent event survival analysis. *Ecosphere* 6(11):236.

Hart, L.B., D.S. Rotstein, R.S. Wells, J. Allen, A. Barleycorn, B.C. Balmer, S.M. Lane, T. Speakman, E.S. Zolman, M. Stolen, W. McFee, T. Goldstein, T.K. Rowles, and L. Schwacke. 2012. Skin lesions on common bottlenose dolphins (*Tursiops truncatus*) from three sites in the Northwest Atlantic, USA. *PLoS ONE* 7(3):e33081.

Harvell, C.D., C.E. Mitchell, J.R. Ward, S. Altizer, A.P. Dobson, R.S. Ostfeld, and M.D. Samuel. 2002. Climate warming and disease risks for terrestrial and marine biota. *Science* 296(5576):2158-2162.

Harvey, B.P., D. Gwynn-Jones, and P.J. Moore. 2013. Meta-analysis reveals complex marine biological responses to the interactive effects of ocean acidification and warming. *Ecology and Evolution* 3(4):1016-1030.

Harvey, P.W. 2016. Adrenocortical endocrine disruption. *Journal of Steroid Biochemistry and Molecular Biology* 155:199-206.

Harwood, J. 1992. Assessing the competitive effects of marine mammal predation on commercial fisheries. *South African Journal of Marine Science* 12(1):689-693.

Harwood, J. 2001. Marine mammals and their environment in the twenty-first century. *Journal of Mammalogy* 82(3):630-640.

Harwood, L.A., T.G. Smith, and H. Melling. 2000. Variation in reproduction and body condition of the ringed seal (*Phoca hispida*) in western Prince Albert Sound, NT, Canada, as assessed through a harvest-based sampling program. *Arctic* 53(4):422-431.

Hatch, L.T., C.W. Clark, S.M. Van Parijs, A.S. Frankel, and D.W. Ponirakis. 2012. Quantifying loss of acoustic communication space for right whales in and around a U.S. National Marine Sanctuary. *Conservation Biology* 26:983-994.

Heithaus, M.R. 2001. Shark attacks on bottlenose dolphins (*Tursiops aduncus*) in Shark Bay, Western Australia: Attack rate, bite scar frequencies, and attack seasonality. *Marine Mammal Science* 17(3):526-539.

Hildebrand, J.A. 2009. Anthropogenic and natural sources of ambient noise in the ocean. *Marine Ecology Progress Series* 395:5-20.

Hobbs, R.C., D.J. Rugh, and D.P. DeMaster. 2000. Abundance of belugas, *Delphinapterus leucas*, in Cook Inlet, Alaska. 1994-2000. *Marine Fisheries Review* 62:37-45.

Hohn, A.A., M.D. Scott, R.S. Wells, J.C. Sweeney, and A.B. Irvine. 1989. Growth layers in teeth from known-age, free-ranging bottlenose dolphins. *Marine Mammal Science* 5(4):315-342.

Holt, M.M., D.P. Noren, V. Veirs, C.K. Emmons, and S. Veirs. 2009. Speaking up: Killer whales (*Orcinus orca*) increase their call amplitude in response to vessel noise. *Journal of the Acoustical Society of America* 125:EL27.

Holt, M.M., D.P. Noren, R.C. Dunkin, and T.M. Williams. 2015. Vocal performance affects metabolic rate in dolphins: Implications for animals communicating in noisy environments. *Journal of Experimental Biology* 218:1647-1654.

Hooker, S.K., I.L. Boyd, M. Jessopp, O. Cox, J. Blackwell, P.L. Boveng, and J.L. Bengtson. 2002. Monitoring the prey-field of marine predators: Combining digital imaging with datalogging tags. *Marine Mammal Science* 18(3):680-697.

Hooker, S.K., A. Fahlman, M.J. Moore, N. Aguilar de Soto, Y. Bernaldo de Quirós, A.O. Brubakk, D.P. Costa, A.M. Costidis, S. Dennison, K.J. Falke, A. Fernandez, M. Ferrigno, J.R. Fitz-Clarke, M.M. Garner, D.S. Houser, P.D. Jepson, D.R. Ketten, P.H. Kvadsheim, P.T. Madsen, N.W. Pollock, D.S. Rotstein, T.K. Rowles, S.E. Simmons, W. Van Bonn, P.K. Weathersby, M.J. Weise, T.M. Williams, and P.L. Tyack. 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.

Horning, M. and R.D. Hill. 2005. Designing an archival satellite transmitter for life-long deployments on oceanic vertebrates: The life history transmitter. *IEEE Journal of Oceanic Engineering* 30(4):807-817.

Horning, M., and J.A.E. Mellish. 2014. In cold blood: Evidence of Pacific sleeper shark *(Somniosus pacificus*) predation on Steller sea lions (*Eumetopias jubatus*) in the Gulf of Alaska. Available at http://hdl.handle.net/1957/54605.

Horning, M., M. Haulena, P.A. Tuomi, and J.A.E. Mellish. 2008. Intraperitoneal implantation of life-long telemetry transmitters in otariids. *BMC Veterinary Research* 4:51.

Houde, M., P.F. Hoekstra, K.R. Solomon, and D.C. Muir. 2005. Organohalogen contaminants in delphinoid cetaceans. *Reviews of Environmental Contamination and Toxicology* 184:1-57.

Houde, M., G. Pacepavicius, C. Darling, P.A. Fair, M. Alaee, G.D. Bossart, K.R. Solomon, R.J. Letcher, A. Bergman, G. Marsh, and D.C. Muir. 2009. Polybrominated diphenyl ethers and their hydroxylated analogs in plasma of bottlenose dolphins (*Tursiops truncatus*) from the United States east coast. *Environmental Toxicology and Chemistry* 28(10):2061-2068.

Houser, D.S., D.A. Helweg, and P.W.B. Moore. 2001. A bandpass filter-bank model of auditory sensitivity in the humpback whale. *Aquatic Mammals* 27:82-91.

Houser, D.S., S.W. Martin, and J.J. Finneran. 2013a. Behavioral responses of California sea lions to mid-frequency (3250-3450 Hz) sonar signals. *Marine Environmental Research* 92:268-278.

Houser, D.S., S.W. Martin, and J.J. Finneran. 2013b. Exposure amplitude and repetition affect bottlenose dolphin behavioral responses to simulated mid-frequency sonar signals. *Journal of Experimental Marine Biology and Ecology* 443:123-133.

Hua, F., K.E. Sieving, R.J. Fletcher, and C.A. Wright. 2014. Increased perception of predation risk to adults and offspring alters avian reproductive strategy and performance. *Behavioral Ecology* 25:509-519.

Huang, S.-L., L.-S. Chou, N.-T. Shih, and I.-H Ni. 2011. Implication of life history strategies for prenatal investment in cetaceans. *Marine Mammal Science* 27:182-194. doi: 10.1111/j.1748-7692.2010.00392.x.

Huber, M., J.A. Knottnerus, L. Green, H. van der Horst, A.R. Jadad, D. Kromhout, B. Leonard, K. Lorig, M.I. Loureiro, J.W. van der Meer, P. Schnabel, R. Smith, C. van Weel, and H. Smid. 2011. How should we define health? *British Medical Journal* 343(7817):235-237.

Hughes, T.P. 1994. Catastrophes, phase shifts, and large-scale degradation of a Caribbean coral reef. *Science* 265(5178):1547-1551.

Hunsicker, M.E, C.V. Kappel, K.A. Selkoe, B.S. Halpern, C. Scarborough, L. Mease, and A. Amrhein. 2016. Characterizing driver-response relationships in marine pelagic ecosystems for improved ocean management. *Ecological Applications* 26(3):651-663.

Hunt, K.E., M.J. Moore, R.M. Rolland, N.M. Kellar, A.J. Hall, J. Kershaw, S.A. Raverty, C.E. Davis, L.C. Yeates, D.A. Fauquier, T.K. Rowles, and S.D. Kraus. 2013. Overcoming the challenges of studying conservation physiology in large whales: A review of available methods. *Conservation Physiology* 1:1-24.

Hunt, K.E., R. Stimmelmayr, C. George, C. Hanns, R. Suydam, H. Brower, and R.M. Rolland. 2014. Baleen hormones: A novel tool for retrospective assessment of stress and reproduction in bowhead whales (*Balaena mysticetus*). *Conservation Physiology* 2(1):cou030.

Hyrenbach, K.D., K.A. Forney, and P.K. Dayton. 2000. Marine protected areas and ocean basin management. *Aquatic Conservation: Marine and Freshwater Ecosystems* 10:437-458.

Inoue, Y., T. Itou, T. Sakai, and T. Oike. 1999. Cloning and sequencing of a bottle-nosed dolphin (*Tursiops truncatus*) interleukin-4-encoding cDNA. *Journal of Veterinary Medical Science* 61(6):693-696.

IOM (Institute of Medicine) and NRC (National Research Council). 2009. *Sustaining Global Surveillance and Response to Emerging Zoonotic Diseases*. Washington, DC: The National Academies Press.

Isaksson, C., B.C. Sheldon, and T. Uller. 2011. The challenges of integrating oxidative stress into life-history biology. *BioScience* 61(3):194-202.

Isojunno, S., C. Curé, P.H. Kvadsheim, F.-P.A. Lam, P.L. Tyack, P.J. Wensveen, and P.J.O. Miller. 2016. Sperm whales reduce foraging effort during exposure to 1–2 kHz sonar and killer whale sounds. *Ecological Applications* 26(1):77-93.

Jackson, M.C., C.J. Loewen, R.D. Vinebrooke, and C.T. Chimimba. 2016. Net effects of multiple stressors in freshwater ecosystems: A meta-analysis. *Global Change Biology* 22(1):180-189.

Jacobson, J.L. and S.W. Jacobson. 1996. Intellectual impairment in children exposed to polychlorinated biphenyls in utero. *New England Journal of Medicine* 335:783-789.

Jadad, A.R., and L. O'Grady. 2008. How should health be defined? *British Medical Journal* 337(7683):1363-1364.

Jefferson, T.A., and B.E. Curry. 1996. Acoustic methods of reducing or eliminating marine mammal interactions with fisheries: Do they work? *Ocean & Coastal Management* 31(1):41-70.

Jefferson, T.A., P.J. Stacey, and R.W. Baird. 1991. A review of killer whale interactions with other marine mammals: Predation to co-existence. *Mammal Review* 21(4):151-180.

Jensen, S.-K., J.-P. Lacaze, G. Hermann, J. Kershaw, A. Brownlow, A. Turner, and A.J. Hall. 2015. Detection and effects of harmful algal toxins in Scottish harbour seals and potential links to population decline. *Toxicon* 97:1-14.

Jenssen, B.M. 2006. Endocrine-disrupting chemicals and climate change: A worst-case combination for arctic marine mammals and seabirds? *Environmental Health Perspectives* 114(Suppl. 1):76-80.

Jepson, P.D., M. Arbelo, R. Deaville, I.A.P. Patterson, P. Castro, J.R. Baker, E. Degollada, H.M. Ross, P. Herraez, A.M. Pocknell, F. Rodriguez, F.E. Howiell, A. Espinosa, R.J. Reid, J.R. Jaber, V. Martin, A.A. Cunningham, and A. Fernández. 2003. Gas-bubble lesions in stranded animals: Was sonar responsible for a spate of whale deaths after an Atlantic military exercise? *Nature* 425(6958):575-576.

Jepson, P.D., P.M. Bennett, R. Deaville, C.R. Allchin, J.R. Baker, and R.J. Law. 2005. Relationships between polychlorinated biphenyls and health status in harbor porpoises (*Phocoena phocoena*) stranded in the United Kingdom. *Environmental Toxicology and Chemistry* 24(1):238-248.

Jewell, R., L. Thomas, C.M. Harris, K. Kaschner, R. Wiff, P.S. Hammond, and N.J. Quick. 2012. Global analysis of cetacean line-transect surveys: Detecting trends in cetacean density. *Marine Ecology Progress Series* 453:227-240.

Jiang, M., M.W. Brown, J.T. Turner, R.D. Kenney, C.A. Mayo, Z. Zhang, and M. Zhou. 2007. Springtime transport and retention of *Calanus finmarchicus* in Massachusetts and Cape Cod Bays, USA, and implications for right whale foraging. *Marine Ecology Progress Series* 349:183-197.

Johnson, A.C., P. Nylen, E. Borg, and G. Hoglund. 1990. Sequence of exposure to noise and toluene can determine loss of auditory sensitivity in the rat. *Acta Otolaryngologica* 109:34-40.

Johnson, C.K., M.T. Tinker, J.A. Estes, P.A. Conrad, M. Staedler, M.A. Miller, D.A. Jessup, and J.A.K. Mazet. 2009. Prey choice and habitat use drive sea otter pathogen exposure in a resource-limited coastal system. *Proceedings of the National Academy of Sciences of the United States of America* 106(7):2242-2247.

Johnson, M., and P.L. Tyack. 2003. A digital acoustic recording tag for measuring the response of wild marine mammals to sound. *IEEE Journal of Oceanic Engineering* 28:3-12.

Kajiwara, N., S. Kamikawa, K. Ramu, D. Ueno, T.K. Yamada, A. Subramanian, P.K. Lam, T.A. Jefferson, M. Prudente, K.H. Chung, and S. Tanabe. 2006. Geographical distribution of polybrominated diphenyl ethers (PBDEs) and organochlorines in small cetaceans from Asian waters. *Chemosphere* 64:287-295.

Kakuschke, A., and A. Prange. 2007. The influence of metal pollution on the immune system a potential stressor for marine mammals in the North Sea. *International Journal of Comparative Psychology* 20:179-193.

Karl, D.M., R. Letelier, D. Hebel, L. Tupas, J. Dore, J. Christian, and C. Winn. 1995. Ecosystem changes in the North Pacific subtropical gyre attributed to the 1991–92 El Niño. *Nature* 373:230-234.

Kastak, D., J. Mulsow, A. Ghoul, and C. Reichmuth. 2008. Noise-induced permanent threshold shift in a harbor seal. *Journal of the Acoustical Society of America* 123:2986.

Kastelein, R.A., N. Steen, C. de Jong, P.J. Wensveen, and W.C. Verboom. 2011. Effect of broadband-noise masking on the behavioral response of a harbor porpoise (*Phocoena phocoena*) to 1-s duration 6–7 kHz sonar upsweeps. *Journal of the Acoustical Society of America* 129:2307-2315.

Kastelein, R.A., N. Steen, R. Gransier, P.J. Wensveen, and C.A.F. de Jong. 2012. Threshold received sound pressure levels of single 1–2 kHz and 6–7 kHz up-sweeps and down-sweeps causing startle responses in a harbor porpoise (*Phocoena phocoena*). *Journal of the Acoustical Society of America* 131:2325-2333.

Kastelein, R.A., R. Gransier, M. van den Hoogen, and L. Hoek. 2013. Brief behavioral response threshold levels of a harbor porpoise (*Phocoena phocoena*) to five helicopter dipping sonar signals (1.33 to 1.43 kHz). *Aquatic Mammals* 39:162-173.

Keenan, R.E. 2000. An introduction to GRAB eigenrays and CASS reverberation and signal excess. Pp. 1065-1070 in *Oceans 2000 MTS/IEEE Conference & Exhibition, Volume 2*. New York: IEEE.

Kellar, N.M., M.L. Trego, C.I. Marks, and A.E. Dizon. 2006. Determining pregnancy from blubber in three species of delphinids. *Marine Mammal Science* 22:1-16.

Kenet, T., R.C. Froemke, C.E. Schreiner, I.N. Pessah, and M.M. Merzenich. 2007. Perinatal exposure to a noncoplanar polychlorinated biphenyl alters tonotopy, receptive fields, and plasticity in rat primary auditory cortex. *Proceedings of the National Academy of Sciences of the United States of America* 104(18):7646-7651.

Ketten, D.R., J. Lien, and S. Todd. 1993. Blast injury in humpback whale ears: Evidence and implication. *Journal of the Acoustical Society of America* 94:1849-1850.

Khudyakov, J.I., C.D. Champagne, L. Preeyanon, R.M. Ortiz, and D.E. Crocker. 2015. Muscle transcriptome response to ACTH administration in a free-ranging marine mammal. *Physiological Genomics* 47(8):318-330.

Kight, C.R., M.S. Saha, and J.P. Swaddle. 2012. Anthropogenic noise is associated with reductions in the productivity of breeding eastern bluebirds (*Sialia sialis*). *Ecological Applications* 22:1989-1996.

Killen, S.S., S. Marras, N.B. Metcalfe, D.J. McKenzie, and P. Domenici. 2013. Environmental stressors alter relationships between physiology and behaviour. *Trends in Ecology and Evolution* 28:623-680.

Kindig, D., and G. Stoddart. 2003. What is population health? *American Journal of Public Health* 93(3):380-383.

King, D.P., M.D. Schrenzel, M.L. McKnight, T.H. Reidarson, K.D. Hanni, J.L. Stott, and D.A. Ferrick. 1996. Molecular cloning and sequencing of interleukin 6 cDNA fragments from the harbor seal (*Phoca vitulina*), killer whale (*Orcinus orca*), and Southern sea otter (*Enhydra lutris nereis*). *Immunogenetics* 43(4):190-195.

King, D.P., J.L. Sanders, C.T. Nomura, R.A. Stoddard, C.L. Ortiz, and S.W. Evans. 1998. Ontogeny of humoral immunity in northern elephant seal (*Mirounga angustirostris*) neonates. *Comparative Biochemistry and Physiology B* 121:363-368.

King, S.L., R.S. Schick, C. Donovan, C.G. Booth, M. Burgman, L. Thomas, and J. Harwood. 2015. An interim framework for assessing the population consequences of disturbance. *Methods in Ecology and Evolution* 6:1150-1158.

Kitaysky, A., J. Piatt, and J. Wingfield. 2007. Stress hormones link food availability and population processes in seabirds. *Marine Ecology Progress Series* 352:245-258.

Klein, S.C., and M.E. Peterson. 2010. Canine hypoadrenocorticism: Part 1. *Canine Veterinary Journal* 51:63-69.

Knowlton, A.R., P.K. Hamilton, M.K. Marx, H.M. Pettis, and S.D. Kraus. 2012. Monitoring North Atlantic right whale *Eubalaena glacialis* entanglement rates: A 30 year retrospective. *Marine Ecology: Progress Series* 466:293-302.

Knowlton, N. 2004. Multiple "stable" states and the conservation of marine ecosystems. *Progress in Oceanography* 60(2):387-396.

Kooyman, G.L. 1985. Physiology without restraint in diving mammals. *Marine Mammal Science* 1(2):166-178.

Kooyman, G.L., and P.J. Ponganis. 1998. The physiological basis of diving to depth: Birds and mammals. *Annual Review of Physiology* 60(1):19-32.

Kovacs, K.M., and C. Lydersen. 2008. Climate change impacts on seals and whales in the North Atlantic Arctic and adjacent shelf seas. *Science Progress* 91(2):117-150.

Kovacs, K.M., C. Lydersen, J.E. Overland, and S.E. Moore. 2011. Impacts of changing sea-ice conditions on Arctic marine mammals. *Marine Biodiversity* 41(1):181-194.

Kraus, S.D., and R. Rolland. 2007. *The Urban Whale: North Atlantic Right Whales at the Crossroads*. Cambridge, MA: Harvard University Press.

Kraus, S.D., M.W. Brown, H. Caswell, C.W. Clark, M. Fujiwara, P.K. Hamilton, R.D. Kenney, A.R. Knowlton, S. Landry, C.A. Mayo, W.A. McLellan, M.J. Moore, D.P. Nowacek, D.A. Pabst, A.J. Read, and R.M. Rolland. 2005. North Atlantic right whales in crisis. *Science* 309:561-562.

Kucklick, J., L. Schwacke, R. Wells, A. Hohn, A. Guichard, J. Yordy, L. Hansen, E. Zolman, R. Wilson, J. Litz, D. Nowacek, T. Rowles, R. Pugh, B. Balmer, C. Sinclair, and P. Rosel. 2011. Bottlenose dolphins as indicators of persistent organic pollutants in the western North Atlantic Ocean and northern Gulf of Mexico. *Environmental Science & Technology* 45:4270-4277.

Kuehl, D.W., R. Haebler, and C. Potter. 1991. Chemical residues in dolphins from the United States Atlantic Coast including Atlantic bottlenose obtained during the 1987/88 mass mortality. *Chemosphere* 22:1071-1084.

Kuitunen, M., E. Rossi, and A. Stenroos. 1998. Do highways influence density of land birds? *Environmental Management* 22:297-302.

Kujawa, S.G., and M.C. Liberman, 2006. Acceleration of age-related hearing loss by early noise exposure: Evidence of a misspent youth. *Journal of Neuroscience* 26(7):2115-2123.

Kvadsheim, P.H., P.J.O. Miller, P.L. Tyack, L.D. Sivle, F.P.A. Lam, and A. Fahlman. 2012. Estimated tissue and blood N_2 levels and risk of in vivo bubble formation in deep-, intermediate- and shallow diving toothed whales during exposure to naval sonar. *Frontiers in Aquatic Physiology* 3(125):1-14.

Lafferty, K.D., J.W. Porter, and S.E. Ford. 2004. Are diseases increasing in the ocean? *Annual Review of Ecology, Evolution, and Systematics* 35:31-54.

Lagardère, J.P. 1982. Effects of noise on growth and reproduction of *Crangon crangon* in rearing tanks. *Marine Biology* 71(2):177-185.

Laidre, K.L., H. Stern, K.M. Kovacs, L. Lowry, S.E. Moore, E.V. Regehr, S.H. Ferguson, Ø.Wiig, P. Boveng, R.P. Angliss, E.W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. *Conservation Biology* 29:724-737.

Laist, D.W., A.R. Knowlton, J.G. Mead, A.S. Collet, and M. Podesta. 2001. Collisions between ships and whales. *Marine Mammal Science* 17:35-75.

Lane, S.M., C.R. Smith, J. Mitchell, B.C. Balmer, K.P. Barry, T. McDonald, C.S. Mori, P.E. Rosel, T.K. Rowles, T.R. Speakman, and F.I. Townsend. 2015. Reproductive outcome and survival of common bottlenose dolphins sampled in Barataria Bay, Louisiana, USA, following the *Deepwater Horizon* oil spill. *Proceedings of the Royal Society B: Biological Sciences* 282(1818):20151944.

Lataye, R., and P. Campo. 1997. Combined effects of a simultaneous exposure to noise and toluene on hearing function. *Neurotoxicology Teratology* 19:373-382.

Lataye, R., P. Campo, and G. Loquet. 2000. Combined effects of noise and styrene exposure on hearing function in the rat. *Hearing Research* 139:86-96.

Le Boeuf, B.J., and D.E. Crocker. 2005. Ocean climate and seal condition. *BMC Biology* 3(1):1.

Leaper, R., J. Cooke, P. Trathan, K. Reid, V. Rowntree, and R. Payne. 2006. Global climate drives southern right whale (*Eubalaena australis*) population dynamics. *Biology Letters* 2(2):289-292.

Lercher, P., G.W. Evans, and M. Meis. 2003. Ambient noise and cognitive processes among primary schoolchildren. *Environment and Behavior* 35:725-735.

Levin, M., S. De Guise, and P.S. Ross. 2005. Association between lymphocyte proliferation and polychlorinated biphenyls in free-ranging harbor seal (*Phoca vitulina*) pups from British Columbia, Canada. *Environmental Toxicology and Chemistry* 24(5):1247-1252.

Levin, P.S., and C. Möllmann. 2015. Marine ecosystem regime shifts: Challenges and opportunities for ecosystem-based management. *Philosophical Transactions of the Royal Society of London B: Biological Sciences* 370(1659): 20130275.

Levin, P.S., M.J. Fogarty, S.A. Murawski, and D. Fluharty. 2009. Integrated ecosystem assessments: Developing the scientific basis for ecosystem-based management of the ocean. *PLoS ONE Biology* 7(1):e1000014.

Levin, P.S., C.R. Kelble, R.L. Shuford, C. Ainsworth, R. Dunsmore, M.J. Fogarty, K. Holsman, E.A. Howell, M.E. Monaco, S.A. Oakes, and F. Werner. 2014. Guidance for implementation of integrated ecosystem assessments: A US perspective. *ICES Journal of Marine Science* 71(5):1198-1204.

Lewison, R.L., L.B. Crowder, B.P. Wallace, J.E. Moore, T. Cox, R. Zydelis, S. McDonald, A. DiMatteo, D.C. Dunn, C.Y. Kot, and R. Bjorkland. 2014. Global patterns of marine mammal, seabird, and sea turtle bycatch reveal taxa-specific and cumulative megafauna hotspots. *Proceedings of the National Academy of Sciences of the United States of America* 111(14):5271-5276.

Lewontin, R.C. 1969. The meaning of stability. Pp. 13-24 in *Diversity and Stability in Ecological Systems*, G.W. Woodwell and H.H. Smith, eds. Upton, NY: Brookhaven National Laboratory.

Lie, E., H.J. Larsen, S. Larsen, G.M. Johansen, A.E. Derocher, N.J. Lunn, R.J. Norstrom, Ø. Wiig, and J.U. Skaare. 2004. Does high organochlorine (OC) exposure impair the resistance to infection in polar bears (*Ursus maritimus*)? Part I: Effect of OCs on the humoral immunity. *Journal of Toxicology and Environmental Health A* 67(7):555-582.

Lie, E., H.J. Larsen, S. Larsen, G.M. Johansen, A.E. Derocher, N.J. Lunn, R.J. Norstrom, Ø. Wiig, and J.U. Skaare. 2005. Does high organochlorine (OC) exposure impair the resistance to infection in polar bears (*Ursus maritimus*)? Part II: Possible effect of OCs on mitogen- and antigen-induced lymphocyte proliferation. *Journal of Toxicology and Environmental Health A* 68(6):457-484.

Lilienthal, H., P. Heikkinen, P.L. Andersson, L.T.M. van der Ven, and M. Viluksela. 2011. Auditory effects of developmental exposure to purity-controlled polychlorinated biphenyls (PCB52 and PCB180) in rats. *Toxicological Sciences* 122(1):100-111.

Lindenmayer, D.B., and G.E. Likens. 2009. Adaptive monitoring: A new paradigm for long-term research and monitoring. *Trends in Ecology & Evolution* 24:482-486.

Lioy, P.J., and S.M. Rappaport. 2011. Exposure science and the exposome: An opportunity for coherence in the environmental health sciences. *Environmental Health Perspectives* 119(11):A466-A467.

Lochmiller, R.L., and C. Deerenberg. 2000. Trade-offs in evolutionary immunology: Just what is the cost of immunity? *Oikos* 88(1):87-98.

Logan, D.C. 2009. Known knowns, known unknowns, unknown unknowns and the propagation of scientific enquiry. *Journal of Experimental Botany* 60(3):712-714.

Løkkeborg, S., E. Ona, A. Vold, and A. Salthaug. 2012. Sounds from seismic air guns: Gear- and species-specific effects on catch rates and fish distribution. *Canadian Journal of Fisheries and Aquatic Science* 69(8):1278-1291.

Lonergan, M., C.D. Duck, D. Thompson, B.L. Mackey, L. Cunningham, and I.L. Boyd. 2007. Using sparse survey data to investigate the declining abundance of British harbour seals. *Journal of Zoology* 271:261-269.

Lucke, K., U. Siebert, P.A. Lepper, and M. A. Blanchet. 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* 125:4060-4070.

Luebke, R.W., C. Parks, and M.I. Luster. 2004. Suppression of immune function and susceptibility to infections in humans: Association of immune function with clinical disease. *Journal of Immunotoxicology* 1(1):15-24.

Lundquist, D., M. Sironi, B. Würsig, V. Rowntree, J. Martino, and L. Lundquist. 2013. Response of southern right whales to simulated swim-with-whale tourism at Península Valdés, Argentina. *Marine Mammal Science* 29(2):E24-E45.

Lusseau, D., and L. Bejder. 2007. The long-term consequences of short-term responses to disturbance experiences from whalewatching impact assessment. *International Journal of Comparative Psychology* 20:228-236.

Lusseau, D., E. Slooten, and R.J.C. Currey. 2006. Unsustainable dolphin-watching tourism in Fiordland, New Zealand. *Tourism in Marine Environments* 3:173-178.

Lusseau, D., L. New, C. Donovan, B. Cheney, G. Hastie, and J. Harwood. 2012. *The Development of a Framework to Understand and Predict the Population Consequences of Disturbances for the Moray Firth Bottlenose Dolphin Population*. Scottish Natural Heritage Commissioned Report No. 468. Available at http://www.snh.org.uk/pdfs/publications/commissioned_reports/468.pdf.

Madsen, P.T., B. Møhl, B.K. Nielsen, and M. Wahlberg. 2002. Male sperm whale behaviour during exposures to distant seismic survey pulses. *Aquatic Mammals* 28(3):231-240.

Madsen, P.T., M. Wahlberg, J. Tougaard, K. Lucke, and P. Tyack. 2006. Wind turbine underwater noise and marine mammals: Implications of current knowledge and data needs. *Marine Ecology: Progress Series* 309:279-295.

Mäkitie, A.A., U. Pirvola, I. Pyykkö, H. Sakakibara, V. Riihimäki, and J. Ylikoski. 2003. The ototoxic interaction of styrene and noise. *Hearing Research* 179(1-2):9-20.

Malme, C.I., P.R. Miles, C.W. Clark, P. Tyack, and J.E. Bird. 1983. *Investigations of the Potential Effects of Underwater Noise from Petroleum Industry Activities on Migrating Gray Whale Behavior*. Report No. 5366. Prepared by: Bolt Beranek and Newman Inc. for Minerals Management Service Alaska OCS Office. NTIS PB86-174174. https://ntrl.ntis.gov/NTRL.

Malme, C.I., P.R. Miles, C.W. Clark, P. Tyack, and J.E. Bird. 1984. *Investigations of the Potential Effects of Underwater Noise from Petroleum Industry Activities on Migrating Gray Whale Behavior*. Phase II: January 1984 Migration. Report No. 5586. Prepared by Bolt Beranek and Newman Inc. for Minerals Management Service Alaska OCS Office. NTIS PB86-218377. https://ntrl.ntis.gov/NTRL.

Mancia, A., M.L. Lundqvist, T.A. Romano, M.M. Peden-Adams, P.A. Fair, M.S. Kindy, B.C. Ellis, S. Gattoni-Celli, D.J. McKillen, H.F. Trent, Y.A. Chen, J.S. Almeida, P.S. Gross, R.W. Chapman, and G.W. Warr. 2007. A dolphin peripheral blood leukocyte cDNA microarray for studies of immune function and stress reactions. *Developmental & Comparative Immunology* 31(5):520-529.

Mann, D. A., Z. Lu, M.C. Hastings, and A.N. Popper. 1998. Detection of ultrasonic tones and simulated dolphin echolocation clicks by a teleost fish, the American shad (*Alosa sapidissima*). *Journal of the Acoustical Society of America* 104(1):562-568.

Marques, T.A., L. Thomas, S.W. Martin, D.K. Mellinger, J.A. Ward, D.J. Moretti, D. Harris, and P.L. Tyack. 2013. Estimating animal population density using passive acoustics. *Biological Reviews* 88(2):287-309.

Martin, L.B., E. Andreassi, W. Watson, and C. Coon. 2011. Stress and animal health: Physiological mechanisms and ecological consequences. *Nature Education Knowledge* 3(6):11.

Martin, T., M. Burgman, F. Fidler, P. Kuhnert, S. Low-Choy, M. McBride, and K. Mengersen. 2012. Eliciting expert knowledge in conservation science. *Conservation Biology* 26:29-38.

Martineau, D. 2007. Potential synergism between stress and contaminants in free-ranging cetaceans. *International Journal of Comparative Psychology* 20:194-216.

Mason, J.T., C.J.W. McClure, and J.R. Barber. 2016. Anthropogenic noise impairs owl hunting behavior. *Biological Conservation* 199: 29-32.

Mate, B., R. Mesecar, and B. Lagerquist. 2007. The evolution of satellite-monitored radio tags for large whales: One laboratory's experience. *Deep Sea Research, Part II: Topical Studies in Oceanography* 54(3): 224-247.

Matthiopoulos, J., S.C. Smout, A. Winship, D. Thompson, I.L. Boyd, and J. Harwood. 2008. Getting beneath the surface of marine mammal-fisheries competition. *Mammal Review* 38(2-3):167-188.

Matthiopoulos, J., L. Cordes, B. Mackey, D. Thompson, C. Duck, S. Smout, M. Caillat, and P. Thompson. 2014. State-space modeling reveals proximate causes of harbour seal population declines. *Oecologia* 174:151-162.

Maxwell, S.M., E.L. Hazen, S.J. Bograd, B.S. Halpern, G.A. Breed, B. Nickel, N.M. Teutschel, L.B. Crowder, S. Benson, P.H. Dutton, H. Bailey, M.A. Kappes, C.E. Kuhn, M.J. Weise, B. Mate, S.A. Shaffer, J.L. Hassrick, R.W. Henry, L. Irvine, B.I. McDonald, P.W. Robinson, and D.P. Costa. 2013. Cumulative human impacts on marine predators. *Nature Communications* 4:2688.

May, R.M. 1972. Will a large complex system be stable? *Nature* 238:413-414.

May, R.M. 1976. Thresholds and breakpoints in ecosystems with a multiplicity of stable states. *Nature* 260:471-477.

Mazet, J. K., I.A. Gardner, D.A. Jessup, L.J. Lowenstine, and W.M. Boyce. 2000. Evaluation of changes in hematologic and clinical biochemical values after exposure to petroleum products in mink (*Mustela vison*) as a model for assessment of sea otters (*Enhydra lutris*). *American Journal of Veterinary Research* 61:1197-1203.

McCarthy, E. 2007. *International Regulation of Underwater Sound: Establishing Rules and Standards to Address Ocean Noise Pollution.* New York: Springer.

McCarthy, E., D. Moretti, L. Thomas, N. DiMarzio, R. Morrissey, S. Jarvis, J. Ward, A. Izzi, and A. Dilley. 2011. Changes in spatial and temporal distribution and vocal behavior of Blainville's beaked whales (*Mesoplodon densirostris*) during multiship exercises with mid-frequency sonar. *Marine Mammal Science* 27:E206-E226.

McCauley, R.D., J. Fewtrell, A.J. Duncan, C. Jenner, M.N. Jenner, J.D. Penrose, R.I.T Prince, A. Adhitya, J. Murdoch, and K. McCabe. 2000. Marine seismic surveys—a study of environmental implications. *APPEA Journal* 40:692-708.

McCauley, R.D., J. Fewtrell, and A.N. Popper. 2003. High intensity anthropogenic sound damages fish ears. *Journal of the Acoustical Society of America* 113:638. doi: 10.1121/1.1527962.

McClure, C.J., H.E. Ware, J. Carlisle, G. Kaltenecker, and J.R. Barber. 2013. An experimental investigation into the effects of traffic noise on distributions of birds: Avoiding the phantom road. *Proceedings of the Royal Society B: Biological Sciences* 280(1773):20132290.

McDonald, B.I., and P.J. Ponganis. 2012. Lung collapse in the diving sea lion: Hold the nitrogen and save the oxygen. *Biology Letters*. doi: 10.1098/rsbl.2012.0743.

McEwen, B.S. 1998. Stress, adaptation, and disease—allostasis and allostatic load. Pp. 33-44 in *Neuroimmunomodulation*, W. Aerpaoli and N.H. Specter, eds. New York: New York Academy of Sciences.

McEwen, B.S., and J.C. Wingfield. 2003. The concept of allostasis in biology and biomedicine. *Hormones and Behavior* 43:2-15.

McMahon, C.R., H.R. Burton, and M.N. Bester. 2000. Weaning mass and the future survival of juvenile southern elephant seals, *Mirounga leonina*, at Macquarie Island. *Antarctic Science* 12:149-153.

McMahon, C.R., H.R. Burton, and M.N. Bester. 2003. A demographic comparison of two elephant seal populations. *Journal of Animal Ecology* 72:61-74.

Meegan, J., J.L. Dunn, S.K. Venn-Watson, C.R. Smith, I. Sidor, E.D. Jensen, W.G. Van Bonn, R. Pugh, T. Ficht, L.G. Adams, and K. Nielsen. 2012. Serologic response in bottlenose dolphins (*Tursiops truncatus*) infected with *Brucella* sp. using a dolphin-specific indirect ELISA. *Diseases of Aquatic Organisms* 102:73-85.

Meir, J.U., C.D. Champagne, D.P. Costa, C.L. Williams, and P.J. Ponganis. 2009. Extreme hypoxemic tolerance and blood oxygen depletion in diving elephant seals. *American Journal of Physiology: Regulatory, Integrative and Comparative Physiology* 297(4):R927-R939.

Mellinger, D.K., and C.W. Clark. 2003. Blue whale (*Balaenoptera musculus*) sounds from the North Atlantic. *Journal of the Acoustical Society of America* 114:1108-1119.

Mennitt, D., K.M. Fristrup, K. Sherrill, and L. Nelson. 2013. Mapping sound pressure levels on continental scales using a geospatial sound model. *Proceedings of Inter-Noise*. Innsbruck, Austria, September 15-18, 2013. 11 pp.

Metcalfe, N.B., and C. Alonso-Alvarez. 2010. Oxidative stress as a life-history constraint: The role of reactive oxygen species in shaping phenotypes from conception to death. *Functional Ecology* 24(5):984-996.

Meyer-Gutbrod, E.L., C.H. Greene, P.J. Sullivan, and A.J. Pershing. 2015. Climate-associated changes in prey availability drive reproductive dynamics of the North Atlantic right whale population. *Marine Ecology: Progress Series* 535:243-258.

Miksis-Olds, J.L., and S.M. Nichols. 2016. Is low frequency ocean sound increasing globally? *Journal of the Acoustical Society of America* 139(1):501-511.

Miller, C.A., P.B. Best, W.L. Perryman, M.F. Baumgartner, and M.J. Moore. 2012b. Body shape changes associated with reproductive status, nutritive condition and growth in right whales *Eubalaena glacialis* and *E. australis*. *Marine Ecology: Progress Series* 459:135-136.

Miller, P.J.O., N. Biassoni, A. Samuels, and P.L. Tyack. 2000. Whale songs lengthen in response to sonar. *Nature* 405:903.

Miller, P.J.O., M.P. Johnson, and P.L. Tyack. 2004a. Sperm whale behaviour indicates the use of echolocation click buzzes "creaks" in prey capture. *Proceedings of the Royal Society B: Biological Sciences* 271(1554):2239-2247.

Miller, P.J.O., M.P. Johnson, P.L. Tyack, and E.A. Terray. 2004b. Swimming gaits, passive drag, and buoyancy of diving sperm whales *Physeter macrocephalus*. *Journal of Experimental Biology* 207:1953-1967.

Miller, P.J.O., M.P. Johnson, P.T. Madsen, N. Biassoni, M.E. Quero, and P.L. Tyack. 2009. Using at-sea experiments to study the effects of airguns on the foraging behavior of sperm whales in the Gulf of Mexico. *Deep-Sea Research* 56:1168-1181.

Miller, P.J.O., R. Antunes, A. C. Alves, P. Wensveen, P. Kvadsheim, L. Kleivane, N. Nordlund, F.-P. Lam, S. van IJsselmuide, F. Visser, and P. Tyack. 2011. *The 3S Experiments: Studying the Behavioural Effects of Naval Sonar on Killer Whales* (Orcinus orca), *Sperm Whales* (Physeter macrocephalus), *and Long-Finned Pilot Whales* (Globicephala melas) *in Norwegian Waters.* Scottish Oceans Institute Technical Report SOI-2011-001.

Miller, P.J.O., P.H. Kvadsheim, F.P.A. Lam, P.J. Wensveen, R. Antunes, A.C. Alves, F. Visser, L. Kleivane, P.L. Tyack, and L. Doksæter. 2012a. 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., R.N. Antunes, P.J. Wensveen, F.I P. Samarra, A.C. Alves, P.L. Tyack, P.H. Kvadsheim, L. Kleivane, F.-P.A. Lam, M.A. Ainslie, and L. Thomas. 2014. Dose-response relationships for the onset of avoidance of sonar by free-ranging killer whales. *Journal of the Acoustical Society of America* 135:975-993.

Mohr, F.C., B. Lasley, and S. Bursian. 2008. Chronic oral exposure to bunker C fuel oil causes adrenal insufficiency in ranch mink (*Mustela vison*). *Archives of Environmental Contamination and Toxicology* 54:337-347.

Mohr, F.C., B. Lasley, and S. Bursian. 2010. Fuel oil-induced adrenal hypertrophy in ranch mink (*Mustela vison*): Effects of sex, fuel oil weathering, and response to adrenocorticotropic hormone. *Journal of Wildlife Diseases* 46(1):103-110.

Möllmann, C., R. Diekmann, B. Müller-Karulis, G. Kornilovs, M. Plikshs, and P. Axe. 2009. Reorganization of a large marine ecosystem due to atmospheric and anthropogenic pressure: A discontinuous regime shift in the Central Baltic Sea. *Global Change Biology* 15:1377–1393.

Monaghan, P., N.B. Metcalfe, and R. Torres. 2009. Oxidative stress as a mediator of life history trade-offs: Mechanisms, measurements and interpretation. *Ecology Letters* 12(1):75-92.

Montgomery, D.C. 1997. *Design and Analysis of Experiments*. New York: Wiley.

Mooney, T.A., M. Yamato, and B.K. Branstetter. 2012. Hearing in cetaceans: From natural history to experimental biology. *Advances in Marine Biology* 63:197-246.

Moore, J.E., and J.P. Barlow. 2013. Declining abundance of beaked whales (family Ziphiidae) in the California current large marine ecosystem. *PLoS ONE* 8(1):e52770. doi: 10.1371/journal.pone.0052770.

Moore, J.E., and A.J. Read. 2008. A Bayesian uncertainty analysis of cetacean demography and bycatch mortality using age-at-death data. *Ecological Applications* 18(8):1914-1931.

Moore, M.J., and G.A. Early. 2004. Cumulative sperm whale bone damage and the bends. *Science* 306(5705):2215.

Moore, M.J., and J.M. van der Hoop. 2012. The painful side of trap and fixed net fisheries: Chronic entanglement of large whales. *Journal of Marine Biology* 2012:230653.

Moore, S.C., and H.P. Huntington. 2008. Arctic marine mammals and climate change: Impacts and resilience. *Ecological Applications* 18(2 Suppl):s157-165.

Moore, S.E., K.E.W. Shelden, L.K. Litzky, B.A. Mahoney, and D.J. Rugh. 2000. Beluga, *Delphinapterus leucas*, habitat associations in Cook Inlet, Alaska. *Marine Fisheries Review* 62(3):60-80.

Morens, D.M., G.K. Folkers, and A.S. Fauci. 2004. The challenge of emerging and re-emerging infectious diseases. *Nature* 430(6996):242-249.

Moretti, D., L. Thomas, T. Marques, J. Harwood, A. Dilley, B. Neales, J. Shaffer, E. McCarthy, L. New, S. Jarvis, and R. Morrissey. 2014. A risk function for behavioral disruption of Blainville's beaked whales (*Mesoplodon densirostris*) from mid-frequency active sonar. *PLoS ONE* 9:e85064.

Mori, C., B. Morsey, M. Levin, P.R. Nambiar, and S. De Guise. 2006. Immunomodulatory effects of in vitro exposure to organochlorines on T-cell proliferation in marine mammals and mice. *Journal of Toxicology and Environmental Health A.* 69:283-302.

Morris, W.F., and D.F. Doak. 2002. *Quantitative Conservation Biology: Theory and Practice of Population Viability Analysis.* Sunderland, MA: Sinauer Associates.

Morton, A.B., and H.K. Symonds. 2002. Displacement of *Orcinus orca* (L.) by high amplitude sound in British Columbia, Canada. *ICES Journal of Marine Science* 59(1):71-80.

Mullin, K.D., K. Barry, C. Sinclair, J. Litz, K. Maze-Foley, E. Fougeres, B. Mass-Guthrie, R. Ewing, A. Gorgone, J. Adams, and M. Tumlin. 2015. *Common Bottlenose Dolphins* (Tursiops truncatus) *in Lake Pontchartrain, Louisiana: 2007 to Mid-2014.* NOAA Technical Memorandum NMFS-SEFSC 673 43.

Munk, W.H., R.C. Spindel, A. Baggeroer, and T.G. Birdsall. 1994. The Heard Island feasibility test. *Journal of the Acoustical Society of America* 96(4):2330-2342.

Nabe-Nielsen, J., R.M. Sibly, J. Tougaard, J. Teilmann, and S. Sveegaard. 2014. Effects of noise and by-catch on a Danish harbour porpoise population. *Ecological Modeling* 272:242-251.

Narayan, E.J., J.F. Cockrem, and J.M. Hero. 2013. Sight of a predator induces a corticosterone stress response and generates fear in an amphibian. *PLoS ONE* 8:e73564.

New, L.F., J. Harwood, L. Thomas, C. Donovan, J.S. Clark, G. Hastie, P.M. Thompson, B. Cheney, L. Scott-Hayward, and D. Lusseau. 2013a. Modeling the biological significance of behavioural change in coastal bottlenose dolphins in response to disturbance. *Functional Ecology* 27:314-322.

New, L.F., D.J. Moretti, S.K. Hooker, D.P. Costa, and S.E. Simmons. 2013b. Using energetic models to investigate the survival and reproduction of beaked whales (family Ziphiidae). *PLoS ONE* 8:e68725.

New, L.F., J.S. Clark, D.P. Costa, E. Fleishman, M A. Hindell, T. Klanjšček, D. Lusseau, S. Kraus, C.R. McMahon, P.W. Robinson, R.S. Schick, L.K. Schwartz, S.E. Simmons, L. Thomas, P. Tyack, and J. Harwood. 2014. Using short-term measures of behaviour to estimate long-term fitness of southern elephant seals. *Marine Ecology Progress Series* 496:99-108.

New, L.F., A.J. Hall, R. Harcourt, G. Kaufman, E.C.M. Parsons, H.C. Pearson, A.M. Cosentino, and R.S. Schick. 2015. The modeling and assessment of whale-watching impacts. *Ocean and Coastal Management* 115:10-16.

Newman, A.E.M., L.Y. Zanette, M. Clinchy, N. Goodenough, and K.K. Soma. 2013. Stress in the wild: Chronic predator pressure and acute restraint affect plasma DHEA and corticosterone levels in a songbird. *Stress* 16:363-367.

Newman, K.B., S.T. Buckland, S.T. Lindley, L. Thomas, and C. Fernandez. 2006. Hidden process models for animal population dynamics. *Ecological Applications* 16(1):74-86.

Newsome, D., A. Milewski, N. Phillips, and R. Annear. 2002. Effects of horse riding on national parks and other natural ecosystems in Australia: Implications for management. *Journal of Ecotourism* 1(1):52-74.

Nichols, J.D., and B.K. Williams. 2006. Monitoring for conservation. *Trends in Ecology and Evolution* 21:668-673.

Nichols, J.W., M. Breen, R.J. Denver, J.J. DiStefano, J.S. Edwards, R.A. Hoke, D.C. Volz, and X. Zhang. 2011. Predicting chemical impacts on vertebrate endocrine systems. *Environmental Toxicology and Chemistry* 30(1):39-51.

NMFS (National Marine Fisheries Service). 2004. *A Requirements Plan for Improving the Understanding of the Status of U.S. Protected Marine Species: Report of the NOAA Fisheries National Task Force for Improving Marine Mammal and Turtle Stock Assessments.* U.S. Department of Commerce, NOAA Technical Memorandum NMFS-F/SPO-63 112. Available at http://www.nmfs.noaa.gov/pr/pdfs/sars/improvement_plan.pdf.

NMFS. 2008. Recovery Plan for the Steller Sea Lion (*Eumetopias jubatus*). Revision. Silver Spring, MD: National Marine Fisheries Service. 325 pages.

NMFS. 2015. Draft Recovery Plan for the Cook Inlet Beluga Whale (*Delphinapterus leucas*). National Marine Fisheries Service, Alaska Regional Office, Protected Resources Division, Juneau, AK.

NMFS. 2016a. *Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing: Underwater Acoustic Thresholds for Onset of Permanent and Temporary Threshold Shifts.* U.S. Department of Commerce, NOAA Technical Memorandum NMFS-OPR-55, 178 pp.

NMFS. 2016b. Species in the Spotlight. Priority Actions: 2016-2020. Hawaiian Monk Seal (*Neomonachus schauinslandi*). NOAA Fisheries, Silver Spring, MD. Available at http://www.nmfs.noaa.gov/stories/2016/02/docs/hawaiian_monk_seal_spotlight_species__5_year_action_plan_final_web.pdf.

NMFS. 2016c. Species in the Spotlight. Priority Actions: 2016-2020. Southern Resident Killer Whale DPS (*Orcinus ocra*). NOAA Fisheries, Silver Spring, MD. Available at http://www.nmfs.noaa.gov/stories/2016/02/docs/southern_resident_killer_whale_spotlight_species_5_year_action_plan_final_web.pdf.

Nordstad, T., M. Børge, J.O. Bustnes, C. Bech, O. Chastel, A. Goutte, K. Sagerup, C. Trouvé, D. Herzke, and G.W. Gabrielsen. 2012. Relationships between POPs and baseline corticosterone levels in black-legged kittiwakes (*Rissa tridactyla*) across their breeding cycle. *Environmental Pollution* 164:219-226.

Northridge, S.P. 1984. *World Review of Interactions Between Marine Mammals and Fisheries*. Rome: Food and Agricultural Organization of the United Nations.

Nowacek, D.P., M.P. Johnson, and P.L. Tyack. 2004. North Atlantic right whales (*Eubalaena glacialis*) ignore ships but respond to alerting stimuli. *Proceedings of the Royal Society B: Biological Sciences* 271:227-231.

Nowacek, D.P., L.H. Thorne, D.W. Johnston, and P.L. Tyack. 2007. Responses of cetaceans to anthropogenic noise. *Mammal Review* 37:81-115. doi: 10.1111/j.1365-2907.2007.00104.x.

Nowacek, S.M., R.S. Wells, and A.R. Solow. 2001. Short-term effects of boat traffic on bottlenose dolphins, *Tursiops truncatus*, in Sarasota Bay, Florida. *Marine Mammal Science* 17(4):673-688.

NRC (National Research Council). 1994. *Low-Frequency Sound and Marine Mammals: Current Knowledge and Research Needs*. Washington, DC: National Academy Press.

NRC. 2000. *Marine Mammals and Low-Frequency Sound: Progress Since 1994*. Washington, DC: National Academy Press.

NRC. 2003a. *Ocean Noise and Marine Mammals*. Washington, DC: The National Academies Press.

NRC. 2003b. *Decline of the Steller Sea Lion in Alaskan Waters: Untangling Food Webs and Fishing Nets*. Washington, DC: The National Academies Press.

NRC. 2005. *Marine Mammal Populations and Ocean Noise: Determining When Noise Causes Biologically Significant Effects*. Washington, DC: The National Academies Press.

NURC (NATO Undersea Research Centre). 2006. *NATO Undersea Research Centre Human Diver and Marine Mammal Risk Mitigation Rules and Procedures*. NURC-SP-2006-008.

Oakley, G.G., U.-S. Devanaboyina, L.W. Robertson, and R.C. Gupta. 1996. Oxidative DNA damage induced by activation of polychlorinated biphenyls (PCBs): Implications for PCB-induced oxidative stress in breast cancer. *Chemical Research in Toxicology* 9(8):1285-1292.

Ockendon, N., D.J. Baker, J.A. Carr, E.C. White, R.E. Almond, T. Amano, E. Bertram, R.B. Bradbury, C. Bradley, S.H. Butchart, and N. Doswald. 2014. Mechanisms underpinning climatic impacts on natural populations: Altered species interactions are more important than direct effects. *Global Change Biology* 20(7):2221-2229.

O'Connell, S.G., L.D. Kincl, and K.A. Anderson. 2014. Silicone wristbands as personal passive samplers. *Environmental Science & Technology* 48(6):3327-3335.

Oleaga, A., R. Casais, A. Balseiro, A. Espi, L. Llaneza, A. Hartasanchez, and C. Gortazar. 2011. New techniques for an old disease: Sarcoptic mange in the Iberian wolf. *Veterinary Parasitology* 181:255-266.

Olesiuk, P.F., L.M. Nicol, M.J. Sowden, and J.B. Ford. 2002. Effects of the sound generated by an acoustic harassment device on the relative abundance and distribution of harbor porpoises (*Phocoena phocoena*) in Retreat Passage, British Columbia. *Marine Mammal Science* 18(4):843-862.

Ortega, C.P. 2012. Effects of noise pollution on birds: A brief review of our knowledge. *Ornithological Monographs* 74:6-22.

Oskam I, E. Ropstad, E. Lie, A. Derocher, Ø. Wiig, E. Dahl, S. Larsen, and J.U. Skaare. 2004. Organochlorines affect the steroid hormone cortisol in free-ranging polar bears (*Ursus maritimus*) at Svalbard, Norway. *Journal of Toxicology and Environmental Health A* 67:959-977.

OSPAR. 2009. *Overview of the Impacts of Anthropogenic Underwater Sound in the Marine Environment*. OSPAR Convention for the Protection of the Marine Environment of the North-East Atlantic. Available at www.ospar.org. 133 pp.

Parks, S.E., C.W. Clark, and P.L. Tyack. 2007a. 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* 122:3725-3731.

Parks, S.E., D.R. Ketten, J.T. O'Malley, and J. Arruda. 2007b. Anatomical predictions of hearing in the North Atlantic right whale. *Anatomical Record* 290:734-744.

Parks, S.E., M. Johnson, D. Nowacek, and P.L. Tyack. 2010. Individual right whales call louder in increased environmental noise. *Biology Letters* 7:33-35.

Parks, S.E., J.D. Warren, K. Stamiezkin, C.A. Mayo, and D. Wiley. 2012. Dangerous dining: Surface foraging of North Atlantic right whales increases risk of vessel collisions. *Biology Letters* 8:57-60.

Patrick, L. 2009. Thyroid disruption: Mechanism and clinical implications in human health. *Alternative Medicine Review* 14:326-346.

Paulik, L.B., B.W. Smith, A.J. Bergmann, G.J. Sower, N.D. Forsberg, J.G. Teeguarden, and K.A. Anderson. 2016. Passive samplers accurately predict PAH levels in resident crayfish. *Science of the Total Environment* 544:782-791.

Payne, R., and D. Webb. 1971. Orientation by means of long range acoustic signalling in baleen whales. *Annals of the New York Academy of Sciences* 188:110-141.

Peck, H.E., D.P. Costa, and D.E. Crocker. 2016. Body reserves influence allocation to immune responses in capital breeding female northern elephant seals. *Functional Ecology* 30:389-397.

Peetchey, O.L., M. Pontarp, T.M. Massie, S. Kéfi, A. Ozgul, M. Weilenmann, G.M. Palamara, F. Altermatt, B. Matthews, J.M. Levine, D.Z. Childs, B.J. McGill, M.E. Schaepman, B. Schmid, P. Spaak, A.P. Beckerman, F. Pennekamp, and I.S. Pearse. 2015. The Ecological Forecast Horizon, and examples of its uses and determinants. *Ecology Letters* 18:597-611.

Perrin, W.F., and J.E. Powers. 1980. Role of a nematode in natural mortality of spotted dolphins. *Journal of Wildlife Management* 44:960-963.

Peterson, S.H., M.G. Peterson, C. Debier, A. Covaci, A.C. Dirtu, G. Malarvannan, D.E. Crocker, L. Schwarz, and D.P. Costa. 2015. Deep-ocean foraging northern elephant seals bioaccumulate persistent organic pollutants. *Science of the Total Environment* 533:144-155.

Pettis, H.M., R.M. Rolland, P.K. Hamilton, S. Brault, A.R. Knowlton, and S.D. Kraus. 2004. Visual health assessment of North Atlantic right whales (*Eubalaena glacialis*) using photographs. *Canadian Journal of Zoology* 82:8-19.

Piggott, J.J., C.R. Townsend, and C.D. Matthaei. 2015. Reconceptualizing synergism and antagonism among multiple stressors. *Ecology and Evolution* 5:1538-1547.

Pimm, S.L. 1979. The structure of food webs. *Theoretical Population Biology* 16:144-158.

Pirotta, E., N.D. Merchant, P.M. Thompson, T.R. Barton, and D. Lusseau. 2015a. Quantifying the effect of boat disturbance on bottlenose dolphin foraging activity. *Biological Conservation* 181:82-89.

Pirotta, E., J. Harwood, P.M. Thompson, L. New, B. Cheney, M. Arso, P.S. Hammond, C. Donovan, and D. Lusseau. 2015b. Predicting the effects of human developments on individual dolphins to understand potential long-term consequences. *Proceedings of the Royal Society B: Biological Sciences* 282:20152109.

Pirotta, E., P.M. Thompson, B. Cheney, C.R Donovan, and D. Lusseau. 2015c. Estimating spatial, temporal and individual variability in dolphin cumulative exposure to boat traffic using spatially explicit capture–recapture methods. *Animal Conservation* 18:20-31.

Plötz, J., H. Bornemann, R. Knust, A. Schröder, and M. Bester. 2001. Foraging behaviour of Weddell seals, and its ecological implications. *Polar Biology* 24:901-909.

Polischuk, S.C., K.A. Hobson, and M.A. Ramsay. 2001. Use of stable-carbon and -nitrogen isotopes to assess weaning and fasting in female polar bears and their cubs. *Canadian Journal of Zoology* 79(3):499-511.

Proulx, S.R., D.E.L. Promislow, and P.C. Phillips. 2005. Network thinking in ecology and evolution. *Trends in Ecology and Evolution* 20:345-353.

Przeslawski, R., M. Byrne, and C. Mellin. 2015. A review and meta-analysis of the effects of multiple abiotic stressors on marine embryos and larvae. *Global Change Biology* 21(6):2122-2140.

Punt, A.E., and D.S. Butterworth. 1995. The effects of future consumption by the Cape fur seal on catches and catch rates of the Cape hakes. 4. Modeling the biological interaction between Cape fur seals *Arctocephalus pusillus pusillus* and the Cape hakes *Merluccius capensis* and *M. paradoxus*. *South African Journal of Marine Science* 16:255-285.

Quick, N.J., S.J. Middlemas, and J.D. Armstrong. 2004. A survey of antipredator controls at marine salmon farms in Scotland. *Aquaculture* 230:69-180.

Quinn, J.L., M.J. Whittingham, S.J. Butler, and W. Cresswell. 2006. Noise, predation risk compensation and vigilance in the chaffinch *Fringilla coelebs*. *Journal of Avian Biology* 37:601-608. doi: 10.1111/j.2006.0908-8857.03781.x.

Rabalais, N.N., R.E. Turner, Q. Dortch, D. Justic, V. J. Bierman, Jr., and W.J. Wiseman, Jr. 2002. Review. Nutrient-enhanced productivity in the northern Gulf of Mexico: Past, present and future. *Hydrobiologia* 475/476:39-63.

Råberg, L., M. Grahn, D. Hasselquist, and E. Svensson. 1998. On the adaptive significance of stress-induced immunosuppression. *Proceedings of the Royal Society of London B: Biological Sciences* 265(1406):1637-1641.

Rappaport, S.M. 2011. Implications of the exposome for exposure science. *Journal of Exposure Science and Environmental Epidemiology* 21(1):5-9.

Ray, G.C., and F.M. Potter, Jr. 2011. The making of the Marine Mammal Protection Act of 1972. *Aquatic Mammals* 37:520-552.

Rayne, S., M.G. Ikonomou, P.S. Ross, G.M. Ellis, and L.G. Barrett-Lennard. 2004. PBDEs, PBBs, and PCNs in three communities of free-ranging killer whales (*Orcinus orca*) from the northeastern Pacific Ocean. *Environmental Science & Technology* 38(16):4293-4299.

Read, A.J., P. Drinker, and S. Northridge. 2006. Bycatch of marine mammals in US and global fisheries. *Conservation Biology* 20(1):163-169.

Redfern, J.V., M.C. Ferguson, E.A. Becker, K.D. Hyrenbach, C. Good, J. Barlow, K. Kaschner, M.F. Baumgartner, K.A. Forney, L.T. Ballance, P. Fauchald, P. Halpin, T. Hamazaki, A.J. Pershing, S.S. Qian, A. Read, S.B. Reilly, L. Torres, and F. Werner. 2006. Techniques for cetacean-habitat modeling. *Marine Ecology: Progress Series* 310:271-295.

Reeves, R.R., R.J. Hofman, G.K. Silber, and D. Wilkinson. 1996. *Acoustic Deterrence of Harmful Marine Mammal-Fishery Interactions*. Proceedings of a workshop held in Seattle, Washington, March 20-22, 1996. NOAA Technical Memorandum NMFS NMFS-OPR-10. 70 pp.

Regnault, M., and J.P. Lagardère. 1983. Effects of ambient noise on the metabolic level of *Crangon crangon* (Decapoda, Natantia). *Marine Ecology Progress Series* 11:71-78.

Reif, J.S., M.M. Kliks, A.A. Aguirre, and D.L. Borjesson. 2006. Gastrointestinal helminths in the Hawaiian monk seal (*Monachus schauinslandi*): Associations with body size, hematology, and serum chemistry. *Aquatic Mammals* 32:157-167.

Reijnders, P.J.H. 1986. Reproductive failure in common seals feeding on fish from polluted coastal waters. *Nature* 324:456-457.

Reijnders, P.J.H. 2003. Reproductive and developmental effects of environmental organochlorines on marine mammals. Pp. 55-66 in *Toxicology of Marine Mammals*. London: Taylor & Francis.

Reijnen, R., R. Foppen, C. Terbraak, and J. Thissen. 1995. The effects of car traffic on breeding bird populations in woodland. 3. Reduction of density in relation to the proximity of main roads. *Journal of Applied Ecology* 32:187-202.

Reyero, M., E. Cacho, A. Martinez, J. Vazquez, A. Marina, S. Fraga, and J.M. Franco. 1999. Evidence of saxitoxin derivatives as causative agents in the 1997 mass mortality of monk seals in the Cape Blanc Peninsula. *Natural Toxins* 7:311-315.

Richardson, W.J., B. Würsig, and C.R. Greene, Jr. 1986. Reactions of bowhead whales, *Balaena mysticetus*, to seismic exploration in the Canadian Beaufort Sea. *Journal of the Acoustical Society of America* 79(4):1117-1128.

Ricklefs, R.E., and M. Wikelski. 2002. The physiology/life-history nexus. *Trends in Ecology & Evolution* 17(10):462-468.

Rider, C.V., M.L. Dourson, R.C. Hertzberg, M.M. Mumtaz, P.S. Price, and J.E. Simmons. 2012. Incorporating nonchemical stressors into cumulative risk assessments. *Toxicological Sciences* 127(1):10-17.

Rigét, F., B. Braune, A. Bignert, S. Wilson, J. Aars, E. Born, M. Dam, R. Dietz, M. Evans, T. Evans, and M. Gamberg. 2011. Temporal trends of Hg in Arctic biota, an update. *Science of the Total Environment* 409(18):3520-3526.

Roberts, J.J., B.D. Best, L. Mannocci, E. Fujioka, P.N. Halpin, D.L. Palka, L.P. Garrison, K.D. Mullin, T.V.N. Cole, C.B. Khan, W.A. McLellan, D.A. Pabst, and G.G. Lockhart. 2016. Habitat-based cetacean density models for the U.S. Atlantic and Gulf of Mexico. *Scientific Reports* 6:22615.

Robinson, P.W., D.P. Costa, D.E. Crocker, J.P. Gallo-Reynoso, C.D. Champagne, M.A. Fowler, C. Goetsch, K.T. Goetz, J.L. Hassrick, L.A. Hückstädt, and C.E. Kuhn. 2012. Foraging behavior and success of a mesopelagic predator in the northeast Pacific Ocean: Insights from a data-rich species, the northern elephant seal. *PLoS ONE* 7:e36728.

Rocha, R.C., P.J. Clapham, and Y.V. Ivashchenko. 2014. Emptying the oceans: A summary of industrial whaling catches in the 20th century. *Marine Fisheries Review* 76(4):37-48.

Rolland, R.M., S.E. Parks, K.E. Hunt, M. Castellote, P.J. Corkeron, D.P. Nowacek, S.K. Wasser, and S.D. Kraus. 2012. Evidence that ship noise increases stress in right whales. *Proceedings of the Royal Society B: Biological Sciences*. 279(1737):2363-2368.

Rolland, R.M., R.S. Schick, H.M. Pettis, A.R. Knowlton, P.K. Hamilton, J.S. Clark, and S.D. Kraus. 2016. Health of North Atlantic right whales (*Eubalaena glacialis*) over three decades: From individual health to demographic and population trends. *Marine Ecology: Progress Series* 542:265-282.

Rolland, V., M. Nevoux, C. Barbraud, and H. Weimerskirch. 2009. Respective impact of climate and fisheries on the growth of an albatross population. *Ecological Applications* 19:1336-1346.

Rollins-Smith, L.A., and P.J. Blair. 1993. The effects of corticosteroid hormones and thyroid hormones on lymphocyte viability and proliferation during development and metamorphosis of *Xenopus laevis*. *Differentiation* 54(3):155-160.

Romano, T.A., M.J. Keogh, C. Kelly, P. Feng, L. Berk, C.E. Schlundt, D.A. Carder, and J.J. Finneran. 2004. Anthropogenic sound and marine mammal health: Measures of the nervous and immune systems before and after intense sound exposure. *Canadian Journal of Fisheries and Aquatic Sciences* 61(7):1124-1134.

Romero, L.M., M.J. Dickens, and N.E. Cyr. 2009. The reactive scope model—a new model integrating homeostasis, allostasis, and stress. *Hormones and Behavior* 55(3):375-389.

Roos, M.M.H., G.M. Wu, and P.J.O. Miller. 2016. The significance of respiration timing in the energetics estimates of free-ranging killer whales (*Orcinus orca*). *Journal of Experimental Biology* 219(13):2066-2077.

Rosen, D.A., and S. Kumagai. 2008. Hormone changes indicate that winter is a critical period for food shortages in Steller sea lions. *Journal of Comparative Physiology B* 178:573-583.

Ross, P., R. De Swart, P. Reijnders, H.V. Loveren, J. Vos, and A. Osterhaus. 1995. Contaminant-related suppression of delayed-type hypersensitivity and antibody responses in harbor seals fed herring from the Baltic Sea. *Environmental Health Perspectives* 103:162-167.

Ross, P., R. De Swart, R. Addison, H. Van Loveren, J. Vos, and A. Osterhaus. 1996a. Contaminant-induced immunotoxicity in harbour seals: Wildlife at risk? *Toxicology* 112(2):157-169.

Ross, P., R. De Swart, H. Timmerman, P. Reijnders, J. Vos, H. V. Loveren, and A. Osterhaus. 1996b. Suppression of natural killer cell activity in harbour seals (*Phoca vitulina*) fed Baltic Sea herring. *Aquatic Toxicology* 34:71-84.

Ross, P.S., G.M. Ellis, M.G. Ikonomou, L.G. Barrett-Lennard, and R.F. Addison. 2000. High PCB concentrations in free-ranging Pacific killer whales, *Orcinus orca*: Effects of age, sex and dietary preference. *Marine Pollution Bulletin* 40:504-515.

Rotander, A., B. van Bavel, A. Polder, F. Rigét, G.A. Auðunsson, G.W. Gabrielsen, G. Víkingsson, D. Bloch, and M. Dam. 2012. Polybrominated diphenyl ethers (PBDEs) in marine mammals from Arctic and North Atlantic regions, 1986-2009. *Environment International* 40:102-109.

Rotstein, D.S., L.G. Burdett, W. McLellan, L. Schwacke, T. Rowles, K.A. Terio, S. Schultz, and A. Pabst. 2009. Lobomycosis in offshore bottlenose dolphins (*Tursiops truncatus*), North Carolina. *Emerging Infectious Diseases* 15(4):588-590.

Royle, J.A., R.B. Chandler, R. Shollmann, and B. Gardner. 2013. *Spatial Capture Recapture*. New York: Elsevier.

Rudd, M.A. 2014. Scientists' perspectives on global ocean research priorities. *Frontiers in Marine Science* 1:36. doi: 10.3389/fmars.2014.00036.

Rugh, D.J., K.E.W. Shelden, and R.C. Hobbs. 2010. Range contraction in a beluga whale population. *Endangered Species Research* 12:69-75.

Runge, M.C., S.J. Converse, and J.E. Lyons. 2011. Which uncertainty? Using expert elicitation and expected value of information to design an adaptive program. *Biological Conservation* 144(4):1214-1223. doi: 10.1016/j.biocon.2010.12.020.

Runge, M.C., C.A. Langtimm, J. Martin, and C.J. Fonnesbeck. 2015. *Status and Threats Analysis for the Florida Manatee (Trichechus manatus latirostris), 2012*. U.S. Geological Survey Open-File Report 2015-1083. doi: 10.3133/ofr20151083.

Ryser-Degiorgis, M.-P. 2013. Wildlife health investigations: Needs, challenges and recommendations. *BMC Veterinary Research* 9:223.

Samhouri, J.F., A.J. Haupt, P.S. Levin, J.S. Link, and R. Shuford. 2014. Lessons learned from developing integrated ecosystem assessments to inform marine ecosystem-based management in the USA. *ICES Journal of Marine Science* 71:1205-1215.

Sapolsky, R.M., L.M. Romero, and A.U. Munck. 2000. How do glucocorticoids influence stress responses? Integrating permissive, suppressive, stimulatory, and preparative actions 1. *Endocrine Reviews* 21(1):55-89.

Scales, K.L., P.I. Miller, L.A. Hawkes, S.N. Ingram, D.W. Sims, and S.C. Votier. 2014. On the front line: Frontal zones as priority at-sea conservation areas for mobile marine vertebrates. *Journal of Applied Ecology* 51(6):1575-1583.

Schaub, A., J. Ostwald, and B. M. Siemers. 2008. Foraging bats avoid noise. *Journal of Experimental Biology* 211:3174-3180.

Scheffer, M. 2009. *Critical Transitions in Nature and Society*. Princeton, NJ: Princeton University Press.

Scheffer, M., S. Carpenter, J.A. Foley, C. Folke, and B. Walker. 2001. Catastrophic shifts in ecosystems. *Nature* 413:591-596.

Scheffer, M., J. Bascompte, W.A. Brock, V. Brovkin, S.R. Carpenter, V. Dakos, H. Held, E.H. van Nes, M. Rietkerk, and G. Sugihara. 2009. Early-warning signals for critical transitions. *Nature* 461:53-59.

Scheffer, V.B. 1950. Growth layers on the teeth of Pinnipedia as an indication of age. *Science* 112(2907):309-311.

Schick, R.S., L.F. New, L. Thomas, D.P. Costa, M.A. Hindell, C.R. McMahon, P.W. Robinson, S.E. Simmons, M. Thums, J. Harwood, and J.S. Clark. 2013. Estimating resource acquisition and at-sea body condition of a marine predator. *Journal of Animal Ecology* 82(6):1300-1315.

Scholin, C.A., F. Gulland, G.J. Doucette, S. Benson, M. Busman, F.P. Chavez, J. Cordaro, R. DeLong, A. De Vogelaere, J. Harvey, M. Haulena, K. Lefebvre, T. Lipscomb, S. Loscutoff, L.J. Lowenstine, R. Marin, P.E. Miller, W.A. McLellan, P.D.R. Moeller, C.L. Powell, T. Rowles, P. Silvagni, M. Silver, T. Spraker, V. Trainer, and F.M. Van Dolah. 2000. Mortality of sea lions along the central California coast linked to a toxic diatom bloom. *Nature* 403:80-84.

Schultner, J., B. Moe, O. Chastel, S. Tartu, C. Bech, and A.S. Kitaysky. 2014. Corticosterone mediates carry-over effects between breeding and migration in the kittiwake *Rissa tridactyla*. *Marine Ecology Progress Series* 496:125-133.

Schwacke, L.H., E.O. Voit, L.J. Hansen, R.S. Wells, G.B. Mitchum, A.A. Hohn, and P.A. Fair. 2002. Probabilistic risk assessment of reproductive effects of polychlorinated biphenyls on bottlenose dolphins (*Tursiops truncatus*) from the Southeast United States coast. *Environmental Toxicology and Chemistry* 21:2752-2764. doi: 10.1002/etc.5620211232.

Schwacke, L.H., M.J. Twiner, S. De Guise, B.C. Balmer, R.S. Wells, F.I. Townsend, D.C. Rotstein, R.A. Varela, L.J. Hansen, E.S. Zolman, T.R. Spradlin, M. Levin, H. Leibrecht, Z.H. Wang, and T.K. Rowles. 2010. Eosinophilia and biotoxin exposure in bottlenose dolphins (*Tursiops truncatus*) from a coastal area impacted by repeated mortality events. *Environmental Research* 110:548-555.

Schwacke, L.H., E.S. Zolman, B.C. Balmer, S. De Guise, R.C. George, J. Hoguet, A.A. Hohn, J.R. Kucklick, S. Lamb, M. Levin, J.A. Litz, W.E. McFee, N.J. Place, F.I. Townsend, R.S. Wells, and T.K. Rowles. 2012. Anaemia, hypothyroidism and immune suppression associated with polychlorinated biphenyl exposure in bottlenose dolphins (*Tursiops truncatus*). *Proceedings of the Royal Society B: Biological Sciences* 279:48-57.

Schwacke, L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, Jr., S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman, and T.K. Rowles. 2014a. Health of common bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana, following the *Deepwater Horizon* oil spill. *Environmental Science and Technology* 48:93-103.

Schwacke, L.H., C.R. Smith, F.I. Townsend, R.S. Wells, L.B. Hart, B.C. Balmer, T.K. Collier, S. De Guise, M.M. Fry, L.J. Guillette, Jr., S.V. Lamb, S.M. Lane, W.E. McFee, N.J. Place, M.C. Tumlin, G.M. Ylitalo, E.S. Zolman, and T.K. Rowles. 2014b. Response to comment on health of common bottlenose dolphins (*Tursiops truncatus*) in Barataria Bay, Louisiana following the *Deepwater Horizon* oil spill. *Environmental Science & Technology* 48:4209-4211.

Schwartz, J.A., B.M. Aldridge, J.L. Stott, and F.C. Mohr. 2004. Immunophenotypic and functional effects of bunker C fuel oil on the immune system of American mink (*Mustela vison*). *Veterinary Immunology and Immunopathology* 101:179-190.

Schwarz, K.B. 1996. Oxidative stress during viral infection: A review. *Free Radical Biology and Medicine* 21(5):641-649.

Scott, M.D., and S.J. Chivers. 2009. Movements and diving behavior of pelagic spotted dolphins. *Marine Mammal Science* 25(1):137-160.

Seixas, T.G., H.D.A. Kehrig, M. Costa, G. Fillmann, A.P.M. Di Beneditto, E.R. Secchi, C.M. Souza, O. Malm, and I. Moreira. 2008. Total mercury, organic mercury and selenium in liver and kidney of a South American coastal dolphin. *Environmental Pollution* 154(1):98-106.

Selkoe, K.A.,T. Blenckner, M.R. Caldwell, L.B. Crowder, A.L. Erickson, T.E. Essington, J.A. Estes, R.M. Fujita, B.S. Halpern, M.E. Hunsicker, C.V. Kappel, R.P. Kelly, J.N. Kittinger, P.S. Levin, J.M. Lynha, M.E. Mach, R.G. Martone, L.A. Mease, A.K. Salomon, J.F. Samhouri, C.S. Scahborough, A.C. Stier, A.C. Stier, C. White, and J. Zedler. 2015. Principles for managing marine ecosystems prone to tipping points. *Ecosystem Health and Sustainability* 1(5):17.

Senzaki, M., Y. Yamaura, C.D. Francis, and F. Nakamura. 2016. Traffic noise reduces foraging efficiency in wild owls. *Scientific Reports* 6:30602.

Sepulveda, M., and D. Oliva. 2005. Interactions between South American sea lions *Otaria flavescens* (Shaw) and salmon farms in southern Chile. *Aquaculture Research* 36:1062-1068.

Shannon, G., L.M. Angeloni, G. Wittemyer, K.M. Fristrup, and K.R. Crooks. 2014. Road traffic noise modifies behaviour of a keystone species. *Animal Behaviour* 94:135-141.

Shannon, G., M.F. McKenna, L.M. Angeloni, K.R. Crooks, K.M. Fristrup, E. Brown, K.A. Warner, M.D. Nelson, C. White, and J. Briggs. 2015. A synthesis of two decades of research documenting the effects of noise on wildlife. *Biological Reviews* 91(4):982-1005. doi: 10.1111/brv.12207.

Shannon, G., K.R. Crooks, G. Wittemyer, K.M. Fristrup, and L.M. Angeloni. 2016. Road noise causes earlier predator detection and flight response in a free-ranging mammal. *Behavioral Ecology*. doi: 10.1093/beheco/arw058.

Shelden, K.E.W., D.J. Rugh, B.A. Mahoney, and M.E. Dahlheim. 2003. Killer whale predation on belugas in Cook Inlet, Alaska: Implications for a depleted population. *Marine Mammal Science* 19(3):529-544.

Shelden, K.E.W., C.L. Sims, L. Vate Brattström, K.T. Goetz, and R.C. Hobbs. 2015. *Aerial Surveys of Beluga Whales (Delphinapterus leucas) in Cook Inlet, Alaska, June 2014.* AFSC Processed Report 2015-03. NOAA Alaska Fisheries Science Center, National Marine Fisheries Service, Seattle, WA.

Sheldon, B.C., and S. Verhulst. 1996. Ecological immunology: Costly parasite defences and trade-offs in evolutionary ecology. *Trends in Ecology & Evolution* 11(8):317-321.

Shephard, R.J., and P.N. Shek. 1998. Cold exposure and immune function. *Canadian Journal of Physiological Pharmacology* 76:828-836.

Shero, M.R., R.T. Krotz, D.P. Costa, J.P. Avery, and J.M. Burns. 2015. How do overwinter changes in body condition and hormone profiles influence Weddell seal reproductive success? *Functional Ecology* 29:1278-1291.

Siemers, B.M., and A. Schaub. 2011. Hunting at the highway: Traffic noise reduces foraging efficiency in acoustic predators. *Proceedings of the Royal Society B: Biological Sciences* 278:1646-1652.

Sih, A., G. Englund, and D. Wooster. 1998. Emergent impacts of multiple predators on prey. *Trends in Ecology & Evolution* 13:350-355.

Simeone, C.A., F.M. Gulland, T. Norris, and T.K. Rowles. 2015. A systematic review of changes in marine mammal health in North America, 1972-2012: The need for a novel integrated approach. *PLoS ONE* 10:e0142105.

Sivle, L.D., P.H. Kvadsheim, A. Fahlman, F.P.A. Lam, P.L. Tyack, and P.J.O. Miller. 2012. Changes in dive behavior during naval sonar exposure in killer whales, long-finned pilot whales, and sperm whales. *Frontiers in Physiology* 3(400):1-11.

Small, R.J., P.L. Boveng, G.V. Byrd, and D.E. Withrow. 2008. Harbor seal population decline in the Aleutian Archipelago. *Marine Mammal Science* 24(4):845-863.

Smith, C.R., T.K. Rowles, L.B. Hart, F.I. Townsend, R.S. Wells, E.S. Zolman, B.C. Balmer, B. Quigley, M. Ivančić, W. McKercher, M.C. Tumlin, K.D. Mullin, J.D. Adams, Q. Wu, W. McFee, T.K. Collier, and L.H. Schwacke. 2017. Slow recovery of Barataria Bay dolphin health following the *Deepwater Horizon* oil spill (2013–2014), with evidence of persistent lung disease and impaired stress response. *Endangered Species Research* 33:127-147. doi: 10.3354/esr00778

Smith, T.D., J. Allen, P.J. Clapham, P.S. Hammond, S. Katona, F. Larsen, J. Lien, D. Mattila, P.J. Palsbøll, J. Sigurjónsson, and P.T. Stevick. 1999. An ocean-basin-wide mark-recapture study of the North Atlantic humpback whale (*Megaptera novaeangliae*). *Marine Mammal Science* 15(1):1-32.

Smout, S., A. Rindorf, P.S. Hammond, J. Harwood, and J. Matthiopoulos. 2014. Modelling prey consumption and switching by UK grey seals. *ICES Journal of Marine Science: Journal du Conseil* 71:81-89.

Sormo, E.G., I. Jussi, M. Jussi, M. Braathen, J.U. Skaare, and B.M. Jenssen. 2005. Thyroid hormone status in gray seal (*Halichoerus grypus*) pups from the Baltic Sea and the Atlantic Ocean in relation to organochlorine pollutants. *Environmental Toxicology and Chemistry* 24:610-616.

Soto, S., A. Alba, L. Ganges, E. Vidal, J.A. Raga, F. Alegre, B. González, P. Medina, I. Zorrilla, J. Martínez, and A. Marco. 2011. Post-epizootic chronic dolphin morbillivirus infection in Mediterranean striped dolphins *Stenella coeruleoalba*. *Diseases of Aquatic Organisms* 96(3):187-194.

Southall, B.L., A.E. Bowles, W.T. Ellison, J.J. Finneran, R.L. Gentry, C.R. Greene, Jr., D. Kastak, D.R. Ketten, J.H. Miller, P.E. Nachtigall, W.J. Richardson, J.A. Thomas, and P.L. Tyack. 2007. Marine mammal noise exposure criteria: Initial scientific recommendations. *Aquatic Mammals* 33:411-521.

Spoon, T.R., and T.A. Romano. 2012. Neuroimmunological response of beluga whales (*Delphinapterus leucas*) to translocation and a novel social environment. *Brain, Behavior, and Immunity* 26:22-131.

Springer, A.M., J.A. Estes, G.B. van Vliet, T.M. Williams, D.F. Doak, E.M. Danner, K.A. Forney, and B. Pfister. 2003. Sequential megafaunal collapse in the North Pacific Ocean: An ongoing legacy of industrial whaling? *Proceedings of the National Academy of Sciences of the United States of America* 100(21):12223-12228.

Springer, A.M., J.A. Estes, G.B. Van Vliet, T.M. Williams, D.F. Doak, E.M. Danner, and B. Pfister. 2008. Mammal-eating killer whales, industrial whaling, and the sequential megafaunal collapse in the North Pacific Ocean: A reply to critics of Springer et al. 2003. *Marine Mammal Science* 24(April):414-442.

Stamation, K.A., D.B. Croft, and P.D. Shaughnessy. 2009. Behavioral responses of humpback whales (*Megaptera novaeangliae*) to whale-watching vessels on the southeastern coast of Australia. *Marine Mammal Science* 26(1):98-122. doi: 10.1111/j.1748-7692.2009.00320.x.

Steele, J.H. 2004. Regime shifts in the ocean: Reconciling observations and theory. *Progress in Oceanography* 60(2):135-141.

Steiger, J.W. 1994. The Consultation Provision of Section 7(a)(2) of the Endangered Species Act and its application to delegable federal programs. *Ecology Law Quarterly* 21:2. doi: 10.15779/Z382K05.

Steneck, R.S., M.H. Graham, B.J. Bourque, D. Corbett, J.M. Erlandson, J.A. Estes, and M.J. Tegner. 2002. Kelp forest ecosystems: Biodiversity, stability, resilience and future. *Environmental Conservation* 29(4):436-459.

Stephen, C. 2014. Toward a modernized definition of wildlife health. *Journal of Wildlife Diseases* 50(3):427-430.

Stephens, P.A., W.J. Sutherland, and R.P. Freckleton. 1999. What is the Allee effect? *Oikos* 87:185-190.

Sternberg, E.M., G.P. Chrousos, R.L. Wilder, and P.W. Gold. 1992. The stress response and the regulation of inflammatory disease. *Annals of Internal Medicine* 117(10):854-866.

Stewart, K.R., R.L. Lewison, D.C. Dunn, R.H. Bjorkland, S. Kelez, P.N. Halpin, and L.B. Crowder. 2010. Characterizing fishing effort and spatial extent of coastal fisheries. *PLoS ONE* 5(12):e14451.

Stewart, P.W., J. Reihman, E. Lonky, T.J. Darvill, and J. Pagano. 2003. Cognitive development in preschool children prenatally exposed to PCBs and MeHg. *Neurotoxicology and Teratology* 25:11-22.

Stewart, P.W., E. Lonky, J. Reihman, J. Pagano, B.B. Gump, and T. Darvill. 2008. The relationship between prenatal PCB exposure and intelligence (IQ) in 9-year-old children. *Environmental Health Perspectives* 116:1416-1422.

Stier, K.S., B. Almasi, J. Gasparini, R. Piault, A. Roulin, and L. Jenni. 2009. Effects of corticosterone on innate and humoral immune functions and oxidative stress in barn owl nestlings. *Journal of Experimental Biology* 212(13):2085-2091.

Stirling, I., and C.L. Parkinson. 2006. Possible effects of climate warming on selected populations of polar bears (*Ursus maritimus*) in the Canadian Arctic. *Arctic* 59(3):261-275.

Stirling, I., C. Spencer, and D. Andriashek. 1989. Immobilization of polar bears (*Ursus maritimus*) with Telazol® in the Canadian Arctic. *Journal of Wildlife Diseases* 25(2):159-168.

Stirling, I., N.J. Lunn, and J. Iacozza. 1999. Long-term trends in the population ecology of polar bears in western Hudson Bay in relation to climatic change. *Arctic* 52(3):294-306.

Stirling, I., N.J. Lunn, J. Iacozza, C. Elliott, and M. Obbard. 2004. Polar bear distribution and abundance on the southwestern Hudson Bay coast during open water season, in relation to population trends and annual ice patterns. *Arctic* 57(1):15-26.

Stohs, S.J. 1990. Oxidative stress induced by 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). *Free Radical Biology and Medicine* 9(1)79-90.

Sugihara, G., R. May, H. Ye, C.H. Hsieh, E. Deyle, M. Fogarty, and S. Munch. 2012. Detecting causality in complex ecosystems. *Science* 338(6106):496-500.

Sutherland, J.P. 1974. Multiple stable points in natural communities. *American Naturalist* 108(964):859-873.

Sutherland, W.J. 2006. Predicting the ecological consequences of environmental change: A review of the methods. *Journal of Applied Ecology* 43:599-616.

Sutherland, W.J., and M.A. Burgman. 2015. Use experts wisely. *Nature (London)* 526:317-318.

Svensson, E., L. Råberg, C. Koch, and D. Hasselquist. 1998. Energetic stress, immunosuppression and the costs of an antibody response. *Functional Ecology* 12(6):912-919.

Swaddle, J.P., C.D. Francis, J.R. Barber, C.B. Cooper, C.M. Kyba, D.M. Dominoni, G. Shannon, E. Aschehoug, S.E. Goodwin, A.Y. Kawahara, D. Luther, K. Spoelstra, M. Voss, and T. Longcore. 2015. A framework to assess evolutionary responses to anthropogenic light and sound. *Trends in Ecology & Evolution* 30(9):550-560.

Symons, J., E. Pirotta, and D. Lusseau. 2014. Sex differences in risk perception in deep-diving bottlenose dolphins leads to decreased foraging efficiency when exposed to human disturbance. *Journal of Applied Ecology* 51:1584-1592.

Tabuchi, M., N. Veldhoen, N. Dangerfield, S. Jeffries, C.C. Helbing, and P.S. Ross. 2006. PCB-related alteration of thyroid hormones and thyroid hormone receptor gene expression in free-ranging harbor seals (*Phoca vitulina*). *Environmental Health Perspectives* 114:1024-1031.

Tartu, S., F. Angelier, J. Bustnes, B. Moe, S. Hanssen, D. Herzke, G.W. Gabrielsen, N. Verboven, J. Jerreault, P. Labadie, H. Budzinski, J.C. Wingfield, and O. Chastel. 2015. Polychlorinated biphenyl exposure and corticosterone levels in seven polar seabird species. *Environmental Pollution* 197:173-180.

Tasker, M.L., M. Amundin, M. Andre, A.D. Hawkins, W. Lang, T. Merck, A. Scholik-Schlomer, J. Teilman, F. Thomsen, S. Werner, and M. Zakharia. 2010. *Marine Strategy Framework Directive: Task Group 11 Report: Underwater Noise and Other Forms of Energy*. JRC Scientific and Technical Report No. EUR 24341 EN-2010, European Commission and International Council for the Exploration of the Sea, Luxembourg.

Taylor, B.L., and D.P. Demaster. 1993. Implications of non-linear density dependence. *Marine Mammal Science* 9(4):360-371.

Taylor, B.L., P.R. Wade, D.P. De Master, and J. Barlow. 2000. Incorporating uncertainty into management models for marine mammals. *Conservation Biology* 14:1243-1252.

Taylor, B.L., M. Martinez, T. Gerrodette, J. Barlow, and Y.N. Hrovat. 2007. Lessons from monitoring trends in abundance of marine mammals. *Marine Mammal Science* 23:157-175.

Tennessen, J.B., S.E. Parks, and T. Langkilde. 2014. Traffic noise causes physiological stress and impairs breeding migration behaviour in frogs. *Conservation Physiology* 2:cou032.

Thomas, C.D., A. Cameron, R.E. Green, M. Bakkenes, L.J. Beaumont, Y.C. Collingham, B.F.N. Erasmus, M. Ferreira de Siqueira, A. Grainger, L. Hannah, L. Hughes, B. Huntley, A.S. van Jaarsveld, G.F. Midgley, L. Miles, M.A. Ortega-Huerta, A.T. Peterson, O.L. Phillips, and S.E. Williams. 2004. Extinction risk from climate change. *Nature* 427:145-148. doi: 10.1038/nature02121.

Thomas, L., S.T. Buckland, K.B. Newman, and J. Harwood. 2005. A unified framework for modeling wildlife population dynamics. *Australian and New Zealand Journal of Statistics* 47:19-34.

Thomas, P.O., R.R. Reeves, and R.L. Brownell. 2016. Status of the world's baleen whales. *Marine Mammal Science* 32(2):682-734.

Thompson, D., J. Onoufriou, A. Brownlow, and A. Bishop. 2015. Preliminary Report on Predation by Adult Grey Seals on Grey Seal Pups as a Possible Explanation for Corkscrew Injury Patterns Seen in the Unexplained Seal Deaths: Addendum. Sea Mammal Research Unit, University of St. Andrews, Report to Scottish Government. Available at http://www.smru.st-andrews.ac.uk/documents/scotgov/USD1and%206_addendum_report_VF2.pdf.

Thompson, P.M., B. Mackey, T.R. Barton, C. Duck, and J.R.A. Butler. 2007. Assessing the potential impact of salmon fisheries management on the conservation status of harbour seals (*Phoca vitulina*) in north-east Scotland. *Animal Conservation* 10:48-56.

Thompson, P.M., D. Lusseau, T. Barton, D. Simmons, J. Rusin, and H. Bailey. 2010. Assessing the responses of coastal cetaceans to the construction of offshore wind turbines. *Marine Pollution Bulletin* 60:1200-1208.

Thompson, P.M., K.L. Brookes, I.M. Graham, T.R. Barton, K. Needham, G. Bradbury, and N.D. Merchant. 2013a. Short-term disturbance by a commercial two-dimensional seismic survey does not lead to long-term displacement of harbour porpoises. *Proceedings of the Royal Society of London B: Biological Sciences* 280:20132001.

Thompson, P.M., G.D. Hastie, J. Nedwell, R. Barham, K.L. Brookes, L.S. Cordes, H. Bailey, and N. Mclean. 2013b. Framework for assessing impacts of pile-driving noise from offshore wind farm construction on a harbour seal population. *Environmental Impact Assessment Review* 43:73-85.

Tinker, M.T., G. Bentall, and J.A. Estes. 2008. Food limitation leads to behavioral diversification and dietary specialization in sea otters. *Proceedings of the National Academy of Sciences of the United States of America* 105:560-565.

Tougaard, J., J. Carstensen, and J. Teilmann. 2009. Pile driving zone of responsiveness extends beyond 20 km for harbor porpoises (*Phocoena phocoena* (L.)). *Journal of the Acoustical Society of America* 126:11-14.

Trickey, J.S., B.K. Branstetter, and J.J. Finneran. 2010. Auditory masking of a 10 kHz tone with environmental, comodulated, and Gaussian noise in bottlenose dolphins (*Tursiops truncatus*). *Journal of the Acoustical Society of America* 128:3799-3804.

Trillmich, F., K.A. Ono, D.P. Costa, R.L. DeLong, S.D. Feldkamp, J.M. Francis, R.L. Gentry, C.B. Heath, B.J. Le Boeuf, P. Majluf, and A.E. York. 1991. The effects of El Niño on pinniped populations in the eastern Pacific. Pp. 247-270 in *Pinnipeds and El Niño.* Berlin: Springer.

Trites, A.W., V.B. Deecke, E.J. Gregr, J.K.B. Ford, and P.F. Olesiuk. 2007. Killer whales, whaling, and sequential megafaunal collapse in the North Pacific: A comparative analysis of the dynamics of marine mammals in Alaska and British Columbia following commercial whaling. *Marine Mammal Science* 23(4):751-765.

Trumble, S.J., E.M. Robinson, M. Berman-Kowalewski, C.W. Potter, and S. Usenko. 2013. Blue whale earplug reveals lifetime contaminant exposure and hormone profiles. *Proceedings of the National Academy of Sciences of the United States of America* 110(42):16922-16926.

Tubelli, A.A., A. Zosuls, D.R. Ketten, M. Yamato, and D.C. Mountain. 2012. A prediction of the minke whale (*Balaenoptera acutorostrata*) middle-ear transfer function. *Journal of the Acoustical Society of America* 132:3263-3272.

Turvey, S.T., R.L. Pitman, B.L. Taylor, J. Barlow, T. Akamatsu, L.A. Barrett, X. Zhao, R.R. Reeves, B.S. Stewart, K. Wang, and Z. Wei. 2007. First human-caused extinction of a cetacean species? *Biology Letters* 3(5):537-540.

Twiner, M.J., S. Fire, L. Schwacke, L. Davidson, Z.H. Wang, S. Morton, S. Roth, B. Balmer, T.K. Rowles, and R.S. Wells. 2011. Concurrent exposure of bottlenose dolphins (*Tursiops truncatus*) to multiple algal toxins in Sarasota Bay, Florida, USA. *PLoS ONE* 6(3):e17394.

Twiner, M.J., L.J. Flewelling, S.E. Fire, S.R. Bowen-Stevens, J.K. Gaydos, C.K. Johnson, J.H. Landsberg, T.A. Leighfield, B. Mase-Guthrie, L. Schwacke, F.M. Van Dolah, Z. Wang, and T.K. Rowles. 2012. Comparative analysis of three brevetoxin-associated bottlenose dolphin (*Tursiops truncatus*) mortality events in the Florida Panhandle region (USA). *PLoS ONE* 7:e42974.

Tyack, P.L., and C.W. Clark. 1998. *Quick-Look Report: Playback of Low-Frequency Sound to Gray Whales Migrating Past the Central California Coast*. Woods Hole, MA: Woods Hole Oceanographic Institution.

Tyack, P.L., and V.J. Janik. 2013. Effects of noise on acoustic signal production. Pp. 251-271 in *Animal Communication and Noise*, H. Brumm, ed. Berlin: Springer.

Tyack, P.L., W.M.X. Zimmer, D. Moretti, B.L. Southall, D.E. Claridge, J.W. Durban, C.W. Clark, A. D'Amico, N. DiMarzio, S. Jarvis, E. McCarthy, R. Morrissey, J. Ward, and I.L. Boyd. 2011. Beaked whales respond to simulated and actual navy sonar. *PLoS ONE* 6(3):e17009.

Uchida, K., M. Muranaka, Y. Horii, N. Murakami, R. Yamaguchi, and S. Tateyama. 1999. Non-purulent meningoencephalomyelitis of a Pacific striped dolphin (*Lagenorhynchus obliquidens*). The first evidence of morbilivirus infection in a dolphin at the Pacific Ocean around Japan. *Journal of Veterinary Medical Science* 61(2):159-162.

Ulbrich, B., and R. Stahlmann. 2004. Developmental toxicity of polychlorinated biphenyls (PCBs): A systematic review of experimental data. *Archives of Toxicology* 78:252-268.

U.S. Department of the Navy. 2013. Hawaiian-Southern California Training and Testing Activities. Final Environmental Impact Statement/Overseas Environmental Impact Statement. Available at http://hstteis.com/Completedin2013//DocumentsandReferences/HSTTDocuments/FinalEISOEIS.aspx.

USFWS (U.S. Fish and Wildlife Service). 2013. Southwest Alaska Distinct Population Segment of the Northern Sea Otter (*Enhydra lutris kenyoni*) Recovery Plan. Available at https://www.fws.gov/alaska/fisheries/mmm/seaotters/pdf/Recovery%20Plan%20SW%20AK%20DPS%20Sea%20Otter%20Aug13.pdf.

Vacquié-Garcia, J., F. Royer, A.-C. Dragon, M. Viviant, F. Bailleul, and C. Guinet. 2012. Foraging in the darkness of the Southern Ocean: Influence of bioluminescence on a deep diving predator. *PLoS ONE* 7(8):e43565.

Valavanidis, A., T. Vlahogianni, M. Dassenakis, and M. Scoullos. 2006. Molecular biomarkers of oxidative stress in aquatic organisms in relation to toxic environmental pollutants. *Ecotoxicology and Environmental Safety* 64(2):178-189.

Van Bressem, M.-F.E., J.A. Raga, G. Di Guardo, P.D. Jepson, P.J. Duignan, U. Siebert, T. Barrett, M.C. de Oliveira Santos, I.B. Moreno, S. Siciliano, and A. Aguilar. 2009. Emerging infectious diseases in cetaceans worldwide and the possible role of environmental stressors. *Diseases of Aquatic Organisms* 86(2):143-157.

Van Bressem, M.-F.E., P.J. Duignan, A. Banyard, M. Barbieri, K.M. Colegrove, S. De Guise, G. Di Guardo, A. Dobson, M. Domingo, D. Fauquier, and A. Fernandez. 2014. Cetacean morbillivirus: Current knowledge and future directions. *Viruses* 6(12):5145-5181.

Van Bressem, M.-F.E., L. Flach, J.C. Reyes, M. Echegaray, M. Santos, F. Viddi, F. Félix, L. Lodi, and K. Van Waerebeek. 2015. Epidemiological characteristics of skin disorders in cetaceans from South American waters. *Latin American Journal of Aquatic Mammals* 10(1):20-32.

Van den Berg, M., L.S. Birnbaum, M. Denison, M. De Vito, W. Farland, M. Feeley, H. Fiedler, H. Hakansson, A. Hanberg, L. Haws, and M. Rose. 2006. The 2005 World Health Organization reevaluation of human and mammalian toxic equivalency factors for dioxins and dioxin-like compounds. *Toxicological Sciences* 93(2):223-241.

Van der Graaf, A.J., M.A. Ainslie, M. André, K. Brensing, J. Dalen, R.P.A. Dekeling, S. Robinson, M.L. Tasker, F. Thomsen, and S. Werner. 2012. European Marine Strategy Framework Directive–Good Environmental Status (MSFD GES): Report of the Technical Subgroup on Underwater Noise and Other Forms of Energy. Available at http://ec.europa.eu/environment/marine/pdf/MSFD_reportTSG_Noise.pdf.

Van der Zande, A.N., W.J. ter Keurs, and W.J. Weijden. 1980. The impact of roads on the densities of four bird species in an open field habitat—evidence of a long-distance effect. *Biological Conservation* 18:299-321.

Van Dolah, F.M. 2005. Effects of harmful algal blooms. Pp. 85-99 in *Marine Mammal Research, Conservation Beyond Crisis*, J. Reynolds, W. Perrin, R. Reeves, S. Montgomery, and T. Ragen, eds. Baltimore, MD: The Johns Hopkins University Press.

Van Dolah, F.M., M.G. Neely, L.E. McGeorge, B.C. Balmer, G.M. Ylitalo, E.S. Zolman, T. Speakman, C. Sinclair, N.M. Kellar, P.E. Rosel, K.D. Mullin, and L.H. Schwacke. 2015. Seasonal variation in the skin transcriptome of common bottlenose dolphins (*Tursiops truncatus*) from the Northern Gulf of Mexico. *PLoS ONE* 10(6):e0130934.

Van Loveren, H., P.S. Ross, A.D. Osterhaus, and J.G. Vos. 2000. Contaminant-induced immunosuppression and mass mortalities among harbor seals. *Toxicology Letters* 112-113:319-324.

Van Neer, A., L.F. Jensen, and U. Siebert. 2014. Grey seal (*Halichoerus grypus*) predation on harbour seals (*Phoca vitulina*) on the island of Helgoland, Germany. *Journal of Sea Research* 97:1-4.

Vanderlaan, A.S.M., and C.T. Taggart. 2007. Vessel collisions with whales: The probability of lethal injury based on vessel speed. *Marine Mammal Science* 23:144-156.

VanWormer, E., P.A. Conrad, M.A. Miller, A.C. Melli, T.E. Carpenter, and J.A.K. Mazet. 2013. *Toxoplasma gondii*, source to sea: Higher contribution of domestic felids to terrestrial parasite loading despite lower infection prevalence. *EcoHealth* 10(3):277-289.

VanWormer, E., M.A. Miller, P.A. Conrad, M.E. Grigg, D. Rejmanek, T.E. Carpenter, and J.A.K. Mazet. 2014. Using molecular epidemiology to track *Toxoplasma gondii* from terrestrial carnivores to marine hosts: Implications for public health and conservation. *PLoS Neglected Tropical Diseases* 8(5):e2852.

Vázquez-Medina, J.P., T. Zenteno-Savín, R. Elsner, and R.M. Ortiz. 2012. Coping with physiological oxidative stress: A review of antioxidant strategies in seals. *Journal of Comparative Physiology B* 182:741-750.

Vecchione, A., M.M. Peden-Adams, T.A. Romano, and P.A. Fair. 2008. Recent cytokine findings and implications toward health assessment of the bottlenose dolphin (*Tursiops truncatus*). *Aquatic Mammals* 34(1):93-101.

Vélez-Espino, A., J.K.B. Ford, H.A. Araujo, G. Ellis, C.K. Parken, and R. Sharma. 2015. Relative importance of chinook salmon abundance on resident killer whale population growth and viability. *Aquatic Conservation: Marine and Freshwater Ecosystems* 25(6):756-780.

Venn-Watson, S., C.R. Smith, E.D. Jensen, and T. Rowles. 2013. Assessing the potential health impacts of the 2003 and 2007 firestorms on bottlenose dolphins (*Tursiops trucatus*) in San Diego Bay. *Inhalation Toxicology* 25:481-491.

Venn-Watson, S., K.M. Colegrove, J. Litz, M. Kinsel, K. Terio, J. Saliki, S. Fire, R. Carmichael, C. Chevis, W. Hatchett, J. Pitchford, M. Tumlin, C. Field, S. Smith, R. Ewing, D. Fauquier, G. Lovewell, H. Whitehead, D. Rotstein, W. McFee, E. Fougeres, and T. Rowles. 2015. Adrenal gland and lung lesions in Gulf of Mexico common bottlenose dolphins (*Tursiops truncatus*) found dead following the *Deepwater Horizon* oil spill. *PLoS ONE* 10:e0126538.

Verborgh, P., R. De Stephanis, S. Pérez, Y. Jaget, C. Barbraud, and C. Guinet. 2009. Survival rate, abundance, and residency of long-finned pilot whales in the Strait of Gibraltar. *Marine Mammal Science* 25(3):523-536.

Vert-pre, K.A., R.O. Amoroso, O.P. Jensen, and R. Hilborn. 2013. Frequency and intensity of productivity regime shifts in marine fish stocks. *Proceedings of the National Academy of Sciences of the United States of America* 110(5):1779-1784.

Villegas-Amtmann, S., L.K. Schwarz, J.L. Sumich, and D.P. Costa. 2015. A bioenergetics model to evaluate demographic consequences of disturbance in marine mammals applied to gray whales. *Ecosphere* 6(10):183.

Wade, P.R., V.N. Burkanov, M.E. Dahlheim, N.A. Friday, L.W. Fritz, T.R. Loughlin, S.A. Mizroch, M.M. Muto, and D.W. Rice. 2007. Killer whales and marine mammal trends in the North Pacific—A re-examination of evidence for sequential megafauna collapse and the prey-switching hypothesis. *Marine Mammal Science* 23(4):766-802.

Wade, P.R., J.M. Ver Hoef, and D.P. Demaster. 2009. Mammal-eating killer whales and their prey—trend data for pinnipeds and sea otters in the North Pacific Ocean do not support the sequential megafaunal collapse hypothesis. *Marine Mammal Science* 25:737-747.

Wagemann, R., S. Innes, and P.R. Richard. 1996. Overview and regional and temporal differences of heavy metals in Arctic whales and ringed seals in the Canadian Arctic. *Science of the Total Environment* 186(1-2):41-46. doi: 10.1016/0048-9697(96)05085-1.

Wale, M.A., S.D. Simpson, and A.N. Radford. 2013. Noise negatively affects foraging and antipredator behaviour in shore crabs. *Animal Behaviour* 86:111-118.

Walters, C.J. 1986. *Adaptive Management of Renewable Resources*. New York: MacMillan.

Ward, E.J., E.E. Holmes, and K.C. Balcomb. 2009. Quantifying the effects of prey abundance on killer whale reproduction. *Journal of Applied Ecology* 46:632-640.

Ward, W.D., E.M. Cushing, and E.M. Burns. 1976. Effective quiet and moderate TTS: Implications for noise exposure standards. *Journal of the Acoustical Society of America* 59(1):160-165.

Ware, H.E., C.J. McClure, J.D. Carlisle, and J.R. Barber. 2015. A phantom road experiment reveals traffic noise is an invisible source of habitat degradation. *Proceedings of the National Academy of Sciences of the United States of America* 112:12105-12109.

Watanabe, Y., E.A. Baranov, K. Sato, Y. Naito, and N. Miyazaki. 2006. Body density affects stroke patterns in Baikal seals. *Journal of Experimental Biology* 209:3269-3280.

Watkins, W.A., P. Tyack, K.E. Moore and J.E. Bird. 1987. The 20-Hz signals of finback whales (*Balaenoptera physalus*). *Journal of the Acoustical Society of America* 82:1901-1912.

Weihe, P., P. Grandjean, F. Debes, and R. White. 1996. Health implications for Faroe Islanders of heavy metals and PCBs from pilot whales. *Science of the Total Environment* 186(1):141-148.

Weinrich, M., and C. Corbelli. 2009. Does whale watching in Southern New England impact humpback whale (*Megaptera novaeangliae*) calf production or calf survival? *Biological Conservation* 142:2931-2940.

Wells, R.S., H.L. Rhinehart, L.J. Hansen, J.C. Sweeney, F.I. Townsend, R. Stone, D.R. Casper, M.D. Scott, A.A. Hohn, and T.K. Rowles. 2004. Bottlenose dolphins as marine ecosystem sentinels: Developing a health monitoring system. *EcoHealth* 1(3):246-254.

Wells, R.S., J.B. Allen, S. Hofmann, K. Bassos-Hull, D.A. Fauquier, N.B. Barros, R.E. DeLynn, G. Sutton, V. Socha, and M. Scott. 2008. Consequences of injuries on survival and reproduction of common bottlenose dolphins (*Tursiops truncatus*) along the west coast of Florida. *Marine Mammal Science* 24(4):774-794.

Westgate, M.J., G.E. Likens, and D.B. Lindenmayer. 2013. Adaptive management of biological systems: A review. *Biological Conservation* 158:129-139.

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., and I.D. Jonsen. 2013. Inferring animal densities from tracking data using Markov chains. *PLoS ONE* 8:e60901.

Wild, C.P. 2005. Complementing the genome with an “exposome”: The outstanding challenge of environmental exposure measurement in molecular epidemiology. *Cancer Epidemiology, Biomarkers & Prevention* 14(8):1847-1850.

Wiley, D.N., M.A. Thompson, R.M. Pace, and J. Levenson. 2011. Modeling speed restrictions to mitigate lethal collisions between ships and whales in the Stellwagen Bank National Marine Sanctuary, USA. *Biological Conservation* 144:2377-2381.

Williams, B.K. 2011a. Adaptive management of natural resources: Framework and issues. *Journal of Environmental Management* 92:1346-1353.

Williams, B.K. 2011b. Passive and active adaptive management: Approaches and an example. *Journal of Environmental Management* 92(5):1371-1378.

Williams, B.K., J.D. Nichols, and M.J. Conroy. 2001. *Analysis and Management of Animal Populations*. San Diego, CA: Academic Press.

Williams, R., S. Gero, L. Bejder, J. Calambokidis, S.D. Kraus, D. Lusseau, A.J. Read, and J. Robbins. 2011. Underestimating the damage: Interpreting cetacean carcass recoveries in the context of the Deepwater Horizon/BP incident. *Conservation Letters* 4:228-233.

Williams, R., G.A. Vikingsson, A. Gislason, C. Lockyer, L. New, L. Thomas, and P.S. Hammond. 2013. Evidence for density-dependent changes in body condition and pregnancy rate of North Atlantic fin whales over four decades of varying environmental conditions. *ICES Journal of Marine Science* 70(6):273-1280.

Williams, R., E. Ashe, L. Bright, M. Jasny, and L. Nowlan. 2014. Viewpoint: Marine mammals and ocean noise: Future directions and information needs with respect to science, policy and law in Canada. *Marine Pollution Bulletin* 86:29-38.

Williams, R., C. Erbe, E. Ashe, and C.W. Clark. 2015. Quiet(er) marine protected areas. *Marine Pollution Bulletin* 100:154-161.

Wilson, B., H. Arnold, G. Bearzi, C.M. Fortuna, R. Gaspar, S. Ingram, C. Liret, S. Probanic, A.J. Read, V. Ridoux, K. Schneider, K.W. Urian, R.S. Wells, C. Wood, P.M. Thompson, and P.S. Hammond. 1999. Epidermal diseases in bottlenose dolphins: Impacts of natural and anthropogenic factors. *Proceedings of the Royal Society B: Biological Sciences* 266:1077-1083.

Wilson, M., H.B. Shack, P.T. Madsen, A. Surlykke, and M. Wahlberg. 2011. Directional escape behavior in allis shad (*Alosa alosa*) exposed to ultrasonic clicks mimicking an approaching toothed whale. *Journal of Experimental Biology* 214:22-29. doi: 10.1242/jeb.043323.

Wilson, R.P., C.R. White, F. Quintana, L.G. Halsey, N. Liebsch, G.R. Martin, and P.J. Butler. 2006. Moving towards acceleration for estimates of activity-specific metabolic rate in free-living animals: The case of the cormorant. *Journal of Animal Ecology* 75(5):1081-1090.

Wilson, R.P., E.L.C. Shepard, and N. Liebsch. 2008. Prying into the intimate details of animal lives: Use of a daily diary on animals. *Endangered Species Research* 4:123-137.

Wiseman, S.B., Y. Wan, H. Chang, X. Zhang, M. Hecker, P.D. Jones, and J.P. Giesy. 2011. Polybrominated diphenyl ethers and their hydroxylated/methoxylated analogs: Environmental sources, metabolic relationships, and relative toxicities. *Marine Pollution Bulletin* 63(2011):179-188.

Wobeser, G.A. 1981. *Diseases of Wild Waterfowl*. New York: Plenum Press.

Wolkers, H., C. Lydersen, and K.M. Kovacs. 2004. Accumulation and lactational transfer of PCBs and pesticides in harbor seals (*Phoca vitulina*) from Svalbard, Norway. *Science of the Total Environment* 319:137-146.

Wright, A.J., and L.A. Kyhn. 2015. Practical management of cumulative anthropogenic impacts with working marine examples. *Conservation Biology* 29:333-340.

Yang, W.C., V.F. Pang, C.R. Jeng, L.S. Chou, and L.L. Chueh. 2006. Morbilliviral infection in a pygmy sperm whale (*Kogia breviceps*) from Taiwanese waters. *Veterinary Microbiology* 116(1):69-76.

Yeates, L.C., and D.S. Houser. 2008. Thermal tolerance in bottlenose dolphins (*Tursiops truncatus*). *Journal of Experimental Biology* 211(20):3249-3257.

Yodzis, P. 1988. The indeterminacy of ecological interactions as perceived through perturbation experiments. *Ecology* 69:508-515.

Yordy, J.E., R.S. Wells, B.C. Balmer, L.H. Schwacke, T.K. Rowles, and J.R. Kucklick. 2010. Life history as a source of variation for persistent organic pollutant (POP) patterns in a community of common bottlenose dolphins (*Tursiops truncatus*) resident to Sarasota Bay, FL. *Science of the Total Environment* 408:2163-2172.

Zanette, L., A. White, M. Allen, and M. Clinchy. Perceived predation risk reduces the number of offspring songbirds produce per year. 2011. *Science* 334:1398-1401.

Zoeller, R.T., A.L.S. Dowling, C.T.A. Herzig, E.A. Iannocone, K.J. Gauger, and R. Bansal. 2002. Thyroid hormone, brain development, and the environment. *Environmental Health Perspectives* 110(Suppl. 3):355-361.

Appendix A

Workshop Agenda

Workshop for the Committee on the Assessment of the Cumulative Effects of
Anthropogenic Stressors on Marine Mammals

Arnold and Mabel Beckman Center of the National Academies of Sciences, Engineering, and Medicine
100 Academy Dr, Irvine, CA 92617 • (949) 721-2200
October 1-2, 2015

OPEN SESSION AGENDA

Thursday, October 1

8:00 a.m.	*Breakfast for committee members and speakers*
8:30 a.m.	**Welcome and Introductions, *Peter Tyack***
9:00 a.m.	**Cumulative Effects – Approaches from Global Health and Ecotoxicology** ***Moderator: Lori Schwacke*** • Jonna Mazet, University of California, Davis
10:15 a.m.	*Break*
10:30 a.m.	**Indirect Effects on Marine Mammals from Predators, Prey, and Competition** ***Moderator: Clint Francis*** • Tim Essington, University of Washington • Jesse Barber, Boise State University
12:30 p.m.	*Lunch for all attendees*
1:30 p.m.	**Application of Biosensors to Marine Mammals** ***Moderator: Dan Crocker*** • Shekhar Bhansali, Florida International University • Kim Anderson, Oregon State University
3:30 p.m.	*Break*
3:45- 5:45 p.m.	**Plenary Discussion of Day 1 Topics**

Friday, October 2

8:00 a.m.	*Breakfast for committee members and speakers*
8:30 a.m.	**Recap of Day 1 and Introductions, *Peter Tyack***
9:00 a.m.	**Cumulative Effects – Review of Ecological Studies** ***Moderator: Jim Estes*** • Carrie Kappel, University of California, Santa Barbara • Sara Maxwell, Old Dominion University
11:00 a.m.	**Long-Term Monitoring and Adaptive Management** ***Moderator: John Harwood*** • Steve Beissinger**,** University of California, Berkeley • Mitch Eaton, U.S. Geological Survey
1:00 p.m.	*Lunch for all attendees*
2:00 p.m.	**Plenary Discussion of Day 2 Topics**
3:30 p.m.	*Adjourn Workshop*

Appendix B

Relevant Laws and Regulations

RELEVANT U.S. LEGISLATION

In the United States, there are many statutes and regulations that are important to the well-being of marine mammals and their habitats. This appendix highlights three primary statutes that provide the general legal framework for addressing impacts to marine mammals. They are the National Environmental Policy Act (NEPA), the Endangered Species Act (ESA), and the Marine Mammal Protection Act (MMPA). The way, and extent to which, these statutes address cumulative impacts or effects varies. In addition, this appendix identifies and briefly discusses four other federal statutes that require or authorize spatial planning and conservation and management measures important to marine mammals and the protection of their habitats. These are the Ports and Waterways Safety Act, the National Marine Sanctuaries Act, the Outer Continental Shelf Lands Act, and the Magnuson-Stevens Fishery Conservation and Management Act. International laws are also discussed briefly. This appendix is not intended to be a comprehensive discussion of all laws and regulations that impact marine mammals, but rather to provide further policy context for the consideration that agencies must give to cumulative impacts of stressors and other noise on marine mammals.

National Environmental Policy Act (NEPA)

Congress enacted NEPA in December 1969, and President Nixon then signed the statute into law on January 1, 1970.[1] The stated purpose of NEPA was "[t]o declare a national policy which will encourage productive and enjoyable harmony between man and his environment; to promote efforts which will prevent or eliminate damage to the environment and biosphere and stimulate the health and welfare of man; to enrich the understanding of the ecological systems and natural resources important to the Nation; and to establish a Council on Environmental Quality."[2] "NEPA itself does not mandate particular results" in order to accomplish these ends.[3] Rather, NEPA imposes only procedural requirements on federal agencies with a particular focus on requiring agencies to undertake analyses of the environmental impact of their proposals and actions.[4] The Council on Environmental Quality (CEQ) was established in the Executive Office of the President and is the primary agency responsible for ensuring that other federal agencies meet the requirements set forth by NEPA. The CEQ regulations promulgated under this act require consideration of cumulative impacts[5] and define cumulative impact as noted above.[6]

At the heart of NEPA is a requirement that federal agencies "include in every recommendation or report on proposals for legislation and other major Federal actions significantly affecting the quality of the human environment, a detailed statement by the responsible official on—(i) the environmental impact of the proposed action, (ii) any adverse environmental effects which cannot be avoided should the proposal be implemented, (iii) alternatives to the proposed action, (iv) the relationship between local short-term uses of man's environment and the maintenance and enhancement of long-term productivity, and (v) any irreversible and irretrievable commitments of resources which would be involved

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[1] (Pub. L. 91-190, 42 U.S.C. 4321-4347, January 1, 1970, as amended by Pub. L. 94-52, July 3, 1975, Pub. L. 94-83, August 9, 1975, and Pub. L. 97-258, § 4(b), Sept. 13, 1982).

[2] 42 U.S.C. § 4321.

[3] *Robertson v. Methow Valley Citizens Council*, 490 U.S. 332, 350, 109 S.Ct. 1835, 104 L.Ed.2d 351 (1989).

[4] See id., at 349-350, 109 S.Ct. 1835.

[5] 40 C.F.R. § 1508.25.

[6] 40 C.F.R. § 1508.7.

in the proposed action should it be implemented."[7] CEQ regulations clarify that "major Federal actions" may include "projects and programs entirely or partially financed, assisted, conducted, regulated, or approved by Federal agencies; new or revised agency rules, regulations, plans, policies, or procedures; and legislative proposals." Significance, according to the regulations, is determined based on the context and intensity of the action, and the regulations require the agency to consider "[w]hether the action is related to other actions with individually insignificant but cumulatively significant impacts."[8] "Significance exists if it is reasonable to anticipate a cumulatively significant impact on the environment. Significance cannot be voided by terming an action temporary or by breaking it down into small component parts."[9]

The detailed statement called for in 42 U.S.C. § 4332(2)(C) is termed an Environmental Impact Statement (EIS). The CEQ regulations allow an agency to prepare a more limited document, an Environmental Assessment (EA), if the agency's proposed action neither is categorically excluded from the requirement to produce an EIS nor would clearly require the production of an EIS.[10] The EA is to be a "concise public document" that "[b]riefly provide[s] sufficient evidence and analysis for determining whether to prepare an [EIS]."[11] If, pursuant to the EA, an agency determines that an EIS is not required under applicable CEQ regulations, it must issue a "finding of no significant impact," which briefly presents the reasons why the proposed agency action will not have a significant impact on the human environment.[12] EISs and EAs developed in accordance with NEPA and the corresponding CEQ regulations are required to consider direct, indirect, and cumulative impacts.[13] It is worth noting that, according to CEQ regulations, NEPA does not require an EA or EIS for those actions that are categorically excluded, meaning that the responsible agency has determined that the action falls within a category of actions that do "not individually or cumulatively have a significant effect on the quality of the environment."[14]

Courts have further considered how well federal agencies implement NEPA and how cumulative impacts should be addressed in environmental documents developed in accordance with NEPA. The Supreme Court has stated that, in light of agencies' broad discretion, the role of the courts with regard to NEPA is to ensure that the agencies take a "hard look" at the environmental consequences of their proposed major actions and alternatives.[15] Multiple circuit courts have weighed in on what constitutes a "hard look."[16]

The Ninth Circuit has held that the analysis of cumulative impacts must "be more than perfunctory; it must provide a useful analysis of the cumulative impacts of past, present and future projects."[17] Courts have also signaled that the analysis must involve more than "generalized, conclusory assertions from agency experts." [18] Instead, the Ninth Circuit requires that agencies provide supporting data in a manner that can be understood by members of the public.[19]

Litigants have also used the NEPA "hard look" mandate to clarify federal agencies' legal duties to consider the habitat impacts of federally licensed extractive activities. For example, in *American Oceans Campaign v. Daley*, 183 F. Supp.2d 1 (D.D.C. 2000) the court found that the environmental assessments for current fishery management plans lacked sufficient analysis of alternative habitat protection measures. Similar rulings have resulted from NEPA litigation over oil and gas development on the continental shelf or the construction of oil and liquefied natural gas terminals. In this litigation, courts may be asked whether the federal agency had a responsibility to find or fund additional research on reasonably foreseeable environmental impacts of its preferred alternative. Courts often find that the duty depends on severity of the potential impacts or the ready availability of simulation studies or models.[20] When scientific experts express conflicting views regarding the scope and significance of potential impacts, the courts have interpreted NEPA as affording the agency with discretion to rely on the reasonable opinions of its own qualified experts.[21]

Access to courts for judicial opinions such as these is most available for species listed as either endangered or threatened because the ESA has a citizen suit provision. For non-ESA-protected species, agency decisions based on insufficient or conflicting scientific evidence may be challenged as a violation of the Administrative Procedures Act.

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[7] 42 U.S.C. § 4332(2)(C).

[8] 40 C.F.R. § 1508.27.

[9] 40 C.F.R. § 1508.27(b)(7).

[10] 40 C.F.R. §§ 1501.4(a)-(b).

[11] 40 C.F.R. § 1508.9(a).

[12] 40 C.F.R. §§ 1501.4(e), 1508.13.

[13] "Effects" and "impacts" are considered synonymous according to the CEQ regulations.

[14] 40 C.F.R. § 1508.4.

[15] *Kleppe, Secretary of the Interior, et al. v. Sierra Club et al.* citing *NRDC v. Morton.*

[16] *Britt v. U.S. Army Corps of Eng'rs*, 769 F.2d 84, 90 (2d Cir. 1985); *Northwest Indian Cemetery Protective Ass'n v. Peterson*, 764 F.2d 581, 588 (9th Cir. 1985), rev'd on other grounds, 485 U.S. 439 (1988); *Maryland Wildlife Fed'n v. Dole*, 747 F.2d 229 (4th Cir. 1984) (reasonable alternatives must be considered but not every alternative conceivable to the mind of man).

[17] *Oregon Natural Resources Council Fund v. Brong citing Klamath-Siskiyou Wildlands Center v. BLM* (2004) citing *Ocean Advocates* 361 F.3d 1108 (2003) quoting *Kern*, 284 F.3d at 1075 (quoting *Muckleshoot Indian Tribe v. United States Forest Serv.*, 177 F.3d 800, 810 (9th Cir. 1999) for the "useful analysis...").

Klamath-Siskiyou also quotes *Neighbors of Cuddy Mountain v. United States Forest Serv.*, 137 F.3d 1372, 1379-80 (9th Cir. 1998).

[18] *Or. Natural Res. Council Fund v. Goodman*, 505 F.3d 884, 893 (9th Cir. 2007).

[19] *Or. Natural Res. Council Fund v. Goodman*, 505 F.3d 884, 893 (9th Cir. 2007).

[20] *Roosevelt Campobello International Park Comm'n v. US EPA*, 684 F.2d 1041 (1st Cir. 1982).

[21] *NRDC v. Evans*, 232 F.Supp.2d 1003 (N.D. Cal. 2002).

Under this law, courts will defer to agencies' expert judgments in interpreting and applying key statutory terms and standards, such as "harassment" or "unmitigable adverse impact." Judicial review is deferential to agency expertise but will entail an examination of information that was presented to the agency prior to its decision. Under this deferential standard of review, the agency's decision will be upheld unless the record shows the agency considered factors, including political pressures, other than those which Congress directed it to consider.[22]

Endangered Species Act (ESA)

The Endangered Species Act (ESA) was passed by the U.S. Congress and signed into law in 1973.[23] The ESA calls for the listing and protection of endangered and threatened species, and the designation of critical habitat for endangered species. According to the ESA, an endangered species is a species that "is in danger of extinction throughout all or a significant portion of its range."[24] The ESA defines threatened species as those species that are "likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range."[25]

The U.S. Fish and Wildlife Service (FWS) is the lead agency for implementing the ESA for most species. However, most threatened or endangered anadromous fish and marine species are managed by the National Marine Fisheries Service (NMFS) with the exception of walrus, polar bear, sea otters, and sirenians, which are managed by FWS under both the ESA and the MMPA. For listing of shared species, for example, sea turtles, or for policies applicable to all species, the two agencies often issue joint listings or joint guidance, for example, on designation of critical habitat or on interagency consultation.

The ESA protects endangered species from both private and public actions. Section 9 of the ESA states that no one, public or private, may "take" any endangered species.[26] The ESA broadly defines "take" to mean "harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect."[27] Section 7 of the ESA also directs federal agencies to carry out programs for the conservation of threatened and endangered species. It further requires federal agencies to ensure that their actions (i.e., all actions authorized, funded, or carried out by the agency) are not likely to jeopardize the existence of a listed species or adversely modify the critical habitat of a listed species. As part of these assurances, Section 7 also requires agencies to consult with FWS or NMFS (Steiger, 1994) regarding any activities that may affect listed species.[28] "Procedurally, before initiating any action in an area that contains threatened or endangered species, federal agencies must consult with the FWS (for land based species and selected marine mammals) or NMFS (for all other marine species) to determine the likely effects of any proposed action on species and their critical habitat."[29]

Although the text of the ESA does not directly address cumulative impacts or effects, the implementing agencies (FWS and NMFS) and the courts have interpreted Section 7 as to require consideration of cumulative effects during the consultation process. The regulations promulgated under the ESA define "cumulative effects" as "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."[30] Guidance produced by the FWS and NMFS regarding Section 7 consultations specifically states that this more narrow definition should not be conflated with the broader definition of "cumulative impacts" used in NEPA and pertains only to ESA Section 7 analyses. [31] The Ninth Circuit in *Conservation Congress v. USFS* has reiterated this point also.

After listing, two other processes under Section 4 of the ESA are important. These are the requirement to prepare and update recovery plans for listed species and the obligation to designate critical habitat. The latter requirement is central to ensuring that under Section 7 federal agencies do not take or approve actions that adversely modify critical habitat or its key components. Failure to do so can be a basis for litigation, which may result in an injunction until further analysis is done. Recent developments suggest the critical habitat provisions are increasingly important in protecting the marine acoustic environment and in incorporating the latest scientific findings and impact assessment methods. In 2015, NMFS made a legal determination that newly available scientific information warranted proceeding with a petition to revise the critical habitat designation for the Southern Resident killer whale (*Orcinus orca*) Distinct Population Segment. The revision would expand the designation to include essential foraging and wintering areas along the

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[22] *Earth Island Institute v. Hogarth*, 494 F.3d 757 (9th Cir. 2007).

[23] This law repealed the earlier legislation aimed at protecting "selected species" and habitats, including the Endangered Species Preservation Act of 1966 and the Endangered Species Conservation Act of 1969. The ESA has since been amended in 1978, 1979, and 1982.

[24] 16 U.S.C. § 1532 (6).

[25] 16 U.S.C. § 1532 (20).

[26] 16 U.S.C. § 1538 (a)(1).

[27] 16 U.S.C. § 1532 (19).

[28] 16 U.S.C. § 1536 (a). The agency first determines whether their proposed action "may affect" a listed species or its habitat. If the agency determines it may, then formal consultation with either FWS or NOAA Fisheries is automatically required. If the agency determines that the action is not likely to affect a listed species or its habitat and the consulting agency agrees with this assessment, then further formal consultation is not necessary. If, however, the consulting agency does not agree with the assessment, then a formal consultation is required. *Conservation Congress v. USFS*, 720 F.3d 1048 (9th Cir. 2013).

[29] *Conservation Congress v. USFS*, 720 F.3d 1048 (9th Cir. 2013) citing *Natural Res. Defense Council v. Houston*, 146 F.3d 1118, 1125 (9th Cir. 1998) and *Forest Guardians v. Johanns*, 450 F.3d 455, 457 n.1.

[30] 50 C.F.R. § 1508.7.

[31] See https://www.fws.gov/ENDANGERED/esa-library/pdf/esa_section7_handbook.pdf.

West Coast and adopt as a "primary constituent element" of that habitat protective underwater noise levels.[32] In the 2008 recovery plan for the Southern Resident killer whale, the National Oceanic and Atmospheric Administration (NOAA) did not include sound levels as a primary constituent element (PCE),[33] likely because of limitations of available information (Williams et al., 2014).

Marine Mammal Protection Act (MMPA)

The MMPA was passed and signed into law in 1972 at a time when environmental issues resonated particularly strongly with the public. By 1971, 42 marine mammal protection and conservation bills had been filed in Congress (Ray and Potter, 2011). The death of hundreds of thousands of pelagic dolphins annually in the tuna fishing industry, where purse seine nets were set on dolphin schools that were associated with tuna below; the apparent impotence of the International Whaling Commission to prevent the continued decline of great whale stocks; and the harvesting of pup and juvenile harp and northern fur seals by clubbing were primary drivers of the public demand for congressional action. The MMPA charted new territory in environmental legislation by focusing on the ecosystem and requiring that marine mammals be maintained at the optimal sustainable population at which they are significant functioning elements of their ecosystem. With few exceptions, the MMPA prohibited the taking or importing any marine mammal or marine mammal product[34] where a "take" was defined as "harass, hunt, capture, or kill, or attempt to harass, hunt, capture, or kill.[35] The rights of Alaskan Natives to take marine mammals for subsistence purposes, however, were preserved under the MMPA.[36]

The Act is enforced in the 200-mile Exclusive Economic Zone of the United States, and any person, vessel, or other conveyance subject to the jurisdiction of the United States is also prohibited from taking any marine mammal on the high seas.[37] Exemptions to these prohibitions may be made in specific cases in which the Secretary of the Interior or Commerce (depending on whether the species in question falls under FWS or NMFS jurisdiction) authorizes a permit for such activity. Permits may be acquired for scientific research; enhancing the survival or aiding in the recovery of a marine mammal stock or species; commercial and educational photography; first-time import for public display; capture of wild animal for public display; and incidental, i.e., nondirected, take.[38] An incidental take permit may be issued provided that the taking would (1) be of small numbers, (2) have no more than a "negligible impact" on those marine mammal species or stocks, and (3) not have an "unmitigable adverse impact" on the availability of the species or stock for subsistence uses.[39] Fisheries are allowed incidental take outside the normal permit process subject to take reduction plans that seek to reduce mortality and serious injury rates to a rate approaching zero.

Takes by harassment account for almost all takes for which permits are issued. The MMPA has defined two levels of harassment with a somewhat different definition when the harassment is caused by a "military readiness activity" or "a scientific research activity conducted by or on behalf of the Federal Government." Level A harassment occurs when the action "has the potential to injure a marine mammal or marine mammal stock in the wild"[40] or for military readiness "any act that injures or has the significant potential to injure a marine mammal or marine mammal stock in the wild."[41] Level B harassment occurs when the action "has the potential to disturb a marine mammal or marine mammal stock in the wild by causing disruption of behavioral patterns, including, but not limited to, migration, breathing, nursing, breeding, feeding, or sheltering."[42] Or for military readiness "any act that disturbs or is likely to disturb a marine mammal or marine mammal stock in the wild by causing disruption of natural behavioral patterns, including, but not limited to, migration, surfacing, nursing, breeding, feeding, or sheltering, to a point where such behavioral patterns are abandoned or significantly altered."[43]

In developing regulations to implement the MMPA in so far as acoustic harassment is concerned, NMFS has determined that injury equates to a permanent threshold shift (PTS), which is a loss of hearing within a particular frequency range that is not reversible. A temporary threshold shift (TTS) is one in which hearing sensitivity within a particular frequency range is reduced for a period of minutes to hours but recovers to its prior level of sensitivity. NMFS recently published acoustic thresholds for the onset of TTS and PTS (NMFS, 2016a) based on the best current available science. These guidelines have separate PTS thresholds for impulsive and nonimpulsive sounds for five categories of marine mammals: low-, mid-, and high-frequency cetaceans, phocids, and otariids.[44] For each marine mammal category

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[32] NOAA, 80 Fed. Reg. 9682-87 (Feb. 24, 2015).

[33] Primary constituent element (PCE): A physical or biological feature essential for conservation upon which a critical habitat is based. See http://www.fws.gov/nc-es/fish/glossary.pdf.

[34] 16 U.S.C. § 1372.

[35] 16 U.S.C. § 1362. See also 50 C.F.R. § 216.3.

[36] 16 U.S.C. § 1371(b).

[37] 16 U.S.C. § 1372.

[38] 16 U.S.C. § 1374.

[39] 50 C.F.R. § 216.102; see also http://www.nmfs.noaa.gov/pr/permits/incidental.

[40] 16 U.S.C. § 1362 Sec. 3(18)(A)(i).

[41] 16 U.S.C. § 1362 Sec. 3(18)(B)(i).

[42] 16 U.S.C. § 1362 Sec. 3(18)(A)(ii).

[43] 16 U.S.C. § 1362 Sec. 3(18)(B)(i).

[44] Low-frequency cetaceans are all the baleen whales. High-frequency cetaceans are all porpoises, river dolphins, pygmy and dwarf sperm whales, all dolphins in the genus *Cephalorhynchus*, and two species of *Lagenorhynchus, L. australis* and *L. cruciger*. Mid-frequency cetaceans are all the odontocetes not in the high-frequency group.

two thresholds are given for impulsive sounds: one for peak sound pressure level (SPL_{pk}) and one for cumulative sound exposure level (SEL_{cum}) accumulated over 24 hours; and one threshold for nonimpulsive sounds: the cumulative sound exposure level (SEL_{cum}) accumulated over 24 hours. The SPL_{pk} ranges from 202 dB re 1 μPa for high-frequency cetaceans to 232 dB re 1 μPa for otariid pinnipeds in water. The SEL values for impulsive sounds range from 155 dB re 1 μPa^2-s for high-frequency cetaceans to 203 dB re 1 μPa^2-s for otariids, and the threshold values for nonimpulsive sounds range from 173 dB re 1 μPa^2-s for high-frequency cetaceans to 219 dB re 1 μPa^2-s for otariids.

NMFS has not proposed any update to their Level B behavioral harassment criteria. They remain SPL_{RMS} of 160 dB for impulsive sounds and 120 dB for nonpulse sounds.[45] Currently NMFS classifies a variety of sonar signals as impulsive for Level B criteria, although the recently released Technical Guidance (NMFS, 2016a) classifies them as nonimpulsive for Level A criteria. The Navy has adopted more conservative criteria for behavioral response thresholds for beaked whales (140 dB re 1 μPa) and for harbor porpoises (120 dB re 1 μPa) exposed to sonar (Finneran and Jenkins, 2012).

Other Important U.S. Laws

The U.S. Coast Guard has responsibility to implement the Ports and Waterways Safety Act as well as to enforce all other marine environmental laws. As the international shipping community continues to address the issue of shipping noise, this law will be the basis for implementing any resulting international standards or regulations for environmentally sensitive "Areas to Be Avoided" aproved by the International Maritime Organization (IMO). The Papahanaumokuakea Marine National Monument in Hawaii is an example of marine mammal habitat subject to such shipping regulations. Standards for ship noise are under consideration by a correspondence working group of the IMO's Marine Environmental Protection Committee in which both the Coast Guard and NOAA participate. In addition, the Coast Guard conducts ship routing and port access studies under the Ports Act; the law proved to be an important authority in reducing deadly ship strikes of endangered North Atlantic right whales through real-time, whale location reporting and reduced speed limits.

The National Marine Sanctuaries Act can also be used to designate as marine protected areas those marine mammal habitats that are currently quiet, with a minimal amount of anthropogenic noise, preserving this protective status quo as a precautionary measure (Williams et al., 2015) and to offset acoustic degradation that cannot be avoided or mitigated. If a marine sanctuary is established and its management plan identifies the in-water sound levels as sanctuary resources, federal agencies will review proposed federal activities, leases, or licenses for their potential impact on these resources. This process would protect all marine mammals that use the marine sanctuary but would be especially valuable for a species that is neither "depleted" under the MMPA nor listed under the ESA and therefore not protected by the "negligible impact" and "adversely modify" habitat provisions of those laws.

Other relevant legislation regulating the introduction of pollution stressors into the ocean are the Rivers and Harbors Act (RHA) and Clean Water Act (CWA). The RHA regulates activity affecting navigation in U.S. waters. Section 13 of the RHA, commonly named The Refuse Act, 33 U.S.C. § 407 (1976), prohibits discharge of "any refuse matter of any kind or description" into navigable waters. In a similar vein Section 404 of the CWA regulates the discharge of dredged or fill material resulting from water resource projects, infrastructure development, and mining projects in U.S. waters. Applying for a permit to discharge requires showing that steps have been taken to avoid impacts on aquatic resources.[46]

Marine resource development laws such as the Outer Continental Shelf Lands Act (OCSLA), as amended, and the Magnuson-Stevens Fishery Conservation and Management Act, as amended, have important environmental planning and permitting processes that are subject to judicial review under the Administrative Procedures Act or NEPA or both. The OCSLA process could be used to identify and exclude from leasing for offshore renewable energy development (e.g., wind farms) those tracts that are acoustically significant marine mammal habitat. In addition, anthropogenic noise can scatter prey and interact with fisheries extractions to reduce the quality of marine mammal habitat, especially in foraging areas near rookeries. NEPA analyses of fishery operations and catch levels provide an opportunity to review these potential impacts. Again, this could prove especially important for marine mammal life stages that are vulnerable to prey disruption but are not yet listed as MMPA-"depleted" or in danger of extinction and do not trigger Section 7 interagency consultation.

INTERNATIONAL SOUND REGULATIONS

Several national and international regulatory bodies have adopted regulations or guidelines for the effects of underwater sound on marine life, including marine mammals. These share the same scientific underpinning as U.S. regulations but may emphasize different effects, different taxa, and different spatial and temporal scales.

McCarthy (2007) pointed out that low-frequency sound travels so far in the ocean that some sound sources create noise that must be treated as a transboundary pollutant.

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[45] See http://www.westcoast.fisheries.noaa.gov/protected_species/marine_mammals/threshold_guidance.html.

[46] See https://www.epa.gov/sites/production/files/2015-03/documents/404_reg_authority_fact_sheet.pdf.

Gillespie (2010) and McCarthy (2007) identify the United Nations Convention on the Law of the Sea (UNCLOS) as the appropriate international body to regulate ocean noise. UNCLOS article 1(4) says "'pollution of the marine environment' means the introduction by man, directly or indirectly, of substances or energy into the marine environment, including estuaries, which results or is likely to result in such deleterious effects as harm to living resources and marine life." This definition includes acoustic energy along with other forms of energy if it harms marine life.

The International Maritime Organization is tasked with regulating pollution by vessels under the International Convention for the Prevention of Pollution by Ships (MARPOL Convention). In 2013, the Marine Environment Protection Committee of the IMO issued voluntary guidelines for the reduction of underwater noise from commercial shipping (MEPC 66/17).

The International Convention on Migratory Species (CMS, also known as the Bonn Convention) was signed by 117 countries (known as Parties to the Convention) under the auspices of the United Nations Environment Programme (UNEP). In 2008 the Parties to the CMS adopted resolution 9.19 on Adverse Anthropogenic Marine/Ocean Noise Impacts on Cetaceans and Other Biota, which urges the Parties to the Convention "to control the impact of emission of man-made noise pollution in habitat of vulnerable species and in areas where marine mammals or other endangered species may be concentrated." Several regional agreements that operate under the auspices of the Bonn convention of UNEP have also established guidelines on ocean noise for their regions. The ACCOBAMS (Agreement on the Conservation of Cetaceans in the Black Sea Mediterranean Sea and Contiguous Atlantic Area) agreement has passed a resolution on "Guidelines to address the impact of anthropogenic noise on cetaceans in the ACCOBAMS area" and the ASCOBANS (Agreement on the Conservation of Small Cetaceans in the Baltic, North East Atlantic, Irish and North Seas) has issued a report on the assessment of acoustic disturbance (Bräger et al., 2009) and passed resolutions on effects of anthropogenic noise on marine mammals. The Convention for the Protection of the Marine Environment of the North-East Atlantic (the OSPAR Convention) involves the European Union (EU) and 15 European nations in support of conservation of the northeastern Atlantic. In 2009 the OSPAR Commission reviewed the effects of underwater sound on marine life, calling for more research on this problem. There are thus many international agreements, especially within Europe, that have addressed the impacts of anthropogenic noise on marine life, including the cumulative effects of noise plus other stressors, but none of these have established regulations to control these impacts.

Explicit guidelines or regulations have been developed by international or national authorities for three intense sources of underwater sound: pile driving, seismic survey, and naval sonar. Erbe (2013) describes how some countries may prohibit seismic surveys in habitats and seasons when marine mammals are concentrated. Some countries stipulate that seismic surveys use the minimum practicable power or that construction of foundations of offshore wind turbines use methods other than pile driving in some settings. Where pile driving is used, some countries require the use of mitigation measures such as bubble curtains to reduce the sound that propagates from pile driving. Other mitigation measures required by some nations for pile driving, seismic survey, and naval sonar include visual and/or acoustic monitoring to make sure that protected animals do not enter a shutdown zone, 30 minutes of monitoring before starting transmissions to reduce the risk that animals are in the shutdown zone, and a ramp-up procedure that starts at low acoustic power and slowly increases to the full power over tens of minutes to allow animals to move away from aversive or harmful sound levels. The North Atlantic Treaty Organization (NATO) Undersea Research Centre (NURC; now called the Center for Maritime Research and Exploration) has for 50 years provided technical and scientific guidance to NATO nations on anti-submarine warfare, including the use of naval sonar. Frantzis (1998) documented an atypical mass stranding of beaked whales in the Mediterranean that coincided with a sonar trial by NURC in 1996. This evidence of adverse impact led NURC to conduct research on the effects of sonar on cetaceans and to develop Marine Mammal Risk Mitigation Rules and Procedures (NURC, 2006) for their own sonar trials that include similar mitigation measures to those listed above. However, each nation maintains its own procedures for operating naval sonar, including risk mitigation.

The EU has developed a very different strategy for protecting the marine environment and maintaining Good Environmental Status. In 2008, the EU adopted a Marine Strategy Framework Directive (MSFD) to protect the marine environment across the EU. The goal of the MSFD is to achieve Good Environmental Status (GES) by 2020 (European Union, 2008). The goals of the MSFD were to be incorporated into national legislation by July 15, 2010. Good Environmental Status represents a resilient ecosystem in which biodiversity is preserved and human effects, including pollution and noise, do not exceed that which is compatible with a functioning marine ecosystem. The Directive identifies 11 qualitative descriptors that assist member states in identifying what a GES ecosystem should look like. Qualitative Descriptor 11 deals with energy and noise. Technical Subgroups prepared implementation guidelines in 2010 and 2012. The 2010 guidelines (Tasker et al., 2010) identified three underwater noise indicators:

1. The proportion of days within a calendar year, over areas of 15°N × 15°E/W in which anthropogenic sound sources exceed either of two levels, 183 dB re 1µPa2-s (i.e., measured as SEL) or 224 dB re 1µPa peak (i.e., measured as peak sound pressure

level) when extrapolated to 1 meter, measured over the frequency band 10 Hz to 10 kHz.

2. The total number of vessels that are equipped with sonar systems generating sonar pulses below 200 kHz should decrease by at least x% per year starting in [2012]. (The x% was to be set by Member States.)
3. The ambient noise level measured by a statistical representative sets of observation stations in Regional Seas where noise within the 1/3 octave bands 63 and 125 Hz (center frequency) should not exceed the baseline values of year [2012] or 100 dB (re 1 µPa RMS; average noise level in these octave bands over 1 year).

The 2012 guidelines (Van der Graaf et al., 2012) defined an impulsive sound as “a sound for which the effective time duration of individual sound pulses is less than ten seconds and whose repetition time exceeds four times this effective time duration.” However, they abandoned the criteria established in 2010 for impulsive sounds and simply noted that “At the moment it is difficult to provide a more specific description of GES beyond the text of the Directive, due to insufficient knowledge on the cumulative impacts of impulsive sound on the marine environment.” In terms of ambient noise, they concluded “At the moment it is impossible to define those elevations of ambient noise from anthropogenic sources that would cause the marine environment to not be at GES. This is mainly due to a lack of knowledge on the impacts of elevated ambient noise on the marine environment. The TSG cannot therefore advise on a level of ambient noise that could be set as a target for this indicator.”

Many of the national regulations and guidelines to protect marine mammals from the effects of underwater sound emphasize short time scales (tens of minutes) and small spatial scales (hundreds of meters) around intense sound sources. However, the EU MSFD takes a much broader (regional sea) and longer (yearly) view of indicators for cumulative effects of noise to maintain good environmental status. This broader scale may be more appropriate for addressing cumulative effects of noise over time, but this approach is vulnerable to gaps in current scientific ability to predict cumulative effects of different combinations of stressors. There is currently little scientific basis for the indicators of GES for noise, but these kinds of large-scale indicators may prove to be important methods for monitoring stressors in a way that can be linked to effects.

Appendix C

Committee and Staff Biographies

COMMITTEE

Dr. Peter L. Tyack (*Chair*) is a professor of marine mammal biology at the University of St. Andrews in Scotland and a senior scientist emeritus at the Woods Hole Oceanographic Institution. His research interests include social behavior and vocalizations of cetaceans, including vocal learning and mimicry in their natural communication systems and their responses to human noise. Dr. Tyack served on the National Academies of Sciences, Engineering, and Medicine's Ocean Studies Board from 2008 to 2013 and was a member of three previous National Research Council studies on marine mammals and sound, including the Committee on Describing Biologically Significant Marine Mammal Behavior, the Committee to Review Results of the Acoustic Thermometry of the Ocean Climate's Marine Mammal Research Program, and the Committee on Low-Frequency Sound and Marine Mammals. He has also served on the Office of Naval Research's Population Consequences of Disturbance Working Group. Dr. Tyack received his Ph.D. in animal behavior from Rockefeller University.

Dr. Helen Bailey is a research assistant professor at the Chesapeake Biological Laboratory, University of Maryland Center for Environmental Science. She has published more than 30 journal articles specializing in marine mammals and sea turtles. She has studied habitat use of whales and dolphins, underwater sound levels and environmental impacts of offshore wind turbines on marine mammals, and migration pathways and hot spots of marine predators at the National Oceanic and Atmospheric Administration as part of the Census of Marine Life's Tagging of Pacific Predators project. She joined the University of Maryland in 2010, where her research focuses on studying patterns of habitat use and behavior of marine species and its application to management and conservation. Dr. Bailey received her Ph.D. in biological sciences at the University of Aberdeen.

Dr. Daniel E. Crocker is a professor of biology at Sonoma State University. His research has focused on both the physiology and behavior of marine mammals. He has published widely on the metabolism, endocrinology, and toxicology of pinnipeds as well as their reproductive and foraging ecology. His current research is focused on the endocrine stress responses of marine mammals and how they vary with foraging success, fasting, and life-history stage. He is examining the interaction of stress responses with the reproductive and immune systems to better understand how stress has demographic impacts. The ultimate goal of this research is to better understand how marine mammals respond to climate variability and anthropogenic stressors. Dr. Crocker received a Ph.D. in biology from the University of California, Santa Cruz.

Dr. James E. Estes is a professor of ecology and marine biology at the University of California, Santa Cruz. He is an internationally known expert on marine mammals and a specialist in the critical role of apex predators in the marine environment. He has conducted field research in Alaska, California, Canada, Mexico, New Zealand, and Russia and has published more than 150 scientific articles, several books, and monographs, and has served on the editorial boards for a variety of professional societies. He is a Pew Fellow in marine conservation, a fellow of the California Academy of Sciences, and a member of the National Academy of Sciences. He received the Western Society of Naturalist's Lifetime Achievement Award in 2011 and the American Society of Mammalogists' C. Hart Merriam Award in 2012. Dr. Estes received his Ph.D. in biology/statistics from the University of Arizona.

Dr. Clinton D. Francis is an assistant professor in the Department of Biological Sciences at California Polytechnic State University. His research spans evolutionary ecology, community ecology, and global change biology, with a focus on avian behavior and ecology. Most of his research seeks to understand how organisms and ecological communities respond to novel environmental conditions created by human activities with an emphasis on how organisms and ecological systems respond directly and indirectly to changes in the acoustical environment. Current work includes (1) revealing links between anthropogenic forces, chronic stress, and fitness; (2) using manipulative field experiments to quantify the costs of anthropogenic noise on reproductive success; and (3) understanding how soundscapes mediate interactions between human and ecological systems. Dr. Francis received his Ph.D. in ecology and evolutionary biology at the University of Colorado.

Dr. John Harwood is a professor of biology at the University of St. Andrews. He is a former director of the Sea Mammal Research Unit, which advises the U.K. and Scottish governments on the conservation of seals and whales. He was also the director of the Centre for Research into Ecological and Environmental Modeling from 2004 to 2009. Currently, his main interest is in developing methods for assessing and mitigating the effects of anthropogenic disturbance on marine ecosystems. Additional research involves exploring the effects of individual variation and spatial structure on the population dynamics, genetics, and epidemiology of vertebrates, particularly marine mammals. He is currently co-chair of the Office of Naval Research's Population Consequences of Disturbance Working Group. Dr. Harwood received his Ph.D. in zoology from the University of Western Ontario.

Dr. Lori H. Schwacke is a biostatistician for the National Oceanic and Atmospheric Administration's National Centers for Coastal Ocean Science and Chief of the Oceans and Human Health Branch. Recognizing the parallels of studying disease in human populations and in populations of marine protected species, her research focuses on the application of statistical models developed for human medicine to assess the risk of stressors such as environmental contaminants, infectious disease, and natural toxins on marine mammals. Most recently, she has been integrally involved in the assessment of injuries to nearshore dolphin populations in the Gulf of Mexico following the *Deepwater Horizon* oil spill. Dr. Schwacke received her Ph.D. in biostatistics, epidemiology, and systems science from the Medical University of South Carolina.

Dr. Len Thomas is an ecological statistician at the University of St. Andrews. He is the director of the Centre for Research into Ecological and Environmental Modeling and a reader in the School of Mathematics and Statistics. He is also part of the U.K. National Centre for Statistical Ecology and the Scottish Oceans Institute. His main research areas focus on the development of methods and software for estimating the size, density, and distribution of wild animal and plant populations, and the use of computer-intensive methods to fit and compare stochastic models of wildlife population dynamics and animal movement. Of relevance to this committee, he has led research projects developing methods for quantifying marine mammal density, distribution, and trends (particularly from passive acoustic data), analyzing cetacean behavioral response studies, and quantifying the population consequences of anthropogenic disturbance. He has also served on the BP-sponsored Working Group on Assessment of Cumulative Effects of Anthropogenic Underwater Sound, as well as the Office of Naval Research's Population Consequences of Disturbance Working Group. Dr. Thomas received his Ph.D. in forestry from the University of British Columbia.

Dr. Douglas Wartzok is a professor of biology at Florida International University, and the former provost, executive vice-president, and chief operating officer. His research on marine mammals has taken him from the Arctic Ocean to Antarctica to study seals, whales, and walrus. His research focuses on behavioral and physiological ecology of marine mammals; sensory systems involved in under-ice navigation by seals; and psychophysiological studies of captive marine mammals. For the past decade he has been involved in the issue of the effects of naval antisubmarine warfare sonar on marine mammals, in particular beaked whales. He recently served as chairman of the Committee of Scientific Advisors for the U.S. Marine Mammal Commission and is a former editor of *Marine Mammal Science*. He is a current member of the Ocean Studies Board, served on the Natonal Research Council Committee on Assessing Ambient Noise in the Ocean with Regard to Potential Impacts on Marine Mammals, and chaired the Committee on Determining Biological Significance of Marine Mammal Responses to Ocean Noise. Dr. Wartzok received his Ph.D. in biophysics (neurophysiology) from Johns Hopkins University.

STAFF

Dr. Kim Waddell is a senior program officer with the Gulf Research Program, after serving 3 years as a study director with the Ocean Studies Board at the National Academies of Sciences, Engineering, and Medicine in Washington, DC. His recently completed reports include *An Ecosystem Services Approach to Assessing the Impacts of the* Deepwater Horizon *Oil Spill in the Gulf of Mexico* and *Evaluating the Effectiveness of Fish Stock Rebuilding Plans in the United States*. Dr. Waddell rejoined the National Academies in 2011 after a 6-year hiatus during which he was a research associate professor at the University of the Virgin Islands and Texas

A&M University working to build marine and environmental research capacity in the Caribbean region. He received his Ph.D. in biological sciences from the University of South Carolina and his B.A. in environmental studies from the University of California, Santa Cruz.

Stacee Karras is an associate program officer with the Ocean Studies Board. She joined the National Academies of Sciences, Engineering, and Medicine in 2012 as a fellow and served as a research associate for the Ocean Studies Board between 2013 and 2015, when she took on her current role. She received her B.A. in marine affairs and policy with concentrations in biology and political science from the University of Miami in 2007. The following year she received an M.A. in marine affairs and policy from the University of Miami's Rosenstiel School of Marine and Atmospheric Science. In 2012, she earned her J.D. from the University of Virginia School of Law.

Payton Kulina joined the Ocean Studies Board in June 2013 as a senior program assistant. He graduated from Dickinson College in 2010 receiving a B.A. in policy management. He is currently pursuing an M.S. degree in finance through the Kogod School of Business at American University. Prior to this position, Mr. Kulina worked as a coordinator with BP Alternative Energy, also in Washington, DC.

Appendix D

Glossary

Accommodation – A response of a biological system to an environmental stressor that restores the system to its normal or baseline condition or establishes a new set point.

Acute Effect – The severe, often lethal, effect of a stressor on an individual that occurs rapidly and is of short duration (see also Chronic Effect).

Acute Exposure – Exposure to a stressor that occurs for a single, discrete period of time (see also Chronic Exposure and Intermittent Exposure).

Adaptive Management – A systematic approach for improving resource management by learning from management outcomes.

Additive Stressor Effect – The combined effect of two or more stressors is considered additive when the shape of the dose–response function of either stressor does not change in the presence of the other stressor (see also Antagonistic Stressor Interaction, Interactions Among Stressors, Stressor, and Synergistic Stressor Interactions).

Adverse Outcome Pathways – A structured representation of biological events leading to adverse effects that is often considered in risk assessments.

Aggregate Exposure – The combined exposure to one stressor from multiple sources or pathways integrated over a defined relevant period: a day, season, year, or lifetime.

Allostatic Load – An organism's cumulative physiological degradation resulting from exposure to stressors, as well as from heightened activity of physiological systems or changes in metabolism.

Antagonistic Stressor Interaction – The interaction of two or more stressors is considered antagonistic if the resulting effects are less than the sum of the effects of the individual stressors (see also Additive Stressor Effect, Stressor, and Synergistic Stressor Interactions).

Bias – The difference between a true population parameter and the expected value of the estimate of that parameter (see also Precision).

Chronic Effect – A stressor effect that does not immediately result in death or reproductive failure, but persists or is irreversible, and may influence long-term survival or reproductive success.

Chronic Exposure – Ongoing or continuously occurring exposure to a stressor (see also Acute Exposure and Intermittent Exposure).

Cumulative Risk – The combined risk from exposures to multiple stressors integrated over a defined relevant period: a day, season, year, or lifetime.

Direct Effects – When considering the influences and interactions among species, and between species and their abiotic environment, direct effects are the proximate impacts that one species or factor has on another species or factor without the effect occurring via an intervening species or factor. In the interaction webs in Chapter 6, these direct effects are depicted as single arrows pointing from one node to another node (see also Indirect Effects and Interaction Web).

Dose – The magnitude or amount of a stressor that is directly experienced or ingested, inhaled, or absorbed by an animal, ideally measured by a dosimeter on the animal.

Dose–p(response) Function – The relationship between the dose or dosage of a particular stressor and the probability of a particular response.

Dose–Response Relationship – The relationship between the amount of exposure (dose) to a stressor and the resulting changes in behavior, physiology, or health (response).

Driver – A biotic or abiotic feature of the environment that affects populations directly and/or indirectly by changing exposure to a single (or multiple) extrinsic stressor.

Ecological Driver – A biotic or abiotic feature of the environment that affects multiple components of an ecosystem directly and/or indirectly by changing exposure to a suite of extrinsic stressors. Ecological drivers may operate on multiple species at varying trophic levels, and may affect multiple ecosystems.

Exposure – Contact with or experience of a stressor, ideally measured in the environment near the animal.

Extrinsic Stressor – A factor in an animal's external environment that creates stress in the animal (see also Intrinsic Stressor and Stressor).

Health – The ability of an organism to adapt and self-manage.

Hearing Threshold – The lowest intensity of a sound at a particular frequency that an organism is able to hear. These thresholds are defined as a function of frequency.

Hearing Threshold Shift – An increase in an organism's hearing threshold (decrease in sensitivity), often caused by a high-intensity sound. This shift can be either temporary (temporary threshold shift, TTS) or permanent (permanent threshold shift, PTS).

Homeostasis – The tendency of the physiological systems of an organism to maintain internal stability in response to stimulus that might disturb its normal condition or function.

Indirect Effects – Interactions between species or between species and the abiotic environment that occur through one or more intervening species or abiotic factor.

Interaction Web – A means of considering the relationships and interactions among species, and between species and their abiotic environment as defined by Dunne et al. (2002). An interaction web is premised on the idea that the distribution and abundance of species in an ecosystem are determined by the interactions among and between species and abiotic environmental elements (see also Direct Effects and Indirect Effects).

Interactions Among Stressors – Interactions occur when the presence of one stressor changes the shape of the dose–response function of the other stressor (see also Additive Stressor Effect).

Intermittent Exposure – Exposure to a stressor that occurs intermittently, repeatedly, or in cycles (see also Acute Exposure and Chronic Exposure).

Intrinsic Stressor – An internal factor or stimulus that results in a significant change to an animal's homeostatic set point. Short-term internal stresses that evoke physiological responses occurring daily to maintain an organism near its homeostatic set points *are not* considered stressors, but natural aspects of an individual's life cycle (e.g., lactation, migration, molting, and fasting) that result in significant changes to homeostasis are considered stressors (see also Extrinsic Stressor and Stressor).

Masking – Acoustic interference that impedes an organism's ability to detect biologically important signals.

Noise – Sounds that are unwanted by or are not useful for a receiver.

Oxidative Stress – Stress to an organism caused by a disturbance in the balance of prooxidants and antioxidants.

Population Health – The distribution of health outcomes in a population or a subset of a population, *as well as* the determinants or factors that influence those outcomes.

Precision – A statistical measure of the repeatability of a sample or an estimate, given by the inverse of the variance (see also Bias).

Recovery – Restoration of normal function after withdrawal of a stressor.

Stressor – Any causal factor or stimulus, occurring in either the animal's internal or external environment, that challenges homeostasis of the animal.

Synergistic Stressor Interactions – The interaction of two or more stressors is considered synergistic if the resulting effects are more than that of the sum of the effects of the individual stressors (see also Additive Stressor Effect, Antagonistic Stressor Interaction, and Stressor).

APPENDIX C

Review of Bryde's Whale Proposed Closure in the Proposed Incidental Take Regulation for Geophysical and Geological Surveys in the Gulf of Mexico

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August, 2018

On June 22, 2018, the National Marine Fisheries Service (NMFS) published a draft Incidental Take Regulation (ITR) for marine mammals affected by geophysical surveys in the Gulf of Mexico (GOM). This report supports the International Association of Geophysical Contractors' (IAGC's) public comment on this draft ITR and supplements the previous report provided by Ecology & Environment, Inc. (E & E). The focus of this report is a review of the Bryde's whale proposed closure area.

The ITR appears to base its proposed Bryde's whale closure on Alternative F of the Programmatic Environmental Impact Statement (PEIS) for geological and geophysical activities in the Gulf of Mexico (BOEM 2017). According to BOEM (2017), the closure area for Bryde's whales is refined from the Area of Concern 2 from the Amended Settlement Agreement to "partially correspond" with the year-round "biologically important area" (BIA) for Bryde's whales from LaBrecque et al. (2015), expanded to the 400 m contour "to incorporate additional Bryde's whale sighitngs" (page 2-19 to 2-20 in BOEM [2017]). LaBrecque et al. (2015) stated that they chose the boundary of the Bryde's whale BIA in the Gulf of Mexico based on the entire area in which Bryde's whales have been sighted in surveys, which, in their dataset, was between the 100 meter (m) and 300 m isobaths from the head of DeSoto Canyon to south of Tampa, Florida. The National Marine Fisheries Service (NMFS) does not describe how this area was defined in the draft ITR, but it appears the PEIS is the baseline for the proposed closure, with some modifications. In the ITR, NMFS frames it as choosing a baseline area of "expected importance" for Bryde's whales and then adding a buffer around it based on the generic 160 dB received distance from the modeling (as described in BOEM [2017] on page 2-17), as well as adding a 6 kilometer (km) buffer around DeSoto Canyon based on what NMFS refers to as Matthews et al. (2016). That reference is confusing because Matthews et al. (2015, 2017) are the communication space reports NMFS has included in its supplementary material, and the reference list for the draft ITR seems to indicate that Matthews et al. (2016) is the same as Matthews et al. (2015) (i.e. the titles are the same). It is not clear if there is a 2016 version that was not included in the supplementary materials. Bryde's whales are not mentioned in Matthews et al. (2017). On page 29280 of the draft ITR, NMFS stated that Matthews et al. (2016) recommended a 5.4 km buffer in DeSoto Canyon. This statement does not appear to be in Matthews et al. (2015). The only place the number 5.4 occurs in the 2015 report is in Tables 13 and 17 which show "time-averaged equivalent sound pressure levels (Leq) at each receiver site with M-weighting," which is not a distance but a sound pressure level, and De Soto Canyon is not specifically mentioned in those tables. Site 8 is DeSoto Canyon in Matthews et al. (2015), and the authors summarize the findings there as follows:

> "Site 8 (De Soto Canyon, 919 m water depth) experienced decreased listening area of up to 54.2% (45.8% remaining) for low-frequency cetaceans. Bryde's whale communication space decreased up to 27% (73% remaining) for Alternative C. The proposed area closures for Alternative F1 further improved the noise conditions at this site since it lies on the eastern edge of the central planning closure area. This led to a change in listening area of 24.1% (75.9% remaining) compared to the no activity alternative (Alternative A).This is an increase of listening area by 30.1% compared to Alternative C."

It would appear that Matthews et al. (2015) analyzed alternatives already proposed in the PEIS (see Figure 1 in Matthews et al. [2015]), so it would be circular to suggest that Matthew et al (2015) is the basis for the closure area around De Soto Canyon as is implied on page 29280.

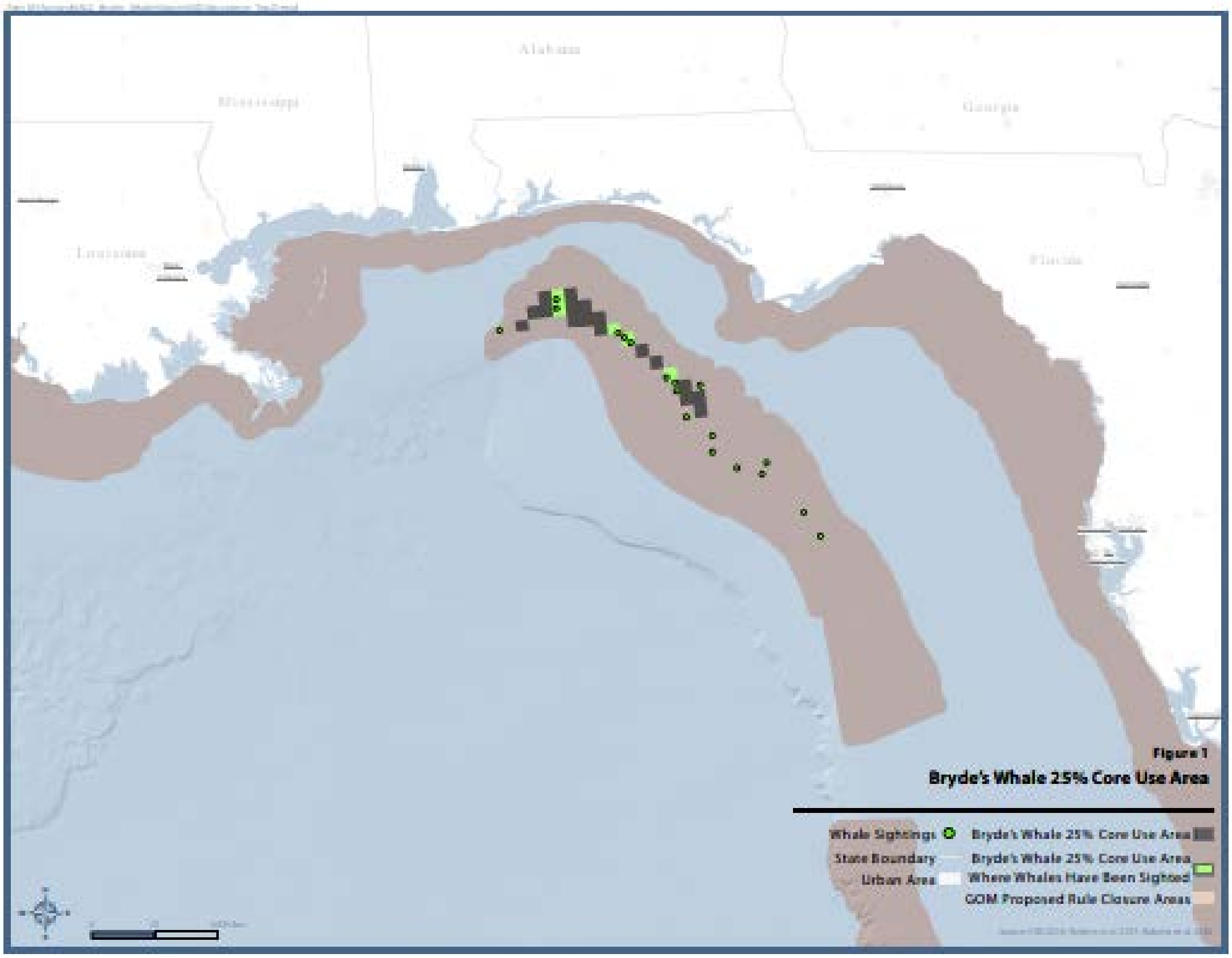
In conclusion, it appears that the Bryde's whale closure area includes an area determined somehow in the settlement agreement, with the BIA (which is all areas in which Bryde's whales have been observed) added, with additional buffers out to the 400 m depth contour and then further out to the 160 dB received level distance (from NMFS' generic seismic propagation model) and an additional 6 km buffer around the BIA in DeSoto Canyon area based on a source that does not appear to be available or is miscited and determined something about the communication space within that area. ***Thus, the proposed closure encompasses what is considered to be the entire known range of Bryde's whales in the U.S. Gulf of Mexico with buffers around it.***

This is stricter than proposed closures for North Atlantic right whales in the Atlantic (82 Federal Register [FR] 26244), which are currently listed under the Endangered Species Act, not just proposed for listing. Most of the closures proposed to protect cetaceans in the Atlantic were based on 25% core use areas predicted by Roberts et al. (2016) (82 FR 26244). These closures can also be argued for various reasons, but given the concerns about Bryde’s whales, it seems like a 25% core use area makes much more sense than 100% of the known range of this species in the U.S. Gulf of Mexico along with additional buffers.

Using the same procedure as described in 82 FR 26244, E & E determined the 25% core use area of Bryde’s whales in the Gulf of Mexico using Roberts et al. (2016) density estimates for the region (Figure 1). If this area is further limited to only grid cells of Roberts et al. (2016) in which Bryde’s whales have been observed, the green area in Figure 1 results. E & E suggests that IAGC consider recommending either a polygon around the general 25% core use area or a polygon around the 25% core use area that includes only grid cells in which Bryde’s whales have been observed. This aligns with the approach taken by NMFS in the Atlantic in analyzing potential closure areas and considers the most important habitat for Bryde’s whales rather than including the entire range of the population within the U.S. Gulf of Mexico.

Figure 1. Bryde’s whale 25% Core Use Area and Proposed Closure Area

Further, with respect to Bryde’s whales, E & E encourages IAGC to use the abundance information provided in



our previous deliverable to address the issue of mean densities artificially inflating Bryde's whale abundance and availability for take in NMFS' model. The total number of available Bryde's whales rises from an estimated 44 in Roberts et al. (2016) to 256 in the take model because of the use of mean densities within designated areas. Thus, in addition to a closure that includes the entire range of the population in the U.S. Gulf of Mexico, the take estimate model allows for take of considerably more Bryde's whales than are predicted to occur in the U.S. Gulf of Mexico, and makes this estimate over multiple surveys, compounding the inflation of the estimate. In addition, if NMFS does enact a closure zone, the zone should be incorporated into final take estimates to decrease the take relative to the density of Bryde's whales in the closure area.

Literature Cited

Bureau of Ocean Energy Management (BOEM). 2017. Gulf of Mexico OCS Proposed Geological and Geophysical Activities Western, Central, and Eastern Planning Areas Final Programmatic Environmental Impact Statement. OCS EIS/EAQ BOEM 2017-051.

LaBrecque, E., C. Curtice, J. Harrison, S.M. Van Parijs, and P.N. Halpin. 2015. Biologically important areas for cetaceans within U.S. waters—Gulf of Mexico region. Aquatic Mammals. Special Issue 41(1):30-38.

Matthews, M-N.R., H. Frouin-Mouy, and D. Zeddies. 2017. Cumulative and chronic effects in the Gulf of Mexico: Addendum: Estimating reduction of listening area and communication space due to seismic activities in support of the BOEM Geological and geophysical activities draft programmatic environmental impact statement. JASCO Document #01510, Version 1.0. Technical report by JASCO Applied Sciences for National Marine Fisheries Service.

Matthews, M-N.R., A. Schlesinger, and D. Hannay. 2015. Cumulative and chronic effects in the Gulf of Mexico: Estimating reduction of listening area and communication space due to seismic activities in support of the BOEM Geological and geophysical activities draft programmatic environmental impact statement. JASCO Document #01073, Version 1.0. Technical report by JASCO Applied Sciences for National Marine Fisheries Service.

Roberts J.J., B.D. Best, L. Mannocci, E. Fujioka, P.N. Halpin, D.L. Palka, L.P. Garrison, K.D. Mullin, T.V.N. Cole, C.B. Khan, W.M. McLellan, D.A. Pabst, G.G. Lockhart. 2016. Habitat-based cetacean density models for the U.S. Atlantic and Gulf of Mexico. Scientific Reports 6:22615. doi: 10.1038/srep22615.

APPENDIX D

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I. Abundance Calculations and Incidental Take Modeling

The following comments address certain aspects of the abundance calculations performed or assumed by NMFS and the related incidental take modeling.

A. In the Roberts et al. (2016) model, Bryde's whales are extended into areas where they have not been observed or recorded, and the population estimate derived from Roberts et al. (2016) suggests that the estimated densities may be too high or the estimated population size of 33 from the 2009 surveys (Hayes et al. 2017) is too low. Using the Roberts et al. (2016) model, without adjusting for the mean densities used in take estimation, predicted Bryde's whale population size from Roberts et al. (2016) is 44, but when adjustment is made to determine available Bryde's whales for take in the seven zones, the available population is 256. This is a discrepancy of 482% between the estimated population of 44 and the population created by using mean densities in the seven chosen zones. Similarly, but not as dramatically, the abundance estimate from Roberts et al. (2016) for short-finned pilot whales is 1,981, but the population available to be taken in the mean density model of seven zones is 4,885, an increase of 147%. In other cases, discrepancies are much lower, but these discrepancies suggest that significant overestimate of take for Bryde's whales and short-finned pilot whales is caused by using mean densities within zones. For all species, a correction for this issue should be used. To make such a correction, NMFS could adjust the mean densities themselves proportional to Roberts et al. (2016) abundance. So for example, for Bryde's whales, a total of 44 individuals is predicted by Roberts et al. (2016), but 256 individuals are predicted by the mean densities in the zones (sum of abundance in each zone). Thus, 44/256 = corrected mean density/original mean density. That would be a "correction factor" of multiplying the original mean density in each zone by 0.172 to get a corrected mean density for each zone.

B. The Proposed Rule states that a new version of the model for Bryde's whales has been created since Zeddies et al. (2015). Based on this new version, NMFS is assuming zero density in Zone 6 "western GOM slope," so NMFS discounted predicted exposures of Bryde's whales in that zone. We support using the results of a model that better reflects the lack of use by Bryde's whales in areas where they have not been observed. Bryde's whales were also not observed in zones 2, 3, 5, 6, or 7 in the survey data from 1994 to 2009 included in Roberts et al. (2016), suggesting that this species is not likely to occur in any of those zones, despite predicted densities by Roberts et al. (2016). Bryde's whales are relatively easy to see, as they are large and do not dive deeply like sperm whales. Thus, although the habitat may be suitable for Bryde's whales elsewhere in the GOM according to modeling, surveys do not support the occurrence of Bryde's whales in zones 2, 3, 5, 6, or 7.

C. Suitability of habitat does not necessarily result in cetaceans following predicted use patterns. For example, Forney et al. (2012) reported that models for three species in their study (sperm whales, killer whales, and coastal spotted dolphins) failed to converge or produced results inconsistent with known occurrence patterns. Only 22 sightings of Bryde's whales contributed to Roberts et al. (2016) modeling; thus, the sample size is less than that suggested for line-transect distance sampling methods (Buckland et al. 2001). Becker et al. (2010) reported that habitat-based cetacean model performance was better when there were more sightings, and the worst model performance in their study was for Pacific white-sided dolphins, with only 25 sightings used in the model. Forney et al. (2012) stated that there are numerous sources of uncertainty in predictive habitat modeling, including survey design, stochasticity in the sighting process, measurement error, model parameter estimation errors, and model selection error. Given the low but predictable sighting of Bryde's whales concentrated in zones 1 and 4, the data suggest that Bryde's whales are mainly found in this region and not in high densities in zones 2 or 3. Because this high density in the model is not reflective of empirical data, we recommend that NMFS also discount take from zones 2 and 3 or reduce the take to reflect a density more similar to the low density zones, such as zones 5 or 7.

D. In Roberts et al. (2016), 17 years of surveys resulted in only the following numbers of sightings in the GOM: Fraser's dolphins, 5; killer whales, 16; melon-headed whales, 29; false killer whales, 19; and pygmy killer whales, 27. These sample sizes are not large enough to be statistically robust for application of distance sampling methods (Buckland et al. 2001). We recommend that NMFS use an alternative method to more accurately reflect the low occurrence of these species in the action area. In addition to issues with density models based on so few sightings, complex animat models require knowledge of actual animal movement parameters that populate the model, so rare sightings also make it difficult to know movement patterns in the region. Rather than using densities that are derived from insufficient data for statistical robustness and complicated animat models that rely on parameters that are not well known for these species in the GOM, NMFS could simplify the modeling to more accurately reflect what we know about seismic survey encounters with these species in the GOM. For example, NOAA could use data from seismic survey observation reports to anticipate encounter rates with rare species and extrapolate those to potential take relative to effort. More sightings of rare species have occurred in seismic surveys than in NMFS abundance surveys, suggesting that these data may be more robust in any case (*compare* Barkaszi et al. 2012 to Roberts et al. 2016). The table below provides estimated observations of these species per year based on Barkaszi et al. (2012). BOEM has considerably more data (2010 to present) that can be used in evaluating encounter rates and potential exposures at various threshold radii. Observations may not extend to the full Level B exposures area, and missed animals must be accounted for through correction factors, but observations do provide a sense of scale of potential exposures. As such, it is highly unlikely that tens of thousands of individuals of rare species would be exposed to Level B harassment as suggested in the Proposed Rule.[1]

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[1] NMFS should also clarify why particular surrogate species were chosen for animat model inputs, as no justification is provided in Zeddies et al. (2015). The species for which surrogates are used include very rare or transient species in the region (Fraser's dolphin, melon-headed whales, false killer whales, and pygmy killer whales). The lack of good data to inform model parameters for these species further highlights the need to consider them in the context of their uncommon occurrence in the GOM.

Observations of Rare Species during Seismic Surveys in the Gulf of Mexico 2002–2008 Based on Barkaszi et al. (2012)

Species	Observations of Species 2002–2008	Observations of Species Per Year	Mean Group Size	Estimated Individuals Per Year
Fraser's Dolphin	16	2.3	16.0	37
Killer Whale	5	0.7	5.0	7
Melon-headed	42	6.0	22.4	134
False Killer Whale	46	6.8	8.3	56
Pygmy Killer Whale	44	6.3	13.8	87

E. Although data on mean group size for GOM species are available from NMFS surveys and seismic survey monitoring, Zeddies et al. (2015) used the highest reported group size mean across various regions in the case that group size across studies was highly variable. This inflates the group size anticipated to occur in the GOM (unless the highest group size is from a GOM study). For example, Zeddies et al. (2015) used a group size of 15 sperm whales, while Mullin and Fulling (2004) reported mean group sizes from 67 sperm whale sightings in the GOM ranging from 1.8 to 2.6, for which a pooled mean would be 2.1. Maze-Foley and Mullin (2006) provide a full analysis of group sizes of GOM marine mammals from surveys spanning 1991-2001. Zeddies et al. (2015) also reported that although the generic outcome of exposure estimates is very similar whether animats are treated as individuals or groups, the likelihood that no animals would have a behavioral response was much higher in species with large group sizes. Thus, taking group size into account is important in terms of likelihood of exposure, particularly as Zeddies et al. (2015) reported likelihoods that no animals would have a behavioral response were in the 95–98% range.

F. NMFS states that mitigation and aversion were not quantified in the take model because of "too much inherent uncertainty" and "too little information." However, the effects of mitigation are not considerably harder to predict than where animals will be located over a five-year period relative to surveys. Models are used to address these questions, and they can be used to address mitigation. As occurred with all the modeling that contributed to the take estimates in the Proposed Rule, assumptions based on the best available science can be made to incorporate consideration for mitigation. The Navy has incorporated quantification of effects of mitigation and aversion in its modeling for marine mammal sound exposures. As examples, Commander (2012) and Blackstock et al. (2017) demonstrate methods of quantitative accounting for mitigation that could be adapted and used in the GOM.

G. Although risk frameworks can assist in qualitatively evaluating impacts, we question the assignment of values of percentages of take relative to a zone-specific population. First, the zone-specific population is not a biological population. There is free exchange among zones (with the potential exception of bottlenose dolphins in some zones). The zones represent a lower resolution density grid than used in Roberts et al. (2016) rather than any barrier to animal movements. Also, Southall et al. (2017) do not provide the numbers that were used for the PBR calculations, which appear to have been made by zone only and not across the full population. Second, Southall et al. (2017) appear to use exposures rather than individuals for their evaluation of impacts of Level A take, and individuals are admittedly overestimated in the Proposed Rule. It is not clear from Southall et al. (2017) how it was determined which percent takes would be assigned to the risk levels of "very high," "high," "moderate," "low," and "very low" or what the calculated PBR values were. It is also unclear how "high" and "moderate" levels of known or future anthropogenic noise are evaluated in each area. It is also unclear how levels 0 to 4 are differentiated for chronic non-noise risk factors. We recommend that NMFS include more explanation to support these approaches and conclusions.

H. NMFS states that *Kogia* spp. are expected to demonstrate greater aversion than NMFS accounted for in the model. *Kogia* spp. are known to avoid ships in general in the GOM, with an estimated 73% avoidance of NMFS survey vessels in 1992–1994 surveys (Würsig et al. 1998). Thus, this species is highly unlikely to approach a vessel and would be likely to avoid a vessel before experiencing Level A take. Barkaszi et al. (2012) reported that only 20 *Kogia* spp. sightings occurred during seismic surveys from 2002 to 2008 (seven years), representing 59 individuals (sighting frequency of 0.10 per 1,000 hours) with a mean closest distance from seismic sources of 773.6 m. Barkaszi et al. (2012) reported that the required shutdown of seismic sources for *Kogia* spp. within 500 m occurred in two instances. Zeddies et al. (2015) reported the distance to the 200 dB peak sound pressure level for an 8,000-in^3 seismic array to be 575 m (the 575 m range to Level A is based on an arguable assignment of *Kogia* to the HF hearing group instead of the MF hearing group and the use of an unrepresentative 8,000 cubic inch array as the representative seismic sound source). Given that the distance to the Level A peak threshold radius for an 8,000-in^3 seismic array is similar to the 500-m shutdown zone from Barkaszi et al. (2012), only two *Kogia* spp. sightings during seismic surveys in the GOM in a seven-year period were within a range that could potentially result in PTS based on NOAA's (2016) criteria. Accordingly, NMFS should use the existing data on *Kogia* spp. distances from seismic vessels in the GOM to adjust the exposure estimate. Given the known aversion to vessels for this genus in general, it is not scientifically supported to make assumptions of low aversion rates in this case.

II. Cumulative and Chronic Effects

Matthews et al (2015) and Matthews et al (2017) reflect two modeling exercises conducted by JASCO at the direction of NMFS to develop metrics for possibly evaluating cumulative and chronic effects of sound through two modeled metrics: a "listening space" metric based on a model developed for birds and other species calling in air (Barber et al, 2009), and a communication space metric developed by Clark et al (2009) based on the sonar equation (*see, e.g.*, Urick, 1983).

Matthews et al. (2015) contains calculated values for changed or "lost" listening space and communication space for each of the proposed PEIS alternatives, as a basis for seismic activity being displaced or reduced through the different time/area closure options and activity reduction options. An additional analysis specifically examines the predicted lost listening space and communication space for the species most like to be listening and hearing at seismic survey frequencies, such as the Bryde's whale. Matthews et al (2017) adds a similar analysis for sperm whales.

Implicit in the Matthews et al. (2015, 2017) models is that each instance of communication opportunity is critical and that a model based on averages and worst case scenarios predicts the outcome of each and every instance of potential communication when there are thousands or millions of such communication opportunities in a year of an animal's life. Moreover, the ability to mathematically calculate averaged or total acoustic energy or sound pressures over a day, a month or a year does not have any biological significance because acoustic energy is transitory and is not stored in any way by the receiver. What can accumulate are phenomena like hearing threshold shifts, physiological stress (or allostatic loading), but these depend on the rate of accumulation of physiological changes and recovery from such changes, and those do not linearly track mathematical measures like Sound Exposure Level (SEL), Energy Equivalent Sound Pressure (L_{eq}), or averaged (Root Mean Square) Sound Pressure Level (SPL) over various time spans. In other words, a mathematical measure of sound that might be correct in terms of the physics is not a substitute for the biological response to that sound. With that said, the following briefly critiques certain aspects of Matthews et al. (2015, 2017).

<u>Array Source Levels</u>**.** The use of an unusually large array with an unusually large number of elements leads to over-prediction of noise sources levels (seismic surveys) by 7 dB or more, not the 2 dB mistakenly assumed by NMFS in the ITR, based on comparing total volume alone, without accounting for the number of elements, which is the greater predictor of relative array amplitude. This over-estimation of source level then cascades through all of the calculations to over-predict listening space reductions and communication space reductions by approximately a factor of four. Moreover, over-estimation of other values in these equations further pushes the more likely loss of listening or communication space close to zero.

<u>Ambient Noise Values.</u> Matthews et al. (2015, p. 13) state that they used NMFS SoundMap data from the NMFS CetMap website, but it is not clear if they used all the data or only "commercial shipping" data, and whether "commercial shipping" includes all non-recreational boating (*e.g.*, fishing vessels, oil field support vessels, etc.), or all vessels over a certain size (*e.g.*, 1000 gross tons) or some other value. This is important because the result has a 5+ dB 'notch' in the baseline ambient noise right where Matthews et al (2015) choose to analyze the increase of noise for their Bryde's whale analysis (p. 14). That notch is an artifact of the way the authors chose to build an artificial ambient baseline, and the notch is coincidentally right at the place that most exaggerates the contribution of seismic to reducing communication space calculations for Bryde's whales.

<u>Choice of Bryde's Whale Signal Parameters</u>**.** Although baleen whale calls can vary significantly in amplitude, a choice of 152 dB SPL (Matthews et al. 2015, p. 20) is definitely at the low end, as, for example, humpbacks, blue, fin and minke whales typically produce calls in the 170-190 dB SPL range. As mentioned above, the frequency arbitrarily chosen for the model matches an ambient noise notch in their artificial ambient noise curve, which causes the impact of seismic sound to be exaggerated relative to any other frequency they might have chosen.

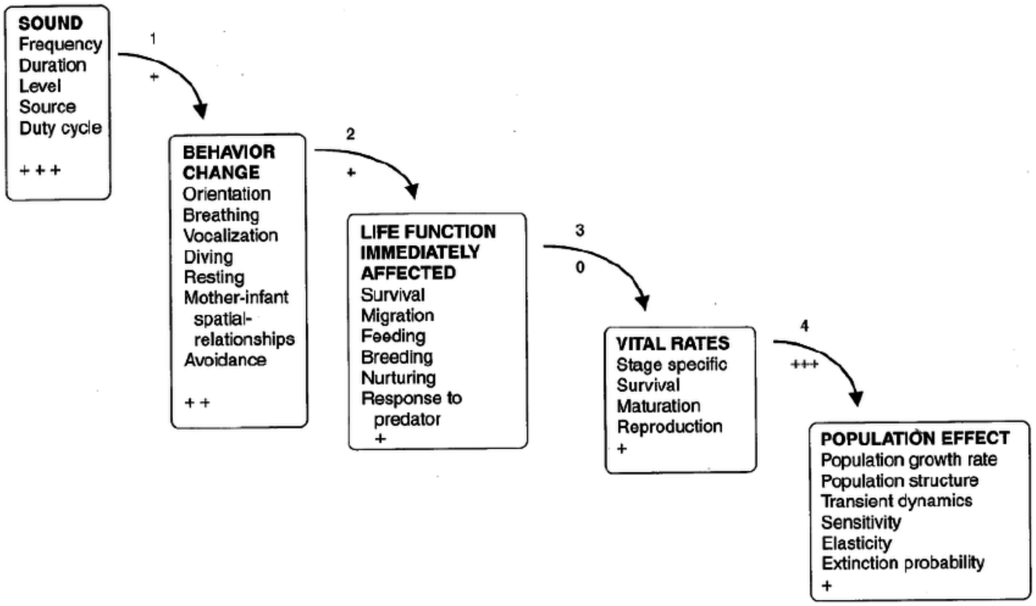
<u>Values Used in the Communication Space Model</u>. Matthews et al. (2015) analyzes only a single 1/3 octave band, omitting call sound outside of that band for consideration (out-of-band listening). Additionally, although a biologically reasonable Directivity Index is in the 5-15 dB range, Matthews et al. (2015) give zero dB of gain from directivity, which is a significant omission leading to huge contributions in over-predicting the lost communication space. A signal gain value of 12.36, while derived from biologically irrelevant engineering signal processing

methods, is approximately in the ballpark of signal gain from time and frequency domain releases from masking, which range from 6 dB to over 20 dB. *See* Erbe et al. (2016).

The use of a simple equal energy model, while common for its simplicity, fails to capture other biological realities of communication. Attentional processes by the receiver can and do lead to detection of some signals well below noise and rejection of other signals above ambient noise, based not on their simple relative amplitude, but on their spatial position or other salient cues. The all-too-common assumption in Clark et al. (2009) and others that any signal that is audible, regardless of distance, should be treated as useful communication ignores a considerable literature showing that context cues like distance and motivational state affect the salience of the cue and thus the likelihood of response (*see e.g.*, Elhilali et al 2009). The assumption that calls tens, hundreds, or even thousands of kilometers away are meaningful and significant to the receiver simply because they are audible runs counter to a greater body of literature from animal and human communication in which many available conspecific communications are rejected because they are too far away or otherwise out of the context that makes them salient and actionable by the receiver.

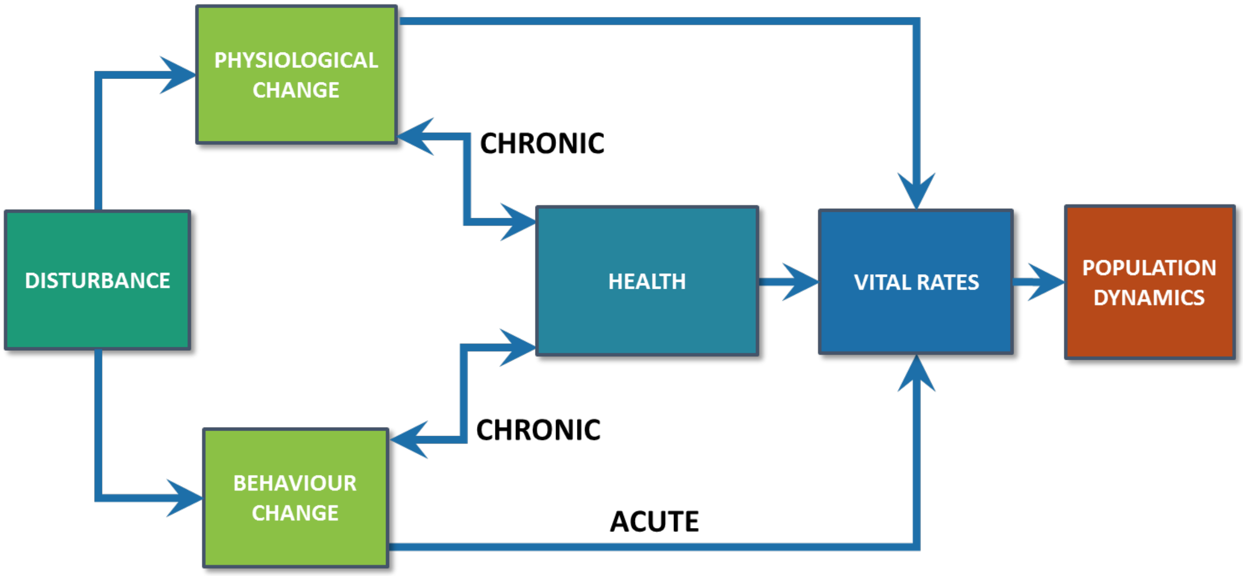
III. Population Consequences of Acoustic Disturbance Model ("PCAD" or "PCOD")

The Population Consequences of Acoustic Disturbance (PCAD) model was the work product of a National Research Council expert panel under the Ocean Studies Board, published in 2005. The expert panel was tasked to develop a model for tracing the consequences of non-injurious acoustic exposure through effects on marine mammal behavior, as well as the consequences for individual health and well-being and marine mammal populations. The following graphic illustrates the PCAD model:

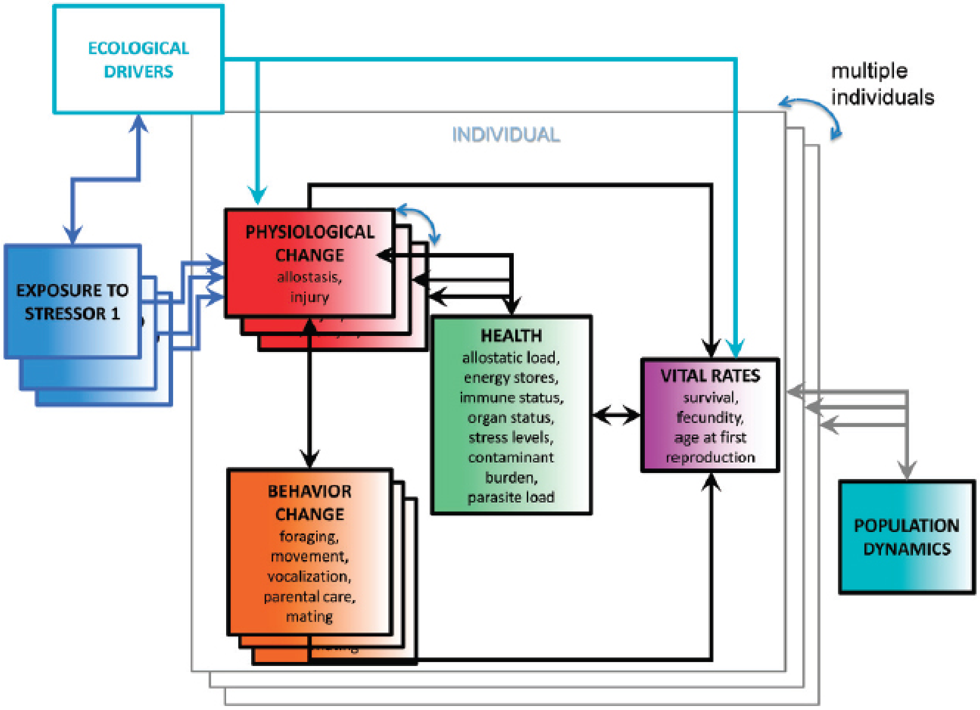


The PCAD model was recognized to be suitable for characterizing any disturbance or perturbation, natural or manmade, and not solely acoustic disturbances. Accordingly the model is now more often referred to as the Population Consequences of Disturbance (PCOD) model. Numerous subsequent studies have used the PCAD/PCOD model to produce testable predictions of how much disturbance would be needed to produce a marine mammal population level change, with general success. Additionally, most model predictions have confirmed the general resilience of animals to disturbances of various kinds, predicting a level of disturbance to individuals that would be hard to produce, especially for a number of individuals sufficient to produce a detectable population level effect.

As the PCOD model has been tested against empirical data, various minor modifications or expansions of the model have been suggested. For example, the version below was generated following work by New et al. (2014) with elephant seals, but is widely applicable to other species and contexts. The New et al (2014) version accounts not only for behavioral effects but also for physiological effects with no apparent outward behavioral component. The consequences of each response modality, behavioral or physiological, is also represented by an immediate or acute consequence on health and life history parameters (survival, growth and reproduction, as well as a chronic or long-term consequence.



In the years since the publication of the initial PCAD model, various groups have expanded the usage and development of the model for translating disturbance at the individual level to population consequences. For example, the NAS (2017) model, the Population Consequences of Multiple Stressors model (PCoMS) is basically a layered build on the foundation of individual PCOD models for specific events. It is a new model for a complex phenomenon or set of phenomena and has not yet been subjected to much research and development or model testing.



The following non-exhaustive list of papers that document the development of, and variations on, the PCAD model and should be taken into account in the rulemaking process:

PCOD References

Bailey, H., B. R. Mate, D. M. Palacios, L. Irvine, S. J. Bograd, and D. P. Costa. 2009. Behavioural estimation of blue whale movements in the Northeast Pacific from state-space model analysis of satellite tracks. Endangered Species Research 10:1.

Bejarano, A. C., R. S. Wells, and D. P. Costa. 2017. Development of a bioenergetic model for estimating energy requirements and prey biomass consumption of the bottlenose dolphin Tursiops truncatus. Ecological Modelling 356:162-172.

Beltran, R. S., J. W. Testa, and J. M. Burns. 2017. An agent-based bioenergetics model for predicting impacts of environmental change on a top marine predator, the Weddell seal. Ecological Modelling 351:36-50.

Christiansen, F. and D. Lusseau. 2015. Linking Behavior to Vital Rates to Measure the Effects of Non-Lethal Disturbance on Wildlife. Conservation Letters 8:424-431.

Costa, D. P., L. A. Hückstädt, L. K. Schwarz, A. S. Friedlaender, B. R. Mate, A. N. Zerbini, A. Kennedy, and N. J. Gales. 2016a. Assessing the exposure of animals to acoustic disturbance: Towards an understanding of the population consequences of disturbance. Proceedings of Meetings on Acoustics 27:010027.

Costa, D. P., L. Schwarz, P. Robinson, R. S. Schick, P. A. Morris, R. Condit, D. E. Crocker, and A. M. Kilpatrick. 2016b. A Bioenergetics Approach to Understanding the Population Consequences of Disturbance: Elephant Seals as a Model System. Pages 161-169 *in* A. N. Popper and A. Hawkins, editors. Effects of Noise on Aquatic Life II.

De Roos, A. M., N. Galic, and H. Heesterbeek. 2009. How resource competition shapes individual life history for nonplastic growth: ungulates in seasonal food environments. Ecology 90:945-960.

Farmer, N. A., D. P. Noren, E. M. Fougères, A. Machernis, and K. Baker. 2018. Resilience of the endangered sperm whale Physeter macrocephalus to foraging disturbance in the Gulf of Mexico, USA: a bioenergetic approach. Marine Ecology Progress Series 589:241-261.

Ferraro, M. S., R. R. Decker, D. P. Costa, P. W. Robinson, D. S. Houser, and D. E. Crocker. 2017. Evaluating gain functions in foraging bouts using vertical excursions in northern elephant seals. Animal Behaviour 129:15-24.

Fiechter, J., L. A. Huckstadt, K. A. Rose, and D. P. Costa. 2016. A fully coupled ecosystem model to predict the foraging ecology of apex predators in the California Current. Marine Ecology Progress Series 556:273-285.

Fleishman, E., D. P. Costa, J. Harwood, S. Kraus, D. Moretti, L. F. New, R. S. Schick, L. K. Schwarz, S. E. Simmons, L. Thomas, and R. S. Wells. 2016. Monitoring population-level responses of marine mammals to human activities. Marine Mammal Science 32:1004-1021.

Gallagher, C. A., S. J. Stern, and E. Hines. 2018. The metabolic cost of swimming and reproduction in harbor porpoises (*Phocoena phocoena*) as predicted by a bioenergetic model. Marine Mammal Science:n/a-n/a.

Harwood, J., S. King, R. Schick, C. Donovan, and C. Booth. 2014. A protocol for implementing the interim population consequences of disturbance (PCoD) approach: Quantifying and assessing the effects of UK offshore renewable energy developmenets on marine mammal populations: Report number SMRUL-TCE-2013-014. Scott Mar Freshwater Sci 5:2.

Hoekendijk, J. P. A., J. Spitz, A. J. Read, M. F. Leopold, and M. C. Fontaine. 2018. Resilience of harbor porpoises to anthropogenic disturbance: Must they really feed continuously? Marine Mammal Science 34:258-264.

Huckstadt, L. A., R. R. Holser, M. S. Tift, and D. P. Costa. 2018. The extra burden of motherhood: reduced dive duration associated with pregnancy status in a deep-diving mammal, the northern elephant seal. . Biol. Lett.

King, S. L., R. S. Schick, C. Donovan, C. G. Booth, M. Burgman, L. Thomas, and J. Harwood. 2015. An interim framework for assessing the population consequences of disturbance. Methods in Ecology and Evolution 6:1150-1158.

McHuron, E. A., D. P. Costa, L. Schwarz, and M. Mangel. 2018. State-dependent behavioural theory for assessing the fitness consequences of anthropogenic disturbance on capital and income breeders. Methods in Ecology and Evolution 8:552-560.

McHuron, E. A., M. Mangel, L. K. Schwarz, and D. P. Costa. 2017b. Energy and prey requirements of California sea lions under variable environmental conditions. Marine Ecology Progress Series 567:235-247.

McHuron, E. A., P. W. Robinson, S. E. Simmons, C. E. Kuhn, M. Fowler, and D. P. Costa. 2016. Foraging strategies of a generalist marine predator inhabiting a dynamic environment. Oecologia 182:995-1005.

McHuron, E. A., L. Schwarz, D. P. Costa, and M. Mangel. in review. A state-dependent model for assessing the population consequences of disturbance on income-breeding mammals. Ecological Modelling.

National Academies of Sciences, E. and Medicine. 2017. Approaches to Understanding the Cumulative Effects of Stressors on Marine Mammals. The National Academies Press, Washington, DC.

New, L. F., J. S. Clark, D. P. Costa, E. Fleishman, M. A. Hindell, T. Klanjšček, D. Lusseau, S. Kraus, C. R. McMahon, P. W. Robinson, R. S. Schick, L. K. Schwarz, S. E. Simmons, L. Thomas, P. Tyack, and J. Harwood. 2014. Using short-term measures of behaviour to estimate long-term fitness of southern elephant seals. Marine Ecology Progress Series 496:99-108.

New, L. F., D. J. Moretti, S. K. Hooker, D. P. Costa, and S. E. Simmons. 2013. Using energetic models to investigate the survival and reproduction of beaked whales (family Ziphiidae). PLoS ONE 8:e68725.

Noad, M. J. and D. H. Cato. 2007. Swimming speeds of singing and non-singing humpback whales during migration. Marine Mammal Science 23:481-495.

NRC. 2005. Marine Mammal Populations and Ocean Noise: Determining When Noise Causes Biologically Significant Effects. The National Academies Press, Washington, DC.

Peck, H. E., D. P. Costa, and D. E. Crocker. 2016. Body reserves influence allocation to immune responses in capital breeding female northern elephant seals. Functional Ecology 30:389-397.

Pirotta, E., M. Mangel, D. P. Costa, B. Mate, J. A. Goldbogen, D. M. Palacios, L. A. Huckstadt, E. A. McHuron, L. Schwarz, and L. New. 2018. A Dynamic State Model of Migratory Behavior and Physiology to Assess the Consequences of Environmental Variation and Anthropogenic Disturbance on Marine Vertebrates. Am Nat 191:E40-E56.

Schwarz, L. K., E. McHuron, M. Mangel, R. S. Wells, and D. P. Costa. 2016. Stochastic dynamic programming: An approach for modelling the population consequences of disturbance due to lost foraging opportunities. Proceedings of Meetings on Acoustics 27:040004.

Villegas-Amtmann, S., B. I. McDonald, D. Paez-Rosas, D. Aurioles-Gamboa, and D. P. Costa. 2017a. Adapted to change: Low energy requirements in a low and unpredictable productivity environment, the case of the Galapagos sea lion. Deep-Sea Research Part Ii-Topical Studies in Oceanography 140:94-104.

Villegas-Amtmann, S., B. I. McDonald, D. Páez-Rosas, D. Aurioles-Gamboa, and D. P. Costa. 2016. Adapted to change: Low energy requirements in a low and unpredictable productivity environment, the case of the Galapagos sea lion. Deep Sea Research Part II: Topical Studies in Oceanography.

Villegas-Amtmann, S., L. K. Schwarz, G. Gailey, O. Sychenko, and D. P. Costa. 2017b. East or west: the energetic cost of being a gray whale and the consequence of losing energy to disturbance. Endangered Species Research 34:167-183.

Villegas-Amtmann, S., L. K. Schwarz, J. L. Sumich, and D. P. Costa. 2015. A bioenergetics model to evaluate demographic consequences of disturbance in marine mammals applied to gray whales. Ecosphere 6.

REFERENCES

Barber, J.R., K.R. Crooks, and K.M. Fristrup. 2009. The costs of chronic noise exposure for terrestrial organisms. Trends in Ecology and Evolution. 25:180-189.

Barkaszi, M.J., M. Butler, R. Compton, A. Unietis, and B. Bennet. 2012. Seismic survey mitigation measures and marine mammal observer reports. U.S. Department of the Interior, Bureau of Ocean Energy Management, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study BOEM 2012-015.

Becker, E.A., K.A. Forney, M.C. Ferguson, D.G. Foley, R.C. Smith, J. Barlow, J.V. Redfern. 2010. Comparing California Current cetacean-habitat models developed using in situ and remotely sensed sea surface temperature data. Marine Ecology Progress Series. 413:163–183.

Blackstock, S.A., J.O. Fayton, P.H. Hulton, T.E. Moll, K.K. Jenkins, S. Kotecki, E. Henderson, S. Rider, C. Martin, and V. Bowman. 2017. Quantifying acoustic impacts on marine mammals and sea turtles: Methods and analytical approach for Phase III training and testing. NUWC-NPT Technical Report 12,242.

Buckland, S.T., D.R. Anderson, K.P. Burnham, J.L. Laake, D.L. Borchers, and L. Thomas. 2001. Introduction to Distance Sampling. Oxford University Press: Oxford, England.

Clark, C.W., W.T. Ellison, B.L. Southall, L. Hatch, S.M. Van Parijs, A. Frankel, and D. Ponirakis. 2009. Acoustic masking in marine ecosystems: Intuitions, analysis, and implication. Marine Ecology Progress Series. 395:201-222.

Commander, United States Pacific Fleet (Commander). 2012. Request for letters of authorization for the incidental harassment of marine mammals resulting from U.S. navy training and testing activities in the Hawaii-Southern California training and testing study area. Retrieved from https://www.fisheries.noaa.gov/action/incidental-take-authorization-us-navy-hawaii-southern-california-training-and-testing-hstt-0 on July 16, 2018.

Elhilali, M., J. Xiang, S.A. Shamma, and J.Z. Simon. 2009. Interaction between attention and bottom-up saliency mediates the representation of foreground and background in an auditory scene. PLOS Biology. 7(6):e1000129. doi:10.1371/journal.pbio.1000129.

Erbe, C., C. Reichmuth, K. Cunningham, K. Lucke, and R. Dooling. 2016. Communication masking in marine mammals: A review and research strategy. Marine Pollution Bulletin. 103:15-38.

Forney, K.A., M.C. Ferguson, E.A. Becker, P.C. Fiedler, J.V. Redfern, J. Barlow, I.L. Vilchis, L.T. Balance. 2012. Habitat-based spatial models of cetacean density in the eastern Pacific Ocean. Endangered Species Research. 16:113–133.

Hayes, S.A., E. Josephson, K. Maze-Foley, and P.E. Rosel (eds.). 2017. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments – 2016. NOAA Technical Memorandum NMFS-NE-241.

Matthews, M-N.R., H. Frouin-Mouy, and D. Zeddies. 2017. Cumulative and chronic effects in the Gulf of Mexico: Addendum: Estimating reduction of listening area and communication space due to seismic activities in support of the BOEM Geological and geophysical activities draft programmatic environmental impact statement. JASCO

Document #01510, Version 1.0. Technical report by JASCO Applied Sciences for National Marine Fisheries Service.

Matthews, M-N.R., A. Schlesinger, and D. Hannay. 2015. Cumulative and chronic effects in the Gulf of Mexico: Estimating reduction of listening area and communication space due to seismic activities in support of the BOEM Geological and geophysical activities draft programmatic environmental impact statement. JASCO Document #01073, Version 1.0. Technical report by JASCO Applied Sciences for National Marine Fisheries Service.

Maze-Foley, K. and K. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distribtuions, group sizes and interspecific associations. Journal of Cetacean Research and Management. 8:203-213.

Mullin, K.D., and G.L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico, 1996-2001. Marine Mammal Science 20(4):787–807.

Mullin, K.D., W. Hoggard, C.L. Roden, R.R. Lohoefener, C.M. Rogers, and B. Taggart. 1994. Cetaceans on the upper continental slope in the north-central Gulf of Mexico. Fishery Bulletin. 92:773-786.

National Academy of Sciences (NAS). 2017. Approaches to Understanding the Cumulative Effects of Stressors on Marine Mammals. Washington DC:The National Academies Press.

National Oceanic and Atmospheric Administration (NOAA). 2016. Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing: Underwater acoustic thresholds for onset of permanent and temporary threshold shifts. NOAA Technical Memorandum NMFS-OPR-55.

National Oceanic and Atmospheric Administration (NOAA) Office of Science and Technology (OST). n.d. Analyzing passive acoustic recordings from potential Bryde's whale habitat and characterizing ocean noise in the Northern Gulf of Mexico. Retrieved from https://www.st.nmfs.noaa.gov/protected-species-science/Research-and-Development/projects/Acoustics/project-example-4 on August 20, 2018.

New, L.F., J.S. Clark, D.P. Costa, E. Fleishman, M.A. Hindell, T. Klanjšček, D. Lusseau, S. Kraus, C.R. McMahon, P.W. Robinson, R.S. Schick, L.K. Schwarz, S.E. Simmons, L. Thomas, P. Tyack, and J. Harwood. Using short-term measures of behaviour to estimate long-term fitness of southern elephant seals. Marine Ecology Progress Series. 496:99-108.

Polk, C. 2018. Trophic Interactions and Habitat Requirements of Gulf of Mexico Bryde´s Whales. Retrieved from https://restoreactscienceprogram.noaa.gov/projects/brydes-whales on August 20, 2018.

Reeves, R.R., J.N. Lund, T.D. Smith, and E.A. Josephson. 2011. Insights from whaling logbooks on whales, dolphins, and whaling in the Gulf of Mexico. Gulf of Mexico Science. 29:41-67.

Rice, A.N., K.J. Palmer, J.T. Tielens, C.A. Muirhead, and C.W. Clark. 2014. Potential Bryde's whale (*Balaenoptera edeni*) calls recorded in the northern Gulf of Mexico. The Journal of the Acoustical Society of America, 135(5):3066-3076.

Roberts, J.J., B.D. Best, L. Mannocci, E. Fujioka, P.N. Halpin, D.L. Palka, L.P. Garrison, K.D. Mullin, T.V.N. Cole, C.B. Khan, W.M. McLellan, D.A. Pabst, and G.G. Lockhart. 2016. Habitat-based cetacean density models for the U.S. Atlantic and Gulf of Mexico. Scientific Reports 6: 22615. doi: 10.1038/srep22615.

Širović A, H.R. Bassett, S.C. Johnson, S.M. Wiggins, and J.A. Hildebrand. 2014. Bryde's whale calls recorded in the Gulf of Mexico. Marine Mammal Science. 30(1):399-409.

Soldevilla, M.S., J.A. Hildebrand, K.E. Frasier, L. Aichinger Dias, A. Martinez, K.D. Mullin, P.E. Rosel, and L.P. Garrison. 2017. Spatial distribution and dive behavior of Gulf of Mexico Bryde's whales: potential risk of vessel strikes and fisheries interactions. Endangered Species Research. 32:533-550.

Southall, B., D. Tollit, C. Clark, and W. Ellison. 2017. Application of an adapted, relativistic risk assessment framework to evaluate modeled marine mammal noise exposures resulting from Gulf of Mexico OCS proposed geological and geophysical activities (programmatic DEIS): Final draft report. Prepared for Bureau of Ocean Energy Management: 26.

Urick, R J. 1983. Principles of Underwater Sound. New York:McGraw-Hill, Inc.

Verfuss, U., D. Gillespie, J. Gordon, T.A. Marques, B. Miller, R. Plunkett, J.A.Theriault, D.J. Tollit, D.P.Zitterbart, P. Hubert, and L. Thomas. 2018. Comparing methods suitable for monitoring marine mammals in low visibility conditions during seismic surveys. Marine Pollution Bulletin 126 (2018), 1-18.

Wiggins, S.M., J.M. Hall, B.J. Thayre, and J.A. Hildebrand. 2016. Gulf of Mexico low-frequency ocean soundscape impacted by airguns. The Journal of the Acoustical Society of America. 140(1):176-183.

Würsig, B., S.K. Lynn, T.A. Jefferson, and K.D. Mullin. 1998. Behaviour of cetaceans in the northern Gulf of Mexico relative to survey ships and aircraft. Aquatic Mammals. 24(1):41–50.

Zeddies, D.G., M. Zykov, H. Yurk, T. Deveau, L. Bailey, I. Gaboury, R. Racca, D. Hannay, and S. Carr. 2015. Acoustic Propagation and Marine Mammal Exposure Modeling of Geological and Geophysical Sources in the Gulf of Mexico: 2016–2025 Annual Acoustic Exposure Estimates for Marine Mammals. JASCO Document 00976, Version 2.0. Technical report by JASCO Applied Sciences for Bureau of Ocean Energy Management (BOEM).

Ziegwied, A.T., V. Dobbin, S. Dyer, C. Pierpoint and N. Sidorovskaia, 2016. Using Autonomous Surface Vehicles for Passive Acoustic Monitoring (PAM). OCEANS 2016 MTS/IEEE Monterey, CA. pp. 1-5. doi: 10.1109/OCEANS.2016.7761380.

APPENDIX E

International Association
IAGC
of Geophysical Contractors

energy API

August 25, 2017

Dr. Walter Cruickshank
Acting Director
Bureau of Ocean Energy Management
1849 C Street, NW
Washington, D.C. 20240

Mr. Chris Oliver
Assistant Administrator
NOAA Fisheries
1315 East-West Highway
Silver Spring, MD 20910

Re: Modeling Analysis for Final Programmatic Environmental Impact Statement for Geological & Geophysical Activities on Gulf of Mexico Outer Continental Shelf

Dear Dr. Cruickshank and Mr. Oliver:

The International Association of Geophysical Contractors and the American Petroleum Institute (the "Associations") respectfully provide the enclosed report, titled "Gulf of Mexico Acoustic Exposure Model Variable Analysis" ("Model Analysis"), for your consideration. We request that the Bureau of Ocean Energy Management ("BOEM") include the Model Analysis in its administrative record for the forthcoming record of decision related to BOEM's Programmatic Environmental Impact Statement evaluating the potential environmental effects of geological and geophysical activities on the Gulf of Mexico Outer Continental Shelf ("PEIS"). We also request that the National Marine Fisheries Service ("NMFS") include the Model Analysis in its administrative record for its Marine Mammal Protection Act ("MMPA") Section 101(a)(5)(a) rulemaking for the Gulf of Mexico. Below, we provide some important context for the Model Analysis.

As explained in our comments on the Draft PEIS, the Associations are very concerned with the repeated application of precautionary assumptions across many variables within the model that was used for the PEIS to estimate marine mammal exposures to certain sound levels. Models are tools, but it is important to remember that models are simplifications of the real world and the parameters of a model are assumptions made by the decision-maker(s). It is the assumptions that lead to overestimates or underestimates of the results. By design, a multivariate model incorporates numerous variables to produce a single predicted result. When "precautionary" values are used for each of those variables—instead of the best available or most likely (*e.g.*, mean or median) values—and the uncertainty, or error (*i.e.*, standard deviation), is not adequately quantified, the predicted outcome from the multivariate model can be inflated by significant orders of magnitude larger than a result based upon the input of the most likely or best available values for each variable. In certain cases, such as marine mammal take modeling, this can be on order of thousands or millions higher. The reason for this phenomenon is that the variables are multiplied within the model and when each variable is given a seemingly innocuous "precautionary" value not supported by proper analysis of variance or error, the multiplicative effect of compounding all those variables produces an extraordinarily unrealistic result.

To illustrate this problem, the Associations requested and received permission from BOEM and NMFS to engage the same contractor that performed the modeling for the PEIS (JASCO Applied Sciences) to run that same model, with the same data, but with alterations to four variables. The alternate values used for these four variables were chosen to attempt to reflect the central or most likely tendency for each value, based upon the best available information or practice. The four altered variables are described as follows:

- Sound Source Size. In the Draft PEIS, an artificial sound source was applied to all surveys, roughly comparable to the largest sound source used in the Gulf of Mexico (8,000 cubic inches). In contrast, the Model Analysis assumes an array of 4,130 cubic inches—a survey sound source used frequently in the Gulf that is near the mean or median size range of arrays used in the Gulf over the past decade. This single change results in a four-fold decrease in exposure estimates. *See* Model Analysis at Tables 15-16 and Appendix B.

- Population Density. The Draft PEIS applies a novel method for estimating animal distribution and abundance (Roberts et al. 2016).[1] The approach used in Roberts et al. (2016) ("Roberts Model") is new and untested, and differs significantly from the official, MMPA-required population data produced by NMFS (NOAA Stock Assessment Reports or "SARs," http://www.nmfs.noaa.gov/pr/sars/species.htm). For some species, SAR values and Roberts Model values have little difference, but for other species, the Roberts Model predicts abundance estimates 8, 16, or even 30 times greater than the SAR estimates. The Roberts Model abundance estimate was smaller than the SAR estimate for only one species. Appendix H of the Model Analysis provides a detailed explanation of how the intermediate values were generated for the analysis. The impact of a slight alteration of animal density data is a decrease in takes of less than 50% by itself, but when combined with the other changes, the more central estimates of population parameters contribute to a much larger reduction to the total take estimates, as illustrated by Tables 15-19 in the Model Analysis.

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[1] Roberts J.J., B.D. Best, L. Mannocci, E. Fujioka, P.N. Halpin, D.L. Palka, L.P. Garrison, K.D. Mullin, T.V.N. Cole, C.B. Khan, W.A. McLellan, D.A. Pabst, G.G. Lockhart. 2016. Habitat-based cetacean density models for the U.S. Atlantic and Gulf of Mexico. Nature: Scientific Reports: 6:22615 | DOI: 10.1038/srep22615. www.nature.com/scientific reports/.

- Aversion. In the Draft PEIS, behavioral avoidance or movement away from the source was acknowledged to be a well-documented and significant factor influencing the number of potential "takes," especially "Level A" takes (as defined under the MMPA). In essence, the animals avoid coming within the zone containing sound levels that may cause Level A take. Avoidance or "aversion" is a well-documented phenomenon across many, if not all, marine mammal species. However, the PEIS modeling did not account for aversion. The Model Analysis includes a relatively slight degree of aversion—a few degrees deviation from course for a few seconds (*see* Model Analysis, Appendix F). Even incorporating a conservatively small amount of aversion results in a predicted reduction of Level A exposures of 40-80%. Stronger aversion that is more consistent with research studies and observer data would further reduce the estimated Level A exposures.

- Mitigation. Although visual and acoustic monitoring and mitigation measures have been required of industry vessels for decades, the Draft PEIS models give zero value to the benefits of these monitoring and mitigation measures. However, mitigation effectiveness likely varies by species and observing conditions, from as low as 5-10% at times to close to 100% for certain species and observing conditions. The Model Analysis includes a modest set of species-dependent mitigation factors (Model Analysis, Section 4.5, Tables 18-19). This has a straightforward impact on reducing predicted takes that scales to the assumed probability of observers detecting the animals, but which, we reiterate, interacts in a multiplicative manner with the other variables to create the highly inflated totals seen in the PEIS.

A fifth variable, the risk threshold criteria, was re-modeled by JASCO under contract to NMFS. This variable has been included along with the four variables selected by IAGC and API, with permission from NMFS, and is consistent with the points made by the other four changed variables: that small movements toward best available science have a greater impact on final model outcome than might be expected from the relatively small change to a single variable, through the multiplicative interactions with the other variables. We note, however, that the NOAA 2016 criteria, while a significant improvement over the criteria used in the Draft PEIS, still contain precautionary assumptions above and beyond the best available science.

We provide the Model Analysis solely to illustrate the substantial overestimation that can result from compounding precautionary assumptions in a multivariate model and to provide quantitative support for the qualitative comments we provided on the Draft PEIS.[2] It is not the structure of the model that is necessarily problematic, but it is the precautionary assumptions allocated to particular variables in the model by BOEM and NMFS that are problematic. The evaluation of alterations to only four of these variables sufficiently demonstrates the significant consequences of redundantly applied precaution in a complex multivariate model. As shown in

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[2] *See* Letter from the Associations to Dr. Jill Lewandowski, dated November 29, 2016.

the Model Analysis, these alterations produce marine mammal exposure estimates that are substantially lower than what are predicted by the model used for the PEIS.

The alternative values used for the Modeling Analysis do not reflect a position by industry about what is or should be considered the best available or most likely values for given variables. Rather, our intent is to demonstrate the importance of having a more thorough and inclusive expert discussion about what are the best available or most likely values for the different variables used in the PEIS model. Additionally, the Model Analysis should not be interpreted as the Associations' agreement with the model generally or a belief that the re-modeled results are indicative of actual effects. For example, we believe the re-modeled results presented in the Model Analysis still substantially overestimate the number of potential "Level B" exposures due to, among other factors, precautionary conservatism applied to the values used for Level B thresholds. Finally, we reiterate that the Model Analysis does not address all of the beneficial effects of mitigation, including benefits that may be qualitatively analyzed. We continue to believe, based upon many years of supporting experience and data, that mitigation measures substantially reduce, if not eliminate, potential takes.

We appreciate your consideration of the Model Analysis and respectfully invite further discussion on this issue. We will contact each of you to schedule a meeting so that we may discuss the Model Analysis in more detail and answer any questions that you or your respective colleagues may have.

Sincerely,

Nikki C. Martin

Nikki Martin
International Association of Geophysical Contractors
President

Andy Radford

Andy Radford
American Petroleum Institute
Sr. Policy Advisor – Offshore

Attachment

cc: David Bernhardt, Deputy Interior Secretary
Kate MacGregor, Deputy Assistant Secretary for Land and Minerals
Vincent DeVito, Counselor to the Secretary for Energy Policy
Chairman Rob Bishop, House Committee on Natural Resources
Chairman John Thune, Senate Commerce Committee
Chairman Lisa Murkowski, Senate Energy and Natural Resources Committee

Gulf of Mexico Acoustic Exposure Model Variable Analysis

Submitted to:
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Director, Marine Environmental Science/Biology
International Association of Geophysical Contractors
and
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Disclaimer:

The results presented herein are relevant within the specific context described in this report. They could be misinterpreted if not considered in the light of all the information contained in this report. Accordingly, if information from this report is used in documents released to the public or to regulatory bodies, such documents must clearly cite the original report, which shall be made readily available to the recipients in integral and unedited form.

Executive Summary

The International Association of Geophysical Contractors (IAGC) and the American Petroleum Institute (API), representing their member companies, are interested in better understanding the effect that various acoustic model parameters or inputs have on the outputs used to estimate numbers of animals exposed to threshold levels of sound from geophysical sources used in the Gulf of Mexico (GoM). JASCO conducted acoustic modeling for the 2016 GoM Outer Continental Shelf Proposed Geological and Geophysical (G&G) Activities Draft Programmatic Environmental Impact Statement (PEIS). One output of the models used in the PEIS work is an estimate of the number of potential animal exposures to a pre-determined acoustic threshold. A number of parameters were used in the model to calculate this estimate for the PEIS.

For this analysis, JASCO was tasked with adjusting several parameters to test their impact on model outcomes, and comparing these outcomes to those found in the PEIS. This comparison provides insight into the relative importance of several variables, individually and in combination, as influencers on model outputs. The parameters discussed in the analysis include:

- seismic sound source array size (including total volume, number of array elements, element air pressure, array geometry and spacing) used in source and propagation models,
- acoustic threshold criteria and associated weighting used to calculate exposures,
- animal densities used for adjusting simulated computer model exposures to potential real-world animal exposures,
- natural aversive behaviors of marine mammals, and
- the addition of mitigative measures that lessen the potential for animals' exposure to threshold levels of seismic sound.

The models and processes used in this analysis are the same, or comparable to those used in the modeling effort for the PEIS. This ensures that comparisons are relevant and meaningful for those parameters tested. The adjusted parameters used in this study for comparison with work completed as part of the PEIS are summarized in the table below.

Parameter	BOEM GOM G&G PEIS		IAGC/API GoM Model Analysis
	Draft PEIS	Final PEIS	
Airgun array volume	8000 in^3	8000 in^3	4130 in^3
Acoustic criteria: injury	180 dB rms SPL re 1 µPa	NOAA Technical Guidance (NMFS 2016)	NOAA Technical Guidance (NMFS 2016)
Acoustic criteria: behavior	160 dB rms SPL re 1 µPa	160 dB rms SPL re 1 µPa	Wood et al. (2012) step function
Frequency weighting	unweighted	Injury: NOAA Technical Guidance (NMFS 2016) Behavior: unweighted	Injury: NOAA Technical Guidance (NMFS 2016) Behavior: Type I (Southall et al. 2007)
Animal density source	PEIS (Roberts et al. 2016a)	PEIS (Roberts et al. 2016a)	PEIS (Roberts et al. 2016a) & Alternate Densities
Animal aversion	not included	not included	included
Mitigation	not applied	not applied	evaluated

For most species, assessment using NOAA's Technical Guidance (NMFS 2016) leads to a substantial decrease in predicted injurious exposures compared to the Draft PEIS. The exception is high-frequency species whose predicted injury rates remain about the same. The Technical Guidance was not available when the Draft PEIS was completed, but injurious exposure estimates using the Technical Guidance are included in the Final PEIS. Exposure estimates from the Final PEIS modeling were used as the baseline values to understand the effects of adjusting the parameters shown in the table.

The reduction in array volume, inclusion of aversion, and use of alternate densities that were introduced in consultation with IAGC, lowered injurious and behavioral exposure estimates for all species. Use of a smaller airgun array volume with lower source level creates a smaller ensonified area resulting in fewer numbers of animals expected to exceed exposure thresholds. Programming simulated animals to avoid loud sounds reduces the number of injurious exposures, though the magnitude of the effect is variable because of statistical variability in re-running the simulations. Use of alternate density estimates changes the exposure rate by the same proportion as the change in the density estimate. Mitigation procedures could further reduce the potential for injury roughly in proportion to the rate at which animals are detected within an exclusion zone.

Contents

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Tables

1. Introduction

The International Association of Geophysical Contractors (IAGC) and the American Petroleum Institute (API), representing their member companies, are interested in better understanding the effect that various acoustic model parameters or inputs have on the outputs used to estimate numbers of animals exposed to threshold levels of sound from geophysical sources used in the Gulf of Mexico (GoM). JASCO conducted acoustic modeling for the 2016 GoM Outer Continental Shelf Proposed Geological and Geophysical (G&G) Activities Draft Programmatic Environmental Impact Statement (PEIS; BOEM 2016). One output of the models used in the PEIS work is an estimate of the number of potential animal exposures to a pre-determined acoustic threshold. A number of parameters were used in the model to calculate this estimate for the PEIS.

For this analysis, JASCO was tasked with adjusting several parameters to test their impact on model outcomes, individually and in combination, and comparing these outcomes to those found in the PEIS. This comparison provides insight into the relative importance of several parameters as influencers on acoustic model outputs. The parameters discussed in the analysis include:

- Seismic sound source array size (including total volume, number of array elements, element air pressure, array geometry and spacing) used in source and propagation models,
- Acoustic threshold criteria and associated weighting used to calculate exposures,
- Animal densities used for adjusting simulated computer model exposures to potential real-world animal exposures,
- Natural aversive behaviors of marine mammals, and
- The addition of mitigative measures that lessen the potential for animals’ exposure to threshold levels of seismic sound.

The models and processes used in this analysis are the same, or comparable to those used in the modeling effort for the PEIS. This ensures that comparisons are relevant and meaningful for those variables tested. Both the PEIS and this analysis also use the same time period, which provides estimates of the annual potential marine mammal acoustic exposure from geological and geophysical exploration sound source activity in the GoM for years 2016 to 2025. Exposure estimates are computed from modeled sound levels received by simulated animals (animats). Because animals and noise sources move relative to the environment and each other, and the sound fields generated by the sound sources are shaped by various physical parameters, the sound levels received by an animal are a complex function of location and time. Acoustic models are used to compute three-dimensional (3-D) sound fields that vary with time. The simulated realistic movements of animats within these fields sample the sound levels in a manner representing how real animals would experience this sound. From the time history of the received sound levels, the number of animats exposed to levels exceeding threshold criteria are determined and then adjusted by the number of animals in the area to estimate the potential number of real animals likely to receive the pre-determined sound levels.

In this analysis, the GoM is divided into seven modeling zones, with four (4) survey types simulated within each zone used to estimate the potential exposures from each survey. The results from each zone were summed to provide Gulf-wide estimates of the potential number of animals exposed to threshold levels of sound capable of causing injurious effects or behavioral disturbance for each marine mammal species, survey type, and year, based on specific assumed levels of survey activities.

2. Project Description and Methods

The Draft GOM PEIS modeling to estimate potential marine mammal exposures to levels of sound capable of causing injury or behavioral disturbance was conducted prior to the release of the final NOAA Technical Guidance (NMFS 2016). Potential injury (Level A) from acoustic exposure in the Draft PEIS was therefore calculated using a National Marine Fisheries Service (NMFS) criteria with a threshold of 180 dB rms SPL (re 1 µPa) (HESS 1999). Modeling for the PEIS used an array volume of 8000 in^3 as the sound source for seismic surveys, and did not include animal aversions to loud sounds or mitigation procedures. Marine mammal density estimates used in the PEIS were the newly-available habitat-based estimates from Duke University's Marine Geospatial Ecology Laboratory (MGEL) (referenced as PEIS densities hereafter) model (Roberts et al. 2016b).

The objective of this study is to assess the level of influence several variables have on predicted, potential animal exposures, which are a key output of acoustic exposure models. To do this, source, propagation and acoustic exposure models were run using inputs provided by IAGC and API. These inputs are then compared to those modeled for the PEIS. Table 1 provides descriptions of model input assumptions used in this analysis and the PEIS.

Table 1. Summary of model inputs used for comparison and analysis of variable influence on predicted potential animal exposures.

Parameter	BOEM GOM G&G PEIS		IAGC/API GoM Model Analysis
	Draft PEIS	Final PEIS	
Airgun array volume	8000 in^3	8000 in^3	4130 in^3
Acoustic criteria: injury	180 dB rms SPL re 1 µPa	NOAA Technical Guidance (NMFS 2016)	NOAA Technical Guidance (NMFS 2016)
Acoustic criteria: behavior	160 dB rms SPL re 1 µPa	160 dB rms SPL re 1 µPa	Wood et al. (2012) step function
Frequency weighting	unweighted	Injury: NOAA Technical Guidance (NMFS 2016) Behavior: unweighted	Injury: NOAA Technical Guidance (NMFS 2016) Behavior: Type I (Southall et al. 2007)
Animal density source	PEIS (Roberts et al. 2016a)	PEIS (Roberts et al. 2016a)	PEIS (Roberts et al. 2016a) & Alternate density
Animal aversion	not included	not included	included
Mitigation	not applied	not applied	included

An overview of potential reduction of injurious exposures when mitigation procedures are employed will also be addressed.

2.1. Survey Locations

2.1.1. Choice of zone boundaries

The size and shape of acoustic footprints from exploration surveys in the Gulf of Mexico are influenced by many parameters, but the strongest influencers are water depth and seabed slope. We divided the project area into three main bathymetric areas Shelf, Slope, and Deep. The Shelf extends from shore to 100–200 m depths, where bathymetric relief is gradual; water depths on the continental shelf off Florida's eastern coast are less than 200 m deep out to ~ 150 km from shore. The Slope starts at the Shelf's outer boundary and extends into deeper water where the seabed relief is steeper and water deepens from 100–

200 m to 1500–2500 m over as little as 50 km horizontal distance. The Slope ends at the Deep area, where, although water depths are more consistent than in the other areas, depths can vary from 2000–3300 m. The subdivision depth definitions are Shelf: 0–200 m, Slope 200–2000 m, and Deep: > 2000 m.

For this analysis, and to maintain consistency with the PEIS, the Gulf was divided into 7 zones: 3 Shelf zones, 3 Slope zones, and 1 Deep zone [see Section 7.2.3 of Appendix D in Volume II of the Draft PEIS (BOEM 2016) for more detail]. These divisions are based on the physical properties of the area and the distribution of its marine inhabitants, The southern edge of the Deep zone is defined by the U.S. Exclusive Economic Zone (EEZ) boundary. The zones boundaries were defined by the 200 and 2000 m depth contours and the east-west boundary lines of BOEM's Planning Areas (except for the Deep zone 7, which included portions of all three Planning Areas). The seven modeling zones, labelled "zones" are shown in Figure 1 along with the seven representative simulation locations (numbered rectangles) discussed below.

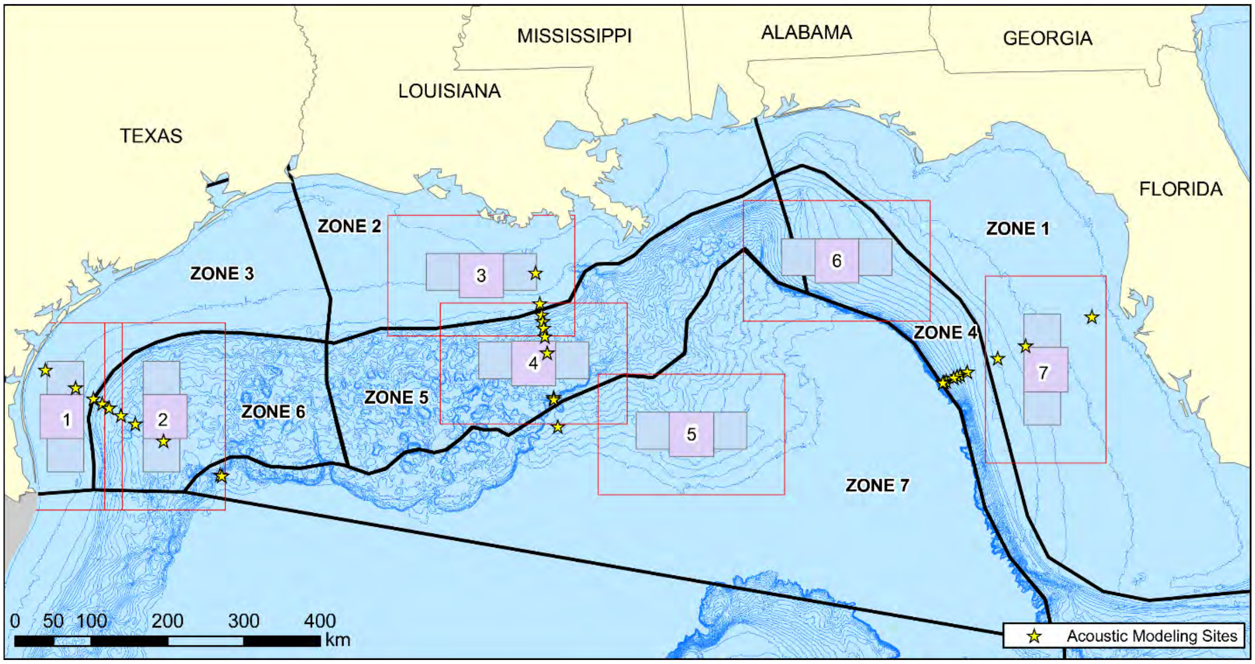


Figure 1. Gulf of Mexico project area. Black lines delineate the zones. Large, red rectangular boxes show the animal simulation extents for seismic surveys. Gray rectangles are the survey area extents for the 2-D and 3-D surveys. Pink squares are the survey extents of coil surveys. Yellow stars show the acoustic modeling sites are along West, Central, and East transects.

2.1.1.1. Survey and simulation locations

Within each of the seven zones, representative survey locations were defined (filled rectangles in Figure 1) for four different survey types described in Section 2.2. During the simulations, the source is moved within these rectangles. The sound produced ensonifies an area larger than the survey rectangle, so the extent of the corresponding animat simulation extents (red boxes in Figure 1) are larger. The animat simulation areas are determined by first finding the range to the lowest sound level which could result in disturbance, or 50 km, (whichever is smaller), and setting a buffer around the survey area of at least this range.

2.1.1.2. Acoustic Modeling Sites

As the acoustic energy from a source propagates, it is subject to a number of marine acoustic effects that depend on the ocean and bottom environment. We selected a set of 30 sites to calculate acoustic propagation loss grids as functions of source, range from the source, azimuth from the source, and receiver depth. We then used these grids as inputs to the acoustic exposure model. The 30 modeling sites (yellow stars in Figure 1) were grouped into three transects—Western, Central, and Eastern. Even though these 30 modeling sites were not all located within the survey extents (boxes) discussed in the previous section, and Boxes 5 and 6 do not contain any individual modeling sites, the environmental parameters and acoustic propagation conditions represented by these 30 modeling sites were chosen to be representative of the prevalent acoustic propagation conditions within the survey extents (boxes). (See Section 7.2.3.2 of Appendix D in Volume II of the Draft PEIS (BOEM 2016) for more detail.)

2.2. Survey Types

Four types of surveys that were included in the PEIS were also modeled for this analysis. These include 2-D, 3-D narrow azimuth (NAZ), 3-D wide azimuth (WAZ), and Coil. Each survey type is described below.

2.2.1. 2-D seismic survey

The 2-D seismic survey is performed with a single vessel towing a single seismic array. The lateral spacing of the production lines is consistent with that modeled in the PEIS, at 4.8 km (Figure 2). The production lines were generated using racetrack infill method, skipping two tracks on the left side turn (15 km wide turn) and transitioning onto the adjacent line on the right side turn (5 km wide turn). Seven days of survey were simulated. The vessel speed was 4.5 kts (2.3 m/s). The shot interval was 21.6 s (50 m). The total length of the simulated track was ~ 1400 km. The number of simulated pulses was ~ 28,000. Constant towing azimuth, parallel to the long side of the survey box, was modeled for all shots.

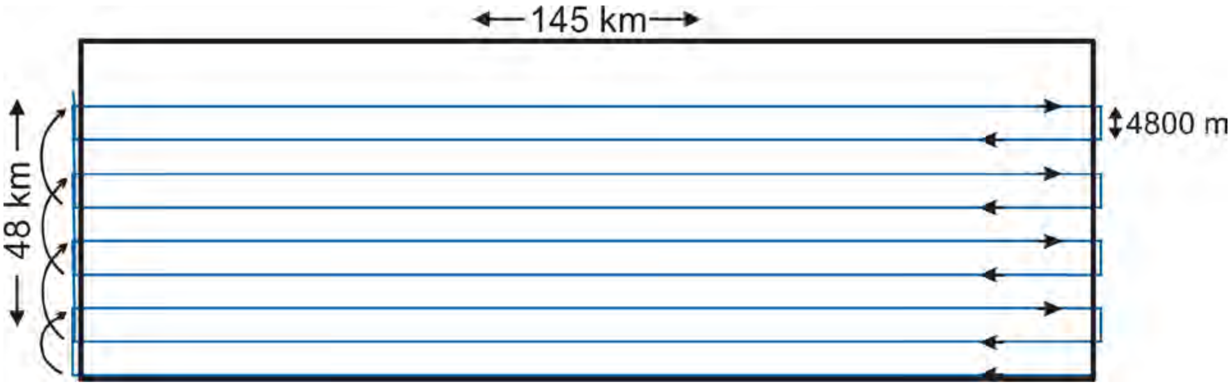


Figure 2. Simulated portion of the track for the 2-D seismic survey.

2.2.2. 3-D narrow azimuth seismic survey

3-D NAZ seismic surveys can be performed with one or two vessels towing two identical seismic source arrays. The source array towed by the same vessel is operated in a flip-flop mode, i.e., for each shot position only one of the two arrays produces a seismic pulse. In the two-vessel option, sources at each vessel produce seismic pulses simultaneously. The two-vessel option was simulated for this analysis. Both vessels follow the same track, separated along the track by 6,000 m. The production lines were laterally spaced by 1 km (Figure 3). The production lines were generated using a racetrack infill-in method with eight loops in each racetrack (7–8 km wide turn). Forty-nine lines were required to fully cover the survey area. The 7-day simulation covered ~ 20% of the complete survey. The vessel speed was 4.9 kts (2.5 m/s). The shot interval was 15 s (37.5 m) for each vessel. The total length of the simulated track is ~ 1500 km, with ~ 80,000 simulated pulses.

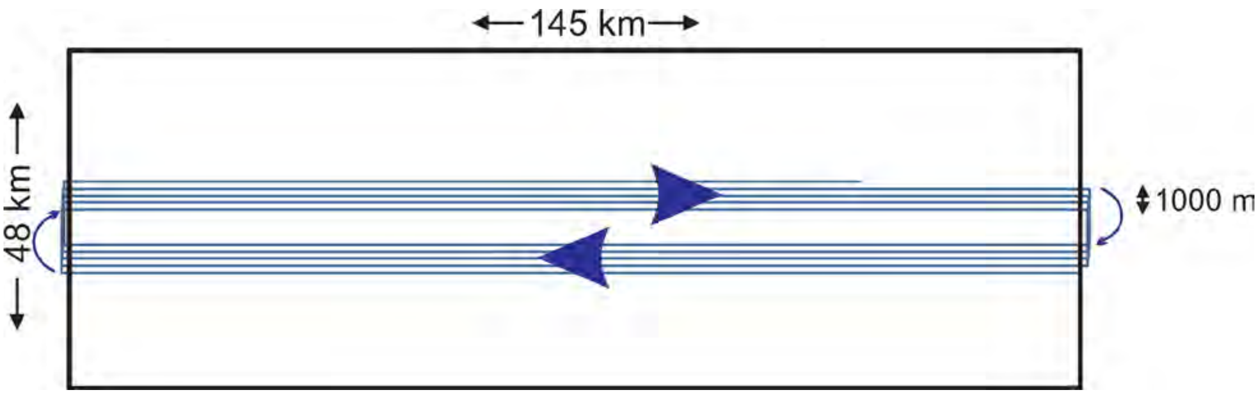


Figure 3. Simulated portion of the track for the 3-D NAZ seismic survey.

2.2.3. 3-D wide azimuth seismic survey

The 3-D WAZ seismic survey was performed with multiple vessels traveling along parallel tracks with some lateral and along the track offsets. The four-vessel option with seismic sources firing sequentially is simulated. The tracks of each vessel have the same geometry with a 1,200 m lateral offset. The vessels also have a 500 m offset along the track. The lateral spacing of the same vessel's production lines is 4.8 km and 1.2 km for the group (Figure 4). The production lines were generated with a racetrack infill method with two loops in each racetrack (9.6 km wide turn). Forty lines are required to fully cover the survey area with the vessel moving at 4.5 kts (2.3 m/s). The 7-day simulation covered ~ 85% of the complete survey. The shot interval was set to 86.4 s (200 m) for each vessel or 21.6 for the group. The total length of the simulated track is ~ 1400 km, with ~ 28,000 simulated pulses.

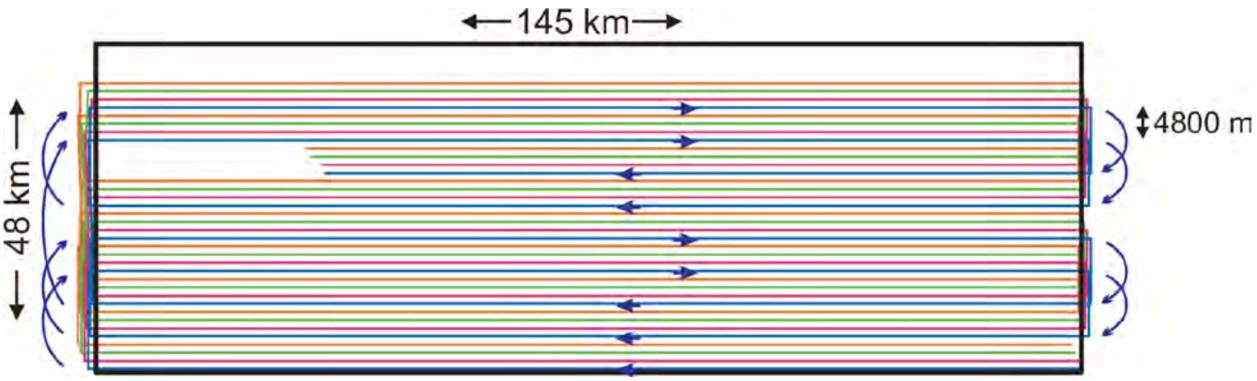


Figure 4. Simulated portion of the track for the 3-D WAZ seismic survey.

2.2.4. Coil seismic survey

The coil seismic survey modeled in both the PEIS and this analysis, is performed by multiple vessels that sail a series of circular tracks with some angular separation while towing sources. The four-vessel option was simulated assuming simultaneous sourcing around a track consisting of a series of circles with 12.5 km diameter (Figure 5). Once the vessel completes a full circle, it advances to the next one along a tangential connection segment. The offset between the center of one circle and the next, either along-swath or between swaths, is 5 km. The full survey geometry consists of two tracks with identical configuration with 1,200 m and 600 m offsets along X and Y directions, respectively. Two of the four vessels follow the first track with 180° separation; the other two vessels follow the second track with 180° separation relative to each other and 90° separation relative to the first pair. One hundred circles per vessel pair were required to fully cover the survey area. The 7-day simulation covered ~ 30% of the complete survey. The vessel speed was 4.9 kts (2.5 m/s). The shot interval was 20 s (50 m) for each vessel. The total length of the simulated track is ~ 1,500 km, with ~ 120,000 simulated pulses.

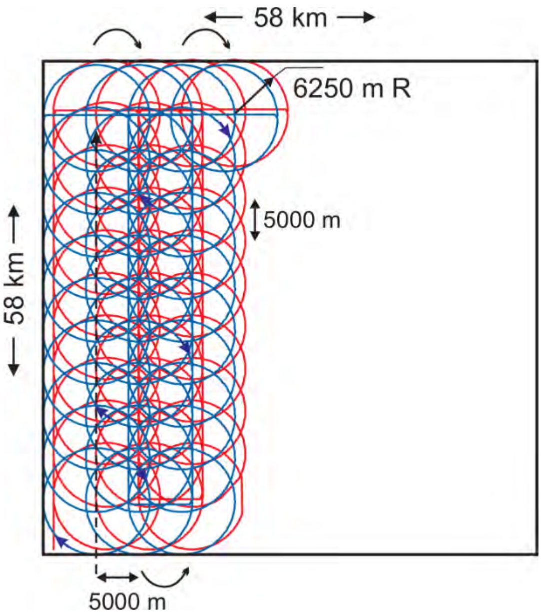


Figure 5. Simulated portion of the track for the coil seismic survey.

2.3. Acoustic Analysis Methods

Acoustic analysis methods used in this study are the same, or similar to those used in the modeling completed for the Draft PEIS, with only model inputs adjusted for comparison as shown in Table 1. To estimate potential direct effects (e.g., injury, behavioral disturbance) to marine life within the sound fields produced by the 4130 in^3 source array in various types of surveys, JASCO performed the following modeling and analysis procedures:

1. Modeled the spectral and temporal characteristics of the sound output from the proposed seismic source using the Airgun Array Source Model (AASM). Model set-up and initialization data for the 4130 in^3 airgun array configuration was provided by IAGC.
2. Acoustic propagation modeling using the Marine Operations Noise Model (MONM) that combines the outputs of the source model with the spatial and temporal environmental context (e.g., location, oceanographic conditions, seabed type) to estimate sound fields (converted to exposure radii for monitoring and mitigation). The lower frequency bands were modeled using MONM-RAM, which is based on the parabolic equation method of acoustic propagation modeling, and the higher

frequencies were modeled using MONM-Bellhop, which is a Gaussian-beam ray-theoretic acoustic propagation model.

3. Integrated the estimated sound fields with species-typical behavioral parameters (e.g., dive patterns, aversion), to estimate received sound levels for the animals that may occur in the operational area using the JASCO Animal Simulation Model Including Noise Exposure (JASMINE).
4. Estimated the number of potential injurious and behavioral level exposures based on pre-defined acoustic thresholds/criteria (NMFS 2016) and density estimates provided by IAGC and API.

Details of the acoustic analysis are provided in Appendix B and Appendix F.

2.3.1. Sound source and sound propagation

Seismic airguns generate pulsed acoustic energy by releasing into the water highly compressed air, which forms air bubbles that undergo a damped volume oscillation and emit an acoustic pressure wave that follows the bubble's oscillating internal pressure. Seismic airguns produce sounds primarily at frequencies from a few hertz to a few kilohertz, but also produce lower level sounds at higher frequencies. Larger airguns with larger internal air volume, produce higher broadband sound levels with sound energy spectrum shifted toward the lower frequencies. Single airguns or multiple airguns arranged in a spatial pattern (referred to as an airgun array) are typically towed by a survey vessel, with shots or impulses generated every 5 to 30 s along survey track lines.

A single airgun produces an approximately omnidirectional sound field, with the acoustic energy initially emitted equally in all directions. The sound signal then reflects from the water's surface and interacts with sounds that travel directly from the airgun. The result of this interaction is that, on average, more sound energy is focused downwardly than horizontally, an effect that is more prominent for lower frequencies. Larger seismic surveys usually use multiple airguns arranged in arrays, with most of the airguns in a horizontal plane. This configuration, combined with the effect of the surface reflection, focuses more sound energy downward, while emitting lower levels of sound horizontally. Airgun arrays generally show significant horizontal directionality patterns due to the phase delay between pulses from horizontally separated lines of airguns.

Sound propagates unevenly through water as it radiates away from the acoustic source due to source characteristics, and variation in area-specific environmental parameters such as water temperature and density (affecting sound velocity), and bottom type and bathymetry. The source characteristics and environmental parameters are all considered in the propagation model. The propagation model is described in Appendix C and the environmental parameters detailed in Appendix D.

For this project a seismic source array with a 4130 in^3 volume was used as the sound source. The source levels and directivity pattern calculations are shown Appendix B.1. The results of the source and propagation model for this array volume are compared to the larger source array (8000 in^3) model results included in the Draft PEIS (Section 6.3.1.1 of Appendix D in Volume II of the Draft PEIS (BOEM 2016)).

2.3.2. Animal movement and exposure modeling

The JASCO Animal Simulation Model Including Noise Exposure (JASMINE) was used to predict the exposure of animats (virtual marine mammals) to sound arising from the surveys. Sound exposure models like JASMINE integrate the predicted sound field with biologically meaningful movement rules for each marine mammal species that result in an exposure history for each animat in the model. Inside JASMINE, the sound source mimics the proposed survey pattern (as described above). As shown in Figure 6, animats are programmed to behave like the marine animals that may be present in the survey area. The parameters used for forecasting realistic behaviors (e.g., diving, foraging, aversion, surface times etc.) are determined and interpreted from marine species studies (e.g., tagging studies) where available, or reasonably extrapolated from related species (see Appendix F for a more detailed explanation of JASMINE and the parameters used in modeling marine mammal movement).

Individual animat's sound exposure levels are summed over the total simulation duration or a shorter time period, such as 24 hours, to determine its total received energy. The maximum exposure sound pressure level during the time period is also determined from the exposure history, and both total energy received and maximum pressure are compared to the pre-determined thresholds (Section 2.4).

The Marine Mammal Movement and Behavior (3MB) model (Houser 2006) was used in the modeling for the PEIS (Section 5.3 of Appendix D in Volume II of the Draft PEIS (BOEM 2016)). JASMINE was used for this study so that behavioral aversion could be included. JASMINE was written by JASCO and is based on the 3MB model. The performance of JASMINE and 3MB are the same except that JASMINE allows for animats to change behavioral states in response to specified received levels, which is necessary for implementation of behavioral aversion (see below).

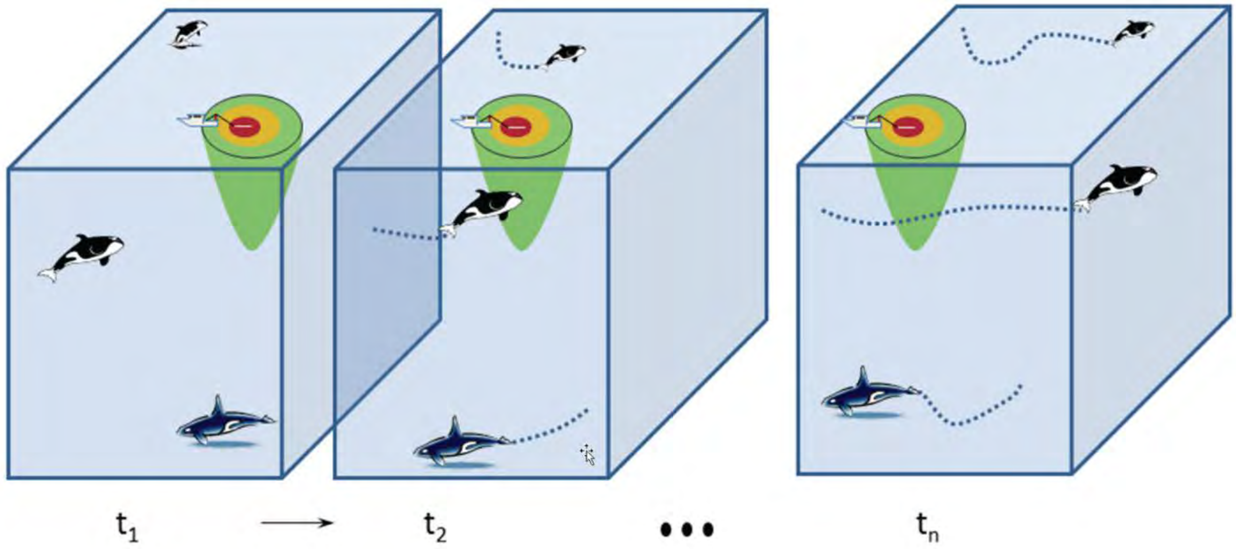


Figure 6. Cartoon of animats in a moving sound field. The acoustic exposure of each animat is determined by where it is in the sound field, and its exposure history is accumulated as the simulation steps through time. In this cartoon the vessel and sound source are moving from right to left, as is the deepest animat. The two upper animats move from left to right. Because the upper and lower animats are far from the source, low levels of sound exposure are expected. The middle animat is nearer the sound source, so its acoustic exposure is expected to be higher than the other two animats, and its highest exposure occurs closest to the sound source at the second time step (t_2).

2.3.2.1. Aversion

Aversion is a common response of animals to sound, particularly at relatively high sound exposure levels (Ellison et al. 2012). As received sound level generally decreases with distance from a source, this aspect of natural behavior can strongly influence the estimated maximum sound levels an animal is predicted to receive and significantly affects the probability of more pronounced direct or subsequent behavioral effects. As part of the revised analysis approach recommended by Southall et al. (2016) aversion parameters to sound level were implemented for all selected acoustic criteria. A scaled aversion response function was created, with the magnitude and probability of an aversion response increasing with increased received sound levels. At the end of each time step, each animat "evaluates" its received sound level and applies the aversion rules. At a given received level, there is a specified probability that an aversion would occur for a specified duration and corresponding course change away from the source. Details of the aversion approach used in JASMINE are provided in Appendix F.1.4. Aversion rules applied in simulation models assume that all animals respond the same way to pre-determined sound levels. Behavioral response of animals is extremely variable (see Southall et al. 2007) and aversion behavior is insufficiently documented in most species.

2.4. Details of Acoustic Criteria Used in this Analysis

To assess the potential impacts, it is necessary to first establish exposure criteria for which sound levels may be expected to have a negative impact on animals. In 2016, after the publication of the Draft PEIS, NOAA issued a Technical Guidance document that provides acoustic thresholds for onset of PTS and TTS in marine mammal hearing for all sound sources (NMFS 2016). NOAA also provided guidance on the use of weighting functions when applying injury criteria. The NOAA Guidance recommends the use of a dual criteria for assessing injurious exposures, including a peak, unweighted sound pressure level metric ($\mathrm{SPL_{pk}}$) and a cumulative sound exposure level (SEL) metric with frequency weighting. Both acoustic criteria and weighting function application are specified by hearing group.

2.4.1. Marine mammal hearing groups

Current data and predictions indicate that not all marine mammal species have equal hearing capabilities, either in absolute hearing sensitivity or frequency band of hearing (Richardson et al. 1995, Wartzok and Ketten 1999, Southall et al. 2007, Au and Hastings 2008). While hearing measurements are available for a small number of species based on captive animal studies, direct measurements of many odontocetes and all mysticetes do not exist. As a result, hearing ranges for many odontocetes are grouped with similar species, and predictions for mysticetes are based on other methods including: anatomical studies and modeling (Houser et al. 2001, Parks et al. 2007, Tubelli et al. 2012, Cranford and Krysl 2015); vocalizations (see reviews in Richardson et al. 1995, Wartzok and Ketten 1999, Au and Hastings 2008); taxonomy; and behavioral responses to sound (Dahlheim and Ljungblad 1990, see review in Reichmuth et al. 2007) In 2007, Southall et al. proposed that marine mammals be divided into hearing groups. This division was updated in 2016 by NMFS using more recent best available science (Table 2).

Table 2. Marine mammal hearing groups (NMFS 2016).

Hearing group	Generalized hearing range*
Low-frequency (LF) cetaceans (mysticetes or baleen whales)	7 Hz to 35 kHz
Mid-frequency (MF) cetaceans (odontocetes: delphinids, beaked whales)	150 Hz to 160 kHz
High-frequency (HF) cetaceans (odontocetes)	275 Hz to 160 kHz

*The generalized hearing range for all species within a group. Individual hearing will vary.

2.4.2. Marine mammal weighting functions

The potential for anthropogenic sounds to impact marine mammals is largely dependent on whether the sound occurs at frequencies that an animal can hear well, unless the sound pressure level is so high that it can cause physical tissue damage regardless of frequency. Auditory (frequency) weighting functions reflect an animal's ability to hear a sound. Sound spectra are weighted at particular frequencies in a manner that reflects an animal's sensitivity to those frequencies (Nedwell and Turnpenny 1998, Nedwell et al. 2007). Auditory weighting functions have been proposed for marine mammals, specifically associated with PTS acoustic thresholds expressed in metrics that consider what is known about marine mammal hearing (e.g., SEL) (Southall et al. 2007, Erbe et al. 2016, Finneran 2016). Marine mammal auditory weighting functions published by Finneran (2016) are included in the NMFS 2016 Technical Guidance for use in conjunction with corresponding SEL PTS (injury) onset acoustic criteria (Table 3). The auditory weighting functions used in this study are described in Appendix E.

The application of marine mammal auditory weighting functions emphasizes the importance of making measurements and characterizing sound sources in terms of their overlap with biologically-important frequencies (e.g., frequencies used for environmental awareness, communication or the detection of predators or prey), and not only the frequencies of interest or concern for the completion of the sound-producing activity (i.e., context of sound source; NMFS 2016).

2.4.3. Injury exposure criteria

Loud and/or sustained sounds may injure the hearing apparatus of animals, resulting in a permanent shift in hearing thresholds. There are no published data on the sound levels that cause PTS in marine mammals. There are data that indicate the received sound levels at which TTS occurs, and PTS onset is typically extrapolated from TTS onset and growth. NMFS 2016 criteria incorporate best available science that indicates injury (PTS) in marine mammals is correlated with both sound exposure level (SEL) that accumulates over time, or very loud, instantaneous peak pressure levels. These dual threshold criteria of SEL and peak SPL are used to calculate marine mammal exposures (Table 3).

Table 3. Summary of relevant PTS and TTS onset acoustic thresholds (NMFS 2016) used in this analysis

Hearing group	PTS onset thresholds* (received level)		TTS onset thresholds* (received level)	
	Impulsive	Non-impulsive	Impulsive	Non-impulsive
Low-frequency (LF) cetaceans	SPL_{pk}, flat: 219 dB SEL_{LF}, 24h: 183 dB	SEL_{LF}, 24h: 199 dB	SPL_{pk}, flat: 213 dB SEL_{LF}, 24h: 168 dB	SEL_{LF}, 24h: 179 dB
Mid-frequency (MF) cetaceans	SPL_{pk}, flat: 230 dB SEL_{MF}, 24h: 185 dB	SEL_{MF}, 24h: 198 dB	SPL_{pk}, flat: 224 dB SEL_{MF}, 24h: 170 dB	SEL_{MF}, 24h: 178 dB
High-frequency (HF) cetaceans	SPL_{pk}, flat: 202 dB SEL_{HF}, 24h: 155 dB	SEL_{HF}, 24h: 173 dB	SPL_{pk}, flat: 196 dB SEL_{HF}, 24h: 140 dB	SEL_{HF}, 24h: 153 dB

* Dual metric acoustic thresholds for impulsive sounds: Use whichever results in the largest isopleth for calculating PTS onset. If a non-impulsive sound has the potential of exceeding the peak sound pressure level thresholds associated with impulsive sounds, these thresholds should also be considered.
SPL_{pk}, flat–peak sound pressure is flat weighted or unweighted and has a reference value of 1 µPa
SEL - denotes cumulative sound exposure over a 24-hour period and has a reference value of 1 $\mu\text{Pa}^2\text{s}$
The subscript associated with cumulative sound exposure level thresholds indicates the designated marine mammal auditory weighting.

2.4.4. Behavioral exposure criteria

Numerous studies on behavioral response have not resulted in consensus in the scientific community on the appropriate sound exposure metric for assessing behavioral reactions, and it is recognized that many variables other than received sound level affect the nature and extent of responses to a stimulus (Southall et al. 2007, Ellison and Frankel 2012). Because of the complexity and variability of marine mammal behavioral responses to acoustic exposure, NMFS has not yet released technical guidance on behavior thresholds for use in calculating animal exposures (NMFS 2016). Based on observations of mysticetes (Malme et al. 1983, Malme et al. 1984, Richardson et al. 1986, Richardson et al. 1990), the NMFS currently uses SPL thresholds for behavioral response of 160 dB re 1 µPa for impulsive sounds and 120 dB re 1 µPa for non-impulsive sounds for all marine mammal species (NMFS 2016). It was noted in early workshops that behavioral responses to sound may occur at lower levels, but significant responses were most likely to occur above an rms SPL of 140 dB re 1 µPa (HESS 1999). An extensive review of behavioral responses to sound was undertaken by Southall et al. (2007, their Appendix B), who found varying responses for most marine mammals between an rms SPL of 140 and 180 dB re 1 µPa, consistent with the HESS (1999) report, but lack of convergence in the data prevented them from suggesting explicit thresholds. Absence of controls, precise measurements, appropriate metrics, and context dependency of responses (including the activity state of the animal) all contribute to variability.

In 2012, Wood et al. proposed a graded probability of response for impulsive sounds using a frequency weighted rms SPL metric. Wood et al. (2012) also designated behavioral response categories for sensitive species (including harbor porpoise and beaked whales) and for migrating mysticetes. For this analysis, the Wood et al. (2012) criteria is used to assess behavioral response to impulsive sounds (Table 4).

Table 4. Behavioral exposure criteria used in this analysis (porpoise and migrating mysticetes are not present in the GoM so are excluded from the table). Probability of behavioral response frequency-weighted sound pressure level (rms SPL dB re 1 µPa). Probabilities are not additive. Adapted from Wood et al. (2012).

Marine mammal group	Probability of response to frequency-weighted rms SPL (dB re 1 µPa)			
	120	140	160	180
Beaked whales and porpoises	50%	90%		
All other species		10%	50%	90%

2.5. Species that May be Present in the Survey Area

Of the approximately 125 species of known marine mammals, 32 cetaceans and one sirenian species are thought to occur in the Gulf of Mexico (Wursig et al. 2000, Jefferson et al. 2008). Seven of the cetacean species are baleen whales (mysticetes) and 25 are toothed whales (odontocetes). Of the seven mysticete species, only the Bryde's whale is resident in the GoM, but its observed range is in the De Soto Canyon area, over 300 km from the proposed survey area. The other six mysticetes, the North Atlantic right whales, and the humpback, minke, sei, fin, and blue whales, are all considered rare or extralimital strays in the GoM. Four of the odontocetes are considered extralimital or rare visitors in the Gulf of Mexico: Sowerby's beaked whales, the long-finned pilot whales, the long-beaked common dolphins, and short-beaked common dolphins (Davis and Fargion 1996, Jefferson and Schiro 1997, Davis et al. 2000). Species that are rare, or are unlikely to occur in the GoM, are not considered further in the environmental analysis. The low frequency Bryde's whales are included in the analysis because the calculated range for behavioral response is larger than that of mid-or high-frequency species.

The one sirenian species present in the Northern GoM is the endangered West Indian Manatee (subspecies Florida manatee, *Trichechus manatus latirostris*). The species occurs mainly along the peninsular Florida coast and southeastern Georgia coasts in the winter and migrates to the North and East during summer. Migration routes and destinations are largely unknown (Pabody et al. 2009). The West Indian manatee is most common in warm, shallow waters of rivers, bays, estuaries, and coastal areas where their primary food source of aquatic plants is abundant (Gannon et al. 2007). A few individuals have been observed in deeper water and as far west as the Texas coast, but these sightings are considered extralimital (Fertl et al. 2005, Pabody et al. 2009). Because manatees are considered rare or absent from the survey areas, they are not included in this analysis.

There are currently no pinniped (sea lions, seals, and walruses) or fissiped (sea otters and polar bears) species known to inhabit the GoM. The Caribbean monk seal (*Monachus tropicalis*) has been extinct since the early 1950s; the last verified sighting in the GoM was made in 1932 (Wursig et al. 2000). There have been no reported sightings of the introduced California sea lion (*Zalophus californianus*) since 1972 (Jefferson et al. 1992, Wursig et al. 2000).

Marine mammal species resident in the GoM are shown in Table 5.

Table 5. Summary of marine mammal species considered in the acoustic exposure analysis.

Species of interest		Hearing group	Estimated auditory bandwidth[1]	Area population status[2]	GoM habitat distribution
Common name	Latin binomial				
Bryde's whales	*Balaenoptera brydei/edeni*	LFC	20–900 Hz	Uncommon	Non-migratory population resident in Northern GoM, especially De Soto Canyon (Schmidly 1981, Leatherwood and Reeves 1983)
Atlantic spotted dolphins	*Stenella frontalis*	MFC	0.1–160 kHz	Common	Occur in coastal and oceanic waters from 40° S to 40° N (Perrin and Hohn 1994, Perrin and Gilpatrick 1994).
Beaked whales[3]					
Blainville's	*Mesoplodon densirostris*	MFC	5–80 kHz	Rare	Occur in Northern GoM, particularly on shelf break (Hildebrand et al. 2015).
Cuvier's	*Ziphius cavirostris*	MFC		Rare	
Gervais'	*Mesoplodon europaeus*	MFC		Uncommon	
Bottlenose dolphins	*Tursiops truncatus*	MFC	150 Hz to 135 kHz	Common	Most widespread and common cetacean species in coastal waters of the GoM. Two genetically distinct geographic varieties (ecotypes) of bottlenose dolphins are known to occur in the GoM: a "coastal" ecotype and an "offshore" ecotype (Hersh and Duffield 1990, LeDuc and Curry 1998).
Clymene dolphins	*Stenella clymene*	MFC	0.1–160 kHz	Common	Occur in coastal and oceanic waters from 40° S to 40° N (Perrin and Hohn 1994, Perrin and Gilpatrick 1994).
False killer whales	*Pseudorca crassidens*	MFC	<1–115 kHz	Uncommon	Sightings of this species in the northern Gulf of Mexico are in oceanic waters, primarily in the eastern Gulf of Mexico (Mullin and Fulling 2004, Maze-Foley and Mullin 2006).
Fraser's dolphins	*Lagenodelphis hosei*	MFC	6.6–23.5 kHz	Rare	Sightings in the northern Gulf of Mexico recorded in all seasons in water depths > 200 m (656 ft) (Leatherwood et al. 1993, Hansen et al. 1996, Mullin and Hoggard 2000, Maze-Foley and Mullin 2006).
Killer whales	*Orcinus orca*	MFC	<500 Hz to 120 kHz	Uncommon	Sightings of killer whales in the northern Gulf of Mexico between 1921 and 1995 occurred primarily in oceanic waters ranging from 840 to 8,700 ft (256 to 2,652 m) (averaging 4,075 ft (1,242 m)), primarily in the North-central region (O'Sullivan and Mullin 1997). Very few killer whales in the Gulf of Mexico have been sighted on the continental shelf.
Melon-headed whales	*Peponocephala electra*	MFC	8–40 kHz	Common	Occur in water depths > 2,625 ft (800 m) and usually west of Mobile Bay, Alabama (Mullin et al. 1994, Mullin and Fulling 2004, Maze-Foley and Mullin 2006).
Pantropical spotted dolphins	*Stenella attenuatus*	MFC	0.1–160 kHz	Common	Found in coastal and oceanic waters from 40° S to 40° N (Perrin and Hohn 1994, Perrin and Gilpatrick 1994).

Species of interest		Hearing group	Estimated auditory bandwidth[1]	Area population status[2]	GoM habitat distribution
Common name	Latin binomial				
Pygmy killer whales	*Feresa attentuata*	MFC	70–85 kHz	Uncommon	Historic sightings of these animals in the northern GoM are in oceanic waters (Mullin and Fulling 2004, Maze-Foley and Mullin 2006).
Risso's dolphins	*Grampus griseus*	MFC	4–80 kHz	Common	Occur throughout oceanic waters of the northern GoM but are concentrated in areas near the continental slope (Baumgartner 1997, Maze-Foley and Mullin 2006).
Rough-toothed dolphins	*Steno bredanesis*	MFC	0.1–200 kHz	Common	Occur in oceanic, and to a lesser extent continental shelf, waters (Fulling et al. 2003, Mullin and Fulling 2004, Maze-Foley and Mullin 2006).
Short-finned pilot whales	*Globicephala macrorhyncus*	MFC	11–50 kHz	Common	Primarily on the continental slope, west of 89° W longitude (Mullin and Fulling 2004, Maze-Foley and Mullin 2006).
Sperm whales	*Physeter macrocephalus*	MFC	2.5–60 kHz	Common	Population surveys indicate that sperm whales are widely distributed during all seasons in continental slope and oceanic waters, particularly along and seaward of the 3,300 ft (1,000 m) isobath and within areas of steep depth gradients (NMFS Mullin et al. 1991, 1994, Hansen et al. 1996, Jefferson and Schiro 1997, Davis et al. 1998, Mullin and Hoggard 2000, Ortega Ortiz 2002, Fulling et al. 2003, Mullin and Fulling 2004, Mullin et al. 2004, Maze-Foley and Mullin 2006, Mullin 2007, Jefferson et al. 2008, 2009).
Spinner dolphins	*Stenella longirostris*	MFC	0.1–160 kHz	Common	Occur in coastal and oceanic waters from 40° S to 40° N (Perrin and Hohn 1994, Perrin and Gilpatrick 1994).
Striped dolphins	*Stenella coeruleoalba*	MFC	0.1–160 kHz	Common	Occur in coastal and oceanic waters from 40° S to 40° N (Perrin and Hohn 1994, Perrin and Gilpatrick 1994).
Kogia spp.[3]					
Dwarf sperm whales	*Kogia sima*	HFC	90–150 kHz	Uncommon	Sightings of these species in the northern GoM are primarily in oceanic waters (Mullin et al. 1991, Mullin and Fulling 2004, Maze-Foley and Mullin 2006).
Pygmy sperm whales	*Kogia breviceps*	HFC		Uncommon	

[1] Estimates of species auditory bandwidth are from many different sources included in the report bibliography

[2] Area population status in the GoM from Wursig et al. (2000). Categories: common–abundant wherever it occurs in the region; uncommon–may or may not be widely distributed but does not occur in large numbers; rare–present in such small numbers throughout the region that it is seldom seen

[3] Species are considered cryptic meaning they are seldom observed at the surface. These species are also difficult to classify from visual observation and are therefore often grouped when estimating population size.

2.5.1. Representative species

Because of the complexity associated with modeling thirty-two cetaceans, four survey types, seven zones, with and without aversion, representative species types were selected for modeling. Exposure results (number of animats exceeding thresholds) are expected to be similar for similarly behaving animals. As a practical measure, six representative species were chosen for full analysis: Bryde's whales, *Kogia spp,* bottlenose dolphins, short-finned pilot whales, sperm whales, and Cuvier's beaked whales. These species were chosen to represent different hearing groups, varying levels of behavioral sensitivity, and general diving patterns of marine mammals in the GOM. Bryde's whales and *Kogia spp* were chosen because they are, respectively, the only low-frequency and high-frequency marine mammals resident in the GOM. The remaining representatives are all mid-frequency species. Bottlenose dolphins in the estuarine stocks are a shallow-diving nearshore species. Short-finned pilot whales represent the relatively shallow diving small pelagic species. Sperm whales are large, deep-diving, and are the only endangered species in the GOM. Cuvier's beaked whales are deep diving and classified as behaviorally sensitive by Wood et al. (2012).

2.5.2. Animal densities

Simulations are run using a constant animat density that is typically much higher than the real-world animal density (see Appendix F). To get the number of real-world animals expected to exceed a threshold the number of animats exceeding the threshold must be scaled by the ratio of the simulation (animat) density and the real world (animal) density. Marine mammal densities used in modeling for the Draft and Final PEIS were from Duke MGEL's habitat-based model for the GoM (Roberts et al. 2016a). Densities for the representative species in each zone are listed in Appendix G. To test the effects of varying the real-world density input to exposure models IAGC/API provided alternate density values for comparison. The alternate density estimates for species in each zone and an explanation of their derivation are provided in Appendix H.

2.5.2.1. Evaluation time period

Animat exposure histories were processed to calculate the number of animats exposed to levels exceeding threshold (the number of exposures). For this analysis, seven-day simulations were run and the exposures estimated in 24 h windows within the seven days. The first 24 h window begins at the start of the simulation and each subsequent window is advanced by 4 h. In this sliding-windows approach, 42 exposure estimate samples are obtained for each seven-day simulation. The mean value is then used as the 24 h exposure estimate for that survey, as was done for the PEIS modeling.

3. Results

3.1. Estimated Sound Fields – 4130 in^3 airgun array

The 4130 in^3 airgun array is modeled (Appendix B) at the 30 sites described in Section 2.1 and Appendix B to determine the single-shot sound fields used in the model simulations. For assessment of potential injury the sound fields were weighted using the functions specified by the NMFS (2016) Technical Guidance, and for aversion and potential behavioral disruption the sound fields were weighted using Type 1 weighting (Southall et al. 2007) (see Appendix E for weighting functions).

3.1.1. Per-pulse peak SPL

To evaluate the risk of acoustic injury, the range to the unweighted, zero-to-peak SPL (dB re 1µPa) is used for the various hearing groups (LF: 219 dB re 1µPa, MF: 230 dB re 1µPa, and HF: 202 dB re 1µPa). The spherical spreading law:

$$L_{pk}(R) = L_{pkSL} - 20 \cdot \log(R) \quad ,$$

where L_{pkSL} is the peak SPL source level of the source and R is the range, was assumed as the propagation model for peak SPL. The ranges to the thresholds were calculated from the peak source level for the 4130 in^3 array and, for comparison, the 8000 in^3 array (Table 6) (see Section 6.3.1.1 of Appendix D in Volume II of the Draft PEIS (BOEM 2016) for details of the 8000 in^3 array).

Table 6. Ranges to hearing group peak SPL threshold.

Source	Source level (peak SPL; dB)	Range (m)		
		LF 219 dB peak SPL	MF 230 dB peak SPL	HF 202 dB peak SPL
4130 in^3 airgun array	247.9	28	8	197
8000 in^3 airgun array	255.2	65	18	457

3.1.2. Per-pulse SEL and SPL

The 3-D per-pulse acoustic fields used as inputs for acoustic exposure analysis were also processed to provide two other products:

- Maps of the acoustic field around the sources.
- Tables of ranges to various isopleths (radii tables) for each source.

The maps and radii tables are, respectively, 2-D and 1-D projections of the 3-D sound fields, which serve as quality assurance checkpoints to verify the acoustic modeling output and control the results of the exposure simulation. Maps were created from the 3-D grid of the acoustic pressure levels by taking the maximum-over-depth value at each horizontal sampling location. The maps therefore represent the maximum received acoustic level over all depths at each location.

The ranges to isopleths in the radii tables are provided as two statistical estimates:

- The maximum range (R_{max}, in meters)
- The 95% range ($R_{95\%}$, in meters)

Given a regularly gridded spatial distribution of sound levels, the $R_{95\%}$ for a given sound level is defined as the radius of the circle, centered on the source, encompassing 95% of the grid points with sound levels at or above the given value. This definition is meaningful in terms of potential effects on animals because, regardless of the shape of the contour for a given sound level, $R_{95\%}$ is the range from the source beyond which only 5% of a uniformly distributed population would be exposed to sounds at or above that level.

The R_{max} for a given sound level is the maximum distance at which the specified received level occurs (equivalent to $R_{100\%}$). It is more conservative than $R_{95\%}$, but could be relevant for defining exclusion zones to avoid any chance of exposures above the specified level. For cases where the volume ensonified to a specific level is discontinuous and small pockets of higher received levels occur beyond the main ensonified volume (e.g., due to convergence), the R_{max} can be much larger than $R_{95\%}$.

Example modeling results of the 4130 in^3 airgun array at site CM3, located in the Central-Slope zone at 750 m water depth, are presented below as maps of unweighted, per-pulse SEL, and SPL fields (Figure 7 to Figure 10). Site CM3 results are presented as example results because that site is centrally located within the Gulf. Maps appear similar as the maximum-over-depth metrics remove fine-scale variability, between the different metrics and seasons. To the south of the source, the maxium-over-depth isopleths of 139-130 dB (SEL) and 149-140 dB (SPL) extend to the modeled extent of 50 km. The corresponding radii tables for the site are shown in Tables 7 to 10 for Seasons 1 (January to March) and 3 (July to September) in SEL and SPL metrics with all applicable M-weighted filtering (see 4.6.Appendix E for auditory weighting functions). It is important to note that these tables show one example from the 30 sites that were modeled for this study, and that these ranges are not directly used in estimating animat exposure. Ranges at other sites could differ and it is the path through the sound field that determines the animat's exposure history. In the case of the SEL metric, even though no range for a single exposure exceeds the threshold, the integration of multiple lower-level exposure could still exceed threshold.

Table 7. 4130 in³ airgun array at Site CM3, Season 1 (February): Ranges to specific threshold levels (SEL).

SEL	Unweighted		Type III M-Weighting					
			LFC		MFC		HFC	
	R_{max}	$R_{95\%}$	R_{max}	$R_{95\%}$	R_{max}	$R_{95\%}$	R_{max}	$R_{95\%}$
210	< 10	< 10						
200	20	20	< 10	< 10				
190	70	60	20	20				
185	120	110	40	30				
183	150	130	60	50				
180	220	190	80	70				
170	740	650	270	220				
160	2700	2400	900	680				
155	7000	4400	2400	1200	< 10	< 10		
150	11000	9500	3900	3200	10	10	< 10	< 10
140	38000	30000	23000	13000	80	80	40	40
130	> 50000	48000	43000	28000	260	240	120	120
120		48000	> 50000	48000	820	780	410	390
110		48000		48000	4400	3000	1300	1200

Maximum (R_{max}, m) and 95% ($R_{95\%}$, m) horizontal distance from the source to modeled broadband maximum-over-depth sound level thresholds, with and without auditory frequency weighting applied for low-frequency cetaceans (LFC), mid-frequency cetaceans (MFC), and high-frequency cetaceans (HFC).
Units: rms SPL (dB re 1 µPa^2·s).

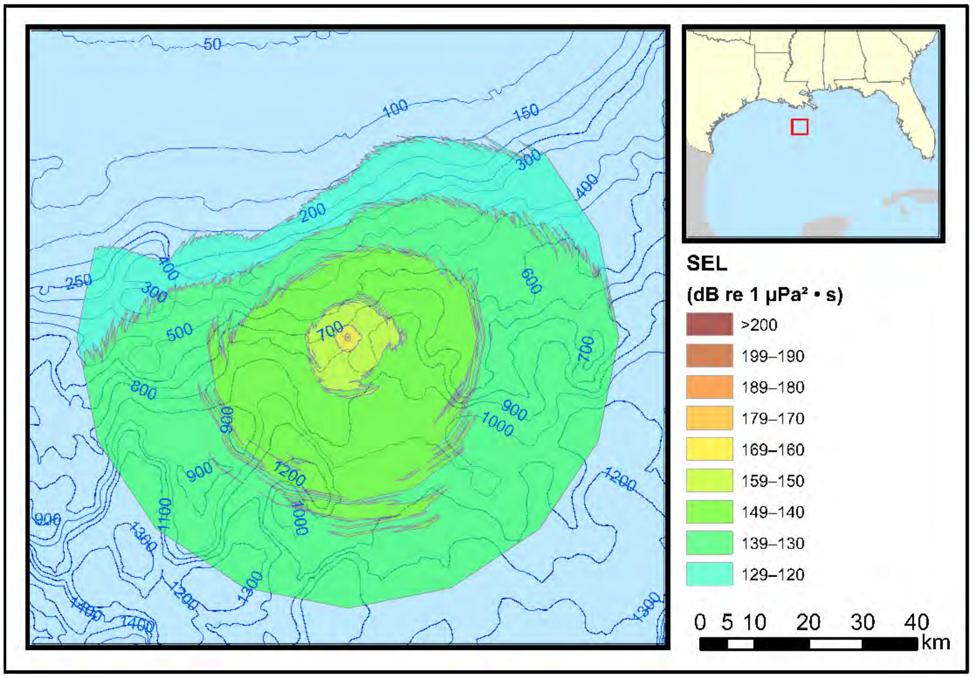


Figure 7. 4130 in³ airgun array at the Central-Slope region (Site CM3), Season 1 (February): Broadband (10–5,000 Hz) maximum-over-depth per-pulse SEL field. Blue contours indicate water depth in meters.

Table 8. 4130 in³ airgun array at Site CM3, Season 3 (September): Ranges to specific threshold levels (SEL).

SEL	Unweighted		Type III M-Weighting					
			LFC		MFC		HFC	
	R_{max}	$R_{95\%}$	R_{max}	$R_{95\%}$	R_{max}	$R_{95\%}$	R_{max}	$R_{95\%}$
210	< 10	< 10						
200	20	20	< 10	< 10				
190	70	60	20	20				
185	120	110	40	30				
183	150	130	60	50				
180	220	190	80	70				
170	730	650	260	210				
160	2700	2300	860	690				
155	6800	4200	2400	1300	< 10	< 10		
150	11000	9000	3700	3100	10	10	< 10	< 10
140	35000	29000	16000	13000	80	80	40	40
130	> 50000	47000	35000	26000	250	240	120	120
120		48000	> 50000	47000	860	820	390	360
110		48000		48000	8200	5800	1500	1300

Maximum (R_{max}, m) and 95% ($R_{95\%}$, m) horizontal distance from the source to modeled broadband maximum-over-depth sound level thresholds, with and without auditory frequency weighting applied for low-frequency cetaceans (LFC), mid-frequency cetaceans (MFC), and high-frequency cetaceans (HFC).
Units: rms SPL (dB re 1 µPa²·s).

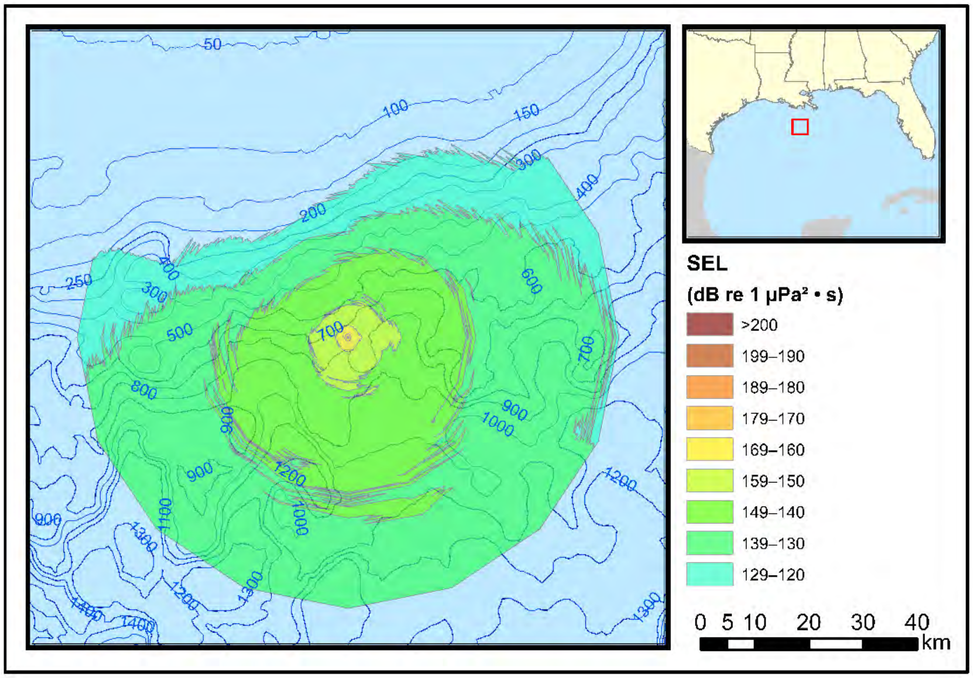


Figure 8. 4130 in³ airgun array at the Central-Slope region (Site CM3), Season 3 (September): Broadband (10–5,000 Hz) maximum-over-depth per-pulse SEL field. Blue contours indicate water depth in meters.

Table 9. 4130 in³ airgun array at Site CM3, Season 1 (February): Ranges to specific threshold levels (SPL).

rms SPL	Unweighted		Type I M-Weighting					
			LFC		MFC		HFC	
	R_{max}	$R_{95\%}$	R_{max}	$R_{95\%}$	R_{max}	$R_{95\%}$	R_{max}	$R_{95\%}$
210	20	20	20	20	< 10	< 10	< 10	< 10
200	80	70	70	60	20	20	10	10
190	240	220	230	210	70	60	60	50
180	830	620	720	570	260	210	190	150
170	2500	2200	2500	2000	770	610	600	470
160	11000	8400	11000	7800	3400	2900	2700	1100
150	34000	24000	31000	23000	14000	9100	11000	8500
140	> 50000	47000	> 50000	47000	27000	20000	26000	17000
130		48000		48000	> 50000	38000	48000	33000
120		48000		48000		48000	> 50000	48000
110		48000		48000		48000		48000

Maximum (R_{max}, m) and 95% ($R_{95\%}$, m) horizontal distance from the source to modeled broadband maximum-over-depth sound level thresholds, with and without auditory frequency weighting applied for low-frequency cetaceans (LFC), mid-frequency cetaceans (MFC), and high-frequency cetaceans (HFC). Units: rms SPL (dB re 1 µPa).

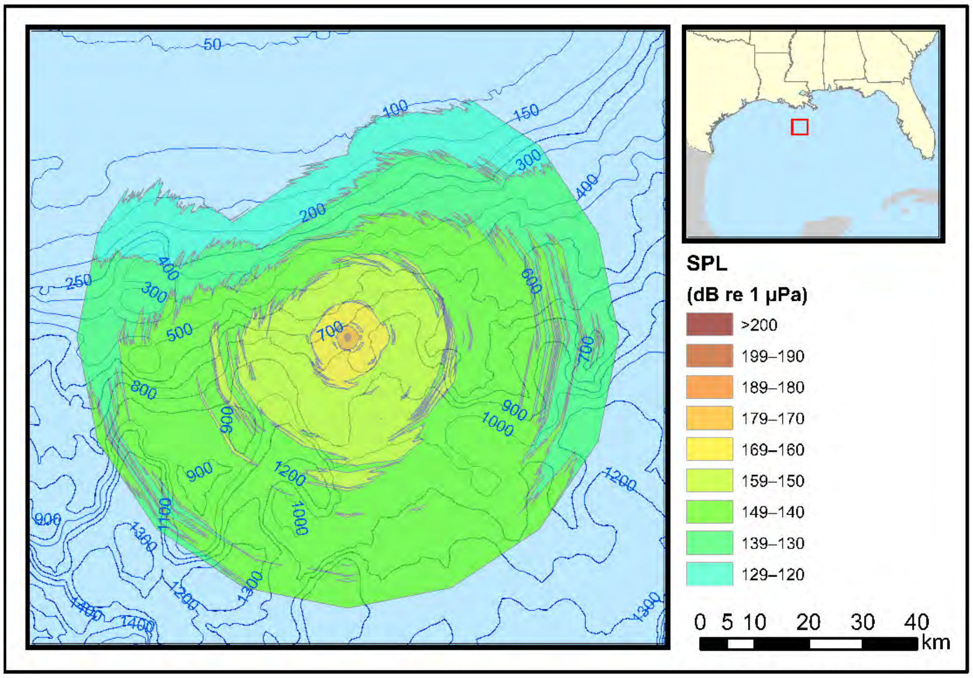


Figure 9. 4130 in³ airgun array at the Central-Slope region (Site CM3), Season 1 (February): Broadband (10–5,000 Hz) maximum-over-depth SPL field. Blue contours indicate water depth in meters.

Table 10. 4130 in³ airgun array at Site CM3, Season 3 (September): Ranges to specific threshold levels (SPL).

rms SPL	Unweighted		Type I M-Weighting					
			LFC		MFC		HFC	
	R_{max}	$R_{95\%}$	R_{max}	$R_{95\%}$	R_{max}	$R_{95\%}$	R_{max}	$R_{95\%}$
210	20	20	20	20	< 10	< 10	< 10	< 10
200	80	70	70	60	20	20	10	10
190	410	360	410	350	120	90	100	80
180	530	430	450	400	180	140	110	90
170	830	620	720	570	260	200	190	150
160	2500	2100	2500	2000	770	600	610	460
150	11000	7700	11000	7200	3300	2800	2800	1200
140	33000	23000	28000	21000	16000	9000	11000	8700
130	> 50000	46000	> 50000	46000	26000	19000	21000	16000
120		48000		48000	44000	36000	43000	31000
110		48000		48000	> 50000	48000	> 50000	48000

Maximum (R_{max}, m) and 95% ($R_{95\%}$, m) horizontal distance from the source to modeled broadband maximum-over-depth sound level thresholds, with and without auditory frequency weighting applied for low-frequency cetaceans (LFC), mid-frequency cetaceans (MFC), and high-frequency cetaceans (HFC). Units: rms SPL (dB re 1 µPa).

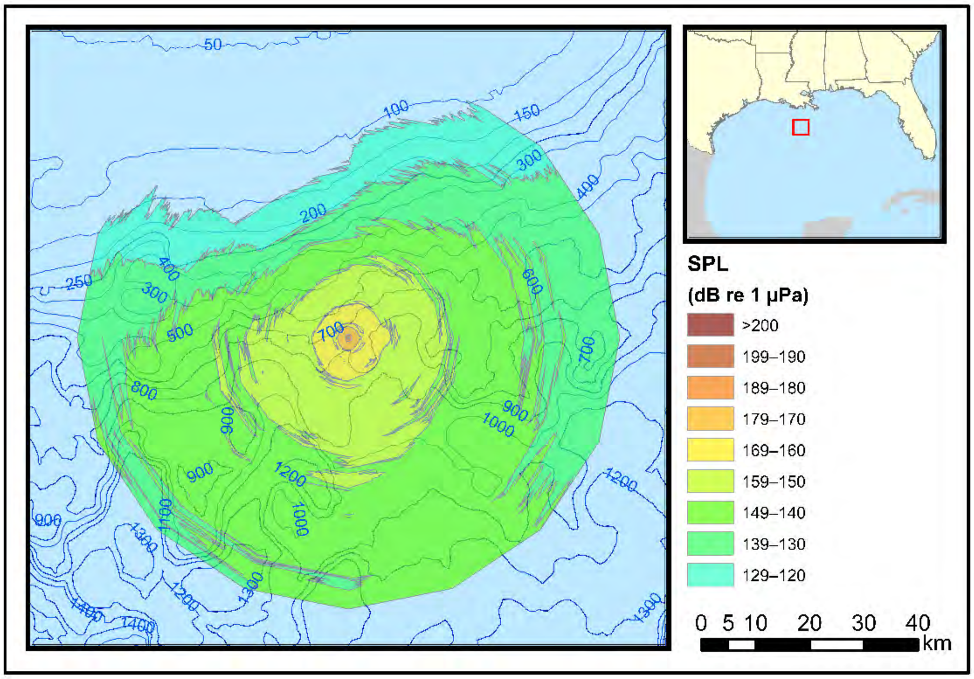


Figure 10. 4130 in³ airgun array at the Central-Slope region (Site CM3), Season 3 (September): Broadband (10–5,000 Hz) maximum-over-depth SPL field. Blue contours indicate water depth in meters.

3.2. 24-hour Exposure Estimates

Simulations were run with and without aversion. It is necessary to run separate simulations for aversion because the animats change their behavior as a function of received level in the model when aversion is included so have different trajectories (and exposure histories) than model runs where no reaction to a received level is included. Both with and without aversion, the number of animats exposed to levels exceeding the specified thresholds were determined in 24-h windows within the seven-day simulations. In a sliding-window approach, the first 24-h window begins at the start of the simulation and each subsequent window is advanced by 4 h, resulting in 42 samples for each survey. The number of individuals exposed to levels exceeding the injury and behavioral thresholds were calculated within each of the 24-h samples. SEL was determined by summing acoustic energy received from the source integrated over 24 h. Slant range was used to determine the zero-to-peak SPL for each animat relative to the source following the spherical spreading law (Section 3.1.1). The number of animats within the range (Table 6) where the received level could exceed threshold were found. The step function proposed by Wood et al. (2012) was used as a metric to evaluate potential behavioral response. The mean value from the 42 24-h estimates was used as the 24-hr exposure estimates for that survey.

Animals are only considered ‘taken’ once during a 24-hr period, and animats are not removed or replaced based on exceeding a threshold. The 24-hr reset was stipulated by BOEM and serves as a recovery mechanism and as a time basis on which survey effort could be based. When scaling up from 24 hours to longer surveys, e.g. 30 days, there is some repeated counting compared to analysis of longer-duration simulations. Overestimate by scaling occurs for single-exposure, SPL-based metrics, and is likely for SEL-based metrics as well. For SEL, the 24 hour duration limits the accumulation of energy but allows for multiple counting of an individual that exceeds threshold on multiple separate days. A fuller evaluation of this issue can be found in the DPEIS (Test Case 1 - Appendix D Section 6.5.1).

To get the real-world individual exposure estimates, the 24-h mean animat exposure estimates were scaled using the mean real-world density estimate in each zone. Two density estimates were used for scaling each representative species: (1) the Duke MGEL model used in the PEIS (Roberts et al. 2016a) (Section 2.5.1, and Tables 1–7) and (2) alternate density estimates supplied by IAGC/API for this study (Appendix H).

3.3. Annual Decade Individual Exposure Estimates

For comparison with exposure estimates from the Draft and Final PEIS, the output of this analysis are estimates of the number of exposures for each species for each year for the entire Gulf using the same methods as the PEIS with selected alternate modeling parameters (seismic array volume, behavioral aversion, alternate densities, and mitigation). Projections of survey level of effort for the different survey types for the Gulf Planning Areas (Eastern, Central, and Western; divided into shallow and deep zones) were the same as those used in the PEIS modeling and were provided by BOEM (Appendix I). Our modeling zones and survey locations were chosen, in part, to coincide with BOEM’s Planning Areas so that the survey projections could be easily used for scaling. The shallow portion of the east, central, and western Planning Areas were the same as our modeling zones 1–3. A portion of each of the deep parts of Planning Areas maps directly to our modeling zones 4–6. The remainder of the deep parts of the Planning Areas were combined as modeling zone 7. The 24-h exposure estimates were scaled by the projected number of survey days to get the annual aggregate exposure estimates. The annual individual estimates using the alternate modeling parameters for each survey type (summed for all zones) are shown in Appendix J with estimates for the two density estimates (PEIS and Alternate), both with and without behavioral aversion. Similarly, the annual individual aggregate estimates (summed for all survey types and zones) are shown in Appendix K with estimates for the two density estimates (PEIS and Alternate), with and without behavioral aversion. The decade aggregate estimates are shown in Tables 11–14.

Table 11. Individual exposure estimates over a decade for all surveys (using 4130 in^3 array volume) and zones using the PEIS densities without aversion.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	1	0	621579
Bottlenose dolphins	436	0	4647116
Bryde's whales	9	62	4103
Kogia spp.	13956	0	60986
Short-finned pilot whales	0	0	72297
Sperm whales	8	0	125607

Table 12. Individual exposure estimates over a decade for all surveys (using 4130 in^3 array volume) and zones using the PEIS densities with aversion.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	574580
Bottlenose dolphins	73	0	4542106
Bryde's whales	7	57	4061
Kogia spp.	8221	8	64238
Short-finned pilot whales	1	0	76184
Sperm whales	7	0	120018

Table 13. Individual exposure estimates over a decade for all surveys (using 4130 in^3 array volume) and zones using alternate densities without aversion.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	310261
Bottlenose dolphins	432	0	4605021
Bryde's whales	2	11	715
Kogia spp.	6963	0	30427
Short-finned pilot whales	0	0	65186
Sperm whales	4	0	62556

Table 14. Individual exposure estimates over a decade for all surveys (using 4130 in^3 array volume) and zones using alternate densities and with aversion.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	286795
Bottlenose dolphins	73	0	4500962
Bryde’s whales	1	10	708
Kogia spp.	4102	4	32050
Short-finned pilot whales	1	0	68328
Sperm whales	3	0	59772

4. Discussion

Exposure estimates for the Draft PEIS were generated by JASCO using SPL criteria with thresholds of 180 dB for potential injury and 160 dB potential behavioral disruption. These thresholds applied to all marine mammals and do not take into account the different hearing ranges of the animals. NOAA released technical guidance (NMFS 2016) for evaluating potential injury due to acoustic exposure after the Draft PEIS was completed. The exposure estimates for potential injury were then updated for NOAA by JASCO using the Technical Guidance for the Final PEIS. Exposure estimates from both the Draft PEIS and Final PEIS are shown here as baseline values (columns 1 and 2 in Tables 15-17) to evaluate the effects of alternate parameter choices on exposure estimates. The parameters investigated (including the use of the NOAA Technical Guidance) were airgun array volume, behavioral aversion, and marine mammal density estimates. Summarized comparisons of the effect alternate parameter choices have on exposure estimates are shown in this section in Tables 15-17.

Table 15. Number individual animals estimated to exceed peak SPL threshold over a decade for all surveys and zones (rounded to nearest integer). The Draft PEIS used 180 dB rms SPL as the threshold for injury, the Final PEIS uses NOAA's Technical Guidance (NMFS 2016).

Species	PEIS		4130 in^3 array, NMFS 2016			
	8000 in^3, no aversion, PEIS densities		PEIS densities		Alternate densities	
	180 dB SPL	NMFS 2016	No aversion	Aversion	No aversion	Aversion
Cuvier's beaked whales	51655	425	1	0	0	0
Bottlenose dolphins	2743723	22841	436	73	432	73
Bryde's whales	589	32	9	7	2	1
Kogia spp.	30620	29171	13956	8221	6963	4102
Short-finned pilot whales	25182	506	0	1	0	1
Sperm whales	81239	350	8	7	4	3

Table 16. Number individual animals estimated to exceed SEL threshold over a decade for all surveys and zones (rounded to nearest integer). The Draft PEIS used 180 dB rms SPL as the threshold for injury, the Final PEIS uses NOAA's Technical Guidance (NMFS 2016).

Species	PEIS		4130 in^3 array, NMFS 2016			
	8000 in^3, no aversion, PEIS densities		PEIS densities		Alternate densities	
	180 dB SPL	NMFS 2016	No aversion	Aversion	No aversion	Aversion
Cuvier's beaked whales	51655	1	0	0	0	0
Bottlenose dolphins	2743723	95	0	0	0	0
Bryde's whales	589	152	62	57	11	10
Kogia spp.	30620	108	0	8	0	4
Short-finned pilot whales	25182	0	0	0	0	0
Sperm whales	81239	0	0	0	0	0

Table 17. Number individual animals estimated to exceed behavioral threshold over a decade for all surveys and zones using (rounded to nearest integer). The Draft and Final PEIS both use 160 dB rms SPL as the threshold for behavioral disruption.

Species	PEIS		4130 in^3 array, Step function (Wood et al. 2012)			
	8000 in^3, no aversion, PEIS densities		PEIS densities		Alternate densities	
	160 dB SPL	Step function*	No aversion	Aversion	No aversion	Aversion
Cuvier's beaked whales	440986	1809109	621579	574580	310261	286795
Bottlenose dolphins	10433991	7860889	4647116	4542106	4605021	4500962
Bryde's whales	6487	5493	4103	4061	715	708
Kogia spp.	275816	127150	60986	64238	30427	32050
Short-finned pilot whales	282759	141502	72297	76184	65186	68328
Sperm whales	680502	322020	125607	120018	62556	59772

* The Draft and Final PEIS did not use the Wood et al. (2012) step function to evaluate potential behavioral disruption but the values were calculated during the modeling and are shown here to aid in comparison.

4.1. NOAA Technical Guidance for injury

For most species, adoption of NOAA's Technical Guidance (NMFS 2016) for evaluating potential injury from acoustic exposure results in a substantial reduction of injurious exposure estimates relative to the Draft PEIS (column one of Table 15 and Table 16). The Technical Guidance uses different metrics (peak SPL and SEL) than the previous criteria (rms SPL) and divides the animals into hearing groups with different threshold levels. With the peak SPL metric, mid-frequency species (beaked whales, bottlenose dolphins, short-finned pilot whales, and sperm whales) have the highest thresholds (230 dB peak SPL re 1 µPa) and the greatest reduction in estimated injurious exposure relative to the previous criteria (column two versus column one in Table 15). The threshold level for low-frequency species (Bryde's whale, 219 dB peak SPL re 1 µPa) is less than the mid-frequency species and the resulting reduction in estimated injurious exposures is less than the mid-frequency species. High-frequency species (*Kogia spp.*) have the lowest thresholds (202 dB peak SPL re 1 µPa) and little reduction in estimated injurious exposure numbers (Table 15) relative to the Draft PEIS. For the SEL metric, the sound fields are weighted for the different hearing groups and each group also has a different threshold level. Most of the acoustic energy emitted by airguns is < 500 Hz, so the auditory (frequency) weighting functions, especially for the mid- and high-frequency species, discount much of the energy. Again, the mid-frequency animals have the highest thresholds and the greatest decrease in exposure estimates (column two versus column one in Table 16). High-frequency species have the lowest thresholds and least reduction, and low-frequency species are in between (Table 16).

4.2. Seismic Sound Source Array Volume

The maximum broadband, far-field, peak source level for the 4130 in^3 array is about 7 dB less than the 8000 in^3 array, and the ranges to the injury threshold for peak SPL are about ½ of those for the 8000 in^3 array (Table 6). The expected reduction in estimated injury due to exceeding peak SPL threshold is ~8 times because the ensonified volume above threshold is reduced in proportion to the cube of the range ($2^3 = 8$). The reductions found are more than a factor of 8 for mid-frequency species but less for low- and mid-frequency species (column 2 divided by column 3 in Table 15). There are a few factors that could explain differences in the expected reduction rates and the observed reduction rates:

1. Exceedance is rare and the summaries across the zones and surveys for a decade can amplify small differences and uncertainty. Simulations where only 0, 1, or 2 animats exceed threshold have less statistical power and more uncertainty than when hundreds or thousands of animats exceed threshold

providing a better mean estimate of exceedance probability -- for example, the difference in behavioral threshold exceedance is more consistently ~1/2 when comparing the use of the 4130 in^3 array to the 8000 in^3 array (column 3 divided by column 2 in Table 17).

2. The sound field modeling resolution is in increments of 5 and 10 meters near the source, which is similar to the range to threshold for mid-frequency animals. This granularity contributes to the noise with few samples.
3. Other factors such as counting only the maximum exposure for each animat and the movement of sources and animats could also contribute to differences between expected and observed outcomes.

For the SEL metric it is difficult to estimate an a priori reduction rate because the acoustic energy is integrated. With the exception of low-frequency Bryde's whales, there are essentially no exceedances of the SEL threshold when using the smaller array (Table 16), and few or none when the larger array is used. Limited examples and granularity remain contributing factors when comparing the effects of array size, but because SEL is so rare and is less than peak SPL, SEL is proving not to be the primary consideration when evaluating the potential injurious impacts of these surveys for most species.

As mentioned above, the number of exposures above behavioral threshold is reduced to ~1/2 for the 4130 in^3 array versus the 8000 in^3 array. The ensonified volume above behavioral threshold is much larger than for injury and there are many samples above threshold. While the factor of ½ is relatively consistent, it should not necessarily be generalized. The number of animats above threshold depends on many factors from sound propagation to animal movement. Very roughly, sound levels decrease logarithmically with distance so all other factors being equal, increasing the source level by 6 dB more than doubles the volume of the ensonified area when no boundaries are present. Depth limitations (boundaries) can limit the increase in the ensonified volume (e.g., depth may be 2 km but the range to a threshold level may be > 40 km), and similarly, animals tend to sample from a limited depth range (e.g., shallow divers may only sample a relatively small portion of the water column).

4.3. Aversion

Animals may avoid loud sounds (F.1.4), and this aversion does appear to decrease the estimated number of injurious exposures (columns 4 and 6 versus columns 3 and 5 in Tables 15 and 16). Because the predicted number of animals exceeding injury thresholds (peak SPL and SEL) are already low, it is difficult to generalize about the effects of aversion on exposure rates. The same factors regarding limited number of samples and granularity (Section 4.2) apply but are compounded by our lack of knowledge in modeling aversive behaviors. For example, injury due to peak SPL exposure in *Kogia spp*. is decreased but the SEL exposure increases. It is noted that the number of peak SPL exceedances is much greater than the number of SEL exceedances, and that the increase in SEL exposures could represent a rare event with a small number of samples. It also suggests a lack of understanding in implementing aversion and highlights the potential for non-intuitive results. In this case animals may turn away from the source and receive a lower maximum exposure level but remain near the source and accumulate greater SEL. Aversive behavior, as implemented, could increase exposure because animats are programmed to ignore the received level for a short period of time and move away from the source. Because animats and sources are moving, ignoring the received level may allow the animats to remain near a source longer than if they had maintained their normal behavior. We do not know if this would occur in the real animals or not, but it is not entirely unrealistic given the natural variability in animal behaviors.

We used the step function proposed by Wood et al. (2012) to implement aversion. The step function was also used to gauge behavioral disruption, so aversion in this case is by definition a behavioral disruption, but as seen in Table 17 the number of behavioral disruptions decrease somewhat with aversion. This result occurs because the step function probability of disruption is graded. 10% of the animats receiving 140-160 dB SPL (all species except beaked whales) are counted as disruption, while 50% for 160-180 dB SPL, and 90% above 180 dB SPL are counted. An animat that receives 140-160 dB SPL and averts to avoid a higher level exposure contributes less to the overall behavioral disruption estimation.

4.4. Alternate Densities

Determining the effects of using different real-world animal density estimates on exposure calculations is straightforward compared to evaluating the effects of other variables. Real-world densities are used to scale the simulation results to obtain the number of real-world individual animals expected to exceed the thresholds. Scaling is done after the simulation and is linear — doubling the density estimate doubles the number of individuals estimated to exceed threshold. The density estimates from the PEIS and the alternate densities provided by IAGC are similar for bottlenose dolphins; IAGC used density estimates from CETMAP for Bryde's whales (~5.8 times lower than the PEIS density estimates). The densities of the rest of the representative species were halved in the IAGC parametrization relative to the PEIS density estimates. A reduction in exposure estimates by these ratios is evident in Tables 15–17 (by comparing column 3 to column 5, and column 4 to column 6).

4.5. Mitigation

In the modeling for the Draft PEIS, a study was undertaken to better understand how mitigation by shutting down the sound source when a protected species enters an exclusion zone of 500 m radius around the source, affects the number of predicted animals exceeding threshold (Section 6.5.3 of Appendix D in Volume II of the Draft PEIS (BOEM 2016). It was shown that detection probability is a primary factor in predicting mitigation effectiveness because shutdowns only occur when the animals are detected. However, detection probability depends on many factors. It is species and weather dependent, and also depends on the skill and equipment of the observer or observing system. Weather is unknown during planning phases and the detection probability varies greatly among species – sperm whales are relatively easy to detect while smaller while cryptic species such as beaked whales are much more difficult to detect. In the modeling study for the Draft PEIS, JASCO evaluated a range of detection probabilities for the same representative species. While a number of factors may contribute to effectiveness, a rough but reasonable summary is that mitigation effectiveness is approximated by the detection probability. That is, if 50% of the animals entering the exclusion zone are detected, then the number of animals exceeding injury threshold is reduced by up to one half. Mitigation effectiveness is roughly predicted by detection probability because exceeding injury threshold in these surveys is usually the result of receiving a small number of pulses close to the source rather than accumulation of energy over longer time period and area. This observation, however, depends on the source, survey design, size of exclusion zone, and is also influenced by the hearing group. Detection probability ranges explored in the Draft PEIS were: beaked whales and *Kogia spp*. 5 – 45%, and bottlenose, short-finned pilot whales, and sperm whales 50-90%, so a reduction in potential injury by up to these detection probabilities could be expected. Tables 18 and 19 respectively shows the peak SPL and SEL decade-long injury that might be expected when shut down is used as a mitigation procedure when an animal is detected within an exclusion zone of 500 m.

Table 18. Number individual animals estimated to exceed peak SPL threshold over a decade for all surveys and zones (rounded to nearest integer), with and without mitigation procedures.

Species	Probability detection range (%)	4130 in³ array, NMFS 2016, Alternate densities, Aversion			
		Mitigation			
		No Mitigation	low	mid	high
Cuvier’s beaked whales	5-45	0	0	0	0
Bottlenose dolphins	50-90	73	33	22	11
Bryde’s whales	50-90	1	0	0	0
Kogia spp.	5-45	4102	3692	3077	2461
Short-finned pilot whales	50-90	1	0	0	0
Sperm whales	50-90	3	1	1	0

Table 19. Number individual animals estimated to exceed SEL threshold over a decade for all surveys and zones (rounded to nearest integer), with and without mitigation procedures.

Species	Probability detection range (%)	4130 in³ array, NMFS 2016, Alternate densities, Aversion			
		Mitigation			
		No Mitigation	Low	Mid	High
Cuvier’s beaked whales	5-45	0	0	0	0
Bottlenose dolphins	50-90	0	0	0	0
Bryde’s whales	50-90	10	5	3	1
Kogia spp.	5-45	4	4	3	2
Short-finned pilot whales	50-90	0	0	0	0
Sperm whales	50-90	0	0	0	0

4.6. Conclusions

For most species, the greatest reduction in injurious exposure estimates relative to the Draft PEIS - but not the Final PEIS - arise from the implementation of NOAA’s Technical Guidance that was released in 2016 (NMFS 2016). Exceptions to this conclusion are high-frequency species whose predicted injury rates remain about the same. The Technical Guidance uses different acoustic metrics (peak SPL and SEL), divides the species into hearing groups with different thresholds, and weights the sound field in accordance with the hearing group for the SEL metric. The Technical Guidance was not released at the time the Draft PEIS was completed (2015), but injurious exposure estimates have since been recalculated for NOAA using the Technical Guidance and will be included in the Final PEIS. While baseline values included here are from both the Draft PEIS using the previous criteria and from the final PEIS using the Technical Guidance, it is important to note that this is for completeness in comparison only. The Final PEIS with significantly decreased estimates of injurious exposure is the best baseline to use in determining the relative influence of model parameters, the stated objective of this study.

New official guidance is not available for estimating potential behavioral disruption, but a step function proposed by Wood et al. (2012) is frequently used in project-specific exposure modeling completed for Environmental Assessments. The step function is a graded probability of response and uses frequency-weighted sound fields to account for hearing ranges of different species. While neither the Draft or Final PEIS use the step function, behavioral disruption for the modeled data from the Draft and Final PEIS was evaluated using the step function for comparison purposes. With the exception of behaviorally sensitive species, such as beaked whales, a reduction in the predicted number of behavioral disruptions was found but not to the same degree as the reduction in injurious exposures when the Technical Guidance is used. For the behaviorally-sensitive beaked whales, behavioral disruption exposures estimates increase using the Wood et al. (2012) step function relative to the unweighted 160 dB rms threshold (HESS 1999).

The parameter changes studied, namely reduction in array volume, inclusion of aversion, and use of alternate densities, reduced injurious and behavioral exposure estimates for all species to varying extents. Combining all parameters results in a cumulative reduction in exposure numbers. Use of a smaller airgun array volume with lower source level creates a smaller ensonified area resulting in fewer numbers of animals expected to exceed a given threshold. Having animals avoid loud sounds (aversion) appears to reduce the number of injurious exposures, though the magnitude of the effect was variable. This variability is likely because, when using the Technical Guidance to assess potential injury, there are few samples of injurious exposure exceedance so the statistical variability of re-running simulations is evident.

Mitigation effectiveness was assessed in the modeling for the Draft PEIS. In this study, the probability of detection rates included in the Draft PEIS were used to assess the influence of this parameter on estimates of injurious exposures. The large range in detection probability reflects the uncertainty associated with this parameter, as not only weather conditions, but also observer experience and height of observation platform can affect detections. Mitigation measures are expected to reduce the potential for injury roughly in proportion to the detection rate. This is observed in the calculations for species that are more easily detected, such as bottlenose dolphins. For cryptic species such as *Kogia spp*. and beaked whales, mitigation parameters have less influence on estimates of injurious exposures relative to other parameters such as frequency weighting and densities.

Literature Cited

[BOEM] Bureau of Ocean Energy Management. 2016. *Gulf of Mexico OCS Proposed Geological and Geophysical Activties. Western, Central, and Eastern Planning Areas. Draft Programmatic Environmental Impact Statement Volume 1: Chapters 1-8*. Prepared by CSA Ocean Sciences, Stuart, Florida. 702 pp. https://www.boem.gov/BOEM-EIS-2016-049-v1/.

[HESS] High Energy Seismic Survey. 1999. *High Energy Seismic Survey Review Process and Interim Operational Guidelines for Marine Surveys Offshore Southern California*. Prepared for the California State Lands Commission and the United States Minerals Management Service Pacific Outer Continental Shelf Region by the High Energy Seismic Survey Team, Camarillo, California. 98 pp.

[NGDC] National Geophysical Data Center. 2014. U.S. Coastal Relief Model, Vol. 3 Florida and Eastern Gulf of Mexico, Vol. 4 Central Gulf of Mexico, Vol. 5 Western Gulf of Mexico. National Geophysical Data Center, National Oceanic and Atmospheric Administration, U.S. Department of Commerce. http://www.ngdc.noaa.gov/mgg/coastal/crm.html (Accessed Dec 2014).

[NMFS] National Marine Fisheries Service. 2009. *Sperm Whale (Physeter macrocephalus). 5-Year Review: Summary and Evaluation*. National Marine Fisheries Service Office of Protected Resources, Silver Spring, MD. 44 pp.

[NMFS] National Marine Fisheries Service. 2016. *Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing: Underwater Acoustic Threshold Levels for Onset Permanent and Temporary Threshold Shifts*. U.S. Department of Commerce, NOAA. NOAA Technical Memorandum NMFS-OPR-55. 178 pp. http://www.nmfs.noaa.gov/pr/acoustics/Acoustic%20Guidance%20Files/opr-55_acoustic_guidance_tech_memo.pdf.

Aerts, L., M. Blees, S. Blackwell, C. Greene, K. Kim, D. Hannay, and M. Austin. 2008. *Marine mammal monitoring and mitigation during BP Liberty OBC seismic survey in Foggy Island Bay, Beaufort Sea, July-August 2008: 90-day report*. Document Number LGL Report P1011-1. Report by LGL Alaska Research Associates Inc., LGL Ltd., Greeneridge Sciences Inc. and JASCO Applied Sciences for BP Exploration Alaska. 199 pp. http://www.nmfs.noaa.gov/pr/pdfs/permits/bp_liberty_monitoring.pdf.

Aguilar Soto, N., M.P. Johnson, P.T. Madsen, F. Diaz, I. Dominguez, A. Brito, and P. Tyack. 2009. Cheetahs of the deep sea: deep foraging sprints in short-finned pilot whales off Tenerife (Canary Islands). *Journal of Animal Ecology* 77: 936-947.

Alves, F., A. Dinis, I. Cascão, and L. Freitas. 2010. Bryde's whale (*Balaenoptera brydei*) stable associations and dive profiles: New insights into foraging behavior. *Marine Mammal Science* 26(1): 202-212.

Amano, M. and M. Yoshioka. 2003. Sperm whale diving behavior monitored using a suction-cup-attached TDR tag. *Marine Ecology Progress Series* 258: 291-295.

Aoki, K., M. Amano, M. Yoshioka, K. Mori, D. Tokuda, and N. Miyazaki. 2007. Diel diving behavior of sperm whales off Japan. *Marine Ecology Progress Series* 349: 277-287.

Au, W.W.L. and M.C. Hastings. 2008. *Principles of Marine Bioacoustics*. Springer. 510 pp. http://link.springer.com/book/10.1007%2F978-0-387-78365-9.

Baird, R.W., G.S. Shorr, D.J. Websater, D.J. McSweeney, and S.D. Mahaffy. 2006a. *Studies of beaked whale diving behavior and odontocete stock structure in Hawai'i in March/April 2006*. Report to

Cascadia Research Collective, Olympia, WA from the Southwest Fisheries Science Center, National Marine Fisheries Service. 30 pp.

Baird, R.W., D.L. Webster, D.J. McSweeney, A.D. Ligon, G.S. Schorr, and J. Barlow. 2006b. Diving behaviour of Cuvier’s (*Ziphius cavirostris*) and Blainville’s (*Mesoplodon densirostris*) beaked whales in Hawai‘i. *Canadian Journal of Zoology* 84(8): 1120-1128.

Baird, R.W., D.L. Webster, S.D. Mahaffy, D.J. McSweeney, G.S. Schorr, and A.D. Ligon. 2008. Site fidelity and association patterns in a deep-water dolphin: Rough-toothed dolphins (*Steno bredanensis*) in the Hawaiian Archipelago. *Marine Mammal Science* 24(3): 535-553.

Baumgartner, M.F. 1997. Distribution of Risso's dolphin (*Grampus griseus*) with respect to the physiography of the northern Gulf of Mexico. *Marine Mammal Science* 13: 614-638.

Buckland, S.T., D.R. Anderson, K.P. Burnham, J.L. Laake, D.L. Borchers, and L. Thomas. 2001. *Introduction to Distance Sampling*. Oxford University Press, Oxford.

Caldwell, D.K., J.H. Prescott, and M.C. Caldwell. 1966. Production of pulsed sounds by the pygmy sperm whale, *Kogia breviceps*. *Bulletin of the Southern California Academy of Science* 65: 245-248.

Caldwell, D.K. and M.C. Caldwell. 1991. A note describing sounds recorded from two cetacean species, *Kogia breviceps* and *Mesoplodon europaeus*, stranded in northeastern Florida. *In* Reynolds, J.E. and D.K. Odell (eds.). *Marine Mammal Strandings in the United States*. NOAA Technical Report NMFS 98. The National Marine Fisheries Service, Seattle, WA. pp 151-154.

Carnes, M.R. 2009. *Description and Evaluation of GDEM-V 3.0*. Document Number NRL Memorandum Report 7330-09-9165. US Naval Research Laboratory, Stennis Space Center, MS. 21 pp.

Collins, M.D. 1993. A split-step Padé solution for the parabolic equation method. *Journal of the Acoustical Society of America* 93(4): 1736-1742.

Collins, M.D., R.J. Cederberg, D.B. King, and S. Chin-Bing. 1996. Comparison of algorithms for solving parabolic wave equations. *Journal of the Acoustical Society of America* 100(1): 178-182.

Cook, M., R. Varela, J. Goldstein, S. Mcculloch, G. Bossart, J. Finneran, D. Houser, and D. Mann. 2006. Beaked whale auditory evoked potential hearing measurements. *Journal of Comparative Physiology A: Neuroethology, Sensory, Neural, and Behavioral Physiology* 192(5): 489-495. http://dx.doi.org/10.1007/s00359-005-0086-1.

Cranford, T.W. and P. Krysl. 2015. Fin whale sound reception mechanisms: Skull vibration enables low-frequency hearing. *PLoS ONE* 10(1): e0116222.

Dahlheim, M.E. and D.K. Ljungblad. 1990. Preliminary hearing study on gray whales (*Eschrichtius robustus*) in the field. *In* Thomas, J. and R. Kastelein (eds.). *Sensory abilities of cetaceans*. Volume 196. Springer US. pp 335-346.

Davis, R.W. and G.S. Fargion. 1996. *Distribution and abundance of cetaceans in the north-central and western Gulf of Mexico: Final report*. Volume 2. US Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.

Davis, R.W., G.S. Fargion, N. May, T.D. Leming, M. Baumgartner, W.E. Evans, L.J. Hansen, and K. Mullin. 1998. Physical habitat of cetaceans along the continental slope in the North-Central and western Gulf of Mexico. *Marine Mammal Science* 14(3): 490-507.

Davis, R.W., W.E. Evan, and B. Würsig. 2000. *Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations*. Volume II: Technical report. OCS Study MMS 2000-003. Prepared by Texas A&M University at Galveston and the National Marine Fisheries Service. U.S. Department of the Interior, Geological Survey, Biological Resources Division. USGS/BRD/CR−1999-0006 and Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA. 346 pp. http://www.data.boem.gov/PI/PDFImages/ESPIS/3/3153.pdf.

Di Sciara, G.N. 1983. *Bryde's whales (Balaenoptera edeni Anderson 1878) off eastern Venezuela (Cetacea, Balaenopteridae)*. Technical Report 83-153. Hubbs-Sea World Research Institute, San Diego, CA. 27 pp.

Dragoset, W.H. 1984. A comprehensive method for evaluating the design of airguns and airgun arrays. *16th Annual Proc. Offshore Tech. Conf.* Volume 3. pp 75–84.

Ellison, W., B. Southall, C. Clark, and A. Frankel. 2012. A new context-based approach to assess marine mammal behavioral responses to anthropogenic sounds. *Conservation Biology* 26(1): 21-28.

Ellison, W.T., C.W. Clark, and G.C. Bishop. 1987. *Potential use of surface reverberation by bowhead whales, Balaena mysticetus, in under-ice navigation: Preliminary considerations*. Report of the International Whaling Commission. Volume 37. 329-332 pp.

Ellison, W.T. and A.S. Frankel. 2012. A common sense approach to source metrics. *In* Popper, A.N. and A. Hawkins (eds.). *The Effects of Noise on Aquatic Life*. Springer. pp 433-438.

Engelhaupt, D., A. Rus Hoelzel, C. Nicholson, A. Frantzis, S. Mesnick, S. Gero, H. Whitehead, L. Rendell, P. Miller, et al. 2009. Female philopatry in coastal basins and male dispersion across the North Atlantic in a highly mobile marine species, the sperm whale (*Physeter macrocephalus*). *Molecular Ecology* 18(20): 4193-4205. http://whitelab.biology.dal.ca/hw/Engelhaupt_et_al%202009.pdf.

Erbe, C., R. McCauley, and A. Gavrilov. 2016. Characterizing marine soundscapes. *In* Popper, N.A. and A. Hawkins (eds.). *The Effects of Noise on Aquatic Life II*. Springer New York, New York, NY. pp 265-271. http://dx.doi.org/10.1007/978-1-4939-2981-8_31.

Fertl, D., A. Schiro, G. Regan, C. Beck, N. Adimey, L. Price-May, A. Amos, G. Worthy, and R. Crossland. 2005. Manatee occurrence in the northern Gulf of Mexico, west of Florida. *Gulf and Caribbean Research* 17: 69-94.

Finneran, J.J., D.S. Houser, B. Mase-Guthrie, R.Y. Ewing, and L. R.G. 2009. Auditory evoked potentials in a stranded Gervais' beaked whale (*Mesoplodon europaeus*). *Journal of the Acoustical Society of America* 126(1): 484-490.

Finneran, J.J. 2015. *Auditory weighting functions and TTS/PTS exposure functions for cetaceans and marine carnivores*. San Diego: SSC Pacific.

Finneran, J.J. 2016. *Auditory weighting functions and TTS/PTS exposure functions for marine mammals exposed to underwater noise*. Technical Report. 49 pp.

Fisher, F.H. and V.P. Simmons. 1977. Sound absorption in sea water. *Journal of the Acoustical Society of America* 62(3): 558-564. http://link.aip.org/link/?JAS/62/558/1.

Frankel, A.S., W.T. Ellison, and J. Buchanan. 2002. Application of the acoustic integration model (AIM) to predict and minimize environmental impacts. *OCEANS'02 MTS/IEEE*. pp 1438-1443.

Fulling, G.L., K.D. Mullin, and C.W. Hubard. 2003. Abundance and distribution of cetaceans in outer continental shelf waters of the US Gulf of Mexico. *Fishery Bulletin* 101(4): 923-932. http://fishbull.noaa.gov/1014/20fullin.pdf.

Funk, D., D. Hannay, D. Ireland, R. Rodrigues, and W. Koski (eds.). 2008. *Marine mammal monitoring and mitigation during open water seismic exploration by Shell Offshore Inc. in the Chukchi and Beaufort Seas, July–November 2007: 90-day report*. LGL Report P969-1. Prepared by LGL Alaska Research Associates Inc., LGL Ltd., and JASCO Research Ltd. for Shell Offshore Inc., National Marine Fisheries Service (US), and US Fish and Wildlife Service. 218 pp.

Gannon, J.G., K.M. Scolardi, J.E. Reynolds, J.K. Koelsch, and T.J. Kessenich. 2007. Habitat selection by manatees in Sarasota Bay, Florida. *Marine Mammal Science* 23(1): 133-143.

Gonçalves, L.R., M. Augustowski, and A. Andriolo. 2016. Occurrence, distribution and behaviour of Bryde's whales (Cetacea: Mysticeti) off south-east Brazil. *Journal of the Marine Biological Association of the United Kingdom* 96(4): 943-954.

Greene, C.R., Jr. 1997. *Physical acoustics measurements*. *In*: Richardson, W.J. (ed.). *Northstar Marine Mammal Monitoring Program, 1996: Marine Mammal and Acoustical Monitoring of a Seismic Program in the Alaskan Beaufort Sea*. LGL Rep. 2121-2. Report from LGL Ltd., King City, ON, and Greeneridge Sciences Inc., Santa Barbara, CA, for BP Exploration (Alaska) Inc., Anchorage, AK, and National Marine Fisheries Services, Anchorage, AK, and Silver Spring, MD. 3-1 to 3-63 pp.

Hannay, D. and R. Racca. 2005. *Acoustic Model Validation*. Document Number 0000-S-90-04-T-7006-00-E, Revision 02. Technical report for Sakhalin Energy Investment Company Ltd. by JASCO Research Ltd. 34 pp.

Hannay, D. 2015. *Summary of Acoustics Effects Studies for BP's Wolfspar Source Operations in the Gulf of Mexico*. Version 1.1 edition. Document Number 1016. Technical report by JASCO Applied Sciences for BP Exploration.

Hansen, L.J., K.D. Mullin, T.A. Jefferson, and G.P. Scott. 1996. Visual surveys aboard ships and aircraft. *In* Davis, R.W. and G.S. Fargion (eds.). *Distribution and abundance of marine mammals in the north-central and western Gulf of Mexico: Final report*. Volume II. OCS Study MMS 96-0027. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region. pp 55-132.

Hastie, G.D., B. Wilson, and P.M. Thompson. 2006. Diving deep in a foraging hotspot: Acoustic insights into bottlenose dolphin dive depths and feeding behaviour. *Marine Biology* 148: 1181-1188.

Hersh, S.L. and D.A. Duffield. 1990. Distinction between northwest Atlantic offshore and coastal bottlenose dolphins based on hemoglobin profile and morphometry. *In* Leatherwood, S. and R. Reeves (eds.). *The bottlenose dolphin*. Academic Press, San Diego. pp 129-139.

Herzing, D.L. and C.R. Elliser. 2016. Opportunistic sightings of cetaceans in nearshore and offshore waters of Southeast Florida. *Journal of Northwest Atlantic Fisheries and Science* 48: 21-31.

Hildebrand, J.A., S. Baumann-Pickering, K.E. Frasier, J.S. Trickey, K.P. Merkens, S.M. Wiggins, M.A. McDonald, L.P. Garrison, D. Harris, et al. 2015. Passive acoustic monitoring of beaked whale densities in the Gulf of Mexico. *Scientific Reports* 5: 16343. http://dx.doi.org/10.1038/srep16343.

Hooker, S.K. and R.W. Baird. 1999. Deep–diving behaviour of the northern bottlenose whale, *Hyperoodon ampullatus* (Cetacea: Ziphiidae). *Proceedings of the Royal Society of London B:*

Biological Sciences 266(1420): 671-676. http://rspb.royalsocietypublishing.org/royprsb/266/1420/671.full.pdf.

Houser, D.S. and M.J. Cross. 1999. *Marine Mammal Movement and Behavior (3MB): A Component of the Effects of Sound on the Marine Environment (ESME) Distributed Model*. Version 8.08, by BIOMIMETICA.

Houser, D.S., D.A. Helweg, and P.W.B. Moore. 2001. A bandpass filter-bank model of auditory sensitivity in the humpback whale. *Aquatic Mammals* 27: 82-91.

Houser, D.S. 2006. A method for modeling marine mammal movement and behavior for environmental impact assessment. *IEEE Journal of Oceanic Engineering* 31(1): 76-81.

Houser, D.S., L.A. Dankiewicz -Talmadge, T.K. Stockard, and P.J. Ponganis. 2010. Investigation of the potential for vascular bubble formation in a repetitively diving dolphin. *Journal of Experimental Biology* 213: 52-62. http://jeb.biologists.org/content/213/1/52.full.pdf+html.

Ireland, D.S., R. Rodrigues, D. Funk, W. Koski, and D. Hannay. 2009. *Marine mammal monitoring and mitigation during open water seismic exploration by Shell Offshore Inc. in the Chukchi and Beaufort Seas, July–October 2008: 90-Day Report*. Document Number LGL Report P1049-1. 277 pp.

Jefferson, T.A., S. Leatherwood, L.K. Shoda, and R.L. Pitman. 1992. *Marine Mammals of the Gulf of Mexico: A field guide for aerial and shipboard observers*. GULFCET Program, Department of Marine Biology, Texas A & M University. 92 pp.

Jefferson, T.A. and A.J. Schiro. 1997. Distributions of cetaceans in the offshore Gulf of Mexico. *Mammal Review* 27(1): 27-50.

Jefferson, T.A., M.A. Webber, and R.L. Pitman. 2008. *Marine Mammals of the World, A Comprehensive Guide to their Identification*. Elsevier, Amsterdam.

Jochens, A.E., D. Biggs, K. Benoit-Bird, D. Engelhaupt, J. Gordon, C. Hu, N. Jaquet, and M. Johnson. 2008. *Sperm Whale Seismic Study in the Gulf of Mexico: Synthesis Report*. Document Number OCS Study MMS 2008-006. US Dept. of Interior, Minerals Management Service, Gulf of Mexico OCS Region. 341 pp.

Kastelein, R.A., A.D. Goodson, J. Lien, and D. De Haan. 1995. The effects of acoustic alarms on harbour porpoise (*Phocoena phocoena*) behaviour. *Harbour Porpoises-laboratory studies to reduce bycatch*: 157-167.

Landro, M. 1992. Modeling of GI gun signatures. *Geophysical Prospecting* 40: 721–747.

Laws, M., L. Hatton, and M. Haartsen. 1990. Computer modeling of clustered airguns. . *First Break* 8: 331–338.

Leatherwood, S. and R.R. Reeves. 1983. *The Sierra Club handbook of whales and dolphins*. Sierra Club Books, San Francisco. 320 pp.

Leatherwood, S., T.A. Jefferson, J.C. Norris, W.E. Stevens, L.J. Hansen, and K.D. Mullin. 1993. Occurrence and sounds of Fraser's dolphins (*Lagenodelphis hosei*) in the Gulf of Mexico. *Texas Journal of Science* 45(4): 349-354.

LeDuc, R.G. and B.E. Curry. 1998. Mitochondrial DNA sequence analysis indicates need for revision of the genus *Tursiops*. *Rep. Int. Whal. Commn* 47: 393.

Lopez, B.D. 2009. The bottlenose dolphin *Tursiops truncatus* foraging around a fish farm: Effects of prey abundance on dolphin's behavior. *Current Zoology* 55(4): 243-248.

Lurton, X. 2002. *An Introduction to Underwater Acoustics: Principles and Applications.* Springer, Chichester, U.K.

MacDonnell, J., D. Hannay, A. Schlesinger, and Z. Li. 2015. *Underwater sound level verification measurements during Wolfspar Source trials in the Gulf of Mexico, December 2014: In support of the "Environmental Assessment for Ocean Trials of the Wolfspar source in the Gulf of Mexico"*. Version 2.0 edition. Document Number 1016. Technical report by JASCO Applied Sciences for BP Exploration.

MacGillivray, A.O. 2006. *Acoustic Modelling Study of Seismic Airgun Noise in Queen Charlotte Basin*. M. Sc. Thesis. University of Victoria, Victoria, BC.

Madsen, P.T., B. Møhl, B.K. Nielsen, and M. Wahlberg. 2002. Male sperm whale behavior during exposures to distant seismic survey pulses. *Aquatic Mammals* 28: 231-240.

Madsen, P.T., M. Johnson, N. Aguilar Soto, W.M.X. Zimmer, and P. Tyack. 2005. Biosonar performance of foraging beaked whales (*Mesoplodon densirostris*). *Journal of Experimental Biology* 208: 181-194.

Malme, C.I., P.R. Miles, C.W. Clark, P. Tyak, and J.E. Bird. 1983. *Investigations of the Potential Effects of Underwater Noise from Petroleum Industry Activities on Migrating Gray Whale Behavior*. Report Number 5366. http://www.boem.gov/BOEM-Newsroom/Library/Publications/1983/rpt5366.aspx.

Malme, C.I., P.R. Miles, C.W. Clark, P. Tyack, and J.E. Bird. 1984. *Investigations of the potential effects of underwater noise from petroleum industry activities on migrating gray whale behavior. Phase II: January 1984 migration*. Report Number BBN Report 5586. Bolt Beranek and Newman Inc. 357 pp.

Marques, F.F.C. and S.T. Buckland. 2004. Covariate models for the detection function. *In* Buckland, S.T., D.R. Anderson, K.P. Burnham, J.L. Laake, D.L. Borchers, and L. Thomas (eds.). *Advanced Distance Sampling*. Oxford University Press, Oxford. pp 31-47.

Mate, B. and J. Ortega-Ortiz. 2004. Status report on satellite-monitored radio tag. *In* Jochens, A.E. and D.C. Biggs (eds.). *Sperm whale seismic study in the Gulf of Mexico; Annual Report: Year 2*. OCS Study MMS 2004-067. pp 85-86.

Maze-Foley, K. and K.D. Mullin. 2006. Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. *Journal of Cetacean Research and Management* 8(2): 203-213.

McCauley, R.D., M. Jenner, C. Jenner, K. McCabe, and J. Murdoch. 1998a. The response of humpback whales (*Megaptera novaeangliae*) to offshore seismic survey noise: Preliminary results of observations about a working seismic vessel and experimental exposures. *Australian Petroleum Production Exploration Association (APPEA) Journal* 38(1): 692-707.

McCauley, R.D., M.N. Jenner, C. Jenner, and D.H. Cato. 1998b. Observations of the movements of humpback whales about an operating seismic survey vessel near Exmouth, Western Australia. *Journal of the Acoustical Society of America* 103: 2909-2909.

Miller, D.L., M.L. Burt, E.A. Rexstad, and L. Thomas. 2013. Spatial models for distance sampling data: Recent developments and future directions. *Methods in Ecology and Evolution* 4(11): 1001-1010.

Miller, P.J.O., M.P. Johnson, P.L. Tyack, and E.A. Terray. 2004. Swimming gaits, passive drag and buoynacy of diving sperm whales *Physeter macrocephalus*. *Journal of Experimental Biology* 207: 1953-1967.

Miller, P.J.O., K. Aoki, L.E. Rendell, and M. Amano. 2008. Stereotypical resting behavior of the sperm whale. *Current Biology* 18(1): 21-23.

Mullin, K., W. Hoggard, C. Roden, R. Lohoefener, and C. Rogers. 1991. *Cetaceans on the upper continental slope in the north-central Gulf of Mexico*. OCS Study/MMS 91-0027. U.S. Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Regional Office, New Orleans, LA. 108 pp.

Mullin, K., W. Hoggard, C. Roden, R. Lohoefener, and C. Rogers. 1994. Cetaceans on the upper continental slope in the north-central Gulf of Mexico. *Fishery Bulletin* 92(4): 773-786. http://fishbull.noaa.gov/924/mullin.pdf.

Mullin, K.D. and W. Hoggard. 2000. Visual surveys of cetaceans and sea turtles from aircraft and ships. (Chapter 4) *In* Davis, R.W., W.E. Evans, and B. Würsig (eds.). *Cetaceans, Sea Turtles and Seabirds in the Northern Gulf of Mexico: Distribution, Abundance and Habitat Associations*. Volume 2. Technical Report by Texas A&M University at Galveston and the National Marine Fisheries Service. U.S. Department of the Interior, U.S. Geological Survey, Biological Resources Division, USGS/BRD/CR-1999-005 and Minerals Management Service, Gulf of Mexico OCS Region. OCS Study MMS 2000-003, New Orleans, LA. pp 96-0027.

Mullin, K.D. and G.L. Fulling. 2004. Abundance of cetaceans in the oceanic northern Gulf of Mexico, 1996-2001. *Marine Mammal Science* 20(4): 787-807.

Mullin, K.D., W. Hoggard, and L.J. Hansen. 2004. Abundance and seasonal occurrence of cetaceans in outer continental shelf and slope waters of the north-central and northwestern Gulf of Mexico. *Gulf of Mexico Science* 22(1): 62-73.

Mullin, K.D. 2007. *Abundance of cetaceans in the oceanic Gulf of Mexico based on 2003-2004 ship surveys*. NMFS, Southeast Fisheries Science Center, Pascagoula, MS. 26 pp.

Murase, H., T. Tamura, S. Otani, and S. Nishiwaki. 2015. Satellite tracking of Bryde's whales Balaenoptera edeni in the offshore western North Pacific in summer 2006 and 2008. *Fisheries science* 82(1): 35-45. http://www.icrwhale.org/pdf/SC-F16-JR45.pdf.

Nedwell, J.R. and A.W. Turnpenny. 1998. The use of a generic frequency weighting scale in estimating environmental effect. *Workshop on Seismics and Marine Mammals*. 23–25th June 1998, London, U.K.

Nedwell, J.R., A.W.H. Turnpenny, J. Lovell, S.J. Parvin, R. Workman, and J.A.L. Spinks. 2007. *A validation of the dB$_{ht}$ as a measure of the behavioural and auditory effects of underwater noise*. Report No. 534R1231 prepared by Subacoustech Ltd. for the UK Department of Business, Enterprise and Regulatory Reform under Project No. RDCZ/011/0004. www.subacoustech.com/information/downloads/reports/534R1231.pdf.

O'Neill, C., D. Leary, and A. McCrodan. 2010. Sound Source Verification. (Chapter 3) *In* Blees, M.K., K.G. Hartin, D.S. Ireland, and D. Hannay (eds.). *Marine mammal monitoring and mitigation during open water seismic exploration by Statoil USA E&P Inc. in the Chukchi Sea, August-October 2010: 90-day report*. LGL Report P1119. Prepared by LGL Alaska Research Associates Inc., LGL Ltd., and JASCO Applied Sciences Ltd. for Statoil USA E&P Inc., National Marine Fisheries Service (US), and US Fish and Wildlife Service. pp 1-34.

O'Sullivan, S. and K.D. Mullin. 1997. Killer whales (*Orcinus orca*) in the northern Gulf of Mexico. *Marine Mammal Science* 13(1): 141-147.

Ortega Ortiz, J.G. 2002. *Multiscale analysis of cetacean distribution in the Gulf of Mexico*. Ph.D. dissertation. Texas A&M University, College Station.

Pabody, C.M., R.H. Carmichael, L. Rice, and M. Ross. 2009. A new sighting network adds to 20 years of historical data on fringe West Indian manatee (*Trichechus manatus*) populations in Alabama waters. *Gulf of Mexico Science* 1: 52-61.

Palka, D. and M. Johnson. 2007. *Cooperative Research to Study Dive Patterns of Sperm Whales in the Atlantic Ocean*. OCS Study MMS 2007-033. Minerals Management. Service, Gulf of Mexico OCS Region, New Orleans, LA. 49 pp.

Parks, S.E., C.W. Clark, and P.L. Tyack. 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.

Perrin, W. and A.A. Hohn. 1994. Pantropical spotted dolphin *Stenella attenuata*. *In* Ridgway, S.H. and R. Harrison (eds.). *Handbook of marine mammals: the first book of dolphins*. Academic Press, London, United Kingdom. pp 71-98.

Perrin, W.F. and J.W.J. Gilpatrick. 1994. Spinner dolphin *Stenella longirostris* (Gray, 1828). *In* Ridgway, S.H. and R. Harrison (eds.). *Handbook of marine mammals: the first book of dolphins*. Volume 5. Academic Press, London. pp 99-128.

Porter, M.B. and Y.-C. Liu. 1994. Finite-element ray tracing. *In*: Lee, D. and M.H. Schultz (eds.). *Proceedings of the International Conference on Theoretical and Computational Acoustics*. Volume 2. World Scientific Publishing Co. pp 947-956.

Quick, N.J., S. Isojunno, D. Sadykova, M. Bowers, D.P. Nowacek, and A.J. Read. 2017. Hidden Markov models reveal complexity in the diving behaviour of short-finned pilot whales. *Scientific Reports* 7: 45765.

Rankin, S. and J. Barlow. 2007. Sounds recorded in the presence of Blainville's beaked whales, *Mesoplodon densirostris*, near Hawai'i. *Journal of the Acoustical Society of America* 122(1): 42-45.

Reeves, R.R., B.S. Stewart, P.J. Clapham, J.A. Powell, and P.A. Folkens. 2002. *Guide to Marine Mammals of the World*. Alfred A. Knopf Inc., New York.

Reichmuth, C., J. Mulsow, J.J. Finneran, D.S. Houser, and A.Y. Supin. 2007. Measurement and response characteristics of auditory brainstem responses in pinnipeds. *Aquatic Mammals* 33(1): 132-150.

Reilly, S.B., J.L. Bannister, P.B. Best, M. Brown, R.L. Brownell Jr., D.S. Butterworth, P.J. Clapham, J. Cooke, G.P. Donovan, et al. 2008. *The IUCN Red List of Threatened Species* (webpage), Version 2014.3. www.iucnredlist.org (Accessed April 1, 2015).

Richardson, W.J., B. Würsig, and C.R. Greene, Jr. 1986. Reactions of bowhead whales, *Balaena mysticetus*, to seismic exploration in the Canadian Beaufort Sea. *Journal of the Acoustical Society of America* 79(4): 1117-1128.

Richardson, W.J., B. Würsig, and C.R. Greene, Jr. 1990. Reactions of bowhead whales, *Balaena mysticetus*, to drilling and dredging noise in the Canadian Beaufort Sea. *Marine Environmental Research* 29(2): 135-160. http://www.sciencedirect.com/science/article/pii/014111369090032J.

Richardson, W.J., C.R. Greene, Jr., C.I. Malme, and D.H. Thomson. 1995. *Marine Mammals and Noise*. Academic Press, San Diego, California. 576 pp.

Ridgway, S.H. and D.A. Carder. 2001. Assessing hearing and sound production in cetaceans not available for behavioral audiograms: Experiences with sperm, pygmy sperm, and gray whales. *Aquatic Mammals* 27(3): 267-276.

Roberts, J.J., B.D. Best, L. Mannocci, E. Fujioka, P.N. Halpin, D.L. Palka, L.P. Garrison, K.D. Mullin, T.V.N. Cole, et al. 2016a. Habitat-based cetacean density models for the U.S. Atlantic and Gulf of Mexico. *Scientific Reports* 6. https://www.ncbi.nlm.nih.gov/pmc/articles/PMC4776172/.

Roberts, J.J., B.D. Best, L. Mannocci, E. Fujioka, P.N. Halpin, D.L. Palka, L.P. Garrison, K.D. Mullin, T.V.N. Cole, et al. 2016b. Habitat-based cetacean density models for the US Atlantic and Gulf of Mexico. *Scientific Reports* 6. https://www.ncbi.nlm.nih.gov/pmc/articles/PMC4776172/pdf/srep22615.pdf.

Sakai, M., K. Aoki, K. Sato, M. Amano, R.W. Baird, D. Webster, G.S. Schorr, and N. Miyazaki. 2011. Swim speed and acceleration measurements of short-finned pilot whales (*Globicephala macrorhynchus*) in Hawai'i. *Mammal Study* 36: 55-59.

Scheer, M. 2013. Call vocalizations recorded among short-finned pilot whales (*Globicephala macrorhynchus*) Off Tenerife, Canary Islands. *Aquatic Mammals* 39(3): 306-313.

Schmidly, D.J. 1981. *Marine mammals of the southeastern United States coast and the Gulf of Mexico*. Report Number FWS/OBS-80/41. Texas A and M University and Department of Wildlife and Fisheries Science. https://www.data.boem.gov/PI/PDFImages/ESPIS/3/4002.pdf.

Schulz, T.M., H. Whitehead, S. Gero, and L. Rendell. 2011. Individual vocal production in a sperm whale (Physeter macrocephalus) social unit. *Marine Mammal Science* 27(1): 149-166. http://dx.doi.org/10.1111/j.1748-7692.2010.00399.x.

Širović, A., H.R. Bassett, S.C. Johnson, S.M. Wiggins, and J.A. Hildebrand. 2014. Bryde's whale calls recorded in the Gulf of Mexico. *Marine Mammal Science* 30(1): 399-409.

Southall, B., W. Ellison, C. Clark, and D. Tollit. 2016. *A risk assessment framework to assess the biological significance of noise exposure on marine mammals*. *Poster presented at the 21st Conference on the Biology of Marine Mammals*, San Francisco. http://sea-inc.net/wp-content/uploads/2016/01/EWG-Framework_SMM-poster.jpg.

Southall, B.L., A.E. Bowles, W.T. Ellison, J.J. Finneran, R.L. Gentry, C.R. Greene, Jr., D. Kastak, D.R. Ketten, J.H. Miller, et al. 2007. Marine mammal noise exposure criteria: Initial scientific recommendations. *Aquatic Mammals* 33(4): 411-521.

Teague, W.J., M.J. Carron, and P.J. Hogan. 1990. A comparison between the Generalized Digital Environmental Model and Levitus climatologies. *Journal of Geophysical Research* 95(C5): 7167-7183.

Thode, A., D.K. Mellinger, S. Stienessen, A. Martinez, and K. Mullin. 2002. Depth-dependent acoustic features of diving sperm whales (*Physeter macrocephalus*) in the Gulf of Mexico. *Journal of the Acoustical Society of America* 112(1): 308-321. http://scitation.aip.org/content/asa/journal/jasa/112/1/10.1121/1.1482077.

Townsend, C.H. 1935. *The distribution of certain whales as shown by logbook records of American whaleships*. New York Zoological Society.

Tubelli, A., A. Zosuls, D. Ketten, and D.C. Mountain. 2012. *Prediction of a mysticete audiogram via finite element analysis of the middle ear*. *The effects of noise on aquatic life*.

Tyack, P., M. Johnson, N. Aguilar Soto, A. Sturlese, and P.T. Madsen. 2006. Extreme diving of beaked whales. *Journal of Experimental Biology* 209: 4238-4253.

Ward, B.G. 1999. *Movement patterns and feeding ecology of the Pacific coast bottlenose dolphin (Tursiops truncatus)*. Master Thesis. San Diego State University. 98 pp.

Waring, G.T., T. Hamazaki, D. Sheehan, G. Wood, and S. Baker. 2001. Characterization of beaked whale (*Ziphiidae*) and sperm whale (*Physeter macrocephalus*) summer habitat in shelf-edge and deeper waters of the northeast U.S. *Marine Mammal Science* 17: 703-717.

Waring, G.T., E. Josephson, K. Maze-Foley, and P.E. Rosel. 2010. *U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments - 2010*. Volume 598. NOAA Tech Memo NMFS NE 219. 475 pp. Available from http://www.nmfs.noaa.gov/pr/sars/region.htm.

Waring, G.T., E. Josephson, K. Maze-Foley, and P.E. Rosel. 2013. US Atlantic and Gulf of Mexico Marine Mammal Stock Assessments--2012. *NOAA Tech Memo NMFS NE* 223(419): 02543-1026. http://www.nmfs.noaa.gov/pr/sars/pdf/ao2012.pdf.

Warner, G., C. Erbe, and D. Hannay. 2010. Underwater Sound Measurements. (Chapter 3) *In* Reiser, C.M., D.W. Funk, R. Rodrigues, and D. Hannay (eds.). *Marine Mammal Monitoring and Mitigation during Open Water Shallow Hazards and Site Clearance Surveys by Shell Offshore Inc. in the Alaskan Chukchi Sea, July-October 2009: 90-Day Report*. LGL Report P1112-1. Report by LGL Alaska Research Associates Inc. and JASCO Applied Sciences for Shell Offshore Inc., National Marine Fisheries Service (US), and US Fish and Wildlife Service. pp 1-54.

Wartzok, D. and D.E. Ketten. 1999. Marine Mammal Sensory Systems. *In* Reynolds, J. and S. Rommel (eds.). *Biology of Marine Mammals*. Smithsonian Institution Press, Washington DC. pp 117-175.

Watwood, S.L., P.J.O. Miller, M.P. Johnson, P.T. Madsen, and P.L. Tyack. 2006. Deep-diving foraging behaviour of sperm whales (*Physeter macrocephalus*). *Journal of Animal Ecology* 75: 814-825.

Wells, R.S., E.M. Fougeres, A.G. Cooper, R.O. Stevens, M. Brodsky, R. Lingenfelser, C. Dold, and D.C. Douglas. 2013. Movements and dive patterns of short-finned pilot whales (*Globicepha macrorhynchus*) released from a mass stranding in the Florid Keys. *Aquatic Mammals* 39(1): 61-72.

Whitehead, H. 2002. Estimates of the current population size and historical trajectory for sperm whales. *Marine Ecology Progress Series* 242: 295-304.

Whitehead, H. and L. Rendell. 2004. Movements, habitat use and feeding success of cultural clans of South Pacific sperm whales. *Journal of Animal Ecology* 73: 190-196. http://onlinelibrary.wiley.com/doi/10.1111/j.1365-2656.2004.00798.x/epdf.

Wood, J., B.L. Southall, and D.J. Tollit. 2012. *PG&E offshore 3 D Seismic Survey Project EIR-Marine Mammal Technical Draft Report*. SMRU Ltd.

Wood, S.N. 2006. *Generalized additive models: An introduction with R*. Chapman and Hall. Boca Raton.

Wursig, B., T.A. Jefferson, and D.J. Schmidly. 2000. *The marine mammals of the Gulf of Mexico*. Texas A&M University Press, College Station, TX.

Würsig, B. and M. Würsig. 1979. Behavior an ecology of the bottlenose dolphin, *Tursiops truncatus* in the South Atlantic. *Fisheries Bulletin* 77: 399-412.

Wynne, K. and M. Schwartz. 1999. *Guide to Marine Mammals and Turtles of the U.S. Atlantic and Gulf of Mexico*. 1st edition. Alaska Sea Grant, University of Alaska Fairbanks.

Zhang, Y. and C. Tindle. 1995. Improved equivalent fluid approximations for a low shear speed ocean bottom. *Journal of the Acoustical Society of America* 98(6): 3391-3396. http://scitation.aip.org/content/asa/journal/jasa/98/6/10.1121/1.413789.

Ziolkowski, A. 1970. A method for calculating the output pressure waveform from an air gun *Geophysical Journal of the Royal Astronomical Society* 21: 137-161.

Appendix A. Sound Metrics Used in Modeling

Underwater sound amplitude is measured in decibels (dB) relative to a fixed reference pressure of p_o = 1 μPa. Because the loudness of impulsive (pulsed) sounds, e.g., shots from seismic source arrays, is not generally proportional to the instantaneous acoustic pressure, several sound level metrics are commonly used to evaluate the loudness of impulsive sound and its effects on marine life.

The zero-to-peak sound pressure level (SPL), or peak SPL (L_{pk}, dB re 1 μPa), is the maximum instantaneous sound pressure level in a stated frequency band attained by an impulse, $p(t)$:

$$L_{\text{pk}} = 10\log_{10}\left[\frac{\max\left(\left|p^2(t)\right|\right)}{p_{\text{o}}^2}\right] \tag{A-1}$$

The peak-to-peak SPL ($L_{\text{pk-pk}}$, dB re 1 μPa) is the difference between the maximum and minimum instantaneous sound pressure level in a stated frequency band attained by an impulse, $p(t)$:

$$L_{\text{pk}-\text{pk}} = 10\log_{10}\left\{\frac{\left[\max\left(p(t)\right)-\min\left(p(t)\right)\right]^2}{p_{\text{o}}^2}\right\} \tag{A-2}$$

The root-mean square (rms) SPL (L_p, dB re 1 μPa) is the rms pressure level in a stated frequency band over a time window (T, s) containing the pulse:

$$L_p = 10\log_{10}\left(\frac{1}{T}\int_T p^2(t)dt \Big/ p_{\text{o}}^2\right) \tag{A-3}$$

The rms SPL can be thought of as a measure of the average pressure or as the “effective” pressure over the duration of an acoustic event, such as the emission of one acoustic pulse. Because the window length, T, is a divisor, pulses more spread out in time have a lower rms SPL for the same total acoustic energy.

By convention, when computing airgun safety radii, T is defined as the “90% energy pulse duration”, containing the central 90% (from 5% to 95% of the total) of the cumulative square pressure (or energy) of the pulse, rather than over a fixed time window (Malme et al. 1983, Greene 1997, McCauley et al. 1998a, McCauley et al. 1998b). The 90% rms SPL (L_{p90}, dB re 1 μPa) in a stated frequency band is calculated over this 90% energy time window, T_{90}:

$$L_{p90} = 10\log_{10}\left(\frac{1}{T_{90}}\int_{T_{90}} p^2(t)dt \Big/ p_{\text{o}}^2\right) \tag{A-4}$$

The sound exposure level (SEL) (L_E, dB re 1 μPa^2·s) is the time integral of the squared pressure in a stated frequency band over a stated time interval or event. The per-pulse SEL is calculated over the time window containing the entire pulse (i.e., 100% of the acoustic energy), T_{100}:

$$L_E = 10\log_{10}\left(\int_{T_{100}} p^2(t)dt \Big/ T_{\text{o}}\, p_{\text{o}}^2\right) \tag{A-5}$$

where T_o is a reference time interval of 1 s. The per-pulse SEL, with units of dB re 1 μPa·√s, or equivalently dB re 1 μPa^2·s, represents the total acoustic energy delivered over the duration of the acoustic event at a receiver location. It is a measure of sound energy (or exposure) rather than sound pressure although it is not measured in energy units.

SEL is a cumulative metric that is calculated over a specified time period that may contain multiple pulses. SEL can be computed by summing (in linear units) the SELs of the *N* individual pulses (L_{Ei}).

$$L_{Ec} = 10\log_{10}\left(\sum_{i=1}^{N} 10^{\frac{L_{Ei}}{10}}\right) \tag{A-6}$$

The cumulative SEL, with units of dB re 1 µPa·√s, or equivalently dB re 1 µPa²·s, represents the total acoustic energy delivered over the duration of the set period of time, i.e., 24 h. It is a representation of the accumulated sound energy (or exposure) delivered by multiple acoustic events.

Because the rms SPL and SEL are both computed from the integral of square pressure, these metrics are related by a simple expression, which depends only on the duration of the 90% energy time window T_{90}:

$$L_E = L_{p90} + 10\log_{10}\left(T_{90}\right) + 0.458 \tag{A-7}$$

where the 0.458 dB factor accounts for the rms SPL containing 90% of the total energy from the per-pulse SEL.

Appendix B. Source and Propagation Modeling

B.1. Acoustic Source Model

B.1.1. 4130 in^3 seismic source array

The source levels and directivity of the 4130 in^3 seismic source array were predicted with JASCO's Airgun Array Source Model (AASM, MacGillivray 2006). This model is based on the physics of oscillation and radiation of airgun bubbles described by Ziolkowski (1970). The model solves the set of parallel differential equations governing bubble oscillations. AASM also accounts for nonlinear pressure interactions among array elements, port throttling, bubble damping, and generator-injector (GI) gun behavior that are discussed by Dragoset (1984), Laws et al. (1990), and Landro (1992). AASM includes four empirical parameters that are tuned so model output matches observed airgun behavior. The model parameters fit to a large library of empirical airgun data using a "simulated annealing" global optimization algorithm. AASM produces a set of "notional" signatures for each array element based on:

- Array layout;
- Volume, tow depth, and firing pressure of each element; and
- Interactions between different elements in the array.

These notional signatures are the pressure waveforms of the individual elements at a standard reference distance of 1 m, and they account for the interactions with the other elements in the array. The signatures are summed with the appropriate phase delays to obtain the far-field source signature of the entire array in all directions. This far-field array signature is filtered into 1/3-octave passbands to compute the source levels (SLs) of the array as a function of frequency band and azimuthal angle in the horizontal plane (at the source depth). It can then be treated as a directional point source in the far field.

A seismic array consists of many sources and the point-source assumption is not valid in the near field where the array elements add incoherently. The maximum extent of the near field of an array (R_{nf}) is:

$$R_{nf} < \frac{l^2}{4\lambda} \,, \tag{B-1}$$

where λ is the sound wavelength and l is the longest dimension of the array (Lurton 2002, §5.2.4). For example, using equation C-1, an array length of $l = 16$ m yields a near-field range of 85 m at 2 kHz and 17 m at 100 Hz. Beyond this R_{nf} range, the array is assumed to radiate like a directional point source and is treated as such for propagation modeling.

The interactions between individual elements of the array create directionality in the overall acoustic emission. Generally, this directionality is prominent mainly at frequencies in the mid-range of several tens to several hundred hertz; at lower frequencies, with acoustic wavelengths much larger than the inter-array separation distances, directivity is small. At higher frequencies, the pattern of lobes is too finely spaced to be resolved and the effective directivity is less.

AASM was used to compute the pressure signatures of the individual source array elements and the composite 1/3-octave-band source levels of the array, as functions of azimuthal angle (in the horizontal plane). While effects of source depth on bubble interactions are accounted for in the AASM source model, the surface-reflected signal (i.e., surface ghost) is not included in the far-field source signatures. The surface reflections, a property of the medium rather than the source, are accounted for by the acoustic propagation models. In this study, the source levels for a 4130 in^3 element array acted as the acoustic source for the MONM sound propagation models.

The horizontal overpressure signatures and corresponding power spectrum levels for the 4130 in^3 element array, at a depth of 8 m (to the vertical center of the element clusters), are shown in Figure B-1 and Table B-1 for the broadside (perpendicular to the tow direction) and endfire (parallel to the tow direction) directions. The signatures consist of a strong primary peak related to the initial firing of the source, followed by a series of pulses associated with bubble oscillations. Most energy is produced at frequencies below 250 Hz (Figure B-2). The spectrum contains peaks and nulls resulting from interference among array elements, where the frequencies at which they occur depend on the volumes of each element and their locations within the array. The maximum (horizontal) 1/3-octave-band sound levels over all directions are plotted in Figure B-2. The horizontal 1/3-octave-band directivities are shown in Figure B-3.

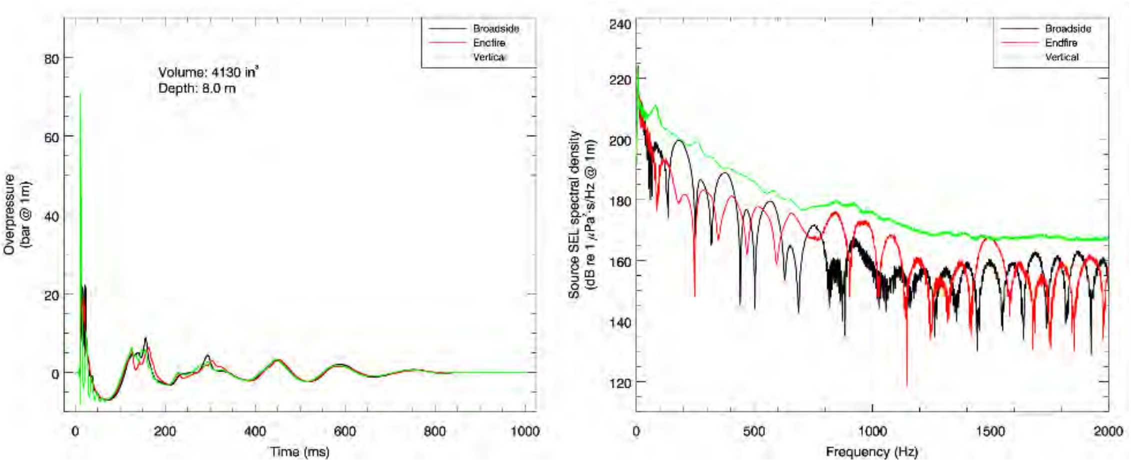


Figure B-1. The 4130 in^3 array: Predicted (a) overpressure signature and (b) power spectrum in the broadside, endfire, and vertical directions. Surface ghosts (effects of the pulse reflection at the water surface) are not included in these signatures as they are accounted for by the MONM propagation model.

Table B-1. Horizontal source level specifications (10–5000 Hz) for the 4130 in^3 seismic airgun array at 8 m depth, computed with AASM in the broadside and endfire directions. Surface ghost effects are not included as they are accounted for by the MONM propagation model.

Direction	Zero-to-peak SPL (dB re 1 µPa @ 1 m)	SEL (0.01–5 kHz) (dB re 1 µPa^2 @ 1 m)
Broadside	247.9	228.9
Endfire	245.6	228.2

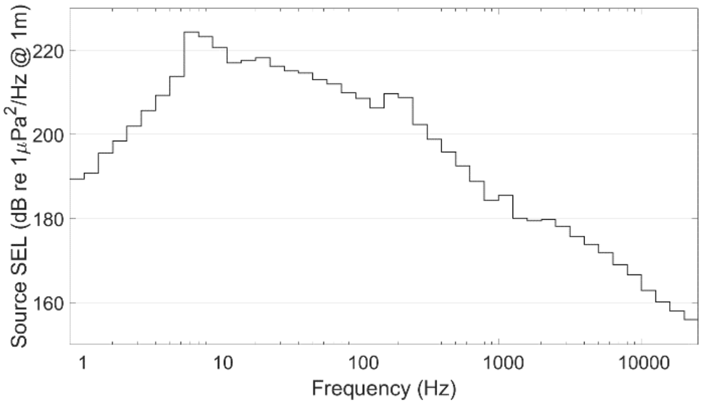


Figure B-2. Maximum directional source level (SL) in the horizontal plane, in each 1/3-octave-band, for the 4130 in^3 airgun array (1–25,000 Hz).

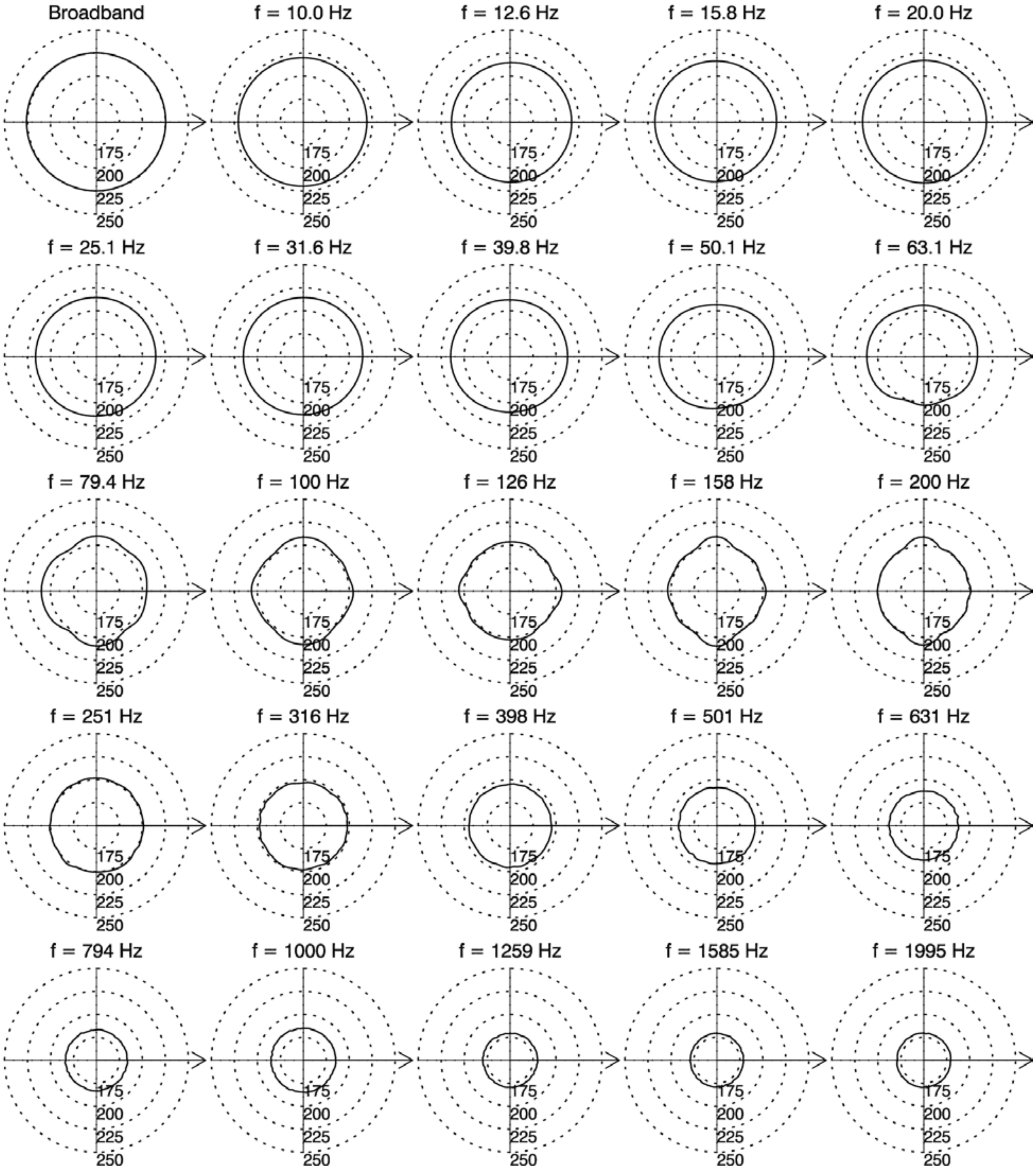


Figure B-3. Horizontal directivity of the 4130 in^3 array. Source levels (SLs, dB re 1 µPa^2·s) in 1/3-octave-bands. The 1/3-octave-band center frequencies are indicated above each plot.

Appendix C. Acoustic Propagation Modeling

C.1. Marine Operations Noise Model (MONM)

Underwater sound propagation (i.e., transmission loss) at frequencies below 4 kHz was predicted with JASCO's Marine Operations Noise Model (MONM). This model computes received sound levels at specified depths. MONM computes acoustic propagation via a wide-angle parabolic equation solution to the acoustic wave equation (Collins 1993) based on a version of the U.S. Naval Research Laboratory's Range-dependent Acoustic Model (RAM), which has been modified to account for an elastic seabed (Zhang and Tindle 1995). The parabolic equation method has been extensively benchmarked and is widely employed in the underwater acoustics community (Collins et al. 1996). MONM accounts for the additional reflection loss at the seabed due to partial conversion of incident compressional waves to shear waves at the seabed and sub-bottom interfaces, and it includes wave attenuations in all layers. MONM incorporates the following site-specific environmental properties: a modeled area bathymetric grid, underwater sound speed as a function of depth, and a geoacoustic profile based on the overall stratified composition of the seafloor.

The accuracy of MONM's predictions have been validated against experimental data from numerous sound source verification programs conducted by JASCO (Hannay and Racca 2005, Aerts et al. 2008, Funk et al. 2008, Ireland et al. 2009, O'Neill et al. 2010, Warner et al. 2010). An inherent variability in measured sound levels is caused by temporal variability in the environment and the variability in the signature of repeated acoustic impulses (sample sound source verification results are presented in Figure C-1.

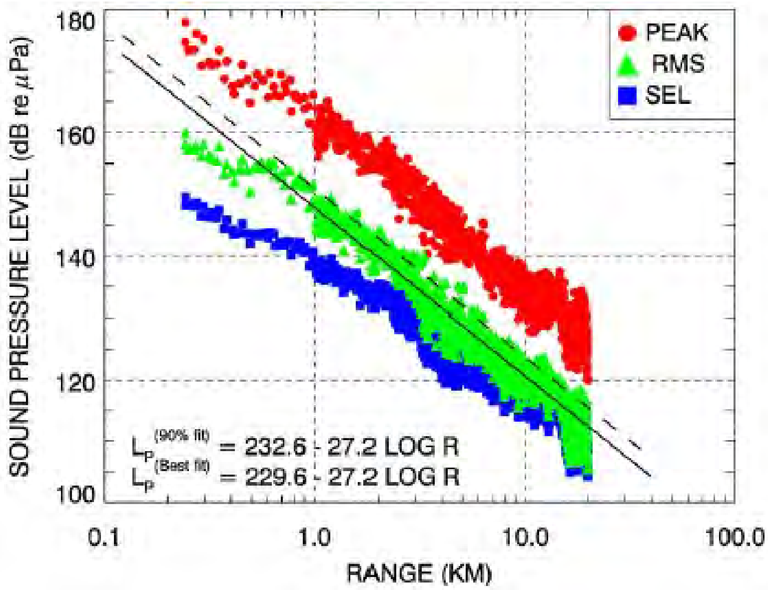


Figure C-1. Field measurements of peak and root-mean-square (rms) sound pressure level (SPL) and sound exposure level (SEL) versus range from a 20 in^3 airgun array. Solid line is the least squares best fit to rms SPL (Ireland et al. 2009).

A model validation assessment was performed between the original modeling study and the SIT measurements. The comparison revealed that the short-range model results exceeded measurements, but at longer distances (> 10 km), the measurements were between 2 and 5 dB above the model. Therefore, a uniform 3 dB was applied to the model to match the longer-range measurements and to be conservative (Figure C-2) (Hannay 2015, MacDonnell et al. 2015).

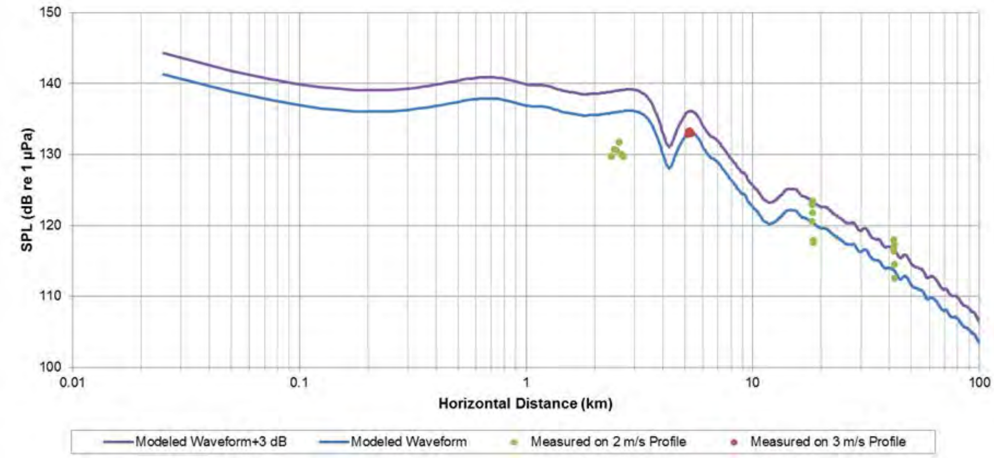


Figure C-2. Modeled results (lines) and measurements (symbols) of SPL for the 2–8 Hz frequency sweep received at a seabed depth of 2490 m (no frequency weighting, maximum 1 second value over the period of the sweep) at several measurement ranges. Two transducer settings are shown (2 m/s with green symbols and 3 m/s with red). The ground-truthed model (mauve line) is derived from the base model (blue line) with a 3 dB positive shift (Hannay 2015).

For frequencies above 4 kHz, MONM model computes sound propagation from high-frequency acoustic sources via the BELLHOP Gaussian beam acoustic ray-trace model (Porter and Liu 1994). This version of MONM accounts for sound attenuation due to energy absorption through ion relaxation and viscosity of water in addition to acoustic attenuation due to reflection at the medium boundaries and internal layers (Fisher and Simmons 1977). The former type of sound attenuation is significant for frequencies higher than 5 kHz and cannot be neglected without noticeably affecting the model results. MONM computes acoustic fields in three dimensions by modeling transmission loss within two-dimensional (2-D) vertical planes aligned along radials covering a 360° swath from the source, an approach commonly referred to as *N*×2-D. These vertical radial planes are separated by an angular step size of 22.5°, yielding 16 planes (Figure C-3).

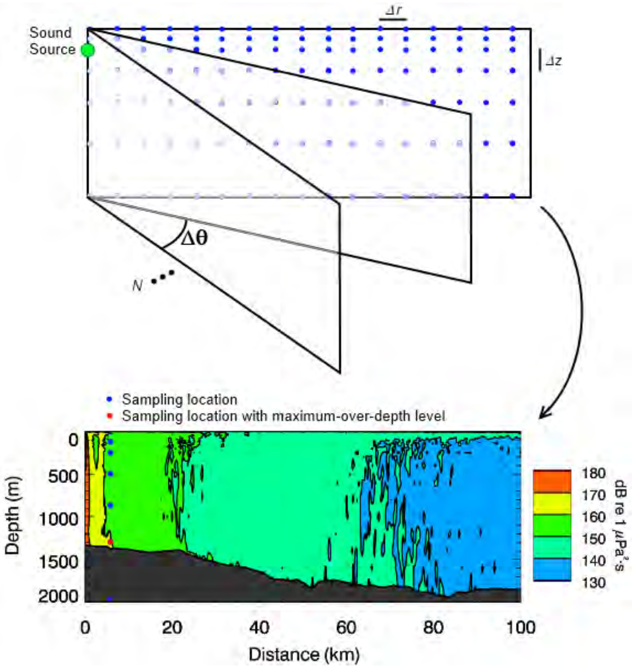
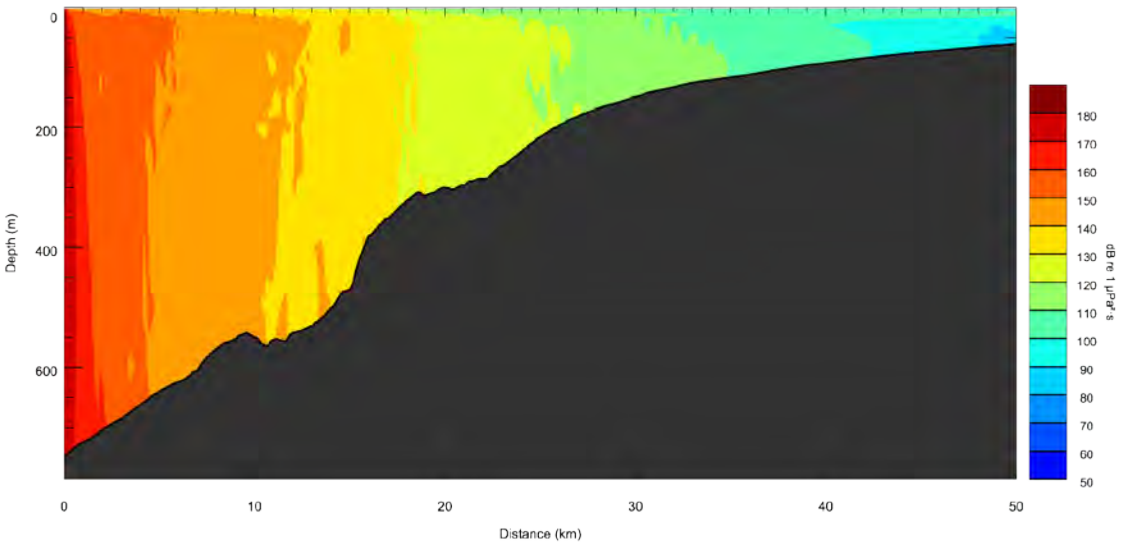


Figure C-3. The *N*×2-D and maximum-over-depth modeling approach.

C.2. Per-pulse Acoustic Field for Input to JASMINE

The transmission loss for exposure simulation is modeled along 16 radial profiles (angular step 22.5°) to a range of at least 100 km from the source location (i.e., to the edge of the larger modeling area). The horizontal step size along the radials is 30 m. At each radial sampling location, the sound field is sampled at up-to 87 depths, from 0.5 m down to the maximum water depth along the profile. The vertical step size in receiver depth is smaller near the surface, gradually increasing to as much as 100 m for the greatest depths. A total of 48 source frequencies (at the center of 1/3-octave-bands), from 1 Hz to 50 kHz were considered for the source array in the calculations of the broadband received levels. The broadband acoustic field passed as input to JASCO's Animal Simulation Model Including Noise Exposure (JASMINE) model is both in SPL and SEL metrics, and it was both range- and depth-dependent (Figure C-4).



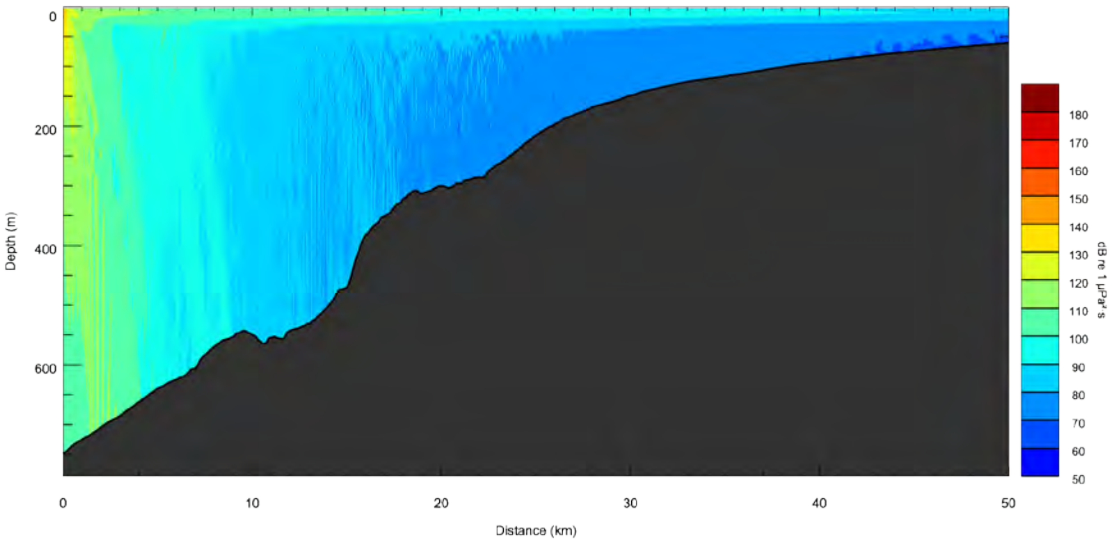


Figure C-4. An example of a per-pulse received sound exposure level (SEL) field along one radial, without frequency weighting (top) and with Type-III weighting for mid-frequency cetaceans (bottom) for the 4130 in^3 source array.

C.3. Frequency Dependence: Summing over 1/3-Octave-Bands

MONM treats frequency dependence by computing acoustic transmission loss at the center frequencies of 1/3-octave-bands. Many 1/3-octave-bands, starting at 10 Hz, are modeled to include most acoustic energy emitted by the source. At each center frequency, the transmission loss is modeled within each of the *N* vertical planes as a function of depth and range from the source. The 1/3-octave-band received per-pulse SELs are computed by subtracting the band transmission loss values from the SL in that frequency band.

Composite broadband received SELs are computed by combining the transmission loss (TL) values obtained from propagation modeling with MONM and SLs obtained from source modeling in each 1/3-octave-band and summing the band levels:

$$\mathrm{RL} = 10 \cdot \log_{10} \sum_{i=1}^{n} 10^{(\mathrm{SL}_i - \mathrm{TL}_i)/10} \qquad \text{(C-1)}$$

where *n* is the number of modeled 1/3-octave-bands, SL_i and TL_i are the source level and transmission loss in the respective 1/3-octave-band.

The frequency weighted received levels ($\mathrm{RL}_{\mathrm{MW}}$) were obtained by adding the relative levels (MW) to the equation:

$$\mathrm{RL}_{\mathrm{MW}} = 10 \cdot \log_{10} \sum_{i=1}^{N} 10^{(\mathrm{SL}_i - \mathrm{TL}_i + \mathrm{MW}_i)/10} \qquad \text{(C-2)}$$

Increasing frequency requires an increasingly finer computational grid, and, therefore, increased computational time. The transmission loss calculation for a single 2 kHz band can take as long as the time required for all other lower frequency bands combined. Transmission loss was modeled in 1/3-octave-bands from 1 Hz up to 50 kHz.

C.4. Converting SEL to rms SPL

The output from the modeling of the source is the sound field value in sound exposure level (SEL) units. A conversion factor is applied to estimate the rms (root-mean-square) sound pressure level (SPL). The rms SPL is conventionally based on an integration interval corresponding to the pulse length of the received signal, generally defined as the shortest time window containing 90% of the pulse energy (90% rms). Computation of rms levels from SEL requires knowledge of this pulse length, which in shallow water can be quite variable and dependent on several factors such as seabed composition, water sound speed profile, and distance from the source. A nominal conversion offset of +10 dB from SEL to rms SPL, corresponding to a pulse arrival duration of ~ 100 ms is commonly used. This value for the conversion offset is expected to be accurate for short-range distances (up to 2 km), based on field measurements. More accurate estimates of the conversion from SEL to rms SPL as a function of distance can be evaluated through full-waveform modeling.

Seismic airgun pulses typically lengthen in duration as they propagate away from their source, due to seabed and surface reflections, as well as other waveguide dispersion effects. The changes in pulse length affect SPL, therefore a full wave model must be used to reproduce the time domain signal and account for the changes in the pulse length. For the current study, JASCO's Full Waveform Range-dependent Acoustic Model (FWRAM) was used to model synthetic airgun pulses along the modeled radials. The synthetic pulses were analyzed to determine pulse length versus depth, distance, and azimuth from the source. The pulse lengths were averaged in 1 km bins along the radials, and the results were used to derive a conversion function between single-pulse SEL and SPL(T90) (Figure 7). The range- and depth-dependent conversion function was applied to predicted SEL per-pulse results from MONM to model SPL values in a 360° field.

Appendix D. Environmental Parameters

Parameters used for this study are the same as were used in modeling for the PEIS, including modeling locations, geoacoustic parameters, and the use of mean sound speed profiles.

D.1. Bathymetry

Water depths throughout the modeled area were obtained from the National Geophysical Data Center's U.S. Coastal Relief Model I (NDGC 2014) that extends up to about 200 km from the U.S. coast. These bathymetry data have a resolution of 3 arc-seconds (~ 80 × 90 m at the studied latitude). Bathymetry data for an area were extracted and re-gridded, using the minimum curvature method, onto a Universal Transverse Mercator (UTM) Zone 15 coordinate projection with a horizontal resolution of 50 × 50 m.

Two bathymetry grids were used for modeling. The first covered the West region (Boxes 1 and 2 in Figure 1); the second covered Central and East regions (Boxes 3–7 in Figure 1).

D.1.1. Multi-layer geoacoustic profile

The top sections of the sediment cover in the Gulf of Mexico are represented by layers of unconsolidated sediments at least several hundred meters thick. The grain size of the surficial sediments follows the general trend for the sedimentary basins: the grain size of the deposited sediments decreases with the distance from the shore. For the Shelf zone, the general surficial bottom type was assumed to be sand, for the Slope zone silt, and for the Deep zone clay. In constructing a geoacoustic model for input to MONM , a median value of φ was selected for each sediment type with the exception of the geoacoustic profile for the East-Shelf area. Because the grain size of the surficial sediment offshore Florida is consistently larger than in other shelf areas, we assumed φ equal to 1 for the sand in this zone.

Four sets of geoacoustic parameters were used in the acoustic propagation modeling:

- Center-West Shelf (Table D-1)
- East Shelf (Table D-2)
- Slope (Table D-3)
- Deep (Table D-4)

Table D-1. Shelf zone Center and West: Geoacoustic properties of the sub-bottom sediments as a function of depth, in meters below the seafloor, for fine sand. Within each depth range, each parameter varies linearly within the stated range.

Depth below seafloor (m)	Material	Density (g/cm^3)	P-wave speed (m/s)	P-wave attenuation **(dB/λ)**	S-wave speed (m/s)	S-wave attenuation **(dB/λ)**
0–20	Sand φ=2	1.61	1610	0.62	200	0.76
20–50		1.7	1900	1.44		
50–200		1.78	2090	1.77		
200–600		1.87	2500	2.31		
> 600		2.04	2500	2.67		

Table D-2. Shelf zone East: Geoacoustic properties of the sub-bottom sediments as a function of depth, in meters below the seafloor (mbsf), for medium-sand. Within each depth range, each parameter varies linearly within the stated range.

Depth below seafloor (m)	Material	Density (g/cm^3)	P-wave speed (m/s)	P-wave attenuation **(dB/λ)**	S-wave speed (m/s)	S-wave attenuation **(dB/λ)**
0–20	Sand *φ=1*	1.7	1660	0.76	200	1.13
20–50		1.78	2040	1.68		
50–200		1.87	2290	2.03		
200–600		1.96	2500	2.56		
> 600		2.04	2500	2.91		

Table D-3. Slope zone: Geoacoustic properties of the sub-bottom sediments as a function of depth, in meters below the seafloor (mbsf), for medium silt. Within each depth range, each parameter varies linearly within the stated range.

Depth below seafloor (m)	Material	Density (g/cm^3)	P-wave speed (m/s)	P-wave attenuation **(dB/λ)**	S-wave speed (m/s)	S-wave attenuation **(dB/λ)**
0–20	Silt *φ=6*	1.44	1515	0.33	150	0.22
20–50		1.7	1670	0.82		
50–200		1.7	1750	1.07		
200–600		1.87	1970	1.48		
> 600		2.04	2260	1.82		

Table D-4. Deep zone: Geoacoustic properties of the sub-bottom sediments as a function of depth, in meters below the seafloor (mbsf), for medium clay. Within each depth range, each parameter varies linearly within the stated range.

Depth below seafloor (m)	Material	Density (g/cm^3)	P-wave speed (m/s)	P-wave attenuation **(dB/λ)**	S-wave speed (m/s)	S-wave attenuation **(dB/λ)**
0–20	Clay *φ=9*	1.52	1472	0.17	100	0.06
20–50		1.7	1560	0.43		
50–200		1.78	1610	0.56		
200–600		1.87	1720	0.83		
> 600		2.04	1890	1.05		

D.1.2. Sound speed profiles

The sound speed profiles for the modeled sites were derived using the same source and method as described in Section 2.

We investigated variation in the sound speed profile throughout the year and produced a set of 12 sound speed profiles, each representing one month, in the Shelf, Slope, and Deep zones (Figure D-1). The set was divided into four seasons:

- Season 1: January, February, and March
- Season 2: April, May, and June
- Season 3: July, August, and September
- Season 4: October, November, and December

For each zone, a month was selected to represent the propagation conditions in the water column in each season (Table D-5).

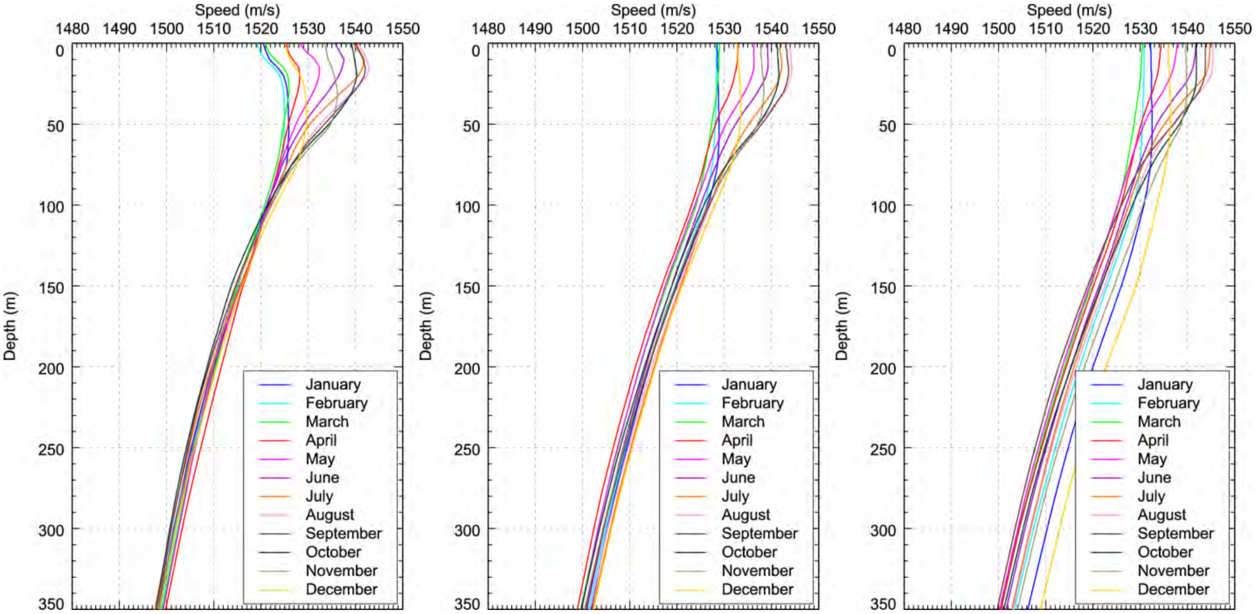


Figure D-1. Sound speed profiles at the (left) Shelf, (center) Slope, and (right) Deep zones, derived from data obtained from GDEM V 3.0 (Teague et al. 1990, Carnes 2009).

Table D-5. Representative months for each season and modeling zone.

Zone	SSP GDEM location	Season 1 (Jan to Mar)	Season 2 (Apr to Jun)	Season 3 (Jul to Sep)	Season 4 (Oct to Dec)
Shelf	25.5° N 90° W	Feb	May	Aug	Oct
Slope	27.25° N 90° W			Sep	Nov
Deep	28.5° N 90° W			Aug	Dec

ssp = sound speed profile

Acoustic fields were modeled using sound speed profiles for Season 1 and Season 3, and all three regions—East, Central, and West—used the same month. Profiles for Season 1 (February) provided the most conservative propagation environment because a surface duct, caused by upward refraction in the top 50–75 m, was present. Although a surface duct of this depth will not be able to prevent leakage of frequencies below 500–250 Hz (respectively), the ducting of frequencies above this cut off is important because these are the frequencies to which most marine mammals are most sensitive and the horizontal far-field acoustic projection from the airgun array seismic sources do have significant energy in this part of the spectrum. The modeling results obtained when the duct was present, therefore, represent the most precautionary propagation environment. Profiles for Season 3 (August or September) provided the least conservative results because they have weak to no sound channels at the surface and are strongly downward refracting in the top 200 m. Only the top 100 m of the water column are affected by the seasonal variation in the sound speed.

The possibility of separately modeling the spring and fall seasons was investigated; however, the results for spring and fall are almost identical to the results for summer, which were used as a proxy for the spring and fall results.

D.1.2.1. Sound speed profiles for box centers

Sound speed profiles were gathered from the center of each modeling box for Seasons 1 and 3. Table D-6 presents the months modeled for each of these seasons. Figure D-2 to Figure D-3 show the sound speed profiles for Seasons 1 and 3, respectively.

Table D-6. Modeling seasons for each box.

Box	Region	Zone	Season 1	Season 3
1	West	Shelf	Feb	Aug
2		Slope		Sep
3	Central	Shelf		Aug
4		Slope		Sep
5		Deep		Aug
6	East	Slope		Sep
7		Shelf		Aug

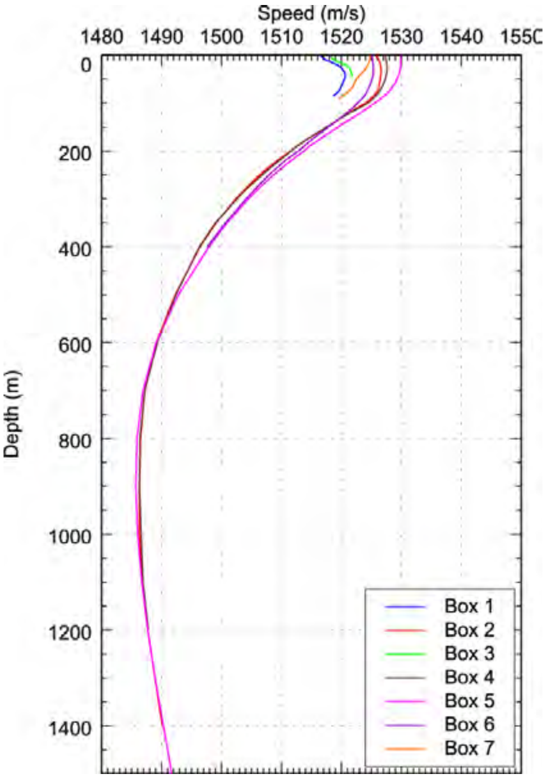


Figure D-2. Sound speed profiles at modeling boxes, Season 1, derived from data obtained from GDEM V 3.0 (Teague et al. 1990, Carnes 2009).

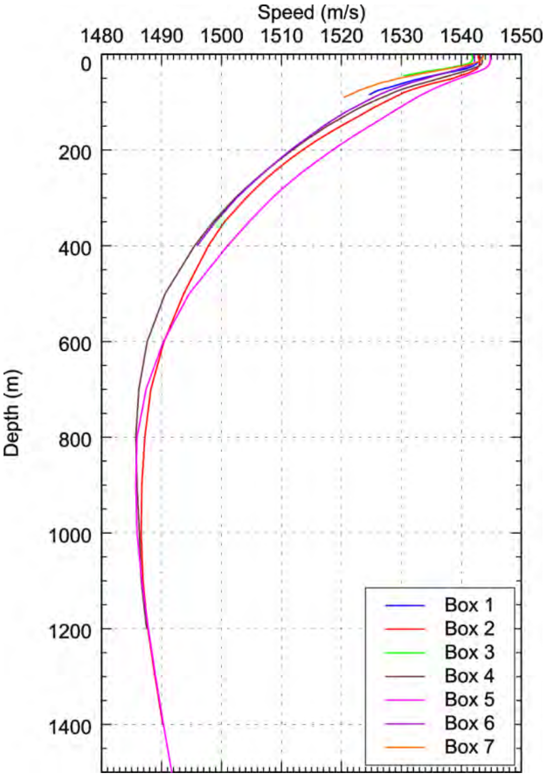


Figure D-3. Sound speed profiles at modeling boxes, Season 3, derived from data obtained from GDEM V 3.0 (Teague et al. 1990, Carnes 2009).

D.1.2.2. Sound speed profiles for acoustic modeling sites along transects

Sound speed profiles were obtained at three locations along each transect. Profiles were selected for Season 1 and Season 3. The months modeled for each season are presented in Table D-7. Figure D-4 to Figure D-6 show the sound speed profiles for transects in the West, Central, and East regions respectively.

Table D-7. Modeling seasons for the sites along transects.

Region	Zone	Season 1	Season 3
West	Shelf	Feb	Aug
	Slope		Sep
	Shelf		Aug
Central	Shelf		Aug
	Slope		Sep
	Shelf		Aug
East	Shelf		Aug
	Slope		Sep
	Deep		Aug

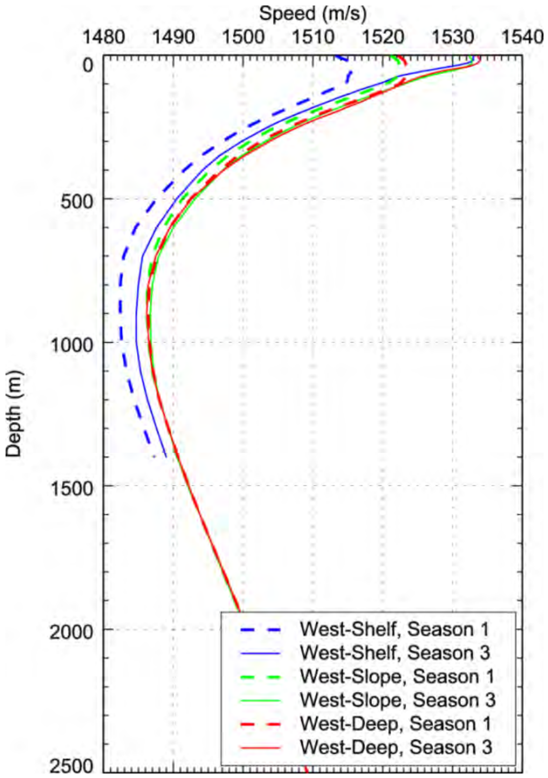


Figure D-4. Sound speed profiles along the West transect, derived from data obtained from GDEM V 3.0 (Teague et al. 1990, Carnes 2009).

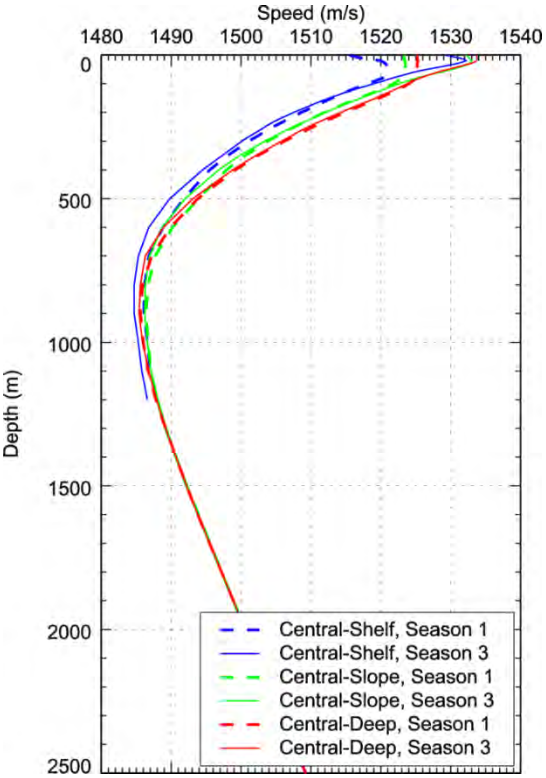


Figure D-5. Sound speed profiles along Central transect, derived from data obtained from GDEM V 3.0 (Teague et al. 1990, Carnes 2009).

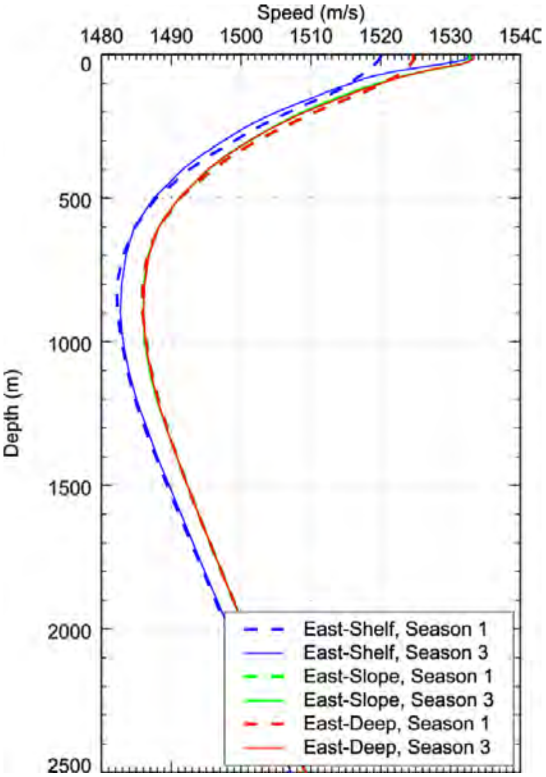


Figure D-6. Sound speed profiles along East transect, derived from data obtained from GDEM V 3.0 (Teague et al. 1990, Carnes 2009).

Appendix E. Auditory (Frequency) Weighting Functions

Described in Section 2.4.2, weighting functions are applied to the sound spectra under consideration to weight the importance of received sound levels at particular frequencies in a manner reflective of an animal's sensitivity to those frequencies (Nedwell and Turnpenny 1998, Nedwell et al. 2007). In this study, multiple weighting functions were used. Type I, also referred to as M-weighting (Southall et. 2007), was used to obtain rms SPL sound fields for gauging potential behavioral disruption and likelihood of aversion (Section E.1.1). Type III weighting (NMFS 2016) was used to assess potential injurious exposure from the sources.

E.1.1. Type I marine mammal frequency weighting functions

Auditory weighting functions for marine mammals—called *M-weighting* functions—were proposed by Southall et al. (2007). Functions were defined for five hearing groups of marine mammals:

- Low-frequency cetaceans (LFCs)—mysticetes (baleen whales)
- Mid-frequency cetaceans (MFCs)—some odontocetes (toothed whales)
- High-frequency cetaceans (HFCs)—odontocetes specialized for using high-frequencies
- Pinnipeds in water—seals, sea lions, and walrus
- Pinnipeds in air (not addressed here)

The M-weighting functions have unity gain (0 dB) through the passband and their high and low frequency roll-offs are approximately –12 dB per octave. The amplitude response in the frequency domain of each M-weighting function is defined by:

$$G(f) = -20\log_{10}\left[\left(1+\frac{a^2}{f^2}\right)\left(1+\frac{f^2}{b^2}\right)\right] \qquad \text{(E-1)}$$

where *G*(f) is the weighting function amplitude (in dB) at the frequency *f* (in Hz), and *a* and *b* are the estimated lower and upper hearing limits, respectively, which control the roll-off and passband of the weighting function. The parameters *a* and *b* are defined uniquely for each hearing group (Table E-1). The auditory weighting functions recommended by Southall et al. (2007) are shown in Figure E-1.

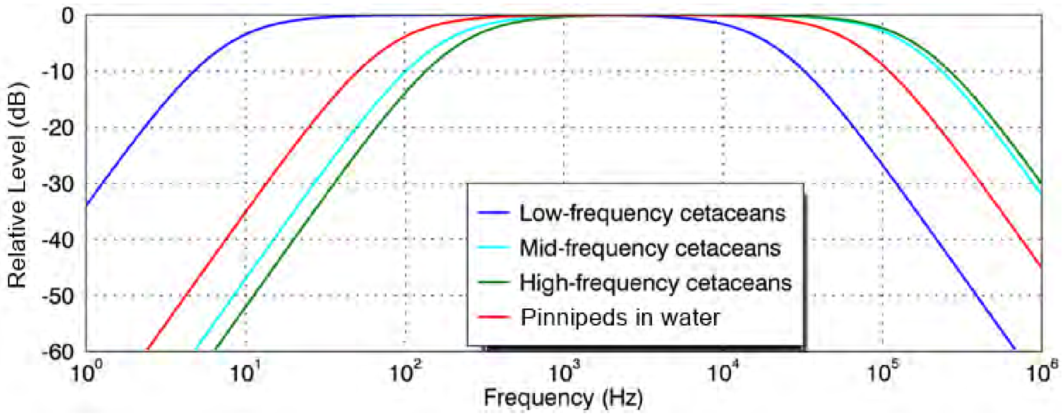


Figure E-1. Auditory weighting functions for functional marine mammal hearing groups as recommended by Southall et al. (2007).

Table E-1. Parameters for the auditory weighting functions recommended by Southall et al. (2007).

Hearing group	Southall et al. (2007)	
	a (Hz)	*b* (Hz)
Low-frequency cetaceans (LFC)	7	22,000
Mid-frequency cetaceans (MFC)	150	160,000
High-frequency cetaceans (HFC)	200	180,000
Pinnipeds in water (Pw)	75	75,000

E.1.2. Type III marine mammal frequency weighting functions

In 2015, a U.S. Navy technical report by Finneran (2015) recommended new auditory weighting functions. The overall shape of the auditory weighting functions is similar to human A-weighting functions, which follows the sensitivity of the human ear at low sound levels. The new frequency-weighting function is expressed as:

$$G(f) = K + 10\log_{10}\left[\left(\frac{(f/f_{lo})^{2a}}{\left[1+(f/f_{lo})^2\right]^a\left[1+(f/f_{hi})^2\right]^b}\right)\right] \tag{E-2}$$

Finneran (2015) proposed five functional hearing groups for marine mammals in water: low-, mid-, and high-frequency cetaceans, phocid pinnipeds, and otariid pinnipeds. The parameters for these frequency-weighting functions were further modified the following year (Finneran 2016) and were adopted in NOAA's technical guidance that assesses noise impacts on marine mammals (NMFS 2016). Table E-2 lists the frequency-weighting parameters for each hearing group; Figure E-2 shows the resulting frequency-weighting curves.

Table E-2. Parameters for the auditory weighting functions recommended by NMFS (2016).

Hearing group	*a*	*b*	f_{lo} (Hz)	f_{hi} (kHz)	*K* (dB)
Low-frequency cetaceans	1.0	2	200	19,000	0.13
Mid-frequency cetaceans	1.6	2	8,800	110,000	1.20
High-frequency cetaceans	1.8	2	12,000	140,000	1.36
Phocid pinnipeds in water	1.0	2	1,900	30,000	0.75
Otariid pinnipeds in water	2.0	2	940	25,000	0.64

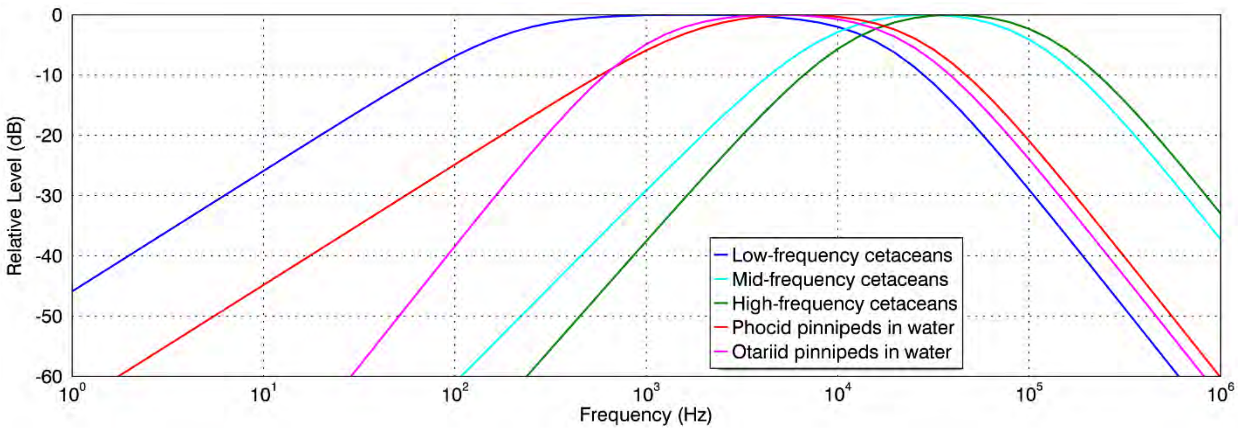


Figure E-2. Auditory weighting functions for functional marine mammal hearing groups as recommended by NMFS (2016).

Appendix F. Animal Simulation and Acoustic Exposure Model

To assess the risk of impacts from exposure, an estimate of received sound levels for the animals in the area during operations is required. Sound sources move and so do animals. The sound fields may be complex and the sound received by an animal is a function of where the animal is at any given time. To a reasonable approximation, the location of the sound source(s) is known and acoustic modeling can be used to predict the 3-D sound field (Appendix B). The location and movement of animals within the sound field, however, is unknown. Realistic animal movement within the sound field can be simulated, and repeated random sampling (Monte Carlo)—achieved by simulating many animals within the operations area—used to estimate the sound exposure history of animals during the operation. Monte Carlo methods provide a heuristic approach for determining the probability distribution function (PDF) of complex situations, such as animals moving in a sound field. The probability of an event's occurrence is determined by the frequency with which it occurs in the simulation. The greater the number of random samples, in this case the more simulated animals (animats), the better the approximation of the PDF. Animats are randomly placed, or seeded, within the simulation boundary at a specified density (animats/km^2). The animat density is much higher than the real-world density to ensure good representation of the PDF. The resulting PDF is scaled using the real-world density.

Several models for marine mammal movement have been developed (Ellison et al. 1987, Frankel et al. 2002, Houser 2006). These models use an underlying Markov chain to transition from one state to another based on probabilities determined from measured swimming behavior. The parameters may represent simple states, such as the speed or heading of the animal, or complex states, such as likelihood of participating in foraging, play, rest, or travel. Attractions and aversions to variables like anthropogenic sounds and different depth ranges can be included in the models.

Analysis in this report uses the JASCO Animal Simulation Model Including Noise Exposure (JASMINE) 2017. JASMINE uses the same animal movement algorithms as the Marine Mammal Movement and Behavior (3MB) model (Houser 2006), but has been extended for use with JASCO-formatted acoustic fields, inclusion of source tracks, and for animats to change behavioral states based on modeled variables such as received level. JASMINE also includes aversion in response to realistic received levels.

F.1. Animal Movement Parameters

JASMINE uses previously measured behavior to forecast behavior in new situations and locations. The parameters used for forecasting realistic behavior are determined (and interpreted) from marine species studies (e.g., tagging studies). Each parameter in the model is described as a probability distribution. When limited or no information is available for a species parameter, a Gaussian or uniform distribution may be chosen for that parameter. For the Gaussian distribution, the user determines the mean and standard deviation of the distribution from which parameter values are drawn. For the uniform distribution, the user determines the maximum and minimum distribution from which parameter values are drawn. When detailed information about the movement and behavior of a species are available, a user-created distribution vector, including cumulative transition probabilities, may be used (referred to here as a vector model; Houser 2006). Different sets of parameters can be defined for different behavior states. The probability of an animat starting out in or transitioning into a given behavior state can in turn be defined in terms of the animat's current behavioral state, depth, and the time of day. In addition, each travel parameter and behavioral state has a termination function that governs how long the parameter value or overall behavioral state persists in simulation.

The parameters used in JASMINE describe animal movement in both the vertical and horizontal planes. The parameters relating to travel in these two planes are briefly described below.

F.1.1. Travel sub-models

Direction–determines the animat's choice of direction in the horizontal plane. Sub-models are available for determining the bearing of animats, allowing for movement to range from strongly biased to undirected. A random walk model can be used for behaviors with no directional preference, such as feeding and playing. In a random walk, all bearings are equally likely at each parameter transition time step. A correlated random walk can be used to smooth the changes in bearing by using the current bearing as the mean of the distribution from which to draw the next heading. An additional variant of the correlated random walk is available that includes a directional bias for use in situations where animals have a preferred absolute direction, such as migration. A user-defined vector of directional probabilities can also be defined to control animat bearing. For more detailed discussion of these parameters, see Houser (2006) and Houser and Cross (1999).

Travel rate–defines the rate of travel of an animat in the horizontal plane. When combined with vertical speed and dive depth, the dive profile of the animat is produced.

F.1.2. Dive sub-models

Ascent Rate–defines the rate of travel of an animat in the vertical plane during the ascent portion of a dive.

Descent Rate–defines the rate of travel of an animat in the vertical plane during the descent portion of a dive.

Depth–defines the maximum depth to which an animat will dive.

Bottom Following–determines whether an animat returns to the surface once reaching the ocean floor, or whether it follows the contours of the bathymetry.

Reversals–determines whether multiple vertical excursions occur once reaching the maximum dive depth. This behavior is used to emulate the foraging behavior of some marine mammal species at depth. Reversal-specific ascent and descent rates may be specified.

Surface Interval–determines the amount of time spent at the surface prior to performing another dive.

F.1.3. Boundaries

Ideally, the simulation area would be large enough to include ranges in which every animal that could approach the survey area during the operation would be included. Similarly, any animat that was exposed could not subsequently reach the boundary of the simulation during the operation. There are limits to the simulation area and computational overhead increases with area. For practical reasons, the simulation area for potential behavioral responses was limited to a maximum range of approximately 55 km from the modeled source tracks. In the simulation, every animat that reaches a border is replaced by another animat entering at the opposing border—e.g., an animat crossing the northern border of the simulation is replaced by one entering the southern border at the same longitude. Where this places the animat in an inappropriate water depth, the animat is randomly placed on the map at a depth suited to its species definition. The exposure history of all animats (including those leaving the simulation and those entering) are kept for exposure analysis. This approach maintains a consistent animat density and allows for longer integration periods with smaller simulation areas. It differs from simulating a larger area in that animats that cross the border are not allowed to re-enter the simulation (they are replaced by new animats) so the possibility of an animat leaving the area after exposure and then re-entering later to be re-exposed is excluded.

F.1.4. Aversion

Animals may avoid loud sounds by moving away from the source. A group of experts was convened to create a framework for assessing acoustic impacts to marine mammals in the GOM (Southall 2016). In this Risk Assessment Framework (RAF), it is suggested that aversion be included in simulations and the results be compared to simulations without aversion. While there are few data on which aversive behavior can be based, the RAF includes some aversion parameters, based on the Wood et al. (2012) behavioral step function. We follow the RAF aversion parameters (Table F-1). Animats avert by changing their headings by a fixed amount away from the source, with higher received levels associated with a greater deflection, and animats remain in the aversive state for a specified amount of time, depending on the level of exposure that triggered aversion (Table F-1). During this time, travel parameters are recalculated periodically as with normal behaviors. At the end of the aversion interval, the animat once again applies the parameters in Table F-1 and, depending on the current level of exposure, either begins another aversion interval or transitions to a non-aversive behavior; while aversion begins immediately, transition to a regular behavior occurs at the end of the next surfacing interval, consistent with regular behavior transitions.

Table F-1. Aversion parameters for the animal movement simulation based on Wood et al. (2012) behavioral response criteria

Probability of aversion	Received sound level (SPL, dB re 1 µPa)			Change in course (°)	Duration of aversion(s)
	Beaked whales	All other marine mammals	Sea turtles		
10%	100	140	146	10	300
50%	120	160	166	20	60
90%	140	180	186	30	30

F.2. Marine Mammal Species-Specific Details

Most marine mammals likely to be near the operations site are mid-frequency odontocetes. Bryde's whales (mysticete) is the only low-frequency animal and the *Kogia* species are the only high-frequency animals. Sperm whale is the only endangered species, although all of the marine mammals are protected. Details for the representative species are listed below.

F.2.1. Bryde's whales (*Balaenoptera edeni*)

Bryde's whales occur in tropical and warm temperate oceans around the world (Atlantic, Indian, and Pacific) from about 40° S to 40° N (Reeves et al. 2002, Jefferson et al. 2008). Southeast Atlantic and northwest Pacific populations migrate seasonally, moving toward higher latitudes during the summer and toward the equator during the winter. Migration patterns of the other populations are poorly known (Reilly et al. 2008). Bryde's whales are usually sighted individually or in pairs, but there are reports of loose aggregations of up to twenty animals associated with feeding areas. They feed on plankton, crustaceans, and schooling fish. Bryde's whales use different methods to feed, including skimming the surface, lunging, and creating bubble nets. They regularly dive for about 5–15 min (maximum of 20 min) and are capable of reaching depths up to 300 m during dives (Reeves et al. 2002, Jefferson et al. 2008).

Few Bryde's whale sightings have been recorded in the Gulf of Mexico. During aerial surveys conducted from summer 1992 through spring 1994, only one Bryde's whale was recorded at ~ 200 m water depth (Mullin et al. 2004). During ship-based spring surveys from 1991–2001 a total of 17 (on- and off-transect) sightings of Bryde's whales with an average group size of 2 animals was recorded, all concentrated along the shelf-edge in water depths ranging from ~200 to ~300 m. About 95% were sighted in the De Soto Canyon area, northeast Gulf of Mexico (Maze-Foley and Mullin 2006). One sighting of two animals was observed in the De Soto Canyon area from similar ship-based surveys in 2003–2004 (Mullin 2007) and three sightings in 2009 (Waring et al. 2013). Three groups of Bryde's whales were observed during the Atlantic Marine Assessment Program for Protected Species (AMAPPS) survey in the Gulf of Mexico, all on 31 July and in the De Soto Canyon (Širović et al. 2014). Because the few sightings of Bryde's whales in the Gulf of Mexico occurred in the De Soto Canyon area, over 300 km from the survey site, Bryde's whales are not expected to receive acoustic energy because of the project.

F.2.1.1. Behavioral parameters for animat modeling

Table F-2. Bryde's whales: Data values and references for inputs in JASMINE software to create diving behavior (number values represent Means (SD) unless otherwise indicated).

Behavior	Variable	Value	Reference
Deep	Travel direction	Correlated random walk	Approximated
	Perturbation value	10	Approximated
	Termination coefficient	0.2	Approximated
	Travel rate (m/s)	Random 0.81–1.53	Murase et al. (2015)
	Ascent rate (m/s)	Gaussian 0.95 (0.55)	Alves et al. (2010)
	Descent rate (m/s)	Gaussian 1.25 (0.4)	Alves et al. (2010)
	Average depth (m)	Gaussian 314 (61.5)	Alves et al. (2010)
	Bottom following	No	Approximated
	Reversals	Gaussian 1.5 (1.5)	Alves et al. (2010)
	Probability of reversal	0.7	Approximated
	Reversal ascent dive rate (m/s)	1.0 (0.2)	Approximated
	Reversal descent dive rate (m/s)	1.0 (0.2)	Approximated
	Time in reversal (s)	Gaussian 50.1 (45.3)	Alves et al. (2010)
	Surface interval (s)	Random, 120 - 300	Alves et al. (2010)
	Bout duration (s)	Gaussian 600 (120) Night Gaussian 3600 (420) Day	Approximated
Shallow	Travel direction	Correlated random walk	Ward (1999)
	Perturbation value	10	Approximated
	Termination coefficient	0.2	Approximated
	Travel rate (m/s)	Random 0.81–1.53	Murase et al. (2015)
	Ascent rate (m/s)	Gaussian 0.95 (0.55)	Alves et al. (2010)
	Descent rate (m/s)	Gaussian 1.25 (0.4)	Alves et al. (2010)
	Average depth (m)	Random, maximum = 40	Alves et al. (2010)
	Bottom following	No	Approximated
	Reversals	No	Approximated
	Surface interval (s)	Random, 141 - 236	Di Sciara (1983)
	Bout duration (s)	Gaussian 3600 (420) Day Gaussian 0 (0) Night	Approximated
General	Shore following (m)	20	Gonçalves et al. (2016)
	Depth limit on seeding (m)	20 (minimum), 3000 (maximum)	Gonçalves et al. (2016)

Approximated: value based on the best fit for diving profile. Those values were not available from literature but were estimated producing a diving profile similar to D-tag results, for example.

F.2.2. Sperm whales (*Physeter macrocephalus*)

The sperm whale is listed as endangered under the Endangered Species Act and depleted under the Marine Mammal Protection Act (MMPA) throughout its entire range. Due to commercial whaling at a large scale from the early 18th to 20th century, sperm whale numbers declined globally. In the Gulf of Mexico, sperm whales were commercially hunted by American whalers until the early 1900s (Townsend 1935). Sperm whale population sizes have increased since commercial whaling ceased, however, they have not reached projected historical numbers (Whitehead 2002).

Sperm whales of all ages and both sexes occur year-round in the Gulf of Mexico, where they are the most common large whale species (Mullin et al. 2004, Waring et al. 2010). Systematic aerial and ship surveys indicate that they inhabit continental slope and oceanic waters and they generally occur in waters deeper than 1000 m (Mullin and Fulling 2004, Mullin et al. 2004, Maze-Foley and Mullin 2006, Mullin 2007). Movements from satellite tagged sperm whales showed that most whales frequented waters of 700–1000 m deep, but that animals were also sighted in waters of 3000 m (Mate and Ortega-Ortiz 2004).

The northern Gulf of Mexico stock is considered by some to be distinct from the U.S. Atlantic stock (Waring et al. 2010). Findings from the Sperm Whale Seismic Study on movement patterns, genetic structure, size, photo-identification data, and vocalizations support the concept of two separate stocks (Jochens et al. 2008). The site fidelity of the Gulf of Mexico sperm whales appears to be high. Although genetic evidence shows that male sperm whales move in and out the Gulf (Engelhaupt et al. 2009), tracks from 39 satellite-tagged northern Gulf sperm whales monitored for up to 607 days displayed no seasonal migrations and tracked only one animal (a male) that left the Gulf of Mexico (Mate and Ortega-Ortiz 2004). During ship-based surveys in continental slope and oceanic waters, 164 groups with an average of 2–3 animals were observed in 1991–2001, and 85 groups with an average of 4 animals were observed in 2003–2004 (Maze-Foley and Mullin 2006, Mullin 2007). In both surveys, sperm whales were frequently observed in the proposed operations area.

Sperm whales feed primarily on squid and occasionally on fish (Wynne and Schwartz 1999). They make deep and long dives reaching depths of ~ 3000 m (Jefferson et al. 2008), but with average diving depths of about 700 m (Watwood et al. 2006). Although dive durations can be as long as 2 h, most recorded dives lasted about 30–45 min (Thode et al. 2002, Watwood et al. 2006, Palka and Johnson 2007). Sperm whales are mid-frequency cetaceans with functional hearing sensitivity estimated to range from 150 Hz to 160 kHz (Southall et al. 2007).

The most prevalent vocalization pattern of sperm whales is the ‘usual' click, which is produced by foraging whales as echolocation to target prey at depth (Watwood et al. 2006). Socializing whales sometimes produce short stereotyped sequences of clicks, termed ‘codas’, which have also been recorded at the beginning of foraging dives and just prior to surfacing. Sperm whale social units have different repertoires or dialects as they show different usage patterns of specific codas (Whitehead and Rendell 2004, Schulz et al. 2011). Most clicks and codas produced by sperm whales are in the 8–25 kHz frequency range (Madsen et al. 2002).

F.2.2.1. Behavioral parameters for animat modeling

Table F-3. Sperm whales: Data values and references for inputs in JASMINE software to create diving behavior (number values represent Means (SD) unless otherwise indicated).

Behavior	Variable	Value	Reference
Deep Foraging Dive	Travel direction	Correlated random walk	Approximated
	Perturbation value	10	Approximated
	Termination coefficient	0.2	Approximated
	Travel rate (m/s)	Gaussian 0.88 (0.27)	Miller et al. (2004)
	Ascent rate (m/s)	Gaussian 1.3 (0.2)	Watwood et al. (2006)
	Descent rate (m/s)	Gaussian 1.1 (0.2)	Watwood et al. (2006)
	Average depth (m)	Gaussian 546.9 (130)	Watwood et al. (2006)

Behavior	Variable	Value	Reference
	Bottom following	No	Approximated
	Reversals	Gaussian 8.2 (4.2)	Aoki et al. (2007)
	Probability of reversal	1	Approximated
	Reversal ascent dive rate (m/s)	1.8 (0.5)	Aoki et al. (2007)
	Reversal descent dive rate (m/s)	1.8 (0.5)	Aoki et al. (2007)
	Time in reversal (s)	Gaussian 141 (82.7)	Aoki et al. (2007) Amano and Yoshioka (2003)
	Surface interval (s)	Gaussian 486 (156)	Watwood et al. (2006)
	Bout duration (s)	Gaussian 42012 (20820)	Approximated
V Dive	Travel direction	Correlated random walk	Approximated
	Perturbation value	10	Approximated
	Termination coefficient	0.2	Approximated
	Travel rate (m/s)	Gaussian 0.88 (0.27)	Miller et al. (2004)
	Ascent rate (m/s)	Gaussian 0.67 (0.43)	Amano and Yoshioka (2003)
	Descent rate (m/s)	Gaussian 0.85 (0.05)	Amano and Yoshioka (2003)
	Average depth (m)	Gaussian 282.7 (69.9)	Amano and Yoshioka (2003)
	Bottom following	No	Approximated
	Reversals	No	Approximated
	Surface interval (s)	Gaussian 408 (114)	Amano and Yoshioka (2003)
	Bout duration (s)	Gaussian 2286 (384)	Approximated
Inactive Bottom Time	Travel direction	Correlated random walk	Approximated
	Perturbation value	10	Approximated
	Termination coefficient	0.2	Approximated
	Travel rate (m/s)	Gaussian 0.88 (0.27)	Miller et al. (2004)
	Ascent rate (m/s)	Gaussian 1.13 (0.07)	Amano and Yoshioka (2003)
	Descent rate (m/s)	Gaussian 1.4 (0.13)	Amano and Yoshioka (2003)
	Average depth (m)	Gaussian 490 (74.6)	Amano and Yoshioka (2003)
	Bottom following	No	Approximated
	Reversals	Gaussian 1 (0)	Approximated
	Probability of reversal	1	Approximated
	Reversal ascent dive rate (m/s)	0.1 (0.1)	Approximated
	Reversal descent dive rate (m/s)	0.1 (0.1)	Approximated
	Time in reversal (s)	Gaussian 1188 (174.6)	Amano and Yoshioka (2003)
	Surface interval (s)	Gaussian 486 (156)	Watwood et al. (2006)
	Bout duration (s)	Gaussian 6192 (4518)	Approximated
Surface active	Travel direction	Correlated random walk	Approximated
	Perturbation value	10	Approximated
	Termination coefficient	0.2	Approximated
	Travel rate (m/s)	Gaussian 0.88 (0.27)	Miller et al. (2004)
	Ascent rate (m/s)	Gaussian 0.67 (0.43)	Amano and Yoshioka (2003)
	Descent rate (m/s)	Gaussian 0.85 (0.05)	Amano and Yoshioka (2003)
	Average depth (m)	Gaussian 25 (25)	Amano and Yoshioka (2003)

Behavior	Variable	Value	Reference
	Bottom following	No	Approximated
	Reversals	No	Approximated
	Surface interval (s)	Gaussian 408 (114)	Amano and Yoshioka (2003)
	Bout duration (s)	Gaussian 3744 (2370)	Approximated
Surface Inactive–Head Up	Travel direction	Correlated random walk	Approximated
	Perturbation value	10	Approximated
	Termination coefficient	0.2	Approximated
	Travel rate (m/s)	Gaussian 0 (0)	Approximated
	Ascent rate (m/s)	Gaussian 0.1 (0.1)	Miller et al. (2008)
	Descent rate (m/s)	Gaussian 0.1 (0.1)	Miller et al. (2008)
	Average depth (m)	Gaussian 8.6 (4.8)	Miller et al. (2008)
	Bottom following	No	Approximated
	Reversals	Gaussian 1 (0)	Approximated
	Probability of reversal	1	Approximated
	Reversal ascent dive rate (m/s)	0 (0)	Miller et al. (2008)
	Reversal descent dive rate (m/s)	0 (0)	Miller et al. (2008)
	Time in reversal (s)	Gaussian 708 (522)	Miller et al. (2008)
	Surface interval (s)	Gaussian 462 (360)	Miller et al. (2008)
	Bout duration	T50 = 486 (s), k=0.9	Approximated
Surface Inactive–Head Down	Travel direction	Correlated random walk	Approximated
	Perturbation value	10	Approximated
	Termination coefficient	0.2	Approximated
	Travel rate (m/s)	Gaussian 0 (0)	Approximated
	Ascent rate (m/s)	Gaussian 0.1 (0.1)	Miller et al. (2008)
	Descent rate (m/s)	Gaussian 0.1 (0.1)	Miller et al. (2008)
	Average depth (m)	Gaussian 16.5 (4.9)	Miller et al. (2008)
	Bottom following	No	Approximated
	Reversals	Gaussian 1 (0)	Approximated
	Probability of reversal	1	Approximated
	Reversal ascent dive rate (m/s)	0 (0)	Miller et al. (2008)
	Reversal descent dive rate (m/s)	0 (0)	Miller et al. (2008)
	Time in reversal (s)	Gaussian 804 (522)	Miller et al. (2008)
	Surface interval (s)	Gaussian 462 (360)	Miller et al. (2008)
	Bout duration	T50 = 486 (s), k=0.9	Approximated
General	Depth limit on seeding (m)	500	Herzing and Elliser (2016)

Approximated: value based on the best fit for diving profile. Those values were not available from literature but were estimated producing a diving profile similar to D-tag results for example.

F.2.3. Beaked whales

Four species of beaked whales could be encountered in the Gulf of Mexico. The Cuvier's beaked whale (*Ziphius cavirostris*), and three of the *Mesoplodon* genus: Blainville's beaked whale (*Mesoplodon densirostris*), Gervais' beaked whale (*Mesoplodon europaeus*), and Sowerby's beaked whale (*Mesoplodon bidens*). Sowerby's beaked whale (*Mesoplodon bidens*), however, is a rare visitor to the area. The only recorded occurrence of the Sowerby's beaked whale in the Gulf of Mexico was a stranded one in Gulf County, Florida (Wursig et al. 2000). Sowerby's beaked whales are not considered further analyzed. Beaked whales are found in temperate, tropical, and subtropical waters. They occur year-round in the Gulf of Mexico where they frequent deep pelagic waters (Wynne and Schwartz 1999). The depth range at which most beaked whale sightings were recorded was 500–3500 m, with an average depth of >1000 m (Maze-Foley and Mullin 2006). Beaked whales make the longest and deepest dives of any whale species, often diving to depths >300 m (Hooker and Baird 1999, Baird et al. 2006a, Baird et al. 2006b, Tyack et al. 2006, Baird et al. 2008). They are dive feeders, usually feeding on squid, but also on fish and crustaceans (Wynne and Schwartz 1999). During eight aerial line-transect surveys conducted from summer 1992 to spring 1994, covering 85,815 km^2 in the north-central and north-western Gulf of Mexico, 11 beaked whales were sighted. One was a Cuvier's beaked whale, four were Mesoplodon spp., and eight were of unidentified beaked whales (Mullin et al. 2004). Ship-based line-transect surveys in 1991–2001 recorded 15 Cuvier's beaked whale sightings, 29 Mesoplodon spp. sightings, and 19 unidentified beaked whale sightings, all with an average groups size of 2 animals (Maze-Foley and Mullin 2006). Observations from similar 2003–2004 ship-based survey data recorded 2 Cuvier's beaked whale sightings and 2 *Mesoplodon spp* sightings, with an average group size of 3 animals, and 15 unidentified beaked whale sightings with an average group size of 2 animals (Mullin 2007).

Information on hearing sensitivity of beaked whales is somewhat limited. Most data are available from stranded whales, using audio evoked potential. The Gervais' beaked whale was found to be most sensitive to high frequency signals between 40 and 80 kHz, but produced smaller evoked potentials to 5 kHz, the lowest frequency tested (Cook et al. 2006, Finneran et al. 2009). Blainville's beaked whale sounds included one frequency-modulated whistle and three frequency- and amplitude-modulated pulsed sounds, with energy between 6 and 16 kHz (Rankin and Barlow 2007). Beaked whale hearing sensitivity measured through audio evoked potential was like those measured in other echolocating odontocetes.

F.2.3.1. Behavioral parameters for animat modeling

Table F-4. Cuvier's beaked whales: Data values and references for inputs in JASMINE software to create diving behavior (number values represent Means (SD) unless otherwise indicated).

Behavior	Variable	Value	Reference
Deep Foraging Dive	Travel direction	Correlated random walk	Approximated
	Perturbation value	10	Approximated
	Termination coefficient	0.2	Approximated
	Travel rate (m/s)	Gaussian 1.5 (0.5)	Approximated
	Ascent rate (m/s)	Gaussian 0.69 (0.19)	Tyack et al. (2006) Baird et al. (2006b)
	Descent rate (m/s)	Gaussian 1.47 (0.13)	Tyack et al. (2006) Baird et al. (2006)
	Average depth (m)	Gaussian 1070 (317)	Tyack et al. (2006)
	Bottom following	No	Approximated
	Reversals	Gaussian 20 (2)	Tyack et al. (2006)
	Probability of reversal	0.95	Approximated
	Reversal ascent dive rate (m/s)	0.8 (0.2)	Madsen et al. (2005)
	Reversal descent dive rate (m/s)	0.8 (0.2)	Madsen et al. (2005)

	Time in reversal (s)	Gaussian 40 (20)	Tyack et al. (2006)
	Surface interval (s)	Gaussian 474 (996)	Tyack et al. (2006)
	Bout duration	T_{50} = 1200 (s), k=10	Approximated
Shallow Dive	Travel direction	Correlated random walk	Approximated
	Perturbation value	10	Approximated
	Termination coefficient	0.2	Approximated
	Travel rate (m/s)	Gaussian 1.5 (0.5)	Approximated
	Ascent rate (m/s)	Gaussian 0.61 (0.2)	Baird et al. (2006b), Tyack et al. (2006)_ENREF_15
	Descent rate (m/s)	Gaussian 0.53 (0.24)	Baird et al. (2006b), Tyack et al. (2006)_ENREF_15
	Average depth (m)	Gaussian 221 (100)	Tyack et al. (2006)
	Bottom following	No	Approximated
	Reversals	No	Approximated
	Surface interval (s)	Gaussian 474 (996)	Tyack et al. (2006)
	Bout duration (s)	Gaussian 3780 (1860)	Tyack et al. (2006)
General	Depth limit on seeding (m)	1381	Baird et al. (2006b)

Approximated: value based on the best fit for diving profile. Those values were not available from literature but were estimated producing a diving profile similar to D-tag results for example.

Table F-5. *Mesoplodon* beaked whales: Data values and references for inputs in JASMINE software to create diving behavior (number values represent Means (SD) unless otherwise indicated).

Behavior	Variable	Value	Reference
Deep Foraging Dive	Travel direction	Correlated random walk	Approximated
	Perturbation value	10	Approximated
	Termination coefficient	0.2	Approximated
	Travel rate (m/s)	Gaussian 1.5 (0.5)	Approximated
	Ascent rate (m/s)	Gaussian 0.79 (0.13)	Tyack et al. (2006) Baird et al. (2006b)
	Descent rate (m/s)	Gaussian 1.45 (0.2)	Tyack et al. (2006) Baird et al. (2006b)
	Average depth (m)	Gaussian 835 (143)	Tyack et al. (2006)
	Bottom following	No	Approximated
	Reversals	Gaussian 20 (2)	Tyack et al. (2006)
	Probability of reversal	0.95	Approximated
	Reversal ascent dive rate (m/s)	0.8 (0.2)	Madsen et al. (2005)
	Reversal descent dive rate (m/s)	0.8 (0.2)	Madsen et al. (2005)
	Time in reversal (s)	Gaussian 40 (20)	Tyack et al. (2006)
	Surface interval (s)	Gaussian 228 (276)	Tyack et al. (2006)
	Bout duration	T_{50} = 1200 (s), k=10	Approximated
	Travel direction	Correlated random walk	Approximated

Behavior	Variable	Value	Reference
Shallow Dive	Perturbation value	10	Approximated
	Termination coefficient	0.2	Approximated
	Travel rate (m/s)	Gaussian 1.5 (0.5)	Approximated
	Ascent rate (m/s)	Gaussian 0.35 (0.2)	Baird et al. (2006b), Tyack et al. (2006)_ENREF_15
	Descent rate (m/s)	Gaussian 0.34 (0.24)	Tyack et al. (2006) Baird et al. (2006)
	Average depth (m)	Gaussian 71 52)	Tyack et al. (2006)
	Bottom following	No	Approximated
	Reversals	No	Approximated
	Surface interval (s)	Gaussian 228 (276)	Tyack et al. (2006)
	Bout duration (s)	Gaussian 3700 (1860)	Tyack et al. (2006)
General	Depth limit on seeding (m)	633	Baird et al. (2006) Waring et al. (2001)

Approximated: value based on the best fit for diving profile. Those values were not available from literature but were estimated producing a diving profile similar to D-tag results for example.

F.2.4. Bottlenose dolphins (*Tursiops truncatus*)

Bottlenose dolphins occur globally in temperate and tropical waters where they inhabit various habitats, such as estuaries, bays, coastal areas, and oceanic environments. Many different stocks have been identified in the Gulf of Mexico, with exact stock definitions still in flux as more information becomes available (Waring et al. 2010). The coastal stock's diet consists of invertebrates and fish, while the oceanic stock feeds mainly on squid and fish (Wynne and Schwartz 1999).

The bottlenose stock most relevant for this survey is the oceanic stock that occurs from the 200 m isobath to the seaward extent of the U.S. Exclusive Economic Zone (Waring et al. 2010). Abundance estimates based on 1996–2001 and 2003–2004 ship-based survey data were very similar (i.e., 2239 and 3708, respectively). During the spring 1991–2001 ship-based surveys with transect lines in waters of >200 m depth, a total of 151 dolphin groups were sighted with average group sizes of about 20 animals (Maze-Foley and Mullin 2006). During the 2003–2004 ship-based surveys in the same general area, 26 groups were observed with an average group size of 25 (Mullin 2007). All these sightings were concentrated in water depths between 200 m and 1000 m.

Bottlenose dolphins produce a variety of sounds, such as whistles, moans, trills, grunts, squeaks, and other. These sounds vary in volume, wavelength, frequency, and pattern. The frequency of the sounds produced by a bottlenose dolphin ranges from 200 Hz to 150 kHz. The lower frequency vocalizations (up to 50 kHz) are likely used in social communication. Social signals have most of their energy at frequencies less than 40 kHz. Higher frequency clicks (40–150 kHz) are primarily used for echolocation (Kastelein et al. 1995).

F.2.4.1. Behavioral parameters for animat modeling

Table F-6. Bottlenose dolphins: Data values and references for inputs in JASMINE software to create diving behavior (number values represent Means (SD) unless otherwise indicated).

Behavior	Variable	Value	Reference
Foraging	Travel direction	Vector model	Ward (1999)
	Travel rate (m/s)	Vector model	Ward (1999)
	Ascent rate (m/s)	Gaussian 2.1 (0.3)	Houser et al. (2010)
	Descent rate (m/s)	Gaussian 1.6 (0.2)	Houser et al. (2010)
	Average depth (m)	Gaussian 25 (5)	Hastie et al. (2006)
	Bottom following	Yes	Approximated
	Reversals	Gaussian 18 (1.1)	Approximated
	Probability of reversal	0.09	Approximated
	Reversal ascent dive rate (m/s)	1.0 (0.2)	Approximated
	Reversal descent dive rate (m/s)	1.0 (0.2)	Approximated
	Time in reversal (s)	Gaussian 1 (0.1)	Approximated
	Surface interval (s)	Gaussian 46.4 (2.5)	Lopez (2009)
	Bout duration (s)	Gaussian 252 (210)	Ward (1999)
Playing	Travel direction	Vector model	Ward (1999)
	Travel rate (m/s)	Vector model	Ward (1999)
	Ascent rate (m/s)	Gaussian 2.1 (0.3)	Houser et al. (2010)
	Descent rate (m/s)	Gaussian 1.6 (0.2)	Houser et al. (2010)
	Average depth (m)	Gaussian 7 (3)	Würsig and Würsig (1979), Hastie et al. (2006)

	Bottom following	Yes	Approximated
	Reversals	No	Approximated
	Surface interval (s)	Gaussian 3 (2)	Approximated
	Bout duration (s)	Gaussian 138 (54)	Ward (1999)
Resting	Travel direction	Vector model	Ward (1999)
	Travel rate (m/s)	Vector model	Ward (1999)
	Ascent rate (m/s)	Gaussian 0.5 (0.1)	Approximated
	Descent rate (m/s)	Gaussian 0.5 (0.1)	Approximated
	Average depth (m)	Random, max = 2	Approximated
	Bottom following	No	Approximated
	Reversals	No	Approximated
	Surface interval (s)	Gaussian 3 (2)	Approximated
	Bout duration (s)	Gaussian 174 (96)	Ward (1999)
Socializing	Travel direction	Vector model	Ward (1999)
	Travel rate (m/s)	Vector model	Ward (1999)
	Ascent rate (m/s)	Gaussian 2.1 (0.3)	Houser et al. (2010)
	Descent rate (m/s)	Gaussian 1.6 (0.2)	Houser et al. (2010)
	Average depth (m)	Random, max = 10	Hastie et al. (2006) Würsig and Würsig (1979)
	Bottom following	Yes	Approximated
	Reversals	No	Approximated
	Surface interval (s)	Gaussian 3 (2)	Approximated
	Bout duration (s)	Gaussian 204 (174)	Ward (1999)
Travel	Travel direction	Vector model	Ward (1999)
	Travel rate (m/s)	Vector model	Ward (1999)
	Ascent rate (m/s)	Gaussian 2.1 (0.3)	Houser et al. (2010)
	Descent rate (m/s)	Gaussian 1.6 (0.2)	Houser et al. (2010)
	Average depth (m)	Gaussian 7 (3)	Hastie et al. (2006) Würsig and Würsig (1979)
	Bottom following	Yes	Approximated
	Reversals	No	Approximated
	Surface interval (s)	Gaussian 3 (2)	Approximated
	Bout duration	Gaussian 306 (276)	Ward (1999)
General	Shore following (m)	2	Würsig and Würsig (1979)
	Depth limit on seeding (m)	2 (minimum), 40 (maximum)	Würsig and Würsig (1979)

Approximated: value based on the best fit for diving profile. Those values were not available from literature but were estimated producing a diving profile similar to D-tag results for example.

F.2.5. Short-finned pilot whales (*Globicephala macrorhynchus*)

Short-finned pilot whale is known to occur year-round in the Gulf of Mexico in coastal to pelagic waters along the continental shelf and over submarine canyons (Wynne and Schwartz 1999, Wursig et al. 2000). They feed primarily on squid (but also fish and octopus), and congregations are often associated with high densities of squid. Maze-Foley and Mullin (2006) reported 18 sightings of short-finned pilot whales over the period 1991–2001, with several sightings within or near the proposed survey. Vocalizations from short-finned pilot whales recorded in the Canary Islands consisted of calls, clicks, and grunts with most energy within frequencies between 280 Hz and 23 kHz (Scheer 2013).

F.2.5.1. Behavioral parameters for animat modeling

Table F-7. Short-finned pilot whales: Data values and references for inputs in JASMINE software to create diving behavior (number values represent Means (SD) unless otherwise indicated).

Behavior	Variable	Value	Reference
State 1	Travel direction	Correlated random walk	Approximated
	Perturbation value	10	Approximated
	Termination coefficient	0.2	Approximated
	Travel rate (m/s)	Gaussian 0.875 (0.572)	Wells et al. (2013)
	Ascent rate (m/s)	Gaussian 2.2 (0.2)	Aguilar Soto et al. (2009)
	Descent rate (m/s)	Gaussian 2 (0.2)	Aguilar Soto et al. (2009)
	Average depth (m)	Gaussian 43 (15)	Quick et al. (2017)
	Bottom following	No	Approximated
	Reversals	No	Approximated
	Surface interval (s)	Gaussian 165 (69)	Sakai et al. (2011)
	Bout duration (s)	T_{50} = 300 (s), k=7	Approximated
State 2	Travel direction	Correlated random walk	Approximated
	Perturbation value	10	Approximated
	Termination coefficient	0.2	Approximated
	Travel rate (m/s)	Gaussian 0.875 (0.572)	Wells et al. (2013)
	Ascent rate (m/s)	Gaussian 3.2 (0.4)	Aguilar Soto et al. (2009)
	Descent rate (m/s)	Gaussian 3 (0.4)	Aguilar Soto et al. (2009)
	Average depth (m)	Gaussian 550 (200)	Quick et al. (2017)
	Bottom following	No	Approximated
	Reversals	No	Approximated
	Surface interval (s)	Gaussian 165 (69)	Sakai et al. (2011)
	Bout duration (s)	T_{50} = 6000 (s), k=7	Approximated
State 3	Travel direction	Correlated random walk	Approximated
	Perturbation value	10	Approximated
	Termination coefficient	0.2	Approximated
	Travel rate (m/s)	Gaussian 0.875 (0.572)	Wells et al. (2013)
	Ascent rate (m/s)	Gaussian 2.2 (0.2)	Aguilar Soto et al. (2009)
	Descent rate (m/s)	Gaussian 2 (0.2)	Aguilar Soto et al. (2009)

Behavior	Variable	Value	Reference
	Average depth (m)	Gaussian 150 (100)	Quick et al. (2017)
	Bottom following	No	Approximated
	Reversals	No	Approximated
	Surface interval (s)	Gaussian 165 (69)	Sakai et al. (2011)
	Bout duration (s)	T_{50} = 3600 (s), k=7	Approximated
State 4	Travel direction	Correlated random walk	Approximated
	Perturbation value	10	Approximated
	Termination coefficient	0.2	Approximated
	Travel rate (m/s)	Gaussian 0.875 (0.572)	Wells et al. (2013)
	Ascent rate (m/s)	Gaussian 3.2 (0.4)	Aguilar Soto et al. (2009)
	Descent rate (m/s)	Gaussian 3 (0.4)	Aguilar Soto et al. (2009)
	Average depth (m)	Gaussian 850 (100)	Quick et al. (2017)
	Bottom following	No	Approximated
	Reversals	No	Approximated
	Surface interval (s)	Gaussian 165 (69)	Sakai et al. (2011)
	Bout duration (s)	T_{50} = 3600 (s), k=7	Approximated
Surface	Travel direction	Vector model	Approximated
	Perturbation value	10	Approximated
	Termination coefficient	0.2	Approximated
	Travel rate (m/s)	Gaussian 0 (0)	Approximated
	Ascent rate (m/s)	Gaussian 0.1 (0.1)	Approximated
	Descent rate (m/s)	Gaussian 0.1 (0.1)	Approximated
	Average depth (m)	Gaussian 12 (5)	Quick et al. (2017)
	Bottom following	No	Approximated
	Reversals	No	Approximated
	Surface interval (s)	Gaussian 165 (69)	Sakai et al. (2011)
	Bout duration	T_{50} = 3600 (s), k=7	Approximated
General	Shore following (m)	200	Approximated
	Depth limit on seeding (m)	200	Approximated

Approximated: value based on the best fit for diving profile. Those values were not available from literature but were estimated producing a diving profile similar to D-tag results for example.

F.2.6. *Kogia* species

The dwarf and pygmy sperm whales (*Kogia breviceps* and *Kogia sima*) are the only species in the Gulf of Mexico that are characterized as high-frequency cetaceans (Southall et al. 2007). Dwarf and pygmy sperm whales were difficult to distinguish during the ship-based surveys and were often reported under the combined name dwarf/pygmy sperm whales or *Kogia* spp. They were most commonly observed in waters of >2000 m depth (Maze-Foley and Mullin 2006, Mullin 2007). During the 1991–2001 ship-based surveys a total of 133 groups with an average size of 2 animals were observed (Maze-Foley and Mullin 2006). Similar surveys conducted in 2003–2004 reported 27 groups with an average of 1.5 animals per group (Mullin 2007).

Sound recordings of stranded *Kogia breviceps* revealed that echolocation clicks for this species ranged from 60 to 200 kHz, with a dominant frequency of 120 to 130 kHz (Caldwell and Caldwell 1991). Almost all energy of low-frequency vocalizations was below 2 kHz (Caldwell et al. 1966). An auditory brainstem response study supports a hearing range of 90–150 kHz (Ridgway and Carder 2001).

F.2.6.1. Behavioral parameters for animat modeling

Table F-8. *Kogia spp*, including Dwarf Sperm Whales and Pygmy Sperm Whales (*Kogia sima* and *K. breviceps*) based on short-finned pilot whale data. Data values and references for inputs in JASMINE software to create diving behavior (number values represent Means (SD) unless otherwise indicated).

Behavior	Variable	Value	Reference
Day dive	Travel direction	Random walk	Approximate
	Termination coefficient	0.2	Approximate
	Travel rate (m/s)	Gaussian 0.875 (0.572)	Short-finned pilot whales
	Ascent rate (m/s)	Gaussian 2.2 (0.2)	Short-finned pilot whales
	Descent rate (m/s)	Gaussian 2 (0.2)	Short-finned pilot whales
	Average depth (m)	Gaussian 30 (20)	Short-finned pilot whales
	Bottom following	No	Approximate
	Reversals	No	Approximate
	Surface interval (s)	Gaussian 165 (69)	Short-finned pilot whales
Night dive	Travel direction	Random walk	Approximate
	Termination coefficient	0.2	Approximate
	Travel rate (m/s)	Gaussian 0.875 (0.572)	Short-finned pilot whales
	Ascent rate (m/s)	Gaussian 3.2 (0.4)	Short-finned pilot whales
	Descent rate (m/s)	Gaussian 3 (0.4)	Short-finned pilot whales
	Average depth (m)	Gaussian 300 (100)	Short-finned pilot whales
	Bottom following	No	Approximate
	Reversals	No	Approximate
	Surface interval (s)	Gaussian 165 (69)	Short-finned pilot whales

Appendix G. Habitat-Density Model by Species

Cetacean density estimates (animals/km^2) were obtained using the Duke University's Marine Geospatial Ecology Laboratory (MGEL) model (Roberts et al. 2016a), preliminary results, which are hereafter referenced as PEIS densities. These estimates were produced with distance sampling methodology (Buckland et al. 2001) from 195,000 linear kilometers of shipboard and aerial surveys conducted by NOAA's Southeast Fisheries Science Center (SEFSC) in the Gulf of Mexico from 1992–2009. For each species, the count of animals per 10 km survey segment was modeled using a Horvitz-Thompson-like estimator (Marques and Buckland 2004, Miller et al. 2013). Species-specific detection functions were fitted using observation-level covariates such as Beaufort sea state, sun glare, and group size. When possible, availability and perception bias were estimated on a per-species basis using results from the scientific literature. After the sightings were corrected for detectability, availability, and perception bias, statistical regressions were used to model counts of animals per segment.

The density of frequently-sighted species were modeled with generalized additive models based on a collection of physiographic, physical oceanographic, and biological productivity predictor variables that plausibly relate to cetacean habitat. Both contemporaneous and climatological predictors were tested. Models were fitted to survey data and insignificant predictors were dropped from the models (Wood 2006). Final models were predicted across a time series of grids at 10 km resolution and averaged to produce a single surface representing mean density at each 10 km × 10 km grid square or cell.

There was insufficient data for infrequently seen species to model density from habitat variables. Instead, the geographic area of probable habitat was delineated from the scientific literature; patterns in the available sightings and density were estimated from the survey segments that occurred there using a statistical model that had no covariates. This model ran over the entire extent of the habitat area, yielding a uniform density estimate for each area.

Marine mammal density estimates for each species in the modeling zones are shown in Table G-1 to Table G-7.

Table G-1. Zone 1 Marine mammal density estimates.

Species	Density estimate			
	Min	Max	Mean	STD
Beaked whales	0.000000	0.004306	0.000107	0.000402
Bottlenose dolphins	10.718610	143.330322	37.130025	20.297288
Bryde's whales	0.000000	0.167721	0.012267	0.035798
Kogia spp	0.000000	0.381413	0.016379	0.046385
Short-finned pilot whales	0.078137	0.017168	0.000262	0.001151
Sperm whales	0.000000	0.004952	0.000150	0.000473

Table G-2. Zone 2 Marine mammal density estimates.

Species	Density estimate			
	Min	Max	Mean	STD
Beaked whales	0.000000	0.000281	0.000003	0.000018
Bottlenose dolphins	8.439063	113.845413	53.082960	22.977138
Bryde’s whales	0.000000	0.028985	0.000164	0.001293
Kogia spp	0.000000	0.043914	0.000937	0.004897
Short-finned pilot whales	0.000000	0.002055	0.000010	0.000086
Sperm whales	0.000000	0.000350	0.000007	0.000035

Table G-3. Zone 3 Marine mammal density estimates.

Species	Density estimate			
	Min	Max	Mean	STD
Beaked whales	0.000000	0.000140	0.000001	0.000012
Bottlenose dolphins	8.936208	79.201904	39.405915	14.535437
Bryde’s whales	0.000000	0.007863	0.000041	0.000375
Kogia spp	0.000000	0.024987	0.000187	0.001645
Short-finned pilot whales	0.000000	0.001161	0.000005	0.000054
Sperm whales	0.000000	0.000212	0.000002	0.000018

Table G-4. Zone 4 Marine mammal density estimates.

Species	Density estimate			
	Min	Max	Mean	STD
Beaked whales	0.000000	4.682173	0.725775	1.107739
Bottlenose dolphins	0.003873	66.720116	11.553444	12.482596
Bryde’s whales	0.000000	0.167727	0.035179	0.055666
Kogia spp	0.000000	2.564462	0.958299	0.613179
Short-finned pilot whales	0.000000	5.891473	0.685525	0.842500
Sperm whales	0.000000	2.049208	0.482223	0.480525

Table G-5. Zone 5 Marine mammal density estimates.

Species	Density estimate			
	Min	Max	Mean	STD
Beaked whales	0.000000	3.432981	1.080930	0.851019
Bottlenose dolphins	0.025899	46.434166	5.728691	8.809752
Bryde’s whales	0.000000	0.167701	0.014526	0.039290
Kogia spp	0.000000	1.972867	0.726706	0.450570
Short-finned pilot whales	0.000000	3.430244	0.639206	0.665957
Sperm whales	0.000000	2.049208	0.725159	0.527590

Table G-6. Zone 6 Marine mammal density estimates.

Species	Density estimate			
	Min	Max	Mean	STD
Beaked whales	0.000000	2.336602	0.832344	0.536911
Bottlenose dolphins	0.030806	24.043407	3.342733	5.111497
Bryde’s whales	0.000000	0.167480	0.013691	0.037372
Kogia spp	0.000000	1.100742	0.411093	0.228572
Short-finned pilot whales	0.000000	5.996468	1.249850	1.434598
Sperm whales	0.000000	1.356392	0.486587	0.286136

Table G-7. Zone 7 Marine mammal density estimates.

Species	Density estimate			
	Min	Max	Mean	STD
Beaked whales	0.222212	3.113844	0.519543	0.286857
Bottlenose dolphins	0.001245	1.554906	0.027482	0.067843
Bryde’s whales	0.000000	0.000004	0.000000	0.000000
Kogia spp	0.151227	0.825459	0.342218	0.062230
Short-finned pilot whales	0.003767	0.771689	0.121555	0.104179
Sperm whales	0.354441	1.140214	0.467025	0.131315

G.1. Marine Mammal Distribution Maps

This section contains distribution maps for representative marine mammal species likely to be affected by geological and geophysical exploration surveys (the remaining species distribution maps can be found in Appendix D of the Draft PEIS). The distributions were obtained from the Duke Marine Geospatial Ecology Laboratory model (Roberts et al. 2016a) as GIS-compatible rasters of density estimates in 100 km^2 areas. These animal distributions guided our selection of modeling zones, which were also patterned on BOEM's planning areas, and to maintain acoustic uniformity throughout zones. The zone boundaries are shown as overlays in the figures.

G.2. Beaked Whales

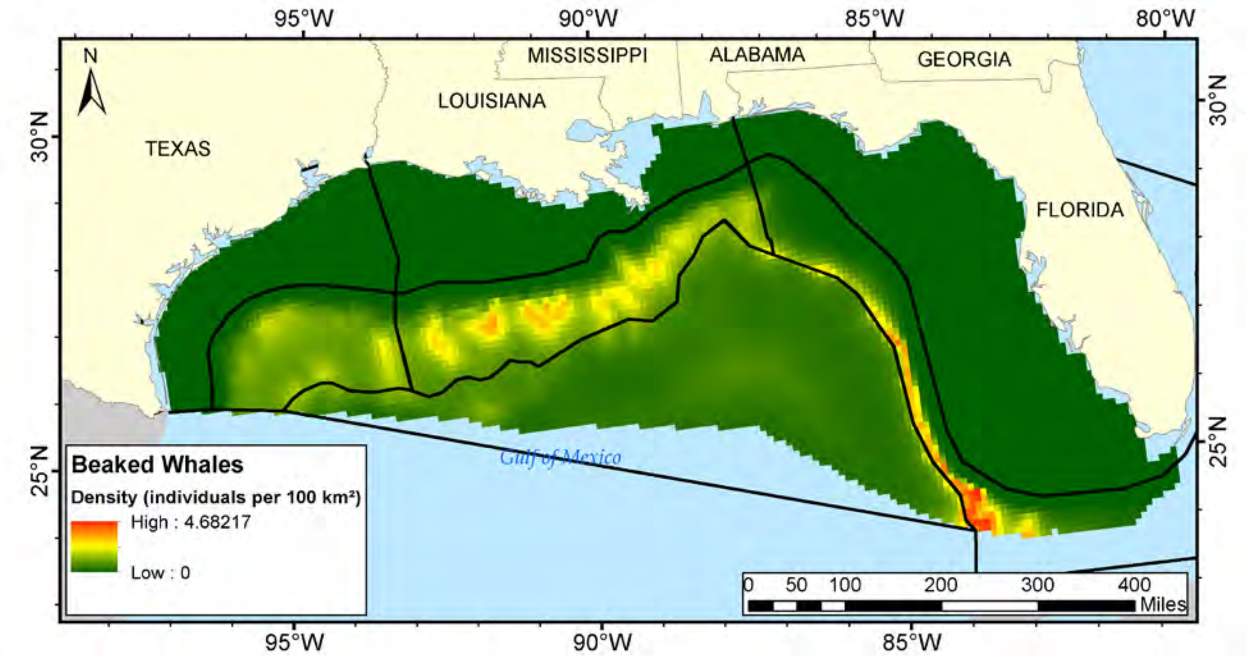


Figure G-1. Beaked whale distribution in the Gulf of Mexico project area. Density estimates were obtained from the Marine Geospatial Ecology Laboratory (Duke University) model (Roberts et al. 2016a), black lines depict the boundaries of the modeling zones.

G.3. Common Bottlenose Dolphins

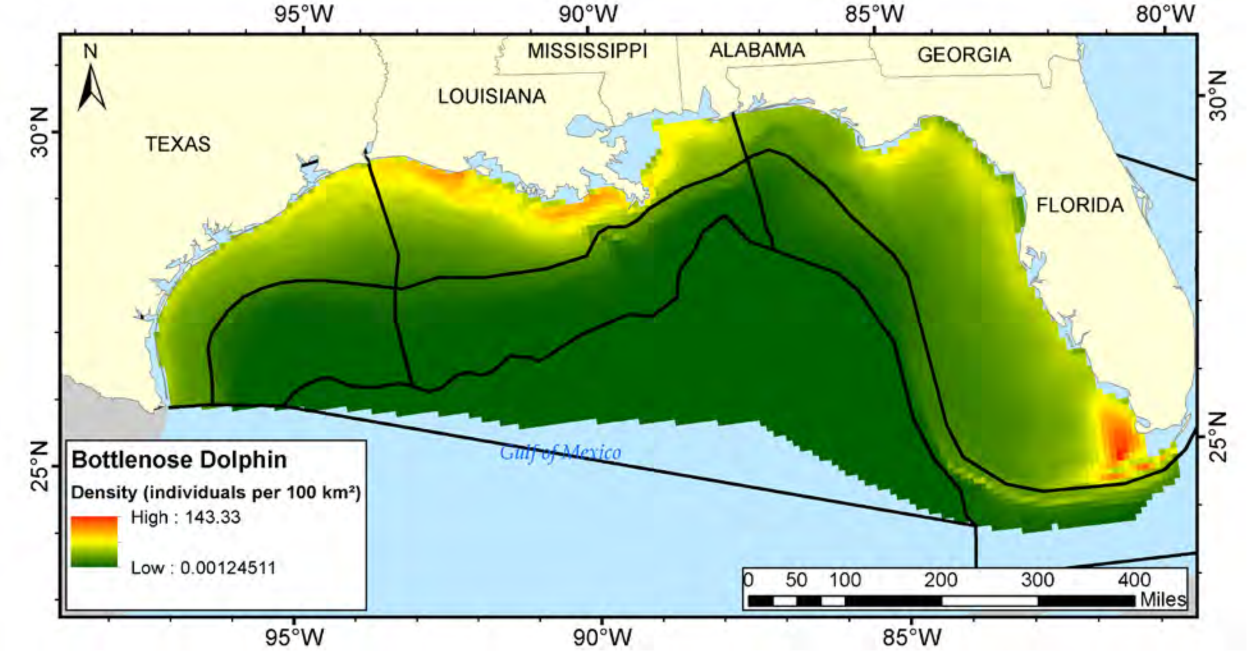


Figure G-2. Common bottlenose dolphin distribution in the Gulf of Mexico project area. Density estimates were obtained from the Marine Geospatial Ecology Laboratory (Duke University) model (Roberts et al. 2016a), black lines depict the boundaries of the modeling zones.

G.4. Bryde’s Whales

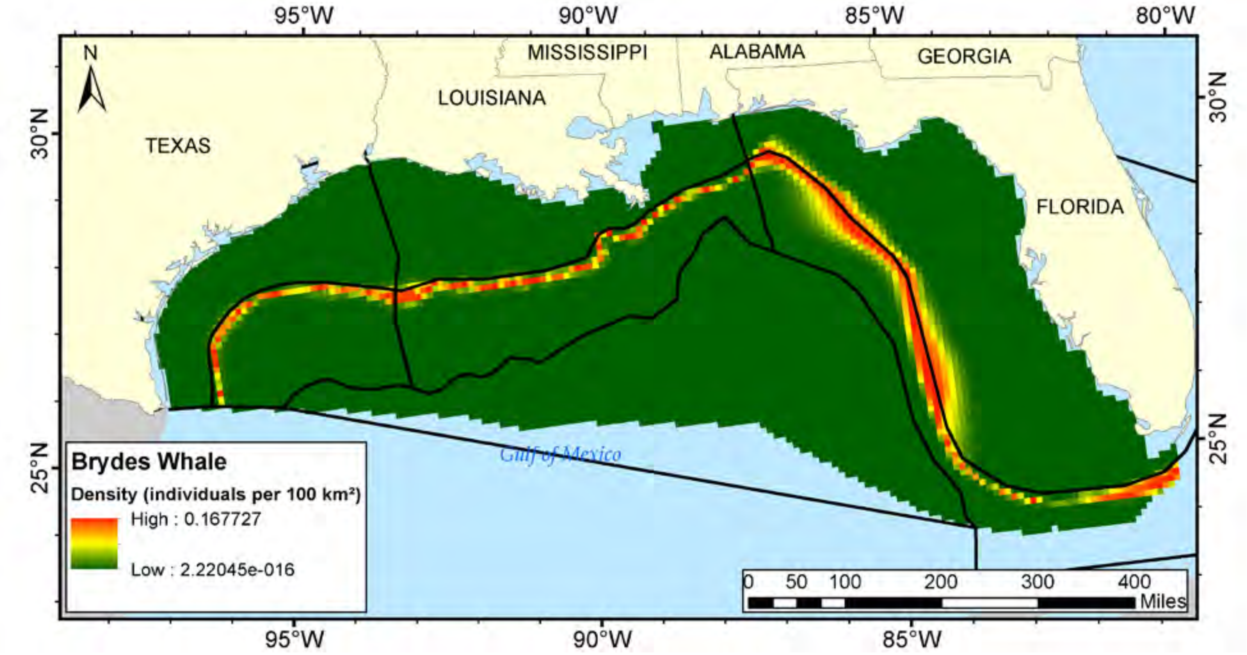


Figure G-3. Bryde’s whale distribution in the Gulf of Mexico project area. Density estimates were obtained from the Marine Geospatial Ecology Laboratory (Duke University) model (Roberts et al. 2016a), black lines depict the boundaries of the modeling zones.

G.5. Kogia Species

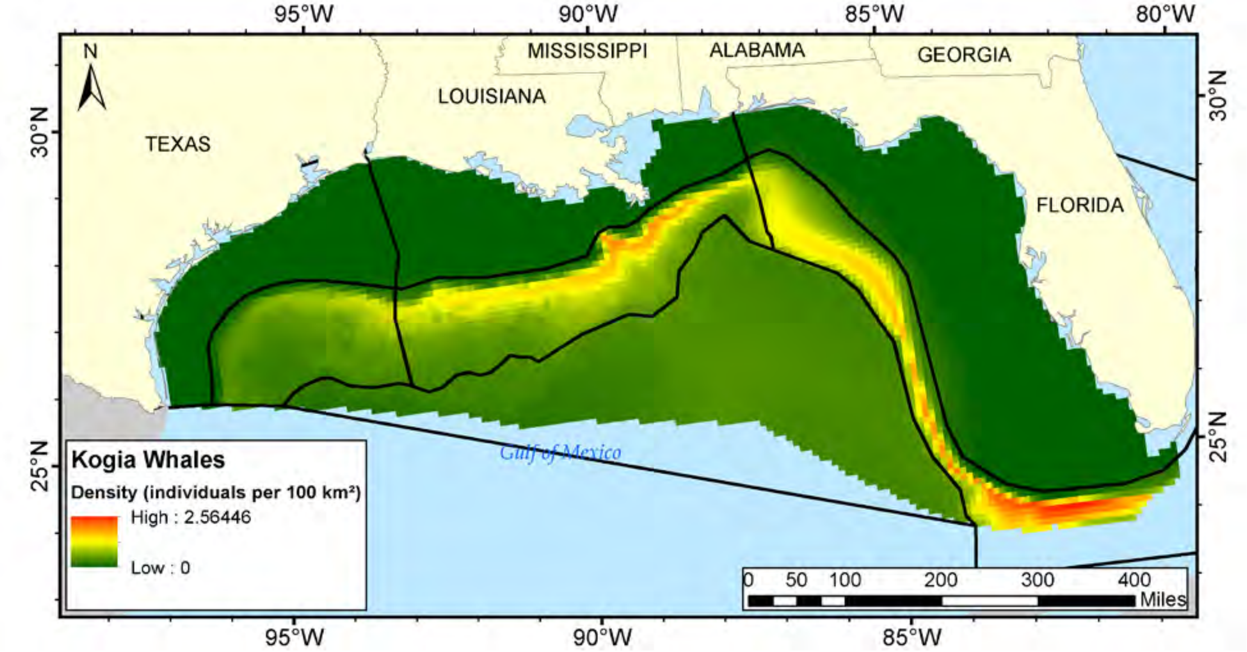


Figure G-4. *Kogia* distribution in the Gulf of Mexico project area. Density estimates were obtained from the Marine Geospatial Ecology Laboratory (Duke University) model (Roberts et al. 2016a), black lines depict the boundaries of the modeling zones.

G.6. Short-finned Pilot Whales

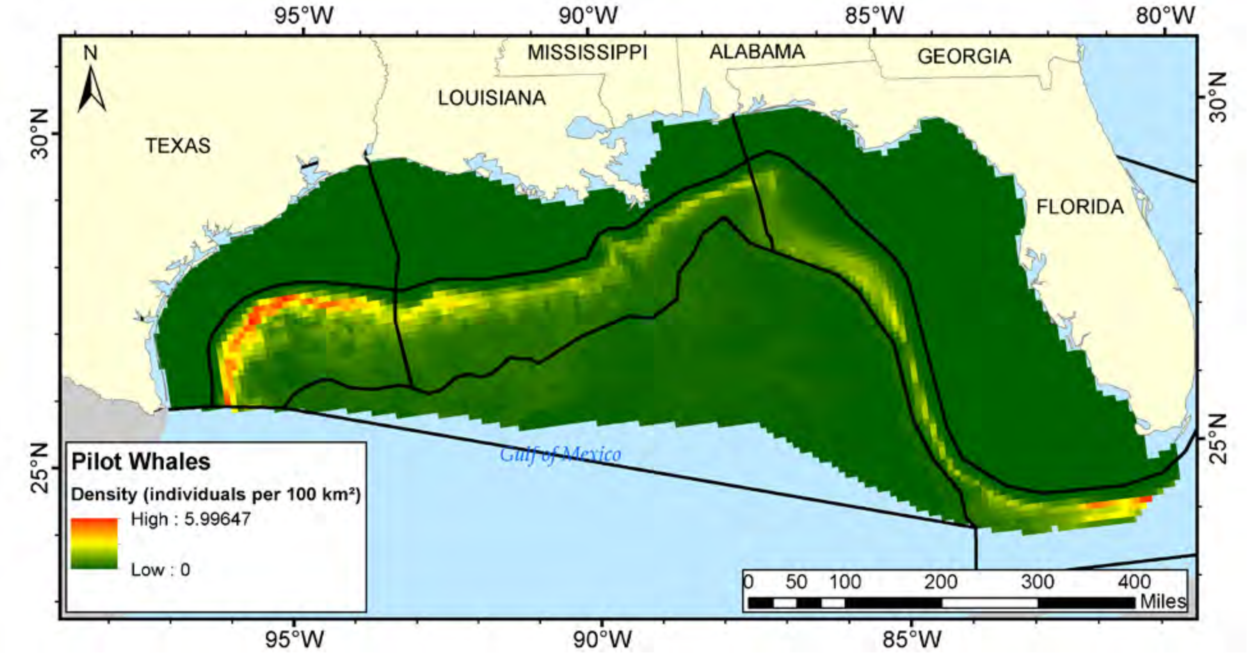


Figure G-5. Short-finned pilot whale distribution in the Gulf of Mexico project area. Density estimates were obtained from the Marine Geospatial Ecology Laboratory (Duke University) model (Roberts et al. 2016a), black lines depict the boundaries of the modeling zones.

G.7. Sperm Whales

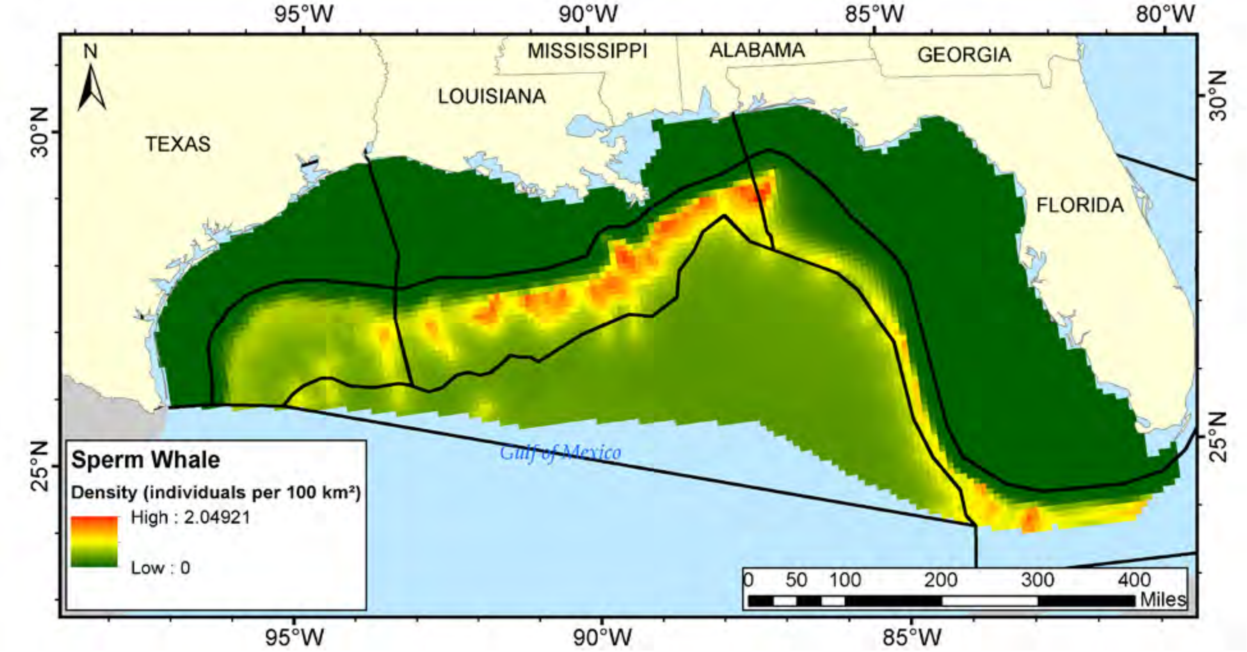


Figure G-6. Sperm whale distribution in the Gulf of Mexico project area. Density estimates were obtained from the Marine Geospatial Ecology Laboratory (Duke University) model (Roberts et al. 2016a), black lines depict the boundaries of the modeling zones.

Appendix H. Alternate Density Estimates

*** From IAGC ***

EXPLANATION OF ALTERNATE DENSITY VALUES

There is general agreement that the NMFS official Stock Assessment Report (SAR) minimum population estimates are probably not the best metric of actual GOM marine mammal population numbers and distribution, based on infrequent data sampling, and conservative assumptions in the Distance modeling (e.g. G_0=1.0). We therefore did not derive our density estimates from SAR data, although the surprisingly unexpected disparity between SAR values and the Duke model in many cases led us to adjust (halve) average regional densities derived from CETMAP/Duke information, at least until the Duke model can be tested and verified, or adjusted with new data. For two species for which there has been no SAR estimate since 2009 (Atlantic spotted dolphins and Frasers dolphins) we used historical NOAA SAR estimates from the 1996-2004 time period.

It may turn out, after additional future survey effort and further model iterations, that the values forecast by the Duke model are closer to the actual numbers of animals than the SARs, but dramatic leaps of as much as 10 to 85 times the previous SAR values for some species (notably Clymene dolphins, and Kogia spp.) led us to halve density values that were more than 3 times historical SAR estimates. We are not taking the position that the CETMAP and Duke estimates are incorrect, but we did note that these dramatic and unprecedented differences would contribute to dramatically increased MMPA take estimates and that the Duke model is not at this time sufficiently verified and validated to employ without some reservations.

Since the purpose of this exercise was to illustrate how the use of alternate numbers would affect model outcome, and was not come up with a “better estimate than any other”, we chose a somewhat arbitrary “middle ground” for the sole purpose of illustrating model sensitivity. We did not choose those numbers to make a statement about what might or might not be a better alternative than the Duke model values. Building confidence in the Duke values or adopting an alternate set of values would require more data collection as well as model refinement.

We concur with JASCO that using the direct Duke model predictions for 100 km^2 density values is vulnerable to sampling errors during the modeling process. Geophysical surveys will not be distributed evenly but their specific locations cannot be predicted, and animal distributions are similarly variable and hard to predict for a given date and location. Use of a smoothed average across a manageable number of oceanographic, acoustic and ecological provinces is therefore the best choice for modeling the likely outcome for any given year’s activities and animal distribution.

Gulf population estimates on the NOAA CETMAP website generally produced population estimates for each species in GOM that are very similar to the JASCO values used in the BOEM DPEIS. Both are derived from the same Duke model geospatial distribution data, though our estimates were derived in a different way than JASCO handled the Duke density values. Specifically, we divided the CETMAP total population estimate in proportion to the relative size of each region and the average density value for each region so that the summed population estimates within each region would add up to the CETMAP total population estimate. In two notable cases, for Brydes whales and for short-finned pilot whales, JASCO’s method of averaging density values within a region without regard to the CETMAP total population estimate yielded total GOM population estimates that greatly differed from CETMAP estimates. For example, CETMAP offers a Bryde’s whale population total for the Gulf of 44 individuals, not far from historical SAR values of 33 individuals. But the JASCO model predicts a population of 256 or almost six times the NOAA CETMAP population estimate. In those unusual cases we relied on the CETMAP-derived regional density estimates, and those will differ significantly from the values previously used in the BOEM DPEIS.

The specific rationales for choosing the density values we offered for each species are presented in the accompanying Excel spreadsheet, along with the regional average density values themselves. The total GOM population estimates for each species from recent Stock Assessment Reports (SAR), CETMAP and the JASCO Appendix D of the BOEM Gulf of Mexico DPEIS are also provided to indicate those species for which SARs, CETMAP and JASCO provide similar numbers, and those species for which the three sources disagree considerably, prompting an adjustment on our part to previously used regional density values.

When differences between SARs, CETMAP, and JASCO population estimates were less than 300% (no value was three times greater than any other) we used the CETMAP-based values, which were very similar to the JASCO values (with the two exceptions noted earlier). But where differences between population estimates were three, four or as much as ten, twelve or even 85 times historical SAR values, we halved the CETMAP-based value. We consider this a conservative compromise until we have a better understanding of whether the new models like the Duke model may have made some seriously incorrect assumptions about the habitat use and ecology of some species, requiring further model refinement and testing. These differences not only have a dramatic effect on the estimated sound exposure risk from geophysical surveys but also imply dramatic consequences for our understanding of the Gulf ecosystem, it’s productivity, trophic dynamics, and vulnerability to anthropogenic or natural perturbations like fishery bycatch and Loop Current dynamics.

It is important to repeat that our choice of regional density values was guided to some degree by the aim of offering numbers sufficiently different from the original JASCO values to produce a discernable difference in model outcome. But our choices were not completely arbitrary, and are based on a consistent rationale, as described above.

Table H-1.

Species	Zone 1	Zone 2	Zone 3	Zone 4	Zone 5	Zone 6	Zone 7	SAR	CetMap	JASCO	Comments
Atlantic spotted dolphins	19.336809	7.370539	8.097456	2.749779	2.007792	1.25886	0.000004	37611	47488	48040	No SAR estimate, so 2000-2004 values were applied. CETMAP-based density values were used
Beaked whales (3)	0.0000535	0.0000015	0.0000005	0.362261	0.5395315	0.4154535	0.259323	223	2910	2915	CetMap estimates are more than 10 times SAR, CETMAP-based density values were halved
Bottlenose dolphins (mult stocks)	36.793692	52.602121	39.048966	11.44879	5.676799	3.312454	0.027233	96732	138602	139869	not surprisingly, SARs, CETMAP and JASCO agree relatively closely on these generally well-characterized stocks (especially estuarine and coastal). CETMAP values were used
Bryde's whales	0.02109	0.000028	0.000007	0.006048	0.002497	0.002354	0	33	44	256	JASCO's use of the Duke density data led to much larger population estimates than either CETMAP or SAR, so CETMAP was used
Clymene dolphins	0.000394	0.000001	0	0.459086	1.71583	2.1406395	1.319643	129	11000	10952	CETMAP and JASCO model were 85 times greater than SAR, CETMAP-based densities were halved
False killer whales	0.0615285	0.0142795	0.0065685	0.361636	0.3611945	0.365652	0.37178	777	3204	3224	CETMAP and JASCO model were four times greater than SAR, CETMAP-based densities were halved
Fraser's dolphins	0.063948	0.014841	0.006827	0.375853	0.375394	0.380026	0.386406	726	1665	1675	CETMAP and JASCO numbers were a little over double the SARs, CETMAP-based densities were used
Killer whales	0.000195	0.000082	0.000085	0.0066	0.010028	0.009839	0.038745	28	185	186	CETMAP and JASCO numbers were five times the SARs, CETMAP-based densities were halved
kogia (2)	0.008172	0.0004675	0.0000935	0.478119	0.3625715	0.2051045	0.170741	186	2234	2239	CETMAP and JASCO numbers were 12 times higher than SAR, CETMAP-based densities were halved
Melon-headed whales	0.001345	0.0000905	0.000031	0.5908705	1.104694	0.9449345	0.766659	2235	6733	6734	CETMAP and JASCO numbers were 3 times higher than SAR, CETMAP-based densities were halved
Pantropical spotted dolphins	0.110796	0.002309	0.000595	21.688002	15.447613	9.828148	25.992595	50880	84014	84322	CETMAP and JASCO numbers were less than double the SAR, CETMAP-based densities were used
Pygmy killer whale	0.000141	0.0000055	0.0000025	0.368834	0.343913	0.672459	0.0654055	152	2126	1976	CETMAP and JASCO numbers were about 13 times higher than SAR, CETMAP-based densities were halved
Risso's dolphins	0.001848	0.000115	0.000043	0.437355	0.673816	0.702498	0.975053	2442	3137	2127	CETMAP and JASCO numbers were less than double the SAR, CETMAP-based densities were used
Rough-toothed dolphins	0.0137505	0.000643	0.0002285	1.093325	0.7489755	0.6119455	0.3233085	624	4853	3151	CETMAP and JASCO numbers were 5 to 7 times higher than SAR, CETMAP-based densities were halved
Short-finned pilot whales	0.164829	0.160061	0.160709	0.390128	0.425843	0.404707	0.323965	2417	1981	4885	JASCO's use of the Duke density data led to much larger population estimates than either CETMAP or SAR, so CETMAP was used
Sperm whales	0.0000795	0.0000035	0.000001	0.240159	0.361147	0.2423325	0.23259	763	2128	2136	CETMAP and JASCO numbers were about 3 times higher than the SAR and the SAR data are stronger for sperm whales than most other GOM species. CETMAP-based densities were halved
Spinner dolphins	0.018356	0	0	11.676661	4.124051	0.234841	0.607681	11441	13485	13584	CETMAP and JASCO numbers were close and near SAR, CETMAP-based densities were used.
Striped dolphins	0.002593	0.000025	0.00003	0.796473	1.329813	1.087864	1.360301	1849	4914	4931	CETMAP and JASCO numbers were about 2.7 times higher than SAR, CETMAP-based densities were used.

Appendix I. Survey Level of Effort

Survey effort in miles by year and zone, supplied by BOEM (same as used in PEIS).

I.1. 2 D Seismic Survey

Year	Eastern Shallow	Central Shallow	Western Shallow	Eastern Deep	Central Deep	Western Deep
2016	0	0	0	0	12,000	0
2017	0	0	0	6,000	0	0
2018	0	0	0	0	0	0
2019	0	0	0	12,000	6,000	0
2020	0	0	0	0	0	0
2021	0	0	0	0	0	0
2022	0	0	0	6,000	6,000	0
2023	0	0	0	2,000	2,000	0
2024	0	0	0	0	0	0
2025	0	0	0	1,000	0	0
Total	0	0	0	27,000	26,000	0

I.2. 3-D Narrow Azimuth Seismic Survey

Year	Eastern Shallow	Central Shallow	Western Shallow	Eastern Deep	Central Deep	Western Deep
2016	0	23,292	2,911	0	84,070	20,583
2017	0	34,938	0	0	84,070	11,000
2018	0	23,292	0	0	73,856	20,588
2019	0	34,938	2,911	11,200	53,428	11,000
2020	0	23,292	0	16,800	63,642	11,000
2021	0	34,938	0	16,800	53,428	20,588
2022	0	23,292	2,911	11,200	53,428	11,000
2023	0	34,938	0	11,200	53,428	11,000
2024	0	23,292	0	11,200	43,214	11,000
2025	0	34,938	2,911	11,200	43,214	11,000
Total	0	291,150	11,644	89,600	605,778	138,759

I.3. 3-D Wide Azimuth Seismic Survey

Year	Eastern Shallow	Central Shallow	Western Shallow	Eastern Deep	Central Deep	Western Deep
2016	0	0	0	0	41,551	5,397
2017	0	4,155	0	0	41,551	0
2018	0	0	0	0	34,626	5,397
2019	0	4,155	0	3,920	20,775	0
2020	0	0	0	0	41,551	0
2021	0	4,155	0	0	34,626	5,397
2022	0	0	0	3,920	34,626	0
2023	0	4,155	0	0	27,700	0
2024	0	0	0	0	41,551	0
2025	0	4,155	0	0	34,626	0
Total	0	20,773	0	7,840	353,180	16,191

I.4. Coil Seismic Survey

Year	Eastern Shallow	Central Shallow	Western Shallow	Eastern Deep	Central Deep	Western Deep
2016	0	0	0	0	17,807	2,313
2017	0	1,781	0	0	17,807	0
2018	0	0	0	0	14,840	2,313
2019	0	1,781	0	1,680	8,904	0
2020	0	0	0	0	17,807	0
2021	0	1,781	0	0	14,840	2,313
2022	0	0	0	1,680	14,840	0
2023	0	1,781	0	0	11,872	0
2024	0	0	0	0	17,807	0
2025	0	1,781	0	0	14,840	0
Total	0	8,903	0	3,360	151,363	6,939

Appendix J. Annual Exposure Estimates

J.1. No Aversion, PEIS Marine Mammal Density Estimates

J.1.1. 2016

Table J-1. 2016 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	5232
Bottlenose dolphins	0	0	3732
Bryde's whales	0	0	27
Kogia spp.	50	0	447
Short-finned pilot whales	0	0	441
Sperm whales	0	0	955

Table J-2. 2016 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	51384
Bottlenose dolphins	34	0	365055
Bryde's whales	1	5	300
Kogia spp.	712	0	4490
Short-finned pilot whales	0	0	6290
Sperm whales	1	0	10383

Table J-3. 2016 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	22258
Bottlenose dolphins	4	0	18884
Bryde's whales	0	0	135
Kogia spp.	715	0	2341
Short-finned pilot whales	0	0	2733
Sperm whales	0	0	4809

Table J-4. 2016 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	5633.6
Bottlenose dolphins	2	0	4395.9
Bryde's whales	0	2	31.0
Kogia spp.	228	0	602.9
Short-finned pilot whales	0	0	710.3
Sperm whales	0	0	1313.6

J.1.2. 2017

Table J-5. 2017 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	1680
Bottlenose dolphins	0	0	2903
Bryde's whales	0	0	32
Kogia spp.	34	0	235
Short-finned pilot whales	0	0	195
Sperm whales	0	0	183

Table J-6. 2017 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	46867
Bottlenose dolphins	39	0	491101
Bryde's whales	1	4	262
Kogia spp.	662	0	4174
Short-finned pilot whales	0	0	5079
Sperm whales	1	0	9367

Table J-7. 2017 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	19775
Bottlenose dolphins	4	0	74562
Bryde's whales	0	0	112
Kogia spp.	651	0	2136
Short-finned pilot whales	0	0	1976
Sperm whales	0	0	4246

Table J-8. 2017 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	4991
Bottlenose dolphins	6	0	15734
Bryde's whales	0	2	26
Kogia spp.	208	0	550
Short-finned pilot whales	0	0	521
Sperm whales	0	0	1151

J.1.3. 2018

Table J-9. 2018 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	0
Bottlenose dolphins	0	0	0
Bryde's whales	0	0	0
Kogia spp.	0	0	0
Short-finned pilot whales	0	0	0
Sperm whales	0	0	0

Table J-10. 2018 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	46322
Bottlenose dolphins	27	0	337980
Bryde’s whales	1	4	274
Kogia spp.	638	0	4027
Short-finned pilot whales	0	0	5843
Sperm whales	1	0	9387

Table J-11. 2018 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	18962
Bottlenose dolphins	4	0	16089
Bryde’s whales	0	0	117
Kogia spp.	607	0	1985
Short-finned pilot whales	0	0	2404
Sperm whales	0	0	4101

Table J-12. 2018 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	4802
Bottlenose dolphins	2	0	3751
Bryde’s whales	0	2	27
Kogia spp.	193	0	511
Short-finned pilot whales	0	0	623
Sperm whales	0	0	1122

J.1.4. 2019

Table J-13. 2019 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	5976
Bottlenose dolphins	0	0	7672
Bryde's whales	0	0	78
Kogia spp.	93	0	693
Short-finned pilot whales	0	0	612
Sperm whales	0	0	844

Table J-14. 2019 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	35223
Bottlenose dolphins	47	0	510215
Bryde's whales	1	4	248
Kogia spp.	548	0	3312
Short-finned pilot whales	0	0	4170
Sperm whales	1	0	6797

Table J-15. 2019 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	11129
Bottlenose dolphins	2	0	68848
Bryde's whales	0	0	81
Kogia spp.	405	0	1285
Short-finned pilot whales	0	0	1161
Sperm whales	0	0	2293

Table J-16. 2019 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	2793
Bottlenose dolphins	6	0	14342
Bryde's whales	0	1	17
Kogia spp.	131	0	336
Short-finned pilot whales	0	0	306
Sperm whales	0	0	609

J.1.5. 2020

Table J-17. 2020 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	0
Bottlenose dolphins	0	0	0
Bryde's whales	0	0	0
Kogia spp.	0	0	0
Short-finned pilot whales	0	0	0
Sperm whales	0	0	0

Table J-18. 2020 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	42062
Bottlenose dolphins	26	0	340812
Bryde's whales	1	5	307
Kogia spp.	674	0	4034
Short-finned pilot whales	0	0	4835
Sperm whales	1	0	8003

Table J-19. 2020 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	19775
Bottlenose dolphins	4	0	16772
Bryde's whales	0	0	112
Kogia spp.	651	0	2135
Short-finned pilot whales	0	0	1976
Sperm whales	0	0	4246

Table J-20. 2020 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	4991
Bottlenose dolphins	2	0	3870
Bryde's whales	0	2	26
Kogia spp.	208	0	550
Short-finned pilot whales	0	0	521
Sperm whales	0	0	1151

J.1.6. 2021

Table J-21. 2021 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	0
Bottlenose dolphins	0	0	0
Bryde's whales	0	0	0
Kogia spp.	0	0	0
Short-finned pilot whales	0	0	0
Sperm whales	0	0	0

Table J-22. 2021 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	41517
Bottlenose dolphins	40	0	493039
Bryde's whales	1	5	319
Kogia spp.	650	0	3892
Short-finned pilot whales	0	0	5599
Sperm whales	1	0	8024

Table J-23. 2021 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	18962
Bottlenose dolphins	4	0	73879
Bryde's whales	0	0	117
Kogia spp.	607	0	1986
Short-finned pilot whales	0	0	2404
Sperm whales	0	0	4101

Table J-24. 2021 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	4802
Bottlenose dolphins	6	0	15615
Bryde's whales	0	2	27
Kogia spp.	193	0	511
Short-finned pilot whales	0	0	623
Sperm whales	0	0	1122

J.1.7. 2022

Table J-25. 2022 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	4296
Bottlenose dolphins	0	0	4769
Bryde's whales	0	0	46
Kogia spp.	59	0	458
Short-finned pilot whales	0	0	416
Sperm whales	0	0	661

Table J-26. 2022 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	35223
Bottlenose dolphins	34	0	357541
Bryde's whales	1	4	248
Kogia spp.	547	0	3310
Short-finned pilot whales	0	0	4169
Sperm whales	1	0	6797

Table J-27. 2022 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	17720
Bottlenose dolphins	4	0	16649
Bryde's whales	0	0	118
Kogia spp.	622	0	1996
Short-finned pilot whales	0	0	1820
Sperm whales	0	0	3708

Table J-28. 2022 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	4456
Bottlenose dolphins	3	0	3768
Bryde's whales	0	2	26
Kogia spp.	201	0	519
Short-finned pilot whales	0	0	479
Sperm whales	0	0	993

J.1.8. 2023

Table J-29. 2023 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	1432
Bottlenose dolphins	0	0	1590
Bryde's whales	0	0	15
Kogia spp.	20	0	153
Short-finned pilot whales	0	0	139
Sperm whales	0	0	220

Table J-30. 2023 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	35223
Bottlenose dolphins	39	0	486721
Bryde's whales	1	4	248
Kogia spp.	548	0	3312
Short-finned pilot whales	0	0	4170
Sperm whales	1	0	6797

Table J-31. 2023 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	13183
Bottlenose dolphins	3	0	68971
Bryde's whales	0	0	75
Kogia spp.	434	0	1424
Short-finned pilot whales	0	0	1318
Sperm whales	0	0	2831

Table J-32. 2023 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	3328
Bottlenose dolphins	5	0	14444
Bryde's whales	0	1	17
Kogia spp.	139	0	367
Short-finned pilot whales	0	0	348
Sperm whales	0	0	767

J.1.9. 2024

Table J-33. 2024 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	0
Bottlenose dolphins	0	0	0
Bryde's whales	0	0	0
Kogia spp.	0	0	0
Short-finned pilot whales	0	0	0
Sperm whales	0	0	0

Table J-34. 2024 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	22230
Bottlenose dolphins	26	0	204336
Bryde's whales	1	4	180
Kogia spp.	474	0	2093
Short-finned pilot whales	0	0	2733
Sperm whales	1	0	4297

Table J-35. 2024 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	14598
Bottlenose dolphins	4	0	12301
Bryde's whales	0	0	95
Kogia spp.	651	0	1584
Short-finned pilot whales	0	0	1460
Sperm whales	0	0	3152

Table J-36. 2024 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	3675
Bottlenose dolphins	2	0	2804
Bryde's whales	0	2	23
Kogia spp.	208	0	407
Short-finned pilot whales	0	0	385
Sperm whales	0	0	850

J.1.10. 2025

Table J-37. 2025 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	280
Bottlenose dolphins	0	0	484
Bryde's whales	0	0	5
Kogia spp.	6	0	39
Short-finned pilot whales	0	0	33
Sperm whales	0	0	31

Table J-38. 2025 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	30159
Bottlenose dolphins	47	0	506633
Bryde's whales	1	4	221
Kogia spp.	474	0	2850
Short-finned pilot whales	0	0	3721
Sperm whales	1	0	5801

Table J-39. 2025 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	16479
Bottlenose dolphins	4	0	71766
Bryde's whales	0	0	93
Kogia spp.	542	0	1780
Short-finned pilot whales	0	0	1647
Sperm whales	0	0	3538

Table J-40. 2025 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	4159
Bottlenose dolphins	6	0	15089
Bryde's whales	0	1	21
Kogia spp.	174	0	459
Short-finned pilot whales	0	0	435
Sperm whales	0	0	959

J.2. Aversion, PEIS Marine Mammal Density Estimates

Table J-41. 2016 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	4982
Bottlenose dolphins	1	0	3724
Bryde's whales	0	0	27
Kogia spp.	27	0	459
Short-finned pilot whales	0	0	470
Sperm whales	0	0	936

Table J-42. 2016 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	47811
Bottlenose dolphins	6	0	357577
Bryde's whales	1	4	297
Kogia spp.	436	1	4790
Short-finned pilot whales	0	0	6694
Sperm whales	1	0	9929

Table J-43. 2016 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	21475
Bottlenose dolphins	1	0	18561
Bryde's whales	0	0	134
Kogia spp.	406	0	2444
Short-finned pilot whales	0	0	2862
Sperm whales	0	0	4665

Table J-44. 2016 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	4797
Bottlenose dolphins	1	0	4256
Bryde's whales	0	2	30
Kogia spp.	119	0	575
Short-finned pilot whales	0	0	654
Sperm whales	0	0	1156

J.2.1. 2017

Table J-45. 2017 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	1392
Bottlenose dolphins	0	0	2901
Bryde's whales	0	0	32
Kogia spp.	23	0	254
Short-finned pilot whales	0	0	213
Sperm whales	0	0	181

Table J-46. 2017 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	43389
Bottlenose dolphins	6	0	480531
Bryde’s whales	0	4	259
Kogia spp.	405	1	4451
Short-finned pilot whales	0	0	5406
Sperm whales	0	0	8953

Table J-47. 2017 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	19012
Bottlenose dolphins	1	0	72451
Bryde’s whales	0	0	112
Kogia spp.	369	0	2226
Short-finned pilot whales	0	0	2078
Sperm whales	0	0	4117

Table J-48. 2017 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	4199
Bottlenose dolphins	0	0	14831
Bryde’s whales	0	2	25
Kogia spp.	108	0	525
Short-finned pilot whales	0	0	481
Sperm whales	0	0	1018

J.2.2. 2018

Table J-49. 2018 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	0
Bottlenose dolphins	0	0	0
Bryde's whales	0	0	0
Kogia spp.	0	0	0
Short-finned pilot whales	0	0	0
Sperm whales	0	0	0

Table J-50. 2018 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	43158
Bottlenose dolphins	6	0	330971
Bryde's whales	1	4	271
Kogia spp.	391	1	4298
Short-finned pilot whales	0	0	6218
Sperm whales	1	0	8978

Table J-51. 2018 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	18306
Bottlenose dolphins	0	0	15817
Bryde's whales	0	0	116
Kogia spp.	344	0	2073
Short-finned pilot whales	0	0	2516
Sperm whales	0	0	3979

Table J-52. 2018 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	4097
Bottlenose dolphins	1	0	3631
Bryde's whales	0	2	26
Kogia spp.	101	0	487
Short-finned pilot whales	0	0	573
Sperm whales	0	0	986

J.2.3. 2019

Table J-53. 2019 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	5274
Bottlenose dolphins	0	0	7664
Bryde's whales	0	0	78
Kogia spp.	60	0	738
Short-finned pilot whales	0	0	661
Sperm whales	0	0	831

Table J-54. 2019 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	32147
Bottlenose dolphins	6	0	499159
Bryde's whales	1	4	245
Kogia spp.	346	1	3551
Short-finned pilot whales	0	0	4457
Sperm whales	0	0	6506

Table J-55. 2019 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	10545
Bottlenose dolphins	0	0	66778
Bryde’s whales	0	0	80
Kogia spp.	240	0	1353
Short-finned pilot whales	0	0	1230
Sperm whales	0	0	2225

Table J-56. 2019 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	2251
Bottlenose dolphins	0	0	13485
Bryde’s whales	0	1	17
Kogia spp.	70	0	320
Short-finned pilot whales	0	0	282
Sperm whales	0	0	543

J.2.4. 2020

Table J-57. 2020 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	0
Bottlenose dolphins	0	0	0
Bryde’s whales	0	0	0
Kogia spp.	0	0	0
Short-finned pilot whales	0	0	0
Sperm whales	0	0	0

Table J-58. 2020 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	38163
Bottlenose dolphins	8	0	333638
Bryde's whales	1	5	303
Kogia spp.	429	1	4330
Short-finned pilot whales	0	0	5176
Sperm whales	0	0	7662

Table J-59. 2020 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	19012
Bottlenose dolphins	1	0	16467
Bryde's whales	0	0	112
Kogia spp.	369	0	2225
Short-finned pilot whales	0	0	2078
Sperm whales	0	0	4117

Table J-60. 2020 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	4199
Bottlenose dolphins	0	0	3747
Bryde's whales	0	2	25
Kogia spp.	108	0	525
Short-finned pilot whales	0	0	481
Sperm whales	0	0	1018

J.2.5. 2021

Table J-61. 2021 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	0
Bottlenose dolphins	0	0	0
Bryde's whales	0	0	0
Kogia spp.	0	0	0
Short-finned pilot whales	0	0	0
Sperm whales	0	0	0

Table J-62. 2021 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	37931
Bottlenose dolphins	7	0	482438
Bryde's whales	1	5	315
Kogia spp.	416	1	4182
Short-finned pilot whales	0	0	5987
Sperm whales	1	0	7687

Table J-63. 2021 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	18306
Bottlenose dolphins	0	0	71800
Bryde's whales	0	0	116
Kogia spp.	344	0	2074
Short-finned pilot whales	0	0	2516
Sperm whales	0	0	3979

Table J-64. 2021 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	4097
Bottlenose dolphins	1	0	14715
Bryde's whales	0	2	26
Kogia spp.	101	0	487
Short-finned pilot whales	0	0	573
Sperm whales	0	0	986

J.2.6. 2022

Table J-65. 2022 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	3883
Bottlenose dolphins	0	0	4763
Bryde's whales	0	0	46
Kogia spp.	37	0	484
Short-finned pilot whales	0	0	448
Sperm whales	0	0	649

Table J-66. 2022 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	32147
Bottlenose dolphins	6	0	349979
Bryde's whales	1	4	245
Kogia spp.	346	1	3549
Short-finned pilot whales	0	0	4457
Sperm whales	0	0	6506

Table J-67. 2022 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	16882
Bottlenose dolphins	0	0	16284
Bryde’s whales	0	0	118
Kogia spp.	362	0	2094
Short-finned pilot whales	0	0	1923
Sperm whales	0	0	3598

Table J-68. 2022 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	3651
Bottlenose dolphins	1	0	3650
Bryde’s whales	0	2	25
Kogia spp.	106	0	495
Short-finned pilot whales	0	0	442
Sperm whales	0	0	882

J.2.7. 2023

Table J-69. 2023 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	1294
Bottlenose dolphins	0	0	1588
Bryde’s whales	0	0	15
Kogia spp.	12	0	161
Short-finned pilot whales	0	0	149
Sperm whales	0	0	216

Table J-70. 2023 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	32147
Bottlenose dolphins	6	0	476116
Bryde's whales	1	4	245
Kogia spp.	346	1	3551
Short-finned pilot whales	0	0	4457
Sperm whales	0	0	6506

Table J-71. 2023 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	12675
Bottlenose dolphins	0	0	66961
Bryde's whales	0	0	74
Kogia spp.	246	0	1484
Short-finned pilot whales	0	0	1385
Sperm whales	0	0	2745

Table J-72. 2023 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	2800
Bottlenose dolphins	0	0	13582
Bryde's whales	0	1	16
Kogia spp.	72	0	350
Short-finned pilot whales	0	0	321
Sperm whales	0	0	679

J.2.8. 2024

Table J-73. 2024 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	0
Bottlenose dolphins	0	0	0
Bryde's whales	0	0	0
Kogia spp.	0	0	0
Short-finned pilot whales	0	0	0
Sperm whales	0	0	0

Table J-74. 2024 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	20301
Bottlenose dolphins	5	0	199285
Bryde's whales	0	3	178
Kogia spp.	301	1	2262
Short-finned pilot whales	0	0	2950
Sperm whales	0	0	4114

Table J-75. 2024 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	14093
Bottlenose dolphins	1	0	12062
Bryde's whales	0	0	96
Kogia spp.	369	0	1655
Short-finned pilot whales	0	0	1542
Sperm whales	0	0	3054

Table J-76. 2024 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	3100
Bottlenose dolphins	0	0	2705
Bryde’s whales	0	2	22
Kogia spp.	108	0	391
Short-finned pilot whales	0	0	355
Sperm whales	0	0	750

J.2.9. 2025

Table J-77. 2025 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	232
Bottlenose dolphins	0	0	484
Bryde’s whales	0	0	5
Kogia spp.	4	0	42
Short-finned pilot whales	0	0	35
Sperm whales	0	0	30

Table J-78. 2025 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	27492
Bottlenose dolphins	5	0	495595
Bryde’s whales	0	3	219
Kogia spp.	302	1	3059
Short-finned pilot whales	0	0	3980
Sperm whales	0	0	5555

Table J-79. 2025 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	15843
Bottlenose dolphins	0	0	69706
Bryde's whales	0	0	93
Kogia spp.	308	0	1855
Short-finned pilot whales	0	0	1732
Sperm whales	0	0	3431

Table J-80. 2025 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	3499
Bottlenose dolphins	0	0	14206
Bryde's whales	0	1	21
Kogia spp.	90	0	437
Short-finned pilot whales	0	0	401
Sperm whales	0	0	849

J.3. No Aversion, Alternate Marine Mammal Density Estimates

Table J-81. 2016 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	2612
Bottlenose dolphins	0	0	3698
Bryde's whales	0	0	5
Kogia spp.	25	0	223
Short-finned pilot whales	0	0	380
Sperm whales	0	0	476

Table J-82. 2016 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	25648
Bottlenose dolphins	34	0	361748
Bryde's whales	0	1	52
Kogia spp.	355	0	2240
Short-finned pilot whales	0	0	5123
Sperm whales	1	0	5171

Table J-83. 2016 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	11110
Bottlenose dolphins	4	0	18713
Bryde's whales	0	0	23
Kogia spp.	357	0	1168
Short-finned pilot whales	0	0	2074
Sperm whales	0	0	2395

Table J-84. 2016 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	2812
Bottlenose dolphins	2	0	4356
Bryde’s whales	0	0	5
Kogia spp.	114	0	301
Short-finned pilot whales	0	0	561
Sperm whales	0	0	654

J.3.1. 2017

Table J-85. 2017 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	838
Bottlenose dolphins	0	0	2877
Bryde’s whales	0	0	5
Kogia spp.	17	0	117
Short-finned pilot whales	0	0	149
Sperm whales	0	0	91

Table J-86. 2017 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	23393
Bottlenose dolphins	39	0	486653
Bryde’s whales	0	1	45
Kogia spp.	330	0	2083
Short-finned pilot whales	0	0	5037
Sperm whales	1	0	4665

Table J-87. 2017 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	9870
Bottlenose dolphins	4	0	73886
Bryde’s whales	0	0	19
Kogia spp.	325	0	1066
Short-finned pilot whales	0	0	1959
Sperm whales	0	0	2115

Table J-88. 2017 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	2491
Bottlenose dolphins	6	0	15592
Bryde’s whales	0	0	4
Kogia spp.	104	0	275
Short-finned pilot whales	0	0	526
Sperm whales	0	0	573

J.3.2. 2018

Table J-89. 2018 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	0
Bottlenose dolphins	0	0	0
Bryde’s whales	0	0	0
Kogia spp.	0	0	0
Short-finned pilot whales	0	0	0
Sperm whales	0	0	0

Table J-90. 2018 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	23121
Bottlenose dolphins	27	0	334919
Bryde's whales	0	1	47
Kogia spp.	318	0	2009
Short-finned pilot whales	0	0	4633
Sperm whales	0	0	4675

Table J-91. 2018 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	9465
Bottlenose dolphins	4	0	15943
Bryde's whales	0	0	20
Kogia spp.	303	0	990
Short-finned pilot whales	0	0	1772
Sperm whales	0	0	2042

Table J-92. 2018 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	2397
Bottlenose dolphins	2	0	3717
Bryde's whales	0	0	5
Kogia spp.	96	0	255
Short-finned pilot whales	0	0	479
Sperm whales	0	0	559

J.3.3. 2019

Table J-93. 2019 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	2983
Bottlenose dolphins	0	0	7603
Bryde's whales	0	0	13
Kogia spp.	46	0	346
Short-finned pilot whales	0	0	489
Sperm whales	0	0	420

Table J-94. 2019 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	17581
Bottlenose dolphins	46	0	505594
Bryde's whales	0	1	43
Kogia spp.	273	0	1653
Short-finned pilot whales	0	0	4277
Sperm whales	0	0	3385

Table J-95. 2019 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	5555
Bottlenose dolphins	2	0	68225
Bryde's whales	0	0	14
Kogia spp.	202	0	641
Short-finned pilot whales	0	0	1193
Sperm whales	0	0	1142

Table J-96. 2019 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	1394
Bottlenose dolphins	6	0	14212
Bryde's whales	0	0	3
Kogia spp.	65	0	168
Short-finned pilot whales	0	0	316
Sperm whales	0	0	303

J.3.4. 2020

Table J-97. 2020 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	0
Bottlenose dolphins	0	0	0
Bryde's whales	0	0	0
Kogia spp.	0	0	0
Short-finned pilot whales	0	0	0
Sperm whales	0	0	0

Table J-98. 2020 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	20995
Bottlenose dolphins	26	0	337725
Bryde's whales	0	1	53
Kogia spp.	336	0	2013
Short-finned pilot whales	0	0	4334
Sperm whales	0	0	3986

Table J-99. 2020 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	9870
Bottlenose dolphins	4	0	16620
Bryde's whales	0	0	19
Kogia spp.	325	0	1065
Short-finned pilot whales	0	0	1811
Sperm whales	0	0	2115

Table J-100. 2020 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	2491
Bottlenose dolphins	2	0	3835
Bryde's whales	0	0	4
Kogia spp.	104	0	274
Short-finned pilot whales	0	0	495
Sperm whales	0	0	573

J.3.5. 2021

Table J-101. 2021 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	0
Bottlenose dolphins	0	0	0
Bryde's whales	0	0	0
Kogia spp.	0	0	0
Short-finned pilot whales	0	0	0
Sperm whales	0	0	0

Table J-102. 2021 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	20722
Bottlenose dolphins	40	0	488573
Bryde's whales	0	1	55
Kogia spp.	324	0	1942
Short-finned pilot whales	0	0	4778
Sperm whales	0	0	3996

Table J-103. 2021 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	9465
Bottlenose dolphins	4	0	73209
Bryde's whales	0	0	20
Kogia spp.	303	0	991
Short-finned pilot whales	0	0	1920
Sperm whales	0	0	2042

Table J-104. 2021 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	2397
Bottlenose dolphins	6	0	15473
Bryde's whales	0	0	5
Kogia spp.	96	0	255
Short-finned pilot whales	0	0	510
Sperm whales	0	0	559

J.3.6. 2022

Table J-105. 2022 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	2144
Bottlenose dolphins	0	0	4726
Bryde’s whales	0	0	8
Kogia spp.	29	0	228
Short-finned pilot whales	0	0	339
Sperm whales	0	0	329

Table J-106. 2022 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	17581
Bottlenose dolphins	33	0	354303
Bryde’s whales	0	1	43
Kogia spp.	273	0	1651
Short-finned pilot whales	0	0	3853
Sperm whales	0	0	3385

Table J-107. 2022 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	8845
Bottlenose dolphins	4	0	16498
Bryde’s whales	0	0	20
Kogia spp.	310	0	996
Short-finned pilot whales	0	0	1649
Sperm whales	0	0	1847

Table J-108. 2022 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	2224
Bottlenose dolphins	3	0	3734
Bryde's whales	0	0	4
Kogia spp.	100	0	259
Short-finned pilot whales	0	0	450
Sperm whales	0	0	495

J.3.7. 2023

Table J-109. 2023 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	715
Bottlenose dolphins	0	0	1575
Bryde's whales	0	0	3
Kogia spp.	10	0	76
Short-finned pilot whales	0	0	113
Sperm whales	0	0	110

Table J-110. 2023 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	17581
Bottlenose dolphins	39	0	482312
Bryde's whales	0	1	43
Kogia spp.	273	0	1653
Short-finned pilot whales	0	0	4186
Sperm whales	0	0	3385

Table J-111. 2023 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	6580
Bottlenose dolphins	3	0	68346
Bryde's whales	0	0	13
Kogia spp.	217	0	711
Short-finned pilot whales	0	0	1355
Sperm whales	0	0	1410

Table J-112. 2023 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	1661
Bottlenose dolphins	5	0	14313
Bryde's whales	0	0	3
Kogia spp.	69	0	183
Short-finned pilot whales	0	0	361
Sperm whales	0	0	382

J.3.8. 2024

Table J-113. 2024 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	0
Bottlenose dolphins	0	0	0
Bryde's whales	0	0	0
Kogia spp.	0	0	0
Short-finned pilot whales	0	0	0
Sperm whales	0	0	0

Table J-114. 2024 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	11103
Bottlenose dolphins	26	0	202485
Bryde's whales	0	1	36
Kogia spp.	236	0	1044
Short-finned pilot whales	0	0	2359
Sperm whales	0	0	2140

Table J-115. 2024 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	7288
Bottlenose dolphins	4	0	12190
Bryde's whales	0	0	20
Kogia spp.	325	0	790
Short-finned pilot whales	0	0	1339
Sperm whales	0	0	1570

Table J-116. 2024 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	1835
Bottlenose dolphins	2	0	2778
Bryde's whales	0	0	5
Kogia spp.	104	0	203
Short-finned pilot whales	0	0	364
Sperm whales	0	0	423

J.3.9. 2025

Table J-117. 2025 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	140
Bottlenose dolphins	0	0	479
Bryde's whales	0	0	1
Kogia spp.	3	0	20
Short-finned pilot whales	0	0	25
Sperm whales	0	0	15

Table J-118. 2025 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	15054
Bottlenose dolphins	46	0	502044
Bryde's whales	0	1	38
Kogia spp.	237	0	1422
Short-finned pilot whales	0	0	3878
Sperm whales	0	0	2889

Table J-119. 2025 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	8225
Bottlenose dolphins	4	0	71116
Bryde's whales	0	0	16
Kogia spp.	271	0	888
Short-finned pilot whales	0	0	1657
Sperm whales	0	0	1762

Table J-120. 2025 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	2076
Bottlenose dolphins	5	0	14952
Bryde's whales	0	0	4
Kogia spp.	87	0	229
Short-finned pilot whales	0	0	443
Sperm whales	0	0	478

J.4. Aversion, Alternate Marine Mammal Density Estimates

Table J-121. 2016 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	2487
Bottlenose dolphins	1	0	3690
Bryde's whales	0	0	5
Kogia spp.	14	0	229
Short-finned pilot whales	0	0	409
Sperm whales	0	0	466

Table J-122. 2016 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	23864
Bottlenose dolphins	6	0	354338
Bryde's whales	0	1	51
Kogia spp.	218	0	2390
Short-finned pilot whales	0	0	5428
Sperm whales	0	0	4945

Table J-123. 2016 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	10719
Bottlenose dolphins	1	0	18393
Bryde's whales	0	0	23
Kogia spp.	202	0	1219
Short-finned pilot whales	0	0	2187
Sperm whales	0	0	2323

Table J-124. 2016 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	2394
Bottlenose dolphins	1	0	4217
Bryde's whales	0	0	5
Kogia spp.	59	0	287
Short-finned pilot whales	0	0	530
Sperm whales	0	0	576

J.4.1. 2017

Table J-125. 2017 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	695
Bottlenose dolphins	0	0	2875
Bryde's whales	0	0	5
Kogia spp.	12	0	127
Short-finned pilot whales	0	0	164
Sperm whales	0	0	90

Table J-126. 2017 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	21657
Bottlenose dolphins	6	0	476179
Bryde's whales	0	1	44
Kogia spp.	202	0	2221
Short-finned pilot whales	0	0	5303
Sperm whales	0	0	4459

Table J-127. 2017 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	9490
Bottlenose dolphins	1	0	71794
Bryde’s whales	0	0	19
Kogia spp.	184	0	1110
Short-finned pilot whales	0	0	2067
Sperm whales	0	0	2051

Table J-128. 2017 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	2096
Bottlenose dolphins	0	0	14696
Bryde’s whales	0	0	4
Kogia spp.	54	0	262
Short-finned pilot whales	0	0	497
Sperm whales	0	0	507

J.4.2. 2018

Table J-129. 2018 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	0
Bottlenose dolphins	0	0	0
Bryde’s whales	0	0	0
Kogia spp.	0	0	0
Short-finned pilot whales	0	0	0
Sperm whales	0	0	0

Table J-130. 2018 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	21542
Bottlenose dolphins	6	0	327973
Bryde's whales	0	1	47
Kogia spp.	195	0	2145
Short-finned pilot whales	0	0	4906
Sperm whales	0	0	4471

Table J-131. 2018 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	9137
Bottlenose dolphins	0	0	15673
Bryde's whales	0	0	20
Kogia spp.	172	0	1034
Short-finned pilot whales	0	0	1868
Sperm whales	0	0	1981

Table J-132. 2018 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	2045
Bottlenose dolphins	1	0	3598
Bryde's whales	0	0	4
Kogia spp.	50	0	243
Short-finned pilot whales	0	0	452
Sperm whales	0	0	491

J.4.3. 2019

Table J-133. 2019 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	2633
Bottlenose dolphins	0	0	7595
Bryde's whales	0	0	13
Kogia spp.	30	0	368
Short-finned pilot whales	0	0	533
Sperm whales	0	0	414

Table J-134. 2019 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	16046
Bottlenose dolphins	6	0	494638
Bryde's whales	0	1	42
Kogia spp.	173	0	1772
Short-finned pilot whales	0	0	4485
Sperm whales	0	0	3240

Table J-135. 2019 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	5263
Bottlenose dolphins	0	0	66173
Bryde's whales	0	0	14
Kogia spp.	120	0	675
Short-finned pilot whales	0	0	1264
Sperm whales	0	0	1108

Table J-136. 2019 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	1124
Bottlenose dolphins	0	0	13363
Bryde’s whales	0	0	3
Kogia spp.	35	0	160
Short-finned pilot whales	0	0	299
Sperm whales	0	0	270

J.4.4. 2020

Table J-137. 2020 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	0
Bottlenose dolphins	0	0	0
Bryde’s whales	0	0	0
Kogia spp.	0	0	0
Short-finned pilot whales	0	0	0
Sperm whales	0	0	0

Table J-138. 2020 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	19048
Bottlenose dolphins	8	0	330616
Bryde’s whales	0	1	52
Kogia spp.	214	0	2160
Short-finned pilot whales	0	0	4606
Sperm whales	0	0	3816

Table J-139. 2020 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	9490
Bottlenose dolphins	1	0	16318
Bryde's whales	0	0	19
Kogia spp.	184	0	1110
Short-finned pilot whales	0	0	1914
Sperm whales	0	0	2051

Table J-140. 2020 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	2096
Bottlenose dolphins	0	0	3713
Bryde's whales	0	0	4
Kogia spp.	54	0	262
Short-finned pilot whales	0	0	468
Sperm whales	0	0	507

J.4.5. 2021

Table J-141. 2021 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	0
Bottlenose dolphins	0	0	0
Bryde's whales	0	0	0
Kogia spp.	0	0	0
Short-finned pilot whales	0	0	0
Sperm whales	0	0	0

Table J-142. 2021 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	18933
Bottlenose dolphins	7	0	478068
Bryde’s whales	0	1	54
Kogia spp.	207	0	2086
Short-finned pilot whales	0	0	5035
Sperm whales	0	0	3828

Table J-143. 2021 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	9137
Bottlenose dolphins	0	0	71149
Bryde’s whales	0	0	20
Kogia spp.	172	0	1035
Short-finned pilot whales	0	0	2022
Sperm whales	0	0	1981

Table J-144. 2021 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	2045
Bottlenose dolphins	1	0	14582
Bryde’s whales	0	0	4
Kogia spp.	50	0	243
Short-finned pilot whales	0	0	480
Sperm whales	0	0	491

J.4.6. 2022

Table J-145. 2022 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	1938
Bottlenose dolphins	0	0	4720
Bryde’s whales	0	0	8
Kogia spp.	18	0	241
Short-finned pilot whales	0	0	369
Sperm whales	0	0	323

Table J-146. 2022 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	16046
Bottlenose dolphins	6	0	346809
Bryde’s whales	0	1	42
Kogia spp.	173	0	1771
Short-finned pilot whales	0	0	4072
Sperm whales	0	0	3240

Table J-147. 2022 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	8426
Bottlenose dolphins	0	0	16136
Bryde’s whales	0	0	20
Kogia spp.	181	0	1045
Short-finned pilot whales	0	0	1748
Sperm whales	0	0	1792

Table J-148. 2022 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	1822
Bottlenose dolphins	1	0	3617
Bryde's whales	0	0	4
Kogia spp.	53	0	247
Short-finned pilot whales	0	0	426
Sperm whales	0	0	439

J.4.7. 2023

Table J-149. 2023 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	646
Bottlenose dolphins	0	0	1573
Bryde's whales	0	0	3
Kogia spp.	6	0	80
Short-finned pilot whales	0	0	123
Sperm whales	0	0	108

Table J-150. 2023 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	16046
Bottlenose dolphins	6	0	471803
Bryde's whales	0	1	42
Kogia spp.	173	0	1772
Short-finned pilot whales	0	0	4395
Sperm whales	0	0	3240

Table J-151. 2023 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	6326
Bottlenose dolphins	0	0	66355
Bryde's whales	0	0	13
Kogia spp.	123	0	740
Short-finned pilot whales	0	0	1429
Sperm whales	0	0	1367

Table J-152. 2023 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	1397
Bottlenose dolphins	0	0	13459
Bryde's whales	0	0	3
Kogia spp.	36	0	175
Short-finned pilot whales	0	0	341
Sperm whales	0	0	338

J.4.8. 2024

Table J-153. 2024 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	0
Bottlenose dolphins	0	0	0
Bryde's whales	0	0	0
Kogia spp.	0	0	0
Short-finned pilot whales	0	0	0
Sperm whales	0	0	0

Table J-154. 2024 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	10134
Bottlenose dolphins	5	0	197480
Bryde's whales	0	1	35
Kogia spp.	150	0	1128
Short-finned pilot whales	0	0	2503
Sperm whales	0	0	2049

Table J-155. 2024 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	7034
Bottlenose dolphins	1	0	11952
Bryde's whales	0	0	21
Kogia spp.	184	0	826
Short-finned pilot whales	0	0	1414
Sperm whales	0	0	1521

Table J-156. 2024 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	1547
Bottlenose dolphins	0	0	2681
Bryde's whales	0	0	5
Kogia spp.	54	0	195
Short-finned pilot whales	0	0	344
Sperm whales	0	0	373

J.4.9. 2025

Table J-157. 2025 annual exposure estimate totals for 2-D survey (4130 in^3 airgun array, 1 vessel).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	116
Bottlenose dolphins	0	0	479
Bryde’s whales	0	0	1
Kogia spp.	2	0	21
Short-finned pilot whales	0	0	27
Sperm whales	0	0	15

Table J-158. 2025 annual exposure estimate totals for 3-D NAZ survey (4130 in^3 airgun array, 2 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	13722
Bottlenose dolphins	5	0	491106
Bryde’s whales	0	1	38
Kogia spp.	150	0	1526
Short-finned pilot whales	0	0	4054
Sperm whales	0	0	2766

Table J-159. 2025 annual exposure estimate totals for 3-D WAZ survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	7908
Bottlenose dolphins	0	0	69075
Bryde’s whales	0	0	16
Kogia spp.	153	0	925
Short-finned pilot whales	0	0	1748
Sperm whales	0	0	1709

Table J-160. 2025 annual exposure estimate totals for Coil survey (4130 in^3 airgun array, 4 vessels).

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	1747
Bottlenose dolphins	0	0	14077
Bryde's whales	0	0	4
Kogia spp.	45	0	218
Short-finned pilot whales	0	0	419
Sperm whales	0	0	423

Appendix K. Annual Aggregate Exposure Estimates

K.1. No Aversion, PEIS Marine Mammal Densities

Table K-1. 2016 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	84508
Bottlenose dolphins	41	0	392066
Bryde's whales	1	7	494
Kogia spp.	1705	0	7880
Short-finned pilot whales	0	0	10175
Sperm whales	1	0	17461

Table K-2. 2017 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	73313
Bottlenose dolphins	50	0	584300
Bryde's whales	1	6	432
Kogia spp.	1555	0	7095
Short-finned pilot whales	0	0	7772
Sperm whales	1	0	14946

Table K-3. 2018 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	70086
Bottlenose dolphins	33	0	357820
Bryde's whales	1	6	417
Kogia spp.	1438	0	6524
Short-finned pilot whales	0	0	8870
Sperm whales	1	0	14610

Table K-4. 2019 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	55121
Bottlenose dolphins	55	0	601078
Bryde's whales	1	6	423
Kogia spp.	1177	0	5626
Short-finned pilot whales	0	0	6248
Sperm whales	1	0	10543

Table K-5. 2020 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	66828
Bottlenose dolphins	33	0	361454
Bryde's whales	1	7	444
Kogia spp.	1533	0	6720
Short-finned pilot whales	0	0	7333
Sperm whales	1	0	13400

Table K-6. 2021 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	65280
Bottlenose dolphins	49	0	582532
Bryde's whales	1	7	462
Kogia spp.	1450	0	6389
Short-finned pilot whales	0	0	8626
Sperm whales	1	0	13247

Table K-7. 2022 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	61696
Bottlenose dolphins	40	0	382727
Bryde’s whales	1	6	437
Kogia spp.	1429	0	6284
Short-finned pilot whales	0	0	6885
Sperm whales	1	0	12159

Table K-8. 2023 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	53166
Bottlenose dolphins	47	0	571726
Bryde’s whales	1	5	355
Kogia spp.	1140	0	5256
Short-finned pilot whales	0	0	5973
Sperm whales	1	0	10615

Table K-9. 2024 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	40504
Bottlenose dolphins	33	0	219440
Bryde’s whales	1	6	298
Kogia spp.	1333	0	4084
Short-finned pilot whales	0	0	4579
Sperm whales	1	0	8299

Table K-10. 2025 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	51077
Bottlenose dolphins	56	0	593973
Bryde’s whales	1	5	341
Kogia spp.	1196	0	5128
Short-finned pilot whales	0	0	5836
Sperm whales	1	0	10329

K.2. Aversion, PEIS Marine Mammal Densities

Table K-11. 2016 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	79064
Bottlenose dolphins	8	0	384117
Bryde’s whales	1	6	488
Kogia spp.	988	1	8269
Short-finned pilot whales	0	0	10680
Sperm whales	1	0	16686

Table K-12. 2017 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	67992
Bottlenose dolphins	7	0	570714
Bryde’s whales	1	5	427
Kogia spp.	905	1	7456
Short-finned pilot whales	0	0	8178
Sperm whales	1	0	14270

Table K-13. 2018 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier’s beaked whales	0	0	65561
Bottlenose dolphins	7	0	350418
Bryde’s whales	1	6	412
Kogia spp.	836	1	6859
Short-finned pilot whales	0	0	9307
Sperm whales	1	0	13943

Table K-14. 2019 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	50217
Bottlenose dolphins	7	0	587086
Bryde's whales	1	5	420
Kogia spp.	717	1	5963
Short-finned pilot whales	0	0	6630
Sperm whales	1	0	10106

Table K-15. 2020 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	61374
Bottlenose dolphins	9	0	353852
Bryde's whales	1	6	439
Kogia spp.	905	1	7080
Short-finned pilot whales	0	0	7734
Sperm whales	1	0	12797

Table K-16. 2021 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	60334
Bottlenose dolphins	8	0	568953
Bryde's whales	1	7	457
Kogia spp.	861	1	6743
Short-finned pilot whales	0	0	9077
Sperm whales	1	0	12651

Table K-17. 2022 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	56563
Bottlenose dolphins	8	0	374676
Bryde's whales	1	6	433
Kogia spp.	852	1	6622
Short-finned pilot whales	0	0	7270
Sperm whales	1	0	11636

Table K-18. 2023 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	48915
Bottlenose dolphins	7	0	558246
Bryde's whales	1	5	351
Kogia spp.	677	1	5547
Short-finned pilot whales	0	0	6312
Sperm whales	1	0	10146

Table K-19. 2024 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	37494
Bottlenose dolphins	6	0	214052
Bryde's whales	1	5	297
Kogia spp.	778	1	4307
Short-finned pilot whales	0	0	4847
Sperm whales	1	0	7918

Table K-20. 2025 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	47066
Bottlenose dolphins	6	0	579991
Bryde's whales	1	5	337
Kogia spp.	703	1	5394
Short-finned pilot whales	0	0	6148
Sperm whales	0	0	9864

K.3. No Aversion, Alternate Marine Mammal Densities

Table K-21. 2016 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	42181
Bottlenose dolphins	40	0	388515
Bryde's whales	0	1	85
Kogia spp.	851	0	3932
Short-finned pilot whales	0	0	8138
Sperm whales	1	0	8696

Table K-22. 2017 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	36593
Bottlenose dolphins	49	0	579007
Bryde's whales	0	1	74
Kogia spp.	776	0	3540
Short-finned pilot whales	0	0	7671
Sperm whales	1	0	7444

Table K-23. 2018 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	34982
Bottlenose dolphins	32	0	354578
Bryde's whales	0	1	72
Kogia spp.	717	0	3255
Short-finned pilot whales	0	0	6884
Sperm whales	0	0	7276

Table K-24. 2019 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	27513
Bottlenose dolphins	54	0	595633
Bryde's whales	0	1	73
Kogia spp.	587	0	2807
Short-finned pilot whales	0	0	6275
Sperm whales	0	0	5251

Table K-25. 2020 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	33356
Bottlenose dolphins	32	0	358180
Bryde's whales	0	1	76
Kogia spp.	765	0	3353
Short-finned pilot whales	0	0	6639
Sperm whales	0	0	6673

Table K-26. 2021 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	32584
Bottlenose dolphins	49	0	577256
Bryde's whales	0	1	79
Kogia spp.	723	0	3188
Short-finned pilot whales	0	0	7208
Sperm whales	0	0	6597

Table K-27. 2022 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	30795
Bottlenose dolphins	40	0	379260
Bryde's whales	0	1	75
Kogia spp.	713	0	3135
Short-finned pilot whales	0	0	6292
Sperm whales	0	0	6055

Table K-28. 2023 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	26537
Bottlenose dolphins	47	0	566547
Bryde's whales	0	1	61
Kogia spp.	569	0	2622
Short-finned pilot whales	0	0	6015
Sperm whales	0	0	5287

Table K-29. 2024 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	20225
Bottlenose dolphins	32	0	217453
Bryde's whales	0	1	61
Kogia spp.	665	0	2038
Short-finned pilot whales	0	0	4061
Sperm whales	0	0	4133

Table K-30. 2025 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	25495
Bottlenose dolphins	55	0	588592
Bryde's whales	0	1	59
Kogia spp.	597	0	2559
Short-finned pilot whales	0	0	6003
Sperm whales	0	0	5144

K.4. Aversion, Alternate Marine Mammal Densities

Table K-31. 2016 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	39464
Bottlenose dolphins	8	0	380638
Bryde's whales	0	1	84
Kogia spp.	493	0	4125
Short-finned pilot whales	0	0	8554
Sperm whales	1	0	8310

Table K-32. 2017 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	33937
Bottlenose dolphins	7	0	565544
Bryde's whales	0	1	73
Kogia spp.	452	0	3720
Short-finned pilot whales	0	0	8031
Sperm whales	0	0	7107

Table K-33. 2018 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	32724
Bottlenose dolphins	6	0	347244
Bryde's whales	0	1	71
Kogia spp.	417	0	3422
Short-finned pilot whales	0	0	7226
Sperm whales	0	0	6944

Table K-34. 2019 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	25065
Bottlenose dolphins	7	0	581768
Bryde's whales	0	1	72
Kogia spp.	358	0	2975
Short-finned pilot whales	0	0	6581
Sperm whales	0	0	5033

Table K-35. 2020 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	30634
Bottlenose dolphins	9	0	350647
Bryde's whales	0	1	76
Kogia spp.	452	0	3532
Short-finned pilot whales	0	0	6988
Sperm whales	0	0	6373

Table K-36. 2021 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	30115
Bottlenose dolphins	8	0	563799
Bryde's whales	0	1	79
Kogia spp.	430	0	3364
Short-finned pilot whales	0	0	7537
Sperm whales	0	0	6301

Table K-37. 2022 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	28232
Bottlenose dolphins	7	0	371282
Bryde's whales	0	1	74
Kogia spp.	425	0	3304
Short-finned pilot whales	0	0	6615
Sperm whales	0	0	5795

Table K-38. 2023 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	24415
Bottlenose dolphins	7	0	553190
Bryde's whales	0	1	60
Kogia spp.	338	0	2767
Short-finned pilot whales	0	0	6288
Sperm whales	0	0	5053

Table K-39. 2024 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	18715
Bottlenose dolphins	6	0	212113
Bryde's whales	0	1	61
Kogia spp.	388	0	2149
Short-finned pilot whales	0	0	4261
Sperm whales	0	0	3943

Table K-40. 2025 annual exposure estimate totals for all sources.

Species	Number of Level A exposures		Number of Level B exposures
	peak SPL	SEL	Step function
Cuvier's beaked whales	0	0	23493
Bottlenose dolphins	6	0	574737
Bryde's whales	0	1	58
Kogia spp.	351	0	2691
Short-finned pilot whales	0	0	6248
Sperm whales	0	0	4913