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VIA EMAIL (ITP.Laws@NOAA.gov)

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 Incidental Harassment Authorizations for the Incidental Taking of Marine Mammals During Geophysical Surveys in the Atlantic Ocean

Dear Ms. Harrison:

This letter provides the comments of the International Association of Geophysical Contractors ("IAGC"), the American Petroleum Institute ("API"), and the National Ocean Industries Association ("NOIA") (collectively, the "Associations") in response to the National Marine Fisheries Service's ("NMFS") request for comments on five proposed Incidental Harassment Authorizations ("Proposed IHAs") for the incidental taking of marine mammals during geophysical surveys on the outer continental shelf ("OCS") of the Atlantic Ocean. *See* 82 Fed. Reg. 26,244 (June 6, 2017). The comments presented in this letter address the Proposed IHAs collectively.

We strongly support geophysical surveying in the Mid- and South Atlantic OCS, which furthers our common interest in the safe and responsible development of domestic oil and gas reserves. As addressed in the comments below, we support NMFS's proposed decision to issue the five IHAs because the best available scientific information demonstrates, *inter alia*, that the proposed surveys will have no more than a negligible impact on marine mammal species or stocks. Our comments also express concerns with the impracticability of, and lack of scientific support for, some of the proposed mitigation measures and with NMFS's substantial overestimation of the number of incidental takes that may occur as result of the surveys. We appreciate NMFS's consideration of our comments.

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. IAGC members have expressed interest in conducting geophysical activities on the Atlantic OCS, and all five of the seismic survey applicants to whom the Proposed IHAs relate are IAGC members.

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 United States' 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.

II. COMMENTS[1]

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

1. Legal context.

The Marine Mammal Protection Act ("MMPA") provides mechanisms for the authorization of the taking of marine mammals incidental to lawful activities. *See* 16 U.S.C. § 1371(a)(5). To issue an incidental take authorization, NMFS must find that the activity is limited to a "specified geographical region," have no more than a "negligible impact" on a marine mammal species or stock, result in the incidental take of "small numbers" of marine mammals, and have the least practicable impact on the affected species or stocks. 16 U.S.C. § 1371(a)(5)(A), (D). NMFS has a long and successful history of issuing such authorizations for seismic surveys in the Beaufort and Chukchi Seas, and in Cook Inlet, Alaska.[2]

NMFS's authorization of marine mammal take incidental to geophysical survey activities in the Atlantic OCS is consistent with the Outer Continental Shelf Lands Act ("OCSLA"), which 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, the President recently signed an Executive Order expressly stating that it "shall be the policy of the United States to encourage energy exploration and production,

[1] The Associations filed comment letters, dated July 2, 2012 and May 7, 2014, in response to the draft and final versions, respectively, of BOEM's Atlantic Geological and Geophysical Activities Programmatic Environmental Impact Statement ("PEIS"). The Associations also filed a comment letter, dated August 28, 2015, in response to NMFS's preliminary notice of the Atlantic IHA applications. Finally, the Associations filed a comment letter, dated April 29, 2015, in response to applications for G&G permits in the Mid- and South Atlantic OCS. We hereby incorporate all of those comment letters by reference, and they are included in Attachment A to this letter to ensure they are included in the administrative record.

[2] API and IAGC recognize that this action relates to oil and gas seismic surveys. However, it bears emphasis that NMFS has issued numerous MMPA authorizations for marine geophysical surveys in the Atlantic and Pacific Oceans, all which have concluded that the impacts would be negligible. *See, e.g.*, 70 Fed. Reg. 52,122 (Sept. 2, 2014) (Incidental Harassment Authorization issued to the USGS, L-DEO and NSF in the Atlantic seaboard).

including on the Outer Continental Shelf . . . while ensuring that any such activity is safe and environmentally responsible."[3]

Here, the geophysical activities to which the Proposed IHAs would apply are authorized by BOEM pursuant to OCSLA. *See* 43 U.S.C. § 1340. Neither OCSLA nor the MMPA requires an applicant for a geological and geophysical ("G&G") permit under OCSLA to obtain an incidental take authorization under the MMPA. However, unlawful incidental takes of marine mammals may be subject to MMPA-based penalties. *See* 16 U.S.C. § 1375.

In the Gulf of Mexico ("GOM"), industry operators have for years complied with measures imposed under the terms of seismic 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 successful. Based on the best available scientific information, there has been no demonstration of any biologically significant negative impacts to marine life from G&G activities in the GOM. *See infra* http://www.boem.gov/BOEM-Science-Note-August-2014/ (*Science Notes*, Aug. 22, 2014); https://www.boem.gov/BOEM-Science-Note-March-2015/ (*Science Notes*, Mar. 9, 2015). In fact, BOEM recently reconfirmed 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."[4]

2. Operational context.

BOEM currently estimates that the Mid- and South Atlantic OCS holds at least 4.59 billion barrels of oil and 38.17 trillion cubic feet of natural gas.[5] Although these estimates are impressive, it is widely believed that modern seismic imaging—the only feasible technology that accurately creates a subsurface image before a well is drilled—will aid in better locating and dissecting prospective areas for exploration and provide more realistic estimates of the potential resource. The pending geophysical survey proposals (for which MMPA incidental take authorizations have been requested) will facilitate the safe and orderly development of oil and gas reserves in the Mid- and South Atlantic OCS.

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[3] Presidential Executive Order Implementing an America-First Offshore Energy Strategy (April 28, 2017), https://www.whitehouse.gov/the-press-office/2017/04/28/presidential-executive-order-implementing-america-first-offshore-energy.

[4] Draft Programmatic Environmental Impact Statement to evaluate potential environmental effects of multiple geological and geophysical activities on the Gulf of Mexico Outer Continental Shelf ("GOM DPEIS") at 4-57 (emphasis added).

[5] *See* https://www.boem.gov/2016-National-Assessment-Fact-Sheet/.

Seismic 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. Industry has made significant improvements in acquisition efficiency in recent years. Using standard hardware, we now acquire more and better quality data due to advancements in vessels, configurations, acquisition planning and execution, and data processing. 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 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 and 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 the most modern technology to make improved evaluations. Moreover, because G&G activities are temporary and transitory, seismic surveying is the least intrusive and most cost-effective means to determine the likely locations of recoverable oil and gas resources in the Atlantic OCS.

Finally, we note that the Marine Mammal Commission ("MMC"), in a letter dated July 6, 2017, recommended that "BOEM and NMFS could seek to reduce the number of surveys authorized such that not more than one survey is conducted in any particular area in a given year . . . . [and] that NMFS work with BOEM to require companies to . . . reduce the potential for multiple overlapping surveys." *See* https://www.mmc.gov/wp-content/uploads/17-07-06-Harrison-NMFS-Atlantic-seismic-surveys-IHAs_with-figure.pdf. Respectfully, the MMC's recommendations are based upon a substantial misunderstanding of important technical, operational, and economic aspects of seismic surveying. *See* Attachment B. In addition, BOEM recently completed a study regarding "duplicative" seismic surveys, which is described in the GOM DPEIS, Appendix L, pp. L-11 – L-39. None of the surveys currently proposed for the Atlantic OCS meet the definition of a "duplicate" survey, as set forth in the GOM DPEIS duplicate survey report. In short, the MMC's recommendations are not supported by the best available information, and are infeasible and impracticable.[6]

[6] The MMC's recommendations, therefore, should not and cannot be included in the terms of MMPA authorizations, which may only include "practicable" mitigation measures. 16 U.S.C. § 1371(a)(5)(D)(ii)(I).

B. The proposed mitigation measures will effectively minimize and avoid incidental takes, but some proposed measures are impracticable and must be revised or eliminated.

The best available scientific data and information demonstrate that mitigation programs can and do effectively minimize and avoid the incidental take of marine mammals associated with offshore geophysical survey operations.[7] Insofar as we are aware, no seismic activities that 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.

The Proposed IHAs incorporate some of the mitigation measures recommended in the preferred alternative of the PEIS. The Associations commented in detail on those recommended measures. *See* Attachment A. For the reasons stated in our previous comments, some of the measures proposed in the PEIS are not consistent with the best available science and are unnecessarily overbroad. We encourage NMFS to apply only those mitigation measures that are appropriate for the specific IHAs requested here and that result in the least practicable adverse impact, as required by the MMPA. In this light, we commend NMFS's decision to not require a 60-minute "all clear" period or a minimum separation distance between surveys. As stated in our previous comments, those measures are unsupported by the best available information, impracticable, and would not provide additional protections for marine mammals.

[7] *See infra* note 17. A study of more than a decade's worth of marine mammal observation data performed by the Joint Nature Conservation Committee ("JNCC") demonstrates that mitigation measures significantly reduce the effects of seismic activities on marine mammals. *See* http://jncc.defra.gov.uk/page-6985. A copy of the JNCC study is provided in Attachment C for inclusion in the administrative record. *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.").

As addressed below, NMFS has proposed some mitigation measures that are not practicable and are without scientific support. These measures will likely result in increased survey duration, which, in turn, can increase the potential exposure of marine mammals to seismic-related effects because shutdowns and delays necessarily result in overall increased surveying time to preserve data quality and integrity. *See* 82 Fed. Reg. at 26,254 ("Increased shutdowns, without a firm idea of the outcome the measure seeks to avoid, simply displace seismic activity in time and increase the total duration of acoustic influence as well as total sound energy in the water. . . ."). Moreover, if implemented, these measures will have substantial adverse effects on offshore geophysical operations, threatening the economic viability of seismic exploration of the Atlantic OCS, contrary to OCSLA's purposes.[8]

1. Dolphin shutdowns.

We appreciate NMFS's inclusion of an exemption from the shutdown requirements for small dolphins approaching a seismic vessel. However, this exemption is too narrow and will not meaningfully alleviate the substantial number of dolphin-related shutdowns that will occur under the IHAs as proposed. Instead, the exemption should apply to all dolphin species regardless of dolphin behavior. Such an exemption is well-supported by the best available science, which shows that seismic surveys do not have any meaningful adverse effects on dolphin species.[9] Our previous comments detail the additional reasons why no shutdown requirement for dolphins is justified. *See* Attachment A.

[8] In some instances, NMFS suggests that it may alter or add mitigation requirements after the IHAs have been issued. *See, e.g.*, 82 Fed. Reg. at 26,250 (mitigation requirement "may be augmented as necessary"); *id*. at 26,252 (mitigation requirement "may be modified on the basis of any new information presented that justifies a different protocol"). However, NMFS may only modify the IHA requirements if the specific regulatory grounds for modifying, withdrawing, or suspending IHAs are satisfied, and after providing public notice and opportunity for comment. *See* 50 C.F.R. § 216.107(f).

[9] *See* 82 Fed. Reg. at 26,253 ("auditory injury is extremely unlikely to occur for mid-frequency cetaceans (*e.g.*, delphinids) as this group is relatively insensitive to sound produced at the predominant frequencies in an airgun pulse while also having a relatively high threshold for the onset of auditory injury (*i.e.*, permanent threshold shift)"); *id*. ("Although other mid-frequency hearing specialists (*e.g.*, large delphinids) are no more likely to incur auditory injury than are small delphinids, they are much less likely to approach vessels."); Finneran J.J., Schlundt C.E., Branstetter, B.K., Trickey, J.S., Bowman, V., and Jenkins, K. Effects of multiple impulses from a seismic air gun on bottlenose dolphin hearing and behavior. 137 J. Acoust. Soc. Am. 1634-46 (April 2015) (no evidence of TTS when bottlenose dolphins exposed to seismic air pulse at cumulative sound exposure levels of 185-196 dB re 1 μPa^2-s).

Indeed, the best available information shows that none of NMFS's three standards for assessing mitigation measures are satisfied by any measure that requires shutdowns for dolphins. Specifically, any dolphin shutdown measure (1) cannot be "expected to minimize adverse impacts" to dolphins based upon the best available science, (2) has no "proven or likely efficacy . . . to minimize adverse impacts" (also based on the best available science), and (3) will be very impracticable, resulting in an inordinate number of shutdowns. *See* 82 Fed. Reg. at 26,267; *see id.* at 26,298 (NMFS recognition that the expected effects from the proposed activities "are considered low for most delphinids, as it is unlikely that disturbance due to survey noise would entail significant disruption of normal behavioral patterns, long-term displacement, or significant potential for masking of acoustic space").[10] Should NMFS require any shutdown measure for dolphins, then it must provide specific support—in the form of known factual and scientific information—addressing each of these three factors. Absent such concrete support, no dolphin shutdown measure is warranted.

2. Shutdowns for certain marine mammal observations "at any distance."

NMFS proposes to require shutdowns for certain types of marine mammal observations "at any distance." *See* 82 Fed. Reg. at 26,254-255. As an initial matter, this requirement is arbitrary and unreasonable because it is unlimited ("at any distance") and therefore contemplates shutdowns for circumstances in which no Level A or Level B harassment will occur. NMFS may not mitigate for non-adverse effects and, instead, must simply ensure the "least practicable impact." As a practical matter, the "at any distance" requirement will cause implementation problems because observers are only required to monitor a 1,000 m zone. For example, if an observer notices an animal beyond 1000 m, he or she will almost certainly feel compelled to look beyond 1000 m to determine whether one of the "at any distance" circumstances is present. The result of these proposed measures is that observers will be constantly monitoring an unlimited zone, which, aside from being unnecessary, may undermine the effectiveness of their monitoring of the 1000 m zone.

Additionally, the specific circumstances to which "at any distance" shutdowns apply are unsupported and will result in an inordinate number of shutdowns to the benefit of neither marine mammals nor seismic operators. For example, at large distances, it will be difficult for observers to determine the presence of calves, whether a sperm whale is diving, or whether six or more animals are present and do not appear to be traveling. Consequently, observers will make frequent "precautionary" shutdown calls for uncertain observations "at any distance." Again, as NMFS has recognized, 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.

[10] Consistent with the best available science, in its recent amended application for MMPA incidental take regulations in the GOM ("GOM ITR Application"), BOEM proposed no shutdown requirements for any dolphin species.

at 26,254. Added survey time also increases safety concerns for operators and ocean users that would otherwise be avoided. In short, the numerous harms and practical implementation difficulties far outweigh any perceived benefit from the overly precautionary "at any distance" shutdown requirements.

Moreover, the circumstances for which NMFS proposes to require "at any distance" shutdowns do not meet NMFS's own criteria because they (1) cannot be "expected to minimize adverse impacts," (2) have no "proven or likely efficacy . . . to minimize adverse impacts," and (3) will be very impracticable. *See* 82 Fed. Reg. at 26,267. We specifically address each of these circumstances as follows:

- Aggregations of marine mammals that do not appear to be traveling: An observer cannot reasonably identify how many marine mammals may be present and whether they are traveling "at any distance" (or at 1000 m). As written, this measure cannot reasonably be implemented because it is vague and unbounded. In addition, NMFS provides no scientific or factual basis for why the potential minor, temporary effects of seismic surveys will have some different adverse effect on aggregations of marine mammals that require measures above and beyond the standard shutdown protocols. In short, this proposed measure cannot reasonably be expected to minimize adverse impacts, has no proven or likely efficacy, and will be very impracticable because of the large number of "precautionary" shutdowns it will generate.

- Large whale with calf: NMFS apparently justifies this proposed measure on the unsupported basis that disturbance of cow-calf pairs "could potentially" result in the separation of the cow-calf pair. We are aware of no evidence, and NMFS cites none, showing that cetacean cow-calf pairs have been separated by seismic surveys or that such separations are likely to occur as a result of already-mitigated seismic surveying.[11] This measure, too, cannot reasonably be expected to minimize adverse impacts, has no proven or likely efficacy, and will be very impracticable because of the large number of "precautionary" shutdowns it will generate.

- Diving sperm whale: An observer cannot reasonably identify whether an animal is a diving sperm whale "at any distance" (or at 1000 m). This proposed measure is

[11] McCauley et al. (2000) conducted seismic exposure trials on humpback whale pods, including cow-calf pairs. None of the pairs were separated when exposed to direct approaches by seismic sounds, and some pods showed an avoidance response. *See* McCauley, R.D., J. Fewtrell, A.J. Duncan, C. Jenner, M-N. Jenner, J.D. Penrose, R.I.T. Prince, A. Adhitya, J. Murdoch, K. MacCabe. 2000. Marine Seismic Surveys: Analysis of propagation of airgun signals; and effects of air-gun exposure on humpback whales, sea turtles, fishes and squid. Centre for Marine Science and Technology, Curtin University of Technology. Prepared for the Australian Petroleum Production Exploration Association. 198 p.

unprecedented and also meets none of NMFS's three mitigation measure criteria. For instance, both seismic surveys and sperm whales are common in the GOM and there is no evidence that detrimental effects to sperm whales from seismic surveys have occurred in the GOM despite the lack of shutdown requirements for diving sperm whales.

- Beaked or *Kogia* species: Beaked whales are known to be sensitive to acoustic disturbance from sonar signals, but no such evidence exists for seismic sounds. Additionally, *Kogia* species are included in the "at any distance" shutdown requirement based on the presumption that they are high-frequency cetaceans and thus would have larger zones of potential auditory injury. However, this presumed higher sensitivity is based on porpoises (*Phocoena phocoena*) with limited relevance to *Kogia* species.[12] NMFS has not provided an adequate basis to support this proposed measure.

- North Atlantic right whale: Although the North Atlantic right whale population is small, the factors limiting its recovery are primarily related to entanglement with fishing gear and ship strikes. Thus, requiring a shutdown of seismic operations for observations of North Atlantic right whales "at any distance" has no impact on the adverse effects responsible for the right whale's decline, nor does it serve to mitigate the impacts, if any, from geophysical surveys.

In sum, the Associations object to the proposed "at any distance" shutdown requirements. In addition to the problems noted above, the proposed "at any distance" requirements directly contradict NMFS's expressed "need for a basic system of seismic mitigation protocols . . . that . . . reduce subjective decision-making for observers to the extent possible." 82 Fed. Reg. at 26,250. We recommend that each of these observational circumstances be subject to the same shutdown requirements as all other marine mammals. Alternatively, the Associations would support a measure in which power-down is required when the observations described above are conclusively made within the 1000 m buffer zone. Under this alternative, full power could resume once the animal(s) leaves the 1000 m zone or, conversely, shutdown protocols would be required if the animal(s) enters the 500 m exclusion zone.[13]

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[12] *See* Southall et al. (2007) (grouping *Kogia* as a high-frequency cetacean based only on similarities in a limited number of recorded *Kogia* echolocation clicks in spite of a substantial anatomical, ecological, and phylogenetic difference from the one high-frequency hearing group species, the harbor porpoise, for which hearing data are available).

[13] Acceptable power-down procedures are a modified version of the procedures described at 79 Fed. Reg. 14,780, 14,797 (Mar. 17, 2014) ("Langseth IHA"). Specifically, the Associations would support power-down procedures similar to those described in the Langseth IHA provided that: (1) power-down would be implemented only if a marine mammal is observed in or entering (not "likely" to enter) the buffer zone; (2) power-down procedures may involve a reduction in the volume and/or pressure of the array; and (3) if a marine mammal is

3. PSO requirements.

In general, the Associations agree that it is helpful to have training requirements and reasonable standards for protected species observers ("PSOs"). However, as addressed below, some of the PSO-related requirements proposed by NMFS are problematic. We are also not clear whether NMFS has considered its existing national standards for PSOs and how those standards apply in this context.[14]

First, NMFS proposes to require NMFS's review and approval of all PSO resumes, accompanied by a "relevant training course information packet." 82 Fed. Reg. at 26,251. Such reviews threaten to delay the planning process for seismic surveys if they are not bounded by some reasonably short time period, with the default being that the observer is approved if NMFS fails to respond within that time period.

Second, NMFS proposes that "a minimum of two PSOs must be on duty and conducting visual observations at all times during daylight hours." *Id*. This requirement, combined with the watch schedule requirements, effectively means that 4-5 PSOs must be onboard the source vessel. It is uncertain whether source vessels can safely accommodate 4-5 PSOs.

Finally, it is infeasible to require that visual PSOs have a minimum of 90 days at-sea experience with no more than 18 months elapsed since the conclusion of the at-sea experience and that PSOs be trained biologists with experience or training in the field identification of marine mammals, including the identification of behaviors. Such rigid restrictions will inevitably eliminate a category of potential PSOs who would otherwise qualify and perform well, which, in turn, will shrink the pool of available PSOs. We recommend that these standards be provided as "guidelines," such that PSOs who do not meet the guidelines may still be approved so long as NMFS determines they are otherwise qualified.

4. Buffers for National Marine Sanctuaries

NMFS has proposed 15 km "buffers" around the boundaries of the Gray's Reef and Monitor National Marine Sanctuaries. However, NMFS has provided no basis for this proposed measure, much less any explanation for how this proposed measure meets the three factors NMFS has identified for the evaluation of mitigation measures. To the contrary, NMFS admits that "[a]ny benefit to marine mammals from these restrictions would likely be minimal." 82 Fed.

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observed within the 500 m exclusion zone, then the reduced array would be shut down and shutdown procedures would apply.

[14] *See* National Standards for a Protected Species Observer and Data Management Program: A Model Using Geological and Geophysical (November 2013), http://www.nmfs.noaa.gov/pr/publications/techmemo/observers_nmfsopr49.pdf

Reg. at 26,266. Neither of these sanctuaries was established for the protection of marine mammals. The Gray's Reef Sanctuary was established to primarily "protect and preserve the live bottom ecosystem," and the Monitor Sanctuary was established to protect historic wreckage. *See* 15 C.F.R. § 938.2; http://monitor.noaa.gov/. There is no reasonable or scientifically supported basis for this proposed mitigation measure and the only supporting information provided by NMFS suggests that the measure will have no or "minimal" benefit.[15] This measure should be eliminated.

5. 1000 m buffer zone for pre-clearance and ramp-up.

NMFS proposes a 1000 m buffer zone applicable during pre-clearance and ramp-up procedures, but provides no support for this novel requirement. Applying a larger exclusion zone (1000 m) during times when the seismic array is not at full power and a smaller one (500 m) when it is at full power is counterintuitive. We assume NMFS believes that applying a larger exclusion zone during ramp-up is precautionary and might improve effectiveness and/or reduce the likelihood of exposing marine mammals during the ramp-up. However, this is not explained in the proposal and, absent a well-supported rationale, this measure should be removed.

6. Closures, SMAs, and DMAs.

We do not agree with the premise for many of the proposed time and area closures or the special management area ("SMA") and dynamic management area ("DMA") requirements, for reasons we have stated in previous comments. *See* Attachment A. Specifically, the DMA measures are very problematic, and unwarranted, for at least the following reasons:

- DMAs were created to address ship strike situations, which involve vessels traveling at high rates of speed (12-20 knots). Indeed, NMFS has indicated that vessel speeds of less than 10 knots are sufficiently protective. *See* 78 Fed. Reg. 73,726 (Dec. 9, 2013). The proposed application of DMAs to seismic operations is therefore contrary to both the original purpose of DMAs (to address ship strikes, not potential acoustic impacts) and NMFS's previous findings.

- Nowhere has NMFS evaluated the operational practicability or effectiveness of applying DMAs to seismic operations. DMAs are unpredictable and the identification of DMAs on short notice will compromise the implementation of seismic survey operations that have been carefully planned over a substantial period of time, with no corresponding benefit.

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[15] *See* Miller, I. and Cripps, E. 2013. Three dimensional marine seismic survey has no measurable effect on species richness or abundance of a coral reef associated fish community. Mar. Pollut. Bull. 77, 63–70. http://refhub.elsevier.com/S0025-326X(16)30958-4/rf0445.

7. Passive acoustic monitoring.

As stated in our previous comments, the Associations continue to believe that the use of Passive Acoustic Monitoring ("PAM") should be optional. PAM is one of several monitoring techniques that offer a monitoring capability during periods of poor visibility or night conditions. PAM complements (rather than replaces) traditional visual monitoring. Mandatory use of PAM may substantially increase survey cost, require the placement of more personnel on vessels, and potentially increase entanglement risk due to more gear being towed in the water. The Associations therefore urge NMFS to propose the use of PAM as a mitigation option that can be elected by an LOA applicant on a case-by-case basis.

For purposes of the Atlantic IHAs, we defer to the positions of the applicants regarding the specific practicability or benefit of the incorporation of PAM as a required element of their respective surveys. We do have practical concerns that (1) the PAM requirements, combined with the watch schedule requirements, mean that 3 PAM operators will need to be onboard the vessel, which could present space and safety concerns and a significant economic impact, especially when considered in the context of the personnel demands of PSO monitoring; and (2) the requirement to diagnose a PAM system malfunction in 30 minutes, with repair in 2 hours, may be unnecessarily rigid and constraining.

8. Vessel strike avoidance.

We also defer to our previous comments, and to the positions of the applicants, regarding the proposed vessel strike avoidance measures. Our primary concern is that the vessel strike avoidance measures be practical, feasible, and safe for the operators. The requirements should be conditioned to apply only in situations "when safety allows" or "to the extent practicable." This would acknowledge the inherent limitations of fully operating seismic vessels and important safety concerns balanced against the very low strike risk posed by seismic vessels that are already transiting at low speeds whenever the acoustic source is in the water (whether activated or not). All available evidence suggests that ship strikes are exceedingly rare when vessels are travelling at less than 10 knots (Knowlton et al. 1995, Clyne 1999, Laist et al. 2001). Because operating seismic vessels generally travel at speeds much lower than this, mitigation measures that attempt to maintain minimum separation distance through steering away from marine mammals are unnecessary, operationally unsafe, and have no scientific basis. Certainly, absent some compelling evidence of heightened risk, vessel strike avoidance requirements imposed upon seismic vessels should not be more stringent than strike avoidance requirements imposed upon other vessels.

Additionally, the Proposed IHAs purport to require a vessel to "reduce speed and shift the engine to neutral" if a right whale is observed within 100 m of the vessel. 82 Fed. Reg. at 26,267. This proposed requirement must be removed. As mentioned in our previous comments, speed alterations, alterations in course, and shifting engines to neutral can present serious safety concerns for seismic vessels. Vessels towing seismic gear typically move at 4-5 knots, which is generally recognized in the U.S. as safe steerage speed and does not require further reduction (or shifting to neutral) in the presence of marine mammals. NMFS's proposal to require operators to shift to neutral may create conflicting obligations for seismic operators as the proposed requirement does not appear to be consistent with the rules set forth under the *Convention on the International Regulations for Preventing Collisions at Sea*.

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

1. The best available scientific information.

For over 40 years, the federal government and academic scientists have studied the potential impacts of G&G 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").[16] These statements accurately summarize the best available scientific information regarding the potential effects of G&G activities on marine mammals. There are no other data to the contrary.

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.[17] More than four decades of worldwide seismic surveying and

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

[17] *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/

scientific research indicate that the risk of physical injury to marine life from seismic survey activities is extremely low. For example, as BOEM concludes in its GOM DPEIS, "within the GOM, there is a long-standing and well-developed OCS [oil and gas] Program (more than 50

("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, 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 ").

years) and there are no data to suggest that activities from the previous OCS Program are significantly impacting marine mammal populations." DPEIS at 4-77.[18]

In addition, a 2016 report from the National Academy of Sciences, Ocean Studies Board (the "NAS Report"),[19] 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);

- "Evidence of the effects of noise on marine mammal populations is largely circumstantial or conjectural" (NAS Report 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" (NAS Report at 35);

- "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. Madsen et al. (2002) studied responses of sperm whales in Norwegian waters to seismic surveys at ranges > 20 km, and reported no responses at exposure ranging up to 123-130 dBRMS." (NAS Report at 56).

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[18] *See also* RPS. 2015. Protected Species Mitigation and Monitoring Report: U.S. Geological Survey 2-D Seismic Reflection Scientific Research Survey Program: Mapping the U.S. Atlantic Seaboard Extended Continental Margin and Investigating Tsunami Hazards, in the northwest Atlantic Ocean, at 37-38, https://www.nsf.gov/geo/oce/envcomp/usgslangseth_2014iha_monrepphase2.pdf ("All potential marine mammal takes for both surveys combined (4) represents <u>0.02 percent of the total takes authorized for marine mammals for the survey</u>.") (emphasis added).

[19] 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 Attachment E to this letter, for NMFS's consideration and for inclusion in the administrative record.

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 where 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, and despite annual lethal takes by Alaska Natives engaged in subsistence activities, bowhead whales have consistently increased in abundance to the point that they are believed to have reached carrying capacity. Similarly, no effects of G&G activities have been observed in Arctic ice seal populations.[20]

Finally, BOEM's Environmental Studies Program has spent more than $50 million on protected species and sound-related research over more than four decades without finding evidence of adverse effects. *See* http://www.boem.gov/BOEM-Science-Note-August-2014/ (*Science Notes*, Aug. 22, 2014) ("Since 1998, BOEM has partnered with academia and other experts to invest more than $50 million on protected species and noise-related research."). The geophysical and oil and gas industries, the National Science Foundation, the U.S. Navy, and others have spent a comparable amount of money on researching potential impacts of anthropogenic sound on marine life and have found no evidence of biologically significant

[20] *See, e.g.*, 84 Fed. Reg. 25,829, 25,834 (May 1, 2012) ("Bowhead whales have continued to travel to the 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."); *see also* 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.").

effects to populations. *See* www.soundandmarinelife.org; https://www.nsf.gov/geo/oce/envcomp/; http://greenfleet.dodlive.mil/environment/lmr/; *see also* http://www.brahss.org.au/content/project.html.

2. NMFS's proposed negligible impact determination.

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, the Arctic, and Cook Inlet, in which no significant effects on *any* marine mammal species or stock have been observed, the Associations concur with NMFS's finding that the Proposed IHAs will have a negligible impact on marine mammal species and stocks. We also emphasize that NMFS's negligible impact determinations are based upon highly conservative, and, in some instances, unrealistic, assumptions about the potential effects of the proposed surveys. For example, as addressed in more detail in Section III.D below, NMFS's estimates of the numbers of potential takes by the proposed surveys are grossly inflated as a result of overly conservative modeling assumptions. The consequence is that NMFS's modeling of potential impacts presumes that far more numbers of animals will be exposed or taken than will actually be taken, based on past and recent observations in the field for similar permitted activities.

In addition, NMFS makes overly conservative "magnitude" and "impact" ratings (such as "high" or "moderate") for many marine mammal stocks or species that cannot be rationally reconciled with the best available scientific data and information. *See* 82 Fed. Reg. at 26,301-306. We are aware of no findings by any agency, including NMFS, that a seismic survey had anything more than an insignificant effect on a marine mammal species or stock. NMFS's "high" and "moderate" magnitude and impact ratings therefore predict potential effects that, quite literally, have never been observed in the multi-decade history of offshore seismic exploration on the U.S. OCS. Accordingly, although the Associations concur with NMFS's conclusion that each Proposed IHA will result in no more than a negligible impact on marine mammal species or stocks, we disagree with NMFS's assumptions, and particularly its inflated "magnitude" and "impact" ratings, because they are inconsistent with the best available scientific information.[21]

[21] 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.

D. NMFS substantially overestimates the number of incidental takes predicted to result from the Proposed IHAs.

NMFS's incidental take estimates for the Proposed IHAs are premised, in substantial part, upon the exposure modeling performed by BOEM in the PEIS.[22] NMFS's reliance on the PEIS exposure modeling results in incidental take estimates far greater than the number of takes that can realistically occur based on past observations and data because the PEIS analysis is premised upon biased modeling that is intentionally designed to overestimate take.

As explained in our PEIS comments, BOEM's evaluation of potential marine mammal impacts at the programmatic level is based on an unrealistic scenario in which seismic activities are projected to result in thousands of incidental takes of marine mammals, which BOEM has definitively stated will not actually occur. *See* Attachment A. BOEM reaffirmed this approach in its response to comments in the Record of Decision associated with the PEIS ("ROD"):

> The take estimates include modeled numbers of both 'Level A' harassment, which is defined as having the potential to injure hearing, and 'Level B' harassment, which is defined as having the potential to disturb. Even as defined to include the sensitive threshold of Level B harassment, the numbers estimated for incidental take are <u>higher than BOEM expects would actually occur</u>. The marine mammal take estimates are estimates of potential take. <u>They do not represent expected levels of actual take</u>. 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. 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. Furthermore, <u>the 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</u>

[22] The IHA applicants cannot be faulted for relying upon the PEIS modeling when estimating the number of the takes in their IHA applications. The DPEIS was issued for public comment before the IHA applications were submitted. Accordingly, the IHA applicants were in the position of having to either submit applications consistent with BOEM's modeling approach or present a new modeling analysis inconsistent with the PEIS and subject to agency scrutiny and objection.

> models is purposely developed to be conservative, and conservative assumptions accumulate throughout the analysis.

ROD at 12 (emphases added). The supposed effects of this worst-case hypothetical scenario are then addressed in the PEIS with mitigation measures, many of which are intended to mitigate the inaccurately presumed effects.[23]

The gist of the agencies' errors is that the PEIS, and therefore of NMFS's take analysis, is based upon a modeling exercise that uses a multiplicative series of conservatively biased assumptions for all uncertain parameter inputs. These assumptions lead to accumulating bias as the cumulative conservative assumptions add up to increasingly unlikely statistical probabilities that are not representative of real-world conditions. Consequently, the results quickly become little more than improbable worst case scenarios—not fair simulations or representations of likely effects.[24]

E. The "small numbers" finding should be thoroughly explained in the final agency record.

MMPA "small numbers" findings have frequently been a topic of litigation and dispute. Given the strong stance in opposition to the Proposed IHAs taken by some advocacy groups, NMFS should assume that its small numbers findings (along with other aspects of the IHAs) will be challenged in court. Based on this reasonable expectation, NMFS should devote special attention to providing well-explained and supported small numbers determinations in its final agency record.

In this light, the Associations direct NMFS to *Ctr. for Biological Diversity v. Salazar*, 695 F.3d 893 (9th Cir. 2012). This decision provides the most recent statement of the law regarding various aspects of MMPA Section 101(a)(5) in the specific context of offshore oil and gas exploration, including application of the MMPA's "small numbers" language. In *Salazar*,

[23] Moreover, the PEIS modeled more than twice as much seismic survey activity as is presented by the activities underlying the Proposed IHAs.

[24] BOEM took a similarly flawed approach in its GOM DPEIS and its GOM ITR Application. *See, e.g.*, GOM DPEIS at 4-47 ("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." (emphasis added)). The Associations provided written comments in response to the GOM DPEIS and the GOM ITR Application. Excerpts of those comments are provided in Attachment F to this comment letter as they are equally relevant to the Proposed IHAs and, accordingly, should be considered by NMFS and included in the administrative record.

the Ninth Circuit held, *inter alia*, that the MMPA authorizing agency is not required "to quantify or estimate the number of mammals that would be taken." *Id*. at 906; *id*. at 906-07 (upholding "small numbers" finding based upon a reasonable qualitative analysis performed by the agency). The court further held that "small numbers" should be analyzed "in relation to the size of the larger population," and that the agency may consider the application of mitigation measures in reaching its finding. *Id*. at 908-09.[25]

As addressed above, the estimates of incidental take used by NMFS for its small numbers analysis substantially overestimate the number of takes. These inflated estimates skew the small numbers analysis by presuming that an unrealistically high percentage of a marine mammal stock's species will be incidentally taken.[26] Although we agree that minor, short-term behavioral modifications of 30% of a marine mammal stock can be reasonably characterized as a "small number," none of the surveys will actually take 30% (or more) of any of the potentially affected species or stocks. It would be helpful for NMFS to address the biased and overly conservative modeling of exposures in its small numbers analysis (and in its negligible impact analysis) and to provide a detailed qualitative explanation for why the Proposed IHAs will affect small numbers of marine mammal species or stocks.

Finally, but very importantly, it is unclear whether NMFS's take estimates include repeated exposures. If so, the estimates do not identify the number of repeated exposures and, instead, simply present a total number of estimated exposures by species (Figure 11). If this generalized presentation of exposures includes underlying repeats, then NMFS must perform additional analysis to eliminate the repeats in order to identify the *actual number* of individual marine mammals that will be incidentally taken. The MMPA's "small numbers" standard is based upon the number of individual marine mammals that are anticipated to be incidentally taken, regardless of how many times each of those marine mammals may be taken. *See* 16

[25] Here, NMFS substantially relies upon the proposed time-area closures to support its small numbers findings. However, all of the proposed mitigation measures (aside from those to which the Associations object) will minimize or avoid take. All mitigation measures included in the final IHAs should be considered by NMFS as part its the small numbers assessment, as NMFS has repeatedly done in other IHAs issued by the agency.

[26] We commend NMFS for its consistent use of the available information to both estimate the density of marine mammal stocks and calculate estimated abundance, for purposes of estimating the affected percentage of the stock. 82 Fed. Reg. at 26,271. It is essential that NMFS use the same information for both density and abundance estimation purposes to ensure that the percentages are fairly and accurately calculated. For example, it would not be fair or accurate to compare estimated takes based upon high density data against a low abundance value derived from a separate dataset.

U.S.C. § 1371(a)(5)(D).[27] Indeed, there are numerous examples of IHAs in which the estimated take percentages (in many cases, much higher than 30%) were found to reflect "small numbers" of marine mammals based on NMFS's explanation that the estimated number of takes included repeat takes, along with other reasons.[28] We encourage NMFS to review each of these examples, and others, as it develops its "small numbers" explanation. In addition, the total number of incidental takes (including repeats) may, at NMFS's discretion, be taken into account as part of the negligible impact analysis.[29]

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[27] The Associations re-emphasize their previous comments regarding Level A harassment and cumulative impacts. *See* Attachment A. Insofar as we are aware, in the history of incidental take authorizations for offshore seismic activities, there has never been a demonstration of Level A harassment from seismic surveys or of population-level effects to marine life from seismic surveys, individually or cumulatively. Moreover, although a cumulative impacts assessment is not required under the MMPA, NMFS may, after issuance of the IHAs, withdraw, modify, or suspend an IHA if it finds that "the authorized taking, either individually or in combination with other authorizations, is having, or may have more than a negligible impact on the species or stock…." 50 C.F.R. § 216.107(f)(2).

[28] *See, e.g.*, 82 Fed. Reg. 32,330, 32,343 (July 13, 2017) (**23.86%** of North Atlantic right whale stock); 82 Fed. Reg. 17,209, 17,223 (April 10, 2017) (**51.8%** of West Coast transient killer whale stock, **48.2%** of Northern resident killer whale stock); 81 Fed. Reg. 66,628, 66,637 (Sept. 28, 2016) (**64%** of Pacific bottlenose dolphin stock); 81 Fed. Reg. 40,852, 40,867 (June 23, 2016) (**27.1%** of West Coast transient killer whale stock and **25.3%** of Northern resident killer whale stock); 79 Fed. Reg. 57,512, 57,538 (Sept. 25, 2014) (**27.34%** of Sei whale stock, **24.9%** of pantropical spotted dolphin stock); 79 Fed. Reg. 65,378, 65,384 (Nov. 4, 2014) (**81%** of Pacific bottlenose dolphin stock).

[29] A recent paper published in *Nature Ecology & Evolution* (22 June 2017, Volume 1; Article Number 0195) purports to demonstrate, but fails to prove, that seismic survey air sources negatively impact zooplankton. The small sample size, variability in the baseline and experimental data, and the large number of speculative conclusions that appear to be inconsistent with the data collected over a two-day period undermine confidence in the reported values for the degree of impact. Both statistically and methodologically, this paper fails to demonstrate a rational basis for concluding that geophysical survey operations cause adverse effects to zooplankton populations. We raise this issue here because we expect that certain advocacy organizations will attempt to misleadingly claim that this paper is somehow relevant to the MMPA process for issuing the Proposed IHAs. To be clear, the paper has no relevance to this process because it creates no reasonable implication regarding the potential effects of seismic surveys on marine mammals. To the extent the paper is relevant to other regulatory processes involving the proposed Atlantic surveys, the Associations will address it with more detail in those processes.

III. CONCLUSION

The extensive record of information regarding the insignificant effects of OCS seismic surveying demonstrates that the Proposed IHAs will have no more than a temporary, localized, and negligible impact on marine mammals and marine mammal species or stocks, and may affect, at most, "small numbers" of individual marine mammals. The proposed seismic surveys are critical to the safe and orderly development of the oil and gas resources of the Atlantic OCS, and can be accomplished with insignificant environmental impacts. The Associations therefore strongly support NMFS's authorization of IHAs to address any incidental harassment of marine mammals that may result from the proposed Atlantic OCS surveys.[30]

We appreciate your consideration of all of the comments and information set forth in this letter (and attached to this letter), which are intended to be constructive and to facilitate the improvement of the scientific and legal integrity of the contemplated IHAs. Should you have any questions, please do not hesitate to contact Nikki Martin (713.957.5068) or Andy Radford (202.682.8584).[31]

Sincerely,

Nikki C. Martin

Nikki Martin
International Association of Geophysical Contractors
President

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[30] We have included a bibliography of references with this letter for inclusion in the administrative record.

[31] NMFS states that it "do[es] not anticipate offering additional discretionary public review of applications should [NMFS] receive further requests for authorization related to proposed geophysical survey activity in the Atlantic Ocean." 82 Fed. Reg. at 26,245. The Associations discourage NMFS from conducting discretionary public reviews of MMPA IHA applications. Such reviews are not required by the MMPA and only serve to delay the permitting process, contrary to U.S. policy. *See* Presidential Executive Order Implementing an America-First Offshore Energy Strategy (April 28, 2017), https://www.whitehouse.gov/the-press-office/2017/04/28/presidential-executive-order-implementing-america-first-offshore-energy. Moreover, full public review and comment is required at the proposal stage of the IHA process.

Andy Radford
American Petroleum Institute
Sr. Policy Advisor – Offshore

Jeff Vorberger
National Ocean Industries Association
Vice President Policy and Government Affairs

Attachments

BIBLIOGRAPHY

Au, W. and J. Oswald. 2010. Development of software for real-time acoustic identification of cetacean species. Final report. 31 p.

Baggenstoss, P.M. 2011. An algorithm for the localization of multiple interfering sperm whales using multi-sensor time difference of arrival. J. Acoust. Soc. Am. 130(1):102-112.

Baggenstoss, P.M. 2011. Separation of sperm whale click-trains for multipath rejection. J. Acoust. Soc. Am. 129(6):3598-3609.

Bakke, S. and J.E. Dyb. 2010. Testing of seismic equipment and monitoring of farmed fish – Storfjorden 2009-2010.Final Report to the E&P Sound & Marine Life Joint Industry Programme. 35 p. + appx.

Barkaszi, M.J., M. Butler, R. Compton, A. Unietis, and B. Bennet. 2012. Seismic survey mitigation measures and marine mammal observer reports. U.S. Dep. Interior, Bureau of Ocean Energy Management Gulf of Mexico OCS Region, New Orleans, LA. OCS Study BOEM 2012-015. 28 p. + appx.

Bartol, S.M. and I.K. Bartol. 2011. Hearing capabilities of loggerhead sea turtles (Caretta caretta) throughout ontogeny: an integrative approach involving behavioral and electrophysiological techniques. Final project report. 35 p.

Barton, C.J.S., R. Jaques, and M. Mason. 2008. Identification of potential utility of and collation of existing marine mammal observer data. Final report to the E&P Sound & Marine Life Joint Industry Programme. 109 p.

Beltran, R.S., S.H. Peterson, E.A. McHuron, C. Reichmuth, L.A. Huckstadt, and D.P. Costa. 2016. Seals and sea lions are what they eat, plus what? Determination of trophic discrimination factors for seven pinniped species. Rapid Commun. Mass Spectrom. 30(9):1115-1122.

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.R. 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.

BOEM (Bureau of Ocean Energy Management). 2016a. Request to the National Oceanic and Atmospheric Administration for incidental take regulations governing geophysical surveys on the Outer Continental Shelf of the Gulf of Mexico (a response to Subpart I — MMPA Request Requirements at 50 CFR § 216.104). Revision to original request package submitted December 20, 2002. Previous revisions provided in 2004, 2007, and 2011. 192 p.

BOEM (Bureau of Ocean Energy Management). 2016b. Gulf of Mexico OCS proposed geological and geophysical activities: Western, Central, and Eastern Planning Areas. Draft programmatic environmental impact statement. OCS EIS/EA BOEM 2016-049. 702 p. + app.

BOEM (Bureau of Ocean Energy Management). 2016c. Notice to lessees and operators (NTL) of federal oil, gas, and sulfur leases in the OCS, Gulf of Mexico OCS region:

implementation of seismic survey mitigation measures and protected species observer program. 10 p. Available online at https://www.boem.gov/BOEM-NTL-2016-G02/.

Bougher, B.B., J. Hood, J. Theriault, and H. Moors. 2012. Generalized marine mammal detection based on improved band-limiting processing. Presented at ECUA 2012 11th European Conference on Underwater Acoustics, Edinburgh, Scotland, 2-6 July 2012.

Boyd, I.L. 2012. Technology requirements to investigate the effects of sound on marine wildlife. Underwater Technol. 30(3):123-133.

Branstetter, B.K., J. St Leger, D. Acton, J. Stewart, D. Houser, J.J. Finneran, and K. Jenkins. 2017. Killer Whale (*Orcinus orca*) behavioral audiograms. J. Acoust. Soc. Am. 141(4): 2387-2398.

Breed, G.A., D.P. Costa, M.E. Goebel, and P.W. Robinson. 2011. Electronic tracking tag programming is critical to data collection for behavioral time-series analysis. Ecosphere 2(1):art10.

Breed, G.A., D.P. Costa, I.D. Jonsen, P.W. Robinson, and J. Mills-Flemming. 2012. State-space methods for more completely capturing behavioral dynamics from animal tracks. Ecol. Model. 235-236:49-58.

Bröker, K., G. Gailey, J. Muir, and R. Racca. 2015. Monitoring and impact mitigation during a 4D seismic survey near a population of gray whales off Sakhalin Island, Russia. Endang. Spec. Res. 28(3):187-208.

Cabrelli, A., J. Harwood, J. Matthiopoulos, L. New, and L. Thomas. 2009. A critical review of the literature on population modelling. CREEM Tech. Rep. 2009-2. 104 p.

Cabrelli, A., J. Harwood, J. Matthiopoulos, L. New, and L. Thomas. 2009. A critical review of the literature on population modelling. Final report to the Joint Industry Project of the International Association of Oil & Gas Producers on contract JIP22 07_20. 105 p.

Cato, D.H., R.A. Dunlop, M.J. Noad, R.D. McCauley, E. Kniest, D. Paton, and A.S. Kavanagh. 2016. Addressing challenges in studies of behavioral responses of whales to noise. p. 145-152 In: A.N. Popper and A. Hawkins (eds.). The effects of noise on aquatic life II. Advances in Experimental Medicine and Biology, vol. 875. Springer Science+Business Media, New York.

Cato, D.H., M. Noad, R. Dunlop, R.D. McCauley, H. Kniest, D. Paton, C.P. Salgado Kent, and C.S. Jenner. 2013a. Behavioral responses of humpback whales to seismic air guns. Presented at ICA 2013, Montreal, Canada, 2-7 June 2013.

Cato, D.H., M.J. Noad, R.A. Dunlop, R.D. McCauley, N.J. Gales, C.P. Salgado Kent, H. Kniest, D. Paton, K.C.S. Jenner, J. Noad, A.L. Maggi, I.M. Parnum, and A.J. Duncan. 2012. Project BRAHSS: Behavioural Response of Australian Humpback whales to Seismic Surveys. Presented at Proceedings of Acoustics, Fremantle, Australia, 21-23 November 2012.

Cato, D.H., M.J. Noad, R.A. Dunlop, R.D. McCauley, N.J. Gales, C.P. Salgado Kent, H. Kniest, D. Paton, K.C.S. Jenner, J. Noad, A.L. Maggi, I.M. Parnum, and A.J. Duncan. 2013b. A study of the behavioral response of whales to the noise of seismic air guns: design, methods and progress. Acoust. Australia 41(1):88-97.

CEF Consultants Ltd. 2011. Report on a workshop on fish behaviour in response to seismic sound held in Halifax, Nova Scotia, Canada, March 28-31, 2011. Environ. Stud. Res. Funds Rep. No. 190. 109 p.
Clyne, H. 1999. Computer simulations of interactions between the North Atlantic Right Whale (Eubalaena glacialis) and shipping. MSc Thesis, Napier University. Edinburgh, Scotland. 53 p.
Costa, D.P. 2012. A bioenergetics approach to developing a population consequences of acoustic disturbance model. p. 423-426 In: A.N. Popper and A. Hawkins (eds.). The effects of noise on aquatic life. Advances in Experimental Medicine and Biology, vol. 730. Springer Science+Business Media, New York.
Costa, D.P. and B. Block. 2011. Use of electronic tag data and associated analytical tools to identify and predict habitat utilization of marine mammals. 13 p.
Costa, D.P., G.A. Breed, and P.W. Robinson. 2012. New Insights into pelagic migrations: implications for ecology and conservation. Annu. Rev. Ecol. Evol. Syst. 43(1):73-96.
Costa, D.P., L.A. Huckstadt, D.E. Crocker, B.I. McDonald, M.E. Goebel, and M.A. Fedak. 2010. Approaches to studying climatic change and its role on the habitat selection of antarctic pinnipeds. Integr. Comp. Biol. 50(6):1018-1030.
Costa, D.P., L. Schwarz, P. Robinson, R.S. Schick, P.A. Morris, R. Condit, D.E. Crocker, and A.M. Kilpatrick. 2016. A bioenergetics approach to understanding the population consequences of disturbance: elephant seals as a model system. p. 161-169 *In:* A. Popper and A. Hawkins (eds.). The effects of noise on aquatic life II. Advances in Experimental Medicine and Biology, vol. 875. Springer, New York.
Davidson, A.D., A.G. Boyer, H. Kim, S. Pompa-Mansilla, M.J. Hamilton, D.P. Costa, G. Ceballos, and J.H. Brown. 2012. Drivers and hotspots of extinction risk in marine mammals. Proc. Natl. Acad. Sci. USA 109(9):3395-3400.
Deng, X.Y., D. Gillespie, J. Gordon, R. McHugh, D. McLaren, D. Mellinger, P. Redmond, A. Thode, and P. Trinder. 2008. PAMGuard 4 final report. 122 p.
DOSITS (Discovery of Sound in the Sea). 2016. Science of sound (video).
DOSITS (Discovery of Sound in the Sea). 2016. Marine mammal hearing (video).
DOSITS (Discovery of Sound in the Sea). 2016. Hearing in marine fishes (video).
DOSITS (Discovery of Sound in the Sea). 2016. Determining mitigation and monitoring through the ecological risk assessment process (video).
Dunlop, R.A. 2016. Changes in vocal parameters with social context in humpback whales: considering the effect of bystanders. Behav. Ecol. Sociobiol. 70:857-870.
Dunlop, R.A., M.J. Noad, and D.H. Cato. 2016a. A spatially explicit model of the movement of humpback whales relative to a source. Presented at Fourth International Conference on the Effects of Noise on Aquatic Life, Dublin, Ireland, 10-16 July 2016.
Dunlop, R.A., M.J. Noad, and D.H. Cato. 2012. Behavioral-response studies: problems with statistical power. p. 293-297 *In:* A.N. Popper and A. Hawkins (eds.). The effects of noise on aquatic life. Advances in Experimental Medicine and Biology, vol. 730. Springer Science+Business Media, New York.

Dunlop, R.A., M.J. Noad, R.D. McCauley, E. Kniest, D. Paton, and D.H. Cato. 2015. The behavioural response of humpback whales (*Megaptera novaeangliae*) to a 20 cubic inch air gun. Aquat. Mamm. 41(4):412-433.

Dunlop, R.A., M.J. Noad, D.H. Cato, E. Kniest, P.J.O. Miller, J.N. Smith, and M.D. Stokes. 2013. Multivariate analysis of behavioural response experiments in humpback whales (*Megaptera novaeangliae*). J. Exp. Biol. 216:759-770.

Dunlop, R.A., M.J. Noad, R.D. McCauley, E. Kniest, R. Slade, D. Paton, and D.H. Cato. 2016b. Response of humpback whales (*Megaptera novaeangliae*) to ramp-up of a small experimental air gun array. Mar. Pollut. Bull. 103:72-83.

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.

Erbe, C., C. Reichmuth, K. Cunningham, K. Lucke, and R. Dooling. 2015. Acoustic masking in marine mammals: a review and research strategy. Report to the International Association of Oil & Gas Producers' Joint Industry Program. 51 p.

Erbe, C., C. Reichmuth, K. Cunningham, K. Lucke, and R. Dooling. 2016. Communication masking in marine mammals: a review and research strategy. Mar. Pollut. Bull. 103:15-38.

Finneran, J.J. 2012. Auditory effects of underwater noise in odontocetes. p. 197-202 In: A.N. Popper and A. Hawkins (eds.). The effects of noise on aquatic life. Advances in Experimental Medicine and Biology, vol. 730. Springer Science+Business Media, New York.

Finneran, J.J. and B.K. Branstetter. 2013. Effects of noise on sound perception in marine mammals. p. 273-308 In: H. Brumm (ed.). Animal communication and noise. Springer-Verlag, Berlin, Germany.

Finneran, J.J. and A.K. Jenkins. 2012. Criteria and thresholds for U.S. Navy acoustic and explosive effects analysis. U.S. Navy, Space and Naval Warfare Systems Center Pacific, San Diego, CA. 65 p.

Finneran, J.J., C.E. Schlundt, B.K. Branstetter, J.S. Trickey, V. Bowman, and K. Jenkins. 2015. Effects of multiple impulses from a seismic air gun on bottlenose dolphin hearing and behavior. J. Acoust. Soc. Am. 137(4):1634-1646.

Finneran, J.J., C.E. Schlundt, B.K. Branstetter, J.S. Trickey, V. Bowman, and K. Jenkins. 2013. Temporary threshold shift (TTS) in odontocetes in response to multiple air gun impulses. Final report. 51 p.

Fowler, M.A., D.P. Costa, D.E. Crocker, W.-J. Shen, and F.B. Kraemer. 2015. Adipose triglyceride lipase, not hormone-sensitive lipase, is the primary lipolytic enzyme in fasting elephant seals (Mirounga angustirostris). Physiol. Biochem. Zool. 88(3):284-294.

Fowler, M.A., C. Debier, C.D. Champagne, D.E. Crocker, and D.P. Costa. 2016. The demands of lactation promote differential regulation of lipid stores in fasting elephant seals. Gen. Comp. Endocrinol. 225:125-132.

Frankel, A., Ellison, W., White, A. and Vigness-Raposa, K. 2016. JIP Aversion Modeling. Final Report. Marine Acoustics, Inc. MAI 947. 46 p.

Frankel, A.S., K. Vigness-Raposa, J. Giard, A. White, and W.T. Ellison. 2016. Exposures v. Individuals: Effects of varying movement patterns and animal behavior on long-term animat model exposure predictions. Proc. Mtgs. Acoust. 27, 010038

Frouin-Mouy, H., K. Kowarski, B. Martin, and K. Bröker. 2017. Seasonal trends in acoustic detection of marine mammals in Baffin Bay and Melville Bay, Northwest Greenland. Arctic 70(1):59-76.

Geokinetics Inc. 2017. Geokinetics acquires 2D seismic data utilizing its AquaVib™ marine vibrator. Multi-client dataset acquired in the Gulf of Mexico. Press release, April 3, 2017. Available online at http://www.geokinetics.com/wp-content/uploads/2017/04/GOK.AquaVib.GOM_.press-release.final_.pdf, accessed April 19, 2017.

Gillespie, D. 2008. PAMGuardIV: final report from SMRU Ltd. Edinburgh, Scotland. 17 p.

Gillespie, D. 2009a. PAMGuard CODA field trial final report. Prepared by SMRU Ltd. Edinburgh, Scotland. 25 p.

Gillespie, D. 2009b. PAMGuard industry field trial 2008. Final report. Prepared by SMRU. Ediburgh, Scotland. 53 p.

Gillespie, D., P. White, M. Caillat, and J. Gordon. 2011. Development and implementation of automatic classification of odontocetes within PAMGuard. Prepared by SMRU Limited. Edinburgh, Scotland. 68 p.

Gillespie, D., C. Blight, M. Caillat, P. White, J. Gordon, P. Trinder, and R. McHugh. 2009. PAMGuard: open source software for real-time acoustic detection and localisation of cetaceans (poster).

Gillespie, D., J. Gordon, R. McHugh, D. McLaren, D.K. Mellinger, P. Redmond, A. Thode, P. Trinder, and X.Y. Deng. 2008. PAMGuard: semiautomated, open source software for real-time acoustic detection and localisation of cetaceans. Proc. Inst. Acoust. 30(Pt. 5).

Godwin, E.M., M.J. Noad, E. Kniest, and R.A. Dunlop. 2016. Comparing multiple sampling platforms for measuring the behavior of humpback whales (Megaptera novaeangliae). Mar. Mammal Sci. 32(1):268-286.

Gordon, J. 2009. A report on a PAMGuard conference and workshops helt at Heriot Watt University, Edinburgh 28th-29th of March 2007. 36 p.

Graham, I. M., E. Pirotta, N.D. Merchant, A. Farcas, T.R. Barton, B. Cheney, G.D. Hastie, and P.M. Thompson. 2017. Responses of bottlenose dolphins and harbor porpoises to impact and vibration piling noise during harbor construction. Ecosphere 8(5):e01793.

Hassrick, J.L., D.E. Crocker, N.M. Teutschel, B.I. McDonald, P.W. Robinson, S.E. Simmons, and D.P. Costa. 2010. Condition and mass impact oxygen stores and dive duration in adult female northern elephant seals. J. Exp. Biol. 213(4):585-592.

Hastings, M.C. 2009. A model for prediction of auditory tissue damage in fish. Final report. Submitted to the Joint Industry Program (JIP) on E&P Sound and Marine Life. 38 p.

Hazen, E.L., S. Jorgensen, R.R. Rykaczewski, S.J. Bograd, D.G. Foley, I.D. Jonsen, S.A. Shaffer, J.P. Dunne, D.P. Costa, L.B. Crowder, and B.A. Block. 2012. Predicted habitat shifts of Pacific top predators in a changing climate. Nature Climate Change 3(3):234-238.

Hood, J. 2009. Mitigation & monitoring: passive acoustic monitoring (PAM) software development – detection, classification and localisation capabilities. Final report: International Association of Oil & Gas Producers (OGP) contract 07-18. Akoostix Inc., Dartmouth, NS. 46 p.

Hood, J.D., D.G. Flogeras, and J.A. Theriault. 2016. Improved passive acoustic band-limited energy detection for cetaceans. Appl. Acoust. 106:36-41.

Hooker, S.K., A. Fahlman, M.J. Moore, N. Aguilar de Soto, Y. Bernaldode Quiros, 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. Proc. R. Soc. B 279(1731):1041-1050.

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. J. Exp. Biol. 213:52-62.

Houser, D.S., W. Yost, R. Burkard, J.J. Finneran, C. Reichmuth, and J. Mulsow. 2017. A review of the history, development and application of auditory weighting functions in humans and marine mammals. J. Acoust. Soc. Am. 141(3):1371-1413.

JIP (Joint Industry Programme). 2015. Understanding the potential impact of repeated exposure to seismic impulses on dolphin hearing (factsheet).

JIP (Joint Industry Programme). 2015. PAMGuard: finding and tracking marine mammals using their sounds (factsheet).

JIP (Joint Industry Programme). 2016. From science to operations: how research is used to manage acoustic E&P industry risks to marine mammal populations (poster).

JIP (Joint Industry Programme). no date. The International Oil and Gas Producers E&P Sound and Marine Life Joint Industry Programme (factsheet).

Jochens, A., D. Biggs, K. Benoit-Bird, D. Engelhardt, J. Gordon, C. Hu, N. Jaquet, M. Johnson, R. Leben, B. Mate, P. Miller, J. Ortega-Ortiz, A. Thode, P. Tyack, and B. Wursig. 2008. Sperm whale seismic study in the Gulf of Mexico: synthesis report. New Orleans, LA. OCS Study MMS 2008-006. U.S. Dept. of the Interior,. 341 p.

Johnson, S.R., W.J. Richardson, S.B. Yazvenko, S.A. Blokhin, G. Gailey, M.R. Jenkerson, S.K. Meier, H.R. Melton, M.W. Newcomer, A.S. Perlov, S.A. Rutenko, B. Wursig, C.R. Martin, and D.E. Egging. 2007. A western gray whale mitigation and monitoring program for a 3-D seismic survey, Sakhalin Island, Russia. Environ. Monitor. Assess. 134(1-3):1-19.

Kaschner, K., N.J. Quick, R. Jewell, R. Williams, and C.M. Harris. 2012. Global coverage of cetacean line-transect surveys: status quo, data gaps and future challenges. PLoS One 7(9):e44075.

Kavanagh, A.S., A.W. Goldizen, S.P. Blomberg, M.J. Noad, and R.A. Dunlop. 2016. Factors affecting the reliability and validity of behavioural datasets: assessing the impact of

observers' experience and native language on studies of wild animals. Aquat. Mamm. 42(1):1-11.
Kavanagh, A.S., M.J. Noad, S.P. Blomberg, A.W. Goldizen, E. Kniest, D.H. Cato, and R.A. Dunlop. 2017. Factors driving the variability in diving and movement behavior of migrating humpback whales (*Megaptera novaeangliae*): implications for anthropogenic disturbance studies. Mar. Mammal Sci. 33(2):413-439.
Kavanagh, A.S., K. Owen, M.J. Williamson, S.P. Blomberg, M.J. Noad, A.W. Goldizen, E. Kniest, D.H. Cato, and R.A. Dunlop. 2017. Evidence for the functions of surface-active behaviors in humpback whales (*Megaptera novaeangliae*). Mar. Mammal Sci. 33(1):313-334.
Keith, E.O. 2008. Critical review of the literature on marine mammal population modeling. JIP 22-07-19. 63 p.
Ketten, D.R. and D.C. Mountain. 2011. Final report: modeling minke whale hearing. Submitted to Joint Industry Program. 30 p.
Knowlton, A.R., F.T. Korsmeyer, J.E. Kerwin, H.Y. Wu, and B. Hynes. 1995. The hydrodynamic effects of large vessels on right whales. NOAA Contract No. 40EANFF400534.
Knudsen, F.R., O.B. Gammelsæter, P.H. Kvadsheim, and L. Nøttestad. 2007. Evaluation of fisheries sonars for whale detection in relation to seismic survey operations. Final report to the E&P Sound & Marine Life Joint Industry Programme. 36 p.
Koski, W.R., P. Abgrall, and S.B. Yazvenko. 2009. A review and inventory of unmanned aerial systems for detection and monitoring of key biological resources and physical parameters affecting marine life during offshore exploration and production activities. IWC Scientific Committee Meeting, Madeira, Portugal, 31 May - 12 June. Working paper SC/61/E9. International Whaling Commission, Cambridge. U.K. 12 p.
Laist, D. W., Knowlton, A. R., Mead, J. G., Collet, A. S. and Podesta, M. 2001. Collisions between ships and whales. Marine Mammal Science, 17: 35–75.
Lavender, A.L., S.M. Bartol, and I.K. Bartol. 2014. Ontogenetic investigation of underwater hearing capabilities in loggerhead sea turtles (Caretta caretta) using a dual testing approach. J. Exp. Biol. 217:2580-2589.
Lawson, G.L., L.A. Hückstädt, A.C. Lavery, F.M. Jaffré, P.H. Wiebe, J.R. Fincke, D.E. Crocker, and D.P. Costa. 2015. Development of an animal-borne "sonar tag" for quantifying prey availability: test deployments on northern elephant seals. Anim. Biotelem. 3:22.
LGL Limited environmental research associates. 2009. Cetacean stock assessment in relation to exploration and production industry sound. Prepared by LGL Limited, environmental research associates for Joint Industry Programme. LGL Report TA4582-1. 319 p.
LGL Limited environmental research associates. 2009. A review and inventory of unmanned aerial systems for detection and monitoring of key biological resources and physical parameters affecting marine life during offshore exploration and production activities. Prepared for Joint Industry Programme on E&P Sound and Marine Life by LGL Limited environmental research associates. LGL Report TA4731-1. 15 p.

LGL Limited environmental research associates and Marine Acoustics Inc. 2011. Environmental assessment of marine vibroseis. Prepared by LGL, Ltd., environmental research associates and Marine Acoustics Inc. for Joint Industry Programme, E&P Sound and Marine Life, International Association of Oil & Gas Producers, London, U.K. JIP contract 22 07-12. LGL Report TA4604-1. xix + 207 p.

Lucke, K., G.D. Hastie, K. Ternes, B. McConnell, S. Moss., D.J.F. Russell, H. Weber, and V.M. Janik. 2016. Aerial low-frequency hearing in captive and free-ranging harbour seals (*Phoca vitulina*) measured using auditory brainstem responses. J. Comp. Physiol. 202:859-868.

Lucke, K., A.N. Popper, A.D. Hawkins, T. Akamatsu, M. André, B.K. Branstetter, M. Lammers, C.A. Radford, A.L. Stansbury, and T.A. Mooney. 2016. Auditory sensitivity in aquatic animals. J. Acoust. Soc. Am. 139(6): 3097-3101.

Lucke, K., J.J. Finneran, J. Almunia, and D.S. Houser. 2016. Variability in Click-Evoked potentials in killer whales (*Orcinus orca*) and determination of a hearing impairment in a rehabilitated killer whale. Aquatic Mammals 42(2): 184-192.

Maresh, J.L., S.E. Simmons, D.E. Crocker, B.I. McDonald, T.M. Williams, and D.P. Costa. 2014. Free-swimming northern elephant seals have low field metabolic rates that are sensitive to an increased cost of transport. J. Exp. Biol. 217:1485-1495.

Maresh, J.L., T. Adachi, A. Takahashi, Y. Naito, D.E. Crocker, M. Horning, T.M. Williams, and D.P. Costa. 2015. Summing the strokes: energy economy in northern elephant seals during large-scale foraging migrations. Mov. Ecol. 3:22.

Marques, T.A., L. Thomas, J. Ward, N. DiMarzio, and P.L. Tyack. 2009. Estimating cetacean population density using fixed passive acoustic sensors: an example with Blainville's beaked whales. J. Acoust. Soc. Am. 125(4):1982-1994.

Marques, T.A., L. Munger, L. Thomas, S. Wiggins, and J.A. Hildebrand. 2011. Estimating North Pacific right whale *Eubalaena japonica* density using passive acoustic cue counting. Endang. Spec. Res. 13(3):163-172.

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. Biol. Rev. 88(2):287-309.

Martin, S.W., T.A. Marques, L. Thomas, R.P. Morrissey, S. Jarvis, N. DiMarzio, D. Moretti, and D.K. Mellinger. 2013. Estimating minke whale (*Balaenoptera acutorostrata*) boing sound density using passive acoustic sensors. Mar. Mammal Sci. 29(1):142-158.

Mate, B.R. 2009. GPS/TDR satellite tracking of sperm whales with 3-axis accelerometers - background. Final report from Oregon State University for JIP sponsored research on GPS/TDR tag testing for 2007. 7 p.

Mate, B. 2012. GPS/TDR satellite tracking of sperm whales with 3-axis accelerometers. Prepared by Oregon State University.

Mate, B.R., L.M. Irvine, and D.M. Palacios. 2017. The development of an intermediate-duration tag to characterize the diving behavior of large whales. Ecol. Evol. 7(2):585-595.

Mathias, D., A.M. Thode, J. Straley, J. Calambokidis, G.S. Schorr, and K. Folkert. 2012. Acoustic and diving behavior of sperm whales (*Physeter macrocephalus*) during natural and depredation foraging in the Gulf of Alaska. J. Acoust. Soc. Am. 132(1):518-532.

Mattsson, A. 2010. Final report. OGP, E&P Sound and Marine Life, JIP Svein Vaage broadband airgun study. 27 p.

McCauley, R.D., J. Fewtrell, A.J. Duncan, C. Jenner, M-N. Jenner, J.D. Penrose, R.I.T. Prince, A. Adhitya, J. Murdoch, K. MacCabe. 2000. Marine Seismic Surveys: Analysis of propagation of airgun signals; and effects of air-gun exposure on humpback whales, sea turtles, fishes and squid. Centre for Marine Science and Technology, Curtin University of Technology. Prepared for the Australian Petroleum Production Exploration Association. 198 p.

McCauley, R.D., A.J. Duncan, A.N. Gavrilov, and D.H. Cato. 2016. Transmission of marine seismic survey, air gun array signals in Australian waters. Presented at Acoustics 2016, Brisbane, Australia, 9-11 November 2016.

McHuron, E.A., S.H. Peterson, J.T. Ackerman, S.R. Melin, J.D. Harris, and D.P. Costa. 2016. Effects of age, colony, and sex on mercury concentrations in California sea lions. Arch. Environ. Contam. Toxicol. 70(1):46-55.

Miller, I. and E. Cripps. 2013. Three dimensional marine seismic survey has no measurable effect on species richness or abundance of a coral reef associated fish community. Mar. Pollut. Bull. 77, 63-70.

Moretti, D., T. Casey, and D.K. Mellinger. 2009. Measuring the health of the field: fixed passive acoustic marine mammal monitoring for estimating species abundance and mitigating the effect of operations on the marine environment. 56 p.

Muir, J.E., M. Gilders, and Y. Bychkov. 2010. A conceptual framework for tiered risk assessment to evaluate the effects of sound from E&P operations on marine mammals. Final report. Prepared by LGL Limited environmental research associates for Joint Industry Program. 88 p. + Appx.

Muir, J.E., L. Ainsworth, R. Racca, Y. Bychkov, G. Gailey, V. Vladimirov, S. Starodymov, and K. Bröker. 2016. Gray whale densities during a seismic survey off Sakhalin Island, Russia. Endang. Spec. Res. 29(3):211-227.

National Academies of Sciences, Engineering, and Medicine. 2016. Approaches to understanding the cumulative effects of stressors on marine mammals. Washington, D.C.: The National Academic Press. Doi: 10.17226/23479

National Research Council. 2005. Marine mammal populations and ocean noise. Determining when noise causes biologically significant effects. The National Academies Press, Washington, D.C. 126 p.

New, L.F., J.S. Clark, D.P. Costa, E. Fleishman, M.A. Hindell, T. Klanjscek, 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. Mar. Ecol. Prog. Ser. 496:99-108.

NOAA Fisheries. 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. U.S. Dep. Commer., Silver Spring, MD. NOAA Tech. Memo. NMFS-OPR-55. 178 p.

Noad, M.J. and R.A. Dunlop. 2016. Marine mammals and the impacts of anthropogenic noise: considerations for the design of large acoustic behavioural response studies such as BRAHSS. Presented at Acoustics 2016, Brisbane, Australia, 9-11 November 2016.

Norris, T.F., J.N. Oswald, and R.S. Sousa-Lima. 2010. A review and inventory of fixed installation passive acoustic monitoring methods and technologies. Final report to the International Association of Oil & Gas Producers Joint Industry Programme on E&P Sound and Marine Life. London, UK. Contract reference # JIP22 08-03. 218 p.

Nowacek, D.P., K. Broker, G. Donovan, G. Gailey, R. Racca, R.R. Reeves, A.I. Vedenev, D.W. Weller, and B.L. Southall. 2013. Responsible practices for minimizing and monitoring environmental impacts of marine seismic surveys with an emphasis on marine mammals. Aquat. Mamm. 39(4):356-377.

Nowacek, D.P., D.W. Johnston, L. Bejder, and D. McElligott. 2010. A review of literature relevant to the PCAD model for guidance in developing transfer functions. 23 p.

ONR (Office of Naval Research). 2017. ONR Marine Mammal & Biology Program Review, 20-24 March 2017. Abstract book. 121 p. Available online at https://www.onr.navy.mil/-/media/Files/32/MB_ProgramReview_AbstractBook2017.pdf.

PAMGuard. 2009. PAMGuard: an overview and gentle introduction. 42 p.

Peck, H.E., D.P. Costa, and D.E. Crocker. 2015. Body reserves influence allocation to immune responses in capital breeding female northern elephant seals. Func. Ecol. 30(3):389-397.

Ponganis, P. 2007. Blood nitrogen uptake & distribution during diving in bottlenose dolphins. Summary report to 'Sound & Marine Life' JIP, August 2007. 3 p.

Pramik, B., M.L. Bell, A. Grier, and A. Lindsay. 2015. Field testing the AquaVib: an alternative marine seismic source. SEG Tech. Prog. Expanded Abstracts 2015:181-185.

Quick, N. 2009. Cetacean stock assessment. Final report. SMRU Limited, Edinburgh. Ref. No. RA1007OGP. 306 p.

Reichmuth, C. 2007. Assessing the hearing capabilities of mysticete whales. A proposed research strategy for the Joint Industry Programme on Sound and Marine Life. 36 p.

Reichmuth, C. 2012. Psychophysical studies of auditory masking in marine mammals: key concepts and new directions. p. 23-27 In: A.N. Popper and A. Hawkins (eds.). The effects of noise on aquatic life. Advances in Experimental Medicine and Biology, vol. 730. Springer Science+Business Media, New York.

Reichmuth, C., A. Ghoul, and J.M. Sills. 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.

Richardson, W.J., C.R. Greene, Jr., C.I. Malme, and D.H. Thomson. 1995. Marine mammals and noise. Academic Press, San Diego, CA. 576 p.

Robinson, P.W., S.E. Simmons, D.E. Crocker, and D.P. Costa. 2010. Measurements of foraging success in a highly pelagic marine predator, the northern elephant seal. J. Anim. Ecol. 79:1146-1156.

Rouse, A.A., P.F. Cook, E.W. Large, and C. Reichmuth. 2016. Beat keeping in a sea lion as coupled oscillation: implications for comparative understanding of human rhythm. Front. Neurosci. 10:257.

Ruser, A., M. Dähne, A. van Neer, K. Lucke, J. Sundermeyer, U. Siebert, D.S. Houser, J.J. Finneran, E. Everaarts, J. Meerbeek, R. Dietz, S. Sveegaard, and J. Teilmann. 2016. Assessing auditory evoked potentials of wild harbor porpoises (*Phocoena phocoena*). J. Acoust. Soc. Am. 140(1): 442-452.

Schlumberger. 2017. eSource Bandwidth Controlled Seismic Source Technology. Available online at: http://www.slb.com/services/seismic/marine/technologies/esource-seismic-source.aspx, accessed April 19, 2017.

Schlundt, C.E., J.J. Finneran, B.K. Branstetter, J.S. Trickey, V. Bowman, and K. Jenkins. 2016. Auditory effects of multiple impulses from a seismic air gun on bottlenose dolphins (*Tursiops truncatus*). p. 987-991 *In:* A.N. Popper and A. Hawkins (eds.). The effects of noise on aquatic life II. Advances in Experimental Medicine and Biology, vol. 875. Springer Science+Business Media, New York.

Schwarz, L.K., M.A. Hindell, C.R. McMahon, and D.P. Costa. 2012. The implications of assuming independent tag loss in southern elephant seals. Ecosphere 3(9):art81.

Schwarz, L.K., M.E. Goebel, D.P. Costa, and A.M. Kilpatrick. 2013. Top-down and bottom-up influences on demographic rates of Antarctic fur seals *Arctocephalus gazella*. J. Anim. Ecol. 82(4):903-911.

Schwarz, L.K., S. Villegas-Amtmann, R.S. Beltran, D.P. Costa, C. Goetsch, L. Huckstadt, J.L. Maresh, and S.H. Peterson. 2015. Comparisons and uncertainty in fat and adipose tissue estimation techniques: the northern elephant seal as a case study. PLoS One 10(6):e0131877.

Sills, J.M. and C. Reichmuth. 2016. Listening for signals in seismic noise: a case study of masking in Arctic seals. Presented at Fourth International Conference on the Effects of Noise on Aquatic Life, Dublin, Ireland, 10-16 July 2016.

Sills, J.M., B.L. Southall, and C. Reichmuth. 2014. Amphibious hearing in spotted seals (Phoca largha): underwater audiograms, aerial audiograms and critical ratio measurements. J. Exp. Biol. 217:726-734.

Sills, J.M., B.L. Southall, and C. Reichmuth. 2015. Amphibious hearing in ringed seals (*Pusa hispida*): underwater audiograms, aerial audiograms and critical ratio measurements. J. Exp. Biol. 218:2250-2259.

Sills, J., B. Southall, and C. Reichmuth. 2015. Amphibious hearing in ringed seals (*Pusa hispida*): underwater audiograms and critial ratio measurements.

Sills, J.M., B.L. Southall, and C. Reichmuth. 2016. Psychoacoustic studies of spotted (*Phoca largha*) and ringed (*Pusa hispida*) seals. p. 1025-1030 *In:* A. Popper and A. Hawkins (eds.). The effects of noise on aquatic life II. Advances in Experimental Medicine and Biology., vol. 875. Springer, New York.

Sills, J.M., B.L. Southall, and C. Reichmuth. 2017. The influence of temporally varying noise from seismic air guns on the detection of underwater sounds by seals. J. Acoust. Soc. Am. 141(2):996-1008.

Sills, J.M., B.L. Southall, and C. Reichmuth. 2017. The influence of temporally varying noise from seismic air guns on the detection of underwater sounds by seals. J. Acoust. Soc. Am. 141(2):996-1008.

SMRU. 2007. Workshop on technology requirements to investigate the effects of sound on marine wildlife. Sponsored by the International Association of Oil and Gas Producers Joint Industry Programme on Sound and Marine Life. St. Andrews, 20-22 March 2007. 53 p.

Sousa-Lima, R.S., T.F. Norris, J.N. Oswald, and D.P. Fernandes. 2013. A review and inventory of fixed autonomous recorders for passive acoustic monitoring of marine mammals. Aquat. Mamm. 39(1):23-53.

Southall, B., D. Nowacek, P. Miller, and P. Tyack. 2016. Experimental field studies to measure behavioral responses of cetaceans to sonar. Endangered Species Res. 31:293-315. doi: 10.3354/esr00764

Southall, B.L., D. Moretti, B. Abraham, J. Calambokidis, S.L. DeRuiter, and P.L. Tyack. 2012. Marine mammal behavioral response studies in southern California: advances in technology and experimental methods. Mar. Technol. Soc. J. 46(4):48-59.

Southall, B.L., A.E. Bowles, W.T. Ellison, J.J. Finneran, R.L. Gentry, C.R.J. Greene, 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. Aquat. Mamm. 33(4):411-522.

Spence, J., R. Fischer, M. Bahtiarian, L. Boroditsky, N. Jones, and R. Dempsey. 2007. Review of existing and future potential treatments for reducing underwater sound from oil and gas industry activities. Prepared by Noise Control Engineering, Inc. for Joint Industry Programme on E&P Sound and Marine Life. London, U.K. 185 p.

Stone, C.J. 2015. Marine mammal observations during seismic surveys from 1994-2010. Joint Nature Conservation Committee, Peterborough, UK. JNCC Report No. 463a. 64 p.

Stone, C.J. and M.L. Tasker. 2006. The effects of seismic airguns on cetaceans in UK waters. J. Cetacean Res. Manage. 8(3):255-263.

Szesciorka, A.R., J. Calambokidis, and J.T. Harvey. 2016. Testing tag attachments to increase the attachment duration of archival tags on baleen whales. Anim. Biotelem. 4(1):18.

Theriault, J., E. Macneil, B. Maranda, L. Gilroy, and J. Hood. 2010. Survey of active acoustic monitoring (AAM) technologies. Vol. 1: survey results. Prepared for International Association of Oil & Gas Producers (OGP). London, U.K. 37 p.

Thode, A., J. Skinner, P. Scott, J. Roswell, J. Straley, and K. Folkert. 2010. Tracking sperm whales with a towed acoustic vector sensor. J. Acoust. Soc. Am. 128(5):2681-2694.

Thode, A., D. Mathias, J. Straley, J. Calambokidis, G. Schorr, and K. Folkert. 2011. Final report, OGP/JIP 22 06/01, Project 3.7.1: testing of potential alerting sound playbacks to sperm whales. OGP/JIP 22 06/01. 61 p.

Thomas, L. and T.A. Marques. 2012. Passive acoustic monitoring for estimating animal density. Acoust. Today 8(3):35-44.

Thomsen, F., S.R. McCully, L.R. Weiss, D.T. Wood, K.J. Warr, J. Barry, and R.J. Law. 2011. Cetacean stock assessments in relation to exploration and production industry activity and other human pressures: review and data needs. Aquat. Mamm. 37(1):1-93.

Thomsen, F., S.R. McCully, L. Weiss, D. Wood, K. Warr, M. Kirby, L. Kell, and R. Law. 2008. Cetacean stock assessment in relation to exploration and porduction industry sound: current knowledge and data needs. 07-11 schedule 01. Submitted to E&P Sound and Marine Life Programme, International Association of Oil and Gas Producers. 134 p.

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. J. Acoust. Soc. Am. 132(5):3263-3272.

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.

University of California Santa Cruz. 2012. Relating behaviour and life functions to populations level effects in marine mammals: an empirical and modeling effort to develop the PCAD model. Final report contract JIP 22 07-23. 21 p.

US Navy. 2016. LMR. U.S. Navy's Living Marine Resources Program report 2016. Naval Facilities Engineering and Expeditionary Warfare Center, Port Hueneme, CA. 65 p.

Verfuss, U.K., D. Gillespie, J. Gordon, T. Marques, B. Miller, R. Plunkett, J. Theriault, D. Tollit, D.P. Zitterbart, P. Hubert, and L. Thomas. 2017. Low visibility real-time monitoring techniques review. Report number SMRUM-OGP2015-002 provided to IOGP, June 2016. 224 p.

Villegas-Amtmann, S., B.I. McDonald, D. Páez-Rosas, D. Aurioles-Gamboa, and D.P. Costa. 2017. Adapted to change: low energy requirements in a low and unpredictable productivity environment, the case of the Galapagos sea lion. Deep-Sea Res. Pt. II 140:94-104.

Ward, J., S. Jarvis, D. Moretti, R. Morrissey, N. DiMarzio, M. Johnson, P. Tyack, L. Thomas, and T. Marques. 2011. Beaked whale (*Mesoplodon densirostris*) passive acoustic detection in increasing ambient noise. J. Acoust. Soc. Am. 129(2):662-669.

Ward, J.A., L. Thomas, S. Jarvis, N. DiMarzio, D. Moretti, T.A. Marques, C. Dunn, D. Claridge, E. Hartvig, and P. Tyack. 2012. Passive acoustic density estimation of sperm whales in the Tongue of the Ocean, Bahamas. Mar. Mammal Sci. 28(4):E444-E455.

Werner, S.R., J.P.G. Spurgeon, G.H. Isaksen, J.P. Smith, N.K. Springer, D.A. Gettleson, L. N'Guessan, and J.M. Dupont. 2014. Rapid prioritization of marine ecosystem services and ecosystem indicators. Mar. Policy 50:178-189.

Williamson, M.J., A.S. Kavanagh, M.J. Noad, E. Kniest, and R.A. Dunlop. 2016. The effect of close approaches for tagging activities by small research vessels on the behavior of humpback whales (*Megaptera novaeangliae*). Mar. Mammal Sci. 32(4):1234-1253.

Winship, A.J., S.J. Jorgensen, S.A. Shaffer, I.D. Jonsen, P.W. Robinson, D.P. Costa, and B.A. Block. 2012. State-space framework for estimating measurement error from double-tagging telemetry experiments. Method. Ecol. Evol. 3(2):291-302.

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. Aquat. Mamm. 43(4): 439-446.

Wyatt, R. 2008. Joint Industry Programme on sound and marine life review of existing data on underwater sounds produced by the oil and gas industry. Issue 1. Submitted by Seiche Measurements Ltd. to Joint Industry Programme on Sound and Marine Life. Seiche Measurements Ltd. Ref. S186. vi + 98 p.

Young, K.E. and E.O. Keith. 2011. A comparative analysis of cetacean vital rates using matrix population modeling analysis of cetacean vital rates. Int. J. Appl. Sci. Technol. 1(6):261.

ATTACHMENT A

to July 21, 2017
Letter of IAGC/API/NOIA

August 28, 2015

VIA EMAIL (ITP.Laws@NOAA.gov)

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 Incidental Harassment Authorization Applications for the Incidental Taking of Marine Mammals During Geophysical Surveys in the Atlantic Ocean

Dear Ms. Harrison:

This letter provides the comments of the International Association of Geophysical Contractors ("IAGC"), the American Petroleum Institute ("API"), and the National Ocean Industries Association ("NOIA") (collectively, the "Associations") in response to the National Marine Fisheries Service's ("NMFS") request for comments on four pending Incidental Harassment Authorization ("IHA") applications for geophysical surveys in the outer continental shelf ("OCS") of the Atlantic Ocean. We appreciate this opportunity to preliminarily comment on the pending applications, and we strongly support geophysical surveying in the Mid- and South Atlantic OCS, which furthers our common interest in the safe and responsible development of domestic oil and gas reserves.

I. THE ASSOCIATIONS

IAGC is the international trade association representing geophysical services companies that support and provide critical data to the oil and natural gas industry. IAGC members (including companies engaged in geophysical data acquisition, processing and interpretation, geophysical information ownership and licensing, and associated services and product providers) play an integral role in the successful exploration and development of offshore hydrocarbon resources through the acquisition and processing of geophysical data. IAGC members have expressed interest in conducting geophysical activities on the Atlantic OCS, and all three of the seismic survey IHA applicants are IAGC members.

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. OCS. The NOIA membership comprises more than 325 companies engaged in a variety of business activities, including seismic surveying, production, drilling, engineering, marine and air transport, offshore construction, equipment manufacture and supply, telecommunications, finance and insurance, and renewable energy.

II. COMMENTS

A. Approval of IHA applications for Atlantic surveys is consistent with the MMPA and furthers Congressional directives to develop oil and gas reserves in the OCS.

The Marine Mammal Protection Act ("MMPA"), 16 U.S.C. §§ 1361-1407, provides mechanisms for the authorization of the incidental taking of small numbers of marine mammals. 16 U.S.C. § 1371(a)(5)(A)(i); 50 C.F.R. § 216.107. To issue an incidental take authorization, NMFS must find that the proposed activity (i) is limited to a "specified geographical region," (ii) would result in the incidental take of "small numbers" of marine mammals, and (iii) have no more than a "negligible impact" on a marine mammal species or stock. 16 U.S.C. § 1371(a)(5)(A). NMFS has a long and successful history of issuing such authorizations for seismic surveys in the Beaufort and Chukchi Seas, and in Cook Inlet, Alaska.

NMFS's authorization of marine mammal take incidental to exploratory activities in the Atlantic OCS is consistent with the Outer Continental Shelf Lands Act ("OCSLA"), which mandates the "expeditious and orderly development" of the OCS "subject to environmental safeguards," such as those provided under the MMPA. 43 U.S.C. § 1332(3). The U.S. Bureau of Ocean Energy Management ("BOEM") currently estimates that the Mid- and South Atlantic OCS holds at least 4.72 billion barrels of oil and 37.51 trillion cubic feet of natural gas.[1] Although these estimates are impressive, it is widely believed that modern seismic imaging—the only feasible technology that accurately creates a subsurface image before a well is drilled—will aid in better locating and dissecting prospective areas for exploration and provide more realistic estimates of the potential resource. The pending geophysical survey proposals will facilitate the safe and orderly development of oil and gas reserves in the Mid- and South Atlantic OCS.

[1] *See* http://www.boem.gov/Assessment-of-Oil-and-Gas-Resources-2014-Update/.

Seismic modeling not only helps to delineate reserves, it also significantly reduces environmental risk by increasing the likelihood that exploratory wells will successfully tap hydrocarbons and decreasing the number of wells that need to be drilled in a given area. This reduces the overall environmental impact of oil and gas development by limiting the footprint of exploration. Because survey activities are temporary and transitory, they are the least intrusive and most cost-effective means to understanding where recoverable oil and gas resources likely exist in the Mid- and South Atlantic OCS.

In addition, more than four decades of worldwide seismic surveying and scientific research indicate that the risk of physical injury to marine life from seismic survey activities is extremely low. Currently, there is no scientific evidence demonstrating biologically significant negative impacts to marine life from seismic surveying. As stated by BOEM in its August 22, 2014, *Science Note*:

> 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.

http://www.boem.gov/BOEM-Science-Note-August-2014/.

Finally, it bears mention that IAGC, API, and the oil and gas industry fund independent research to further our understanding of the potential effects of seismic surveys on marine animals including mammals. This helps to reduce uncertainties about the possible effects of seismic surveys. Some of this research, in addition to other frequently cited references regarding the effects of sound on marine life, is reviewed in the annotated bibliography included as Attachment A to the April 29, 2015 comment letter of IAGC, API, and NOIA (which is included in the Appendix attached hereto).

B. The best available science demonstrates that seismic surveys do not cause Level A harassment and, therefore, authorization of Level A harassment is not required.

Under the MMPA, Level A harassment is defined as "any act of pursuit, torment, or annoyance which . . . has the potential to injure a marine mammal or marine mammal stock in the wild." 16 U.S.C. § 1362(18)(A)(i) (emphasis added); *see also* 50 C.F.R. § 216.3. In addition, NMFS is required to base marine mammal incidental take authorizations on the "best scientific evidence available." 50 C.F.R. § 216.102(a). We are aware of no scientific evidence demonstrating that seismic activities have resulted in the injury of marine mammals. To the contrary, the history of incidental take authorizations for offshore seismic activities shows that seismic operations have negligible impacts to individual marine mammals and to marine mammal stocks, and that levels of actual incidental take (Level B) are far smaller than even the most balanced pre-operation estimates of incidental take.[2]

[2] *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, 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 ").

Given this well-established scientific record, the Associations firmly take the position that the authorization of Level A harassment incidental to seismic surveys is not consistent with the best available science and, therefore, is not warranted or appropriate. In this context, the Associations note that one of the four Atlantic IHA applications requests authorization for Level A harassment. For the reasons stated above and below, the Associations disagree with the projections of Level A harassment set forth in that application.

As a general matter, the Level A take estimates described in the application improperly equate projected received sound levels to take. Potential exposure to certain sound levels does not necessitate that injury may occur. For example, the application estimates 9,017 Level A takes of bottlenose dolphins based only on potential exposures. However, even if 9,017 exposures to 180 dB SPL rms occurs, the best available science demonstrates that temporary threshold shift ("TTS") will not occur to bottlenose dolphins at this level of exposure. *See infra* § II.C.1. Moreover, it is well-accepted that the assumption that exposure to 180 dB SPL rms causes injury to marine mammals is incorrect and contrary to the best available science.[3] NMFS is not bound by this outdated acoustic criteria and, instead, must determine the potential type and levels of take that are "reasonably likely" or "reasonably expected" to occur based on the best scientific evidence available. 50 C.F.R. §§ 216.102(a), 216.103.[4]

More specifically, the subject IHA application appears to contain a number of incorrect assumptions that contribute to incorrect estimates of Level A harassment. Some of these assumptions are as follows:

- The application does not take into account the fact that many, if not all, animals will react to sound and leave an area before they enter areas with sounds levels exceeding the threshold that NMFS assumes will result in Level A harassment. The models used in the application do not appear to incorporate animal behaviors, such as avoidance to "ramping up" sound sources, which would substantially reduce the

[3] *See* Southall, B.L., Bowles, A.E., Ellison, W.T., Finneran, J.J., Gentry, R.L., Greene, Jr., C.R., Kastak, D., Ketten, D.R., Miller, J.H., Nachtigall, P.E., Richardson, W.J., Thomas, J.A., and Tyack P.L. 2007. Marine mammal noise exposure criteria: Initial scientific recommendations. *Aquatic Mammals,* 33:411-521; Finneran, J.J., and Jenkins, A.K. 2012. Criteria and thresholds for U.S. Navy acoustic and explosive effects analysis. San Diego, California: SPAWAR Systems Center Pacific.

[4] In fact, NMFS has used other criteria as the basis for recent MMPA incidental take authorizations. *See* 80 Fed. Reg. 46,112, 46,148-49 (Aug. 3, 2015); 80 Fed. Reg. 13,264, 13,280-81 (Mar. 13, 2015).

estimated number of exposures (which, in any event, do not equate to take, as described above).[5]

- The application assumes that Level A take will occur beyond 500 meters from the sound source, but does not propose to power down or shut down operations for detections beyond 500 meters. It is well-established that marine mammal observations can be made well past 500 meters and seismic operators have a longstanding history of successfully employing power down and shut down procedures for marine mammal observations beyond 500 meters and, thereby, avoiding exposure at levels that NMFS incorrectly assumes will result in Level A harassment.

- The application appears to make overly conservative assumptions in its source characterization, which result in abnormally large acoustic propagation ranges. In some cases, these assumed acoustic propagation ranges are more than double the size of the ranges calculated in the other two seismic survey applications, which increases the assumed affected area by a factor of four.[6]

Finally, except for very limited exceptions,[7] incidental take authorizations have been issued for seismic survey operations for only Level B harassment, not Level A harassment. The extensive record from these authorizations, including substantial monitoring documentation, demonstrates that commonly employed avoidance and mitigation measures (that are less stringent than those proposed in the pending applications) are effective in avoiding Level A harassment and minimizing the amount of Level B harassment. Again, we are aware of no information demonstrating that seismic survey operations have resulted in documented Level A harassment. Based on the extensive scientific record, multiple agency findings, and well-documented monitoring records, the Associations firmly take the position that (1) with the use of

[5] *See, e.g.*, *Issuance of IHA to Apache Alaska Corp. for Seismic Survey in Cook Inlet*, 79 Fed. Reg. 13,626, 13,636-37 (Mar. 11, 2014); *Issuance of IHA to TGS-Nopec for Seismic Survey in Chukchi Sea*, 78 Fed. Reg. 51,147, 51,160 (Aug. 20, 2013).

[6] We note that the applicant may correct these, and other, assumptions by submitting a revised IHA application for NMFS's consideration. Such a revised application would appropriately request authorization for only Level B harassment and propose mitigation measures that effectively avoid Level A harassment.

[7] *See, e.g.*, 80 Fed. Reg. 40,016 (July 13, 2015) (SAExploration IHA for Beaufort Sea survey); 77 Fed. Reg. 65,060 (Oct. 24, 2012) (ION Geophysical IHA for Beaufort Sea and Chukchi Sea survey). In both of these instances, the applicant requested authorization for only Level B harassment, but NMFS nonetheless authorized Level A harassment in the IHA.

proper mitigation measures, seismic survey operations can and do avoid Level A harassment; and (2) the authorization of Level A take incidental to seismic survey operations is therefore not warranted or appropriate.

C. Mitigation programs are effective in limiting and preventing the incidental take of marine mammals.

The best available scientific data and information demonstrate that mitigation programs can effectively minimize and avoid the incidental take of marine mammals as a result of offshore geophysical survey operations. Insofar as we are aware, no seismic activities that have received MMPA incidental take authorizations have caused impacts beyond a temporary change in behavior and there are no known injuries, mortalities, or other adverse consequences to any marine mammal species or stocks.

The majority of IHA applications currently under consideration by NMFS incorporate some of the mitigation measures recommended in the preferred alternative of BOEM's Atlantic Geological and Geophysical Activities Programmatic Environmental Impact Statement ("PEIS").[8] The Associations commented in detail on these proposed measures. *See* Appendix. For the reasons stated in our previous comment letters, some of the measures proposed by BOEM are not consistent with the best available science and/or are unnecessarily overbroad. Notably, however, BOEM has stated that it will not apply those measures uniformly, but rather will apply certain mitigation measures to fit specific circumstances. We encourage NMFS to also apply only those mitigation measures that are appropriate for specific circumstances and that result in the least practicable adverse impact. Although the IHA applicants are free to voluntarily propose some of the mitigation measures recommended by BOEM, we restate below the reasons why some of those measures are either overly broad or not based on the best available science. We also adopt by reference our previous comments with respect to mitigation measures (*see* Appendix).

1. Exclusion zones

All of the IHA applicants commit to using exclusion zones to prevent marine mammal exposure to sound pressure levels of 180 dB re 1 µPa rms or more for cetaceans and 190 dB re 1 µPa rms for pinnipeds. Although the PEIS recommends a minimum exclusion zone of 500 m, exclusion zones should be based on the best available science and modeling, and if that modeling demonstrates that exclusion zones of less than 500 meters are warranted, then there is no basis for arbitrarily requiring a minimum exclusion zone of 500 m. This flexibility is consistent with both NMFS's and BOEM's commitments to adaptive management.

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[8] *See* Record of Decision, BOEM PEIS, available at http://www.boem.gov/Record-of-Decision-Atlantic-G-G/. The full PEIS, including appendices, is available at http://www.boem.gov/BOEM-2014-001-v1/.

The applicants also commit to shutting down seismic arrays where marine mammals are detected in the exclusion zone. The PEIS contains one exception to its proposed mandatory shut down policy—for dolphins that voluntarily enter the exclusion zone. Although this measure is adopted by multiple IHA applicants, we would like to emphasize, for reasons stated in our previous comments, that any shutdown for dolphins that enter the exclusion zone is unwarranted. A recently published study that investigated whether bottlenose dolphin exposure to seismic air pulse at cumulative sound exposure levels of 185-196 dB re 1 μPa^2-s results in a noise-induced TTS found that, even at that level of exposure, there was no evidence of TTS.[9] Additionally, observation reports continue to indicate that there is no significant difference between the frequency of dolphin sightings and acoustic detections during seismic operations, whether the source is active or silent.[10] In sum, mandatory dolphin shutdown mitigation measures, even when the animal does not "voluntarily" enter the exclusion zone, would broadly and substantially impact seismic operations without any corresponding environmental benefit and without any scientific support.

2. Buffer zones between concurrent surveys

Generally, the IHA applicants propose 40 km buffer zones between seismic operations (as recommended in the PEIS), and one applicant proposes a 60 km buffer zone between concurrent surveys. Consistent with our comments on the PEIS, we reiterate here that the best available scientific information does not support buffer zones of 40 km. This measure was not included in NMFS's Biological Opinion (associated with the PEIS), and BOEM has offered no evidence to support its underlying assumption that marine mammals would utilize the "corridor" that the separation requirement is designed to create. Indeed, in its Record of Decision, BOEM acknowledges "uncertainty about [the] effectiveness of this measure." Record of Decision at 6.

The IHA applicants are, of course, free to propose mitigation buffer zones that are appropriate for their specific surveys, and to the extent they propose the 40 km buffer zone recommended in the PEIS, they are agreeing to mitigation measures that go above and beyond what is necessary based upon the best available scientific information. The Associations also wish to clarify that they do not support the proposal for 60 km buffer zones, which clearly are not required based on the extensive scientific record. As stated in previous comments, the

[9] Finneran J.J., Schlundt C.E., Branstetter, B.K., Trickey, J.S., Bowman, V., and Jenkins, K. Effects of multiple impulses from a seismic air gun on bottlenose dolphin hearing and behavior. 137 J. Acoust. Soc. Am. 1634-46 (April 2015).

[10] *See* Barkaszi, M.J., M. Butler, R. Compton, A. Unietis, and B. Bennet. 2012. Seismic survey mitigation measures and marine mammal observer reports. U.S. Dept. of the Interior, Bureau of Ocean Energy Management, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study BOEM 2012-015. *See also* Attachment D to April 29, 2015 letter of IAGC, API, and NOIA (included in Appendix).

Associations recommend either no buffer zone or, alternatively, a 17.5 km buffer zone consistent with standard industry practice.

3. Mandatory "all clear" periods

All of the IHA applicants propose mandatory "all clear" periods, but two of the applicants propose a 30-minute window as opposed to the 60-minute "all clear" period proposed by BOEM in the PEIS. As a practical matter, expanding the standard 30-minute "all clear" period to 60 minutes would substantially increase the duration and cost of seismic surveys, which, in turn, increases safety and environmental risks. Increased survey time will also increase the amount of time that protected species are exposed to the potential effects associated with the presence of vessels.

Moreover, a mandatory 60-minute "all clear" period would be both novel and not supported by the best available science. To our knowledge, a 60-minute "all clear" period has never been required as a condition of any offshore seismic authorization in the United States. In fact, the routine and proven practice is to require a 30-minute or less "all clear" period for marine mammals.[11] There is no available information suggesting that the standard practice has not been effective and, to the contrary, all available information demonstrates that the standard practice has been very successful in protecting marine mammals. *See* footnotes 2 and 11. Mitigation measures required in an IHA must be supported by the best available science and limited to those that effect the "least practicable adverse" impact. A 60-minute "all clear" period is not supported

[11] Since the ROD was issued, additional MMPA incidental take authorizations that include 15- and 30-minute "all clear" periods have been proposed by NMFS. *See Issuance of IHA to Apache Alaska Corp. for Seismic Survey in Cook Inlet*, 79 Fed. Reg. 13,626, 13,636-37 (Mar. 11, 2014) (requiring 30-minute observation period before startup and after sightings of killer and ESA-listed beluga whales and large odontocetes, but only 15-minute period after sightings of pinnipeds and small odontocetes); *Issuance of IHA to Apache Alaska Corp. for Seismic Survey in Cook Inlet*, 78 Fed. Reg. 12,720, 12,732-33 (Feb. 25, 2013) (providing same requirements, and specifying that the shorter 15-minute clearance period applies to harbor porpoises); *Issuance of IHA to TGS-Nopec for Seismic Survey in Chukchi Sea*, 78 Fed. Reg. 51,147, 51,154, 51,160 (Aug. 20, 2013) (same); *Issuance of IHA to Shell and WesternGeco for Seismic Surveys in the Beaufort and Chukchi Seas*, 73 Fed. Reg. 66,106, 66,135-36 (Nov. 6, 2008) (requiring 30-minute observation period before ramp-up and 15- or 30-minute delay of ramp-up for sightings of small odontocetes and pinnipeds, or baleen whales and large odontocetes, including ESA-listed species, respectively); *Issuance of ITR for Oil and Gas Activity in Chukchi Sea*, 78 Fed. Reg. 35,364, 35,424, 35,425 (June 12, 2013) (requiring monitoring period of 30 minutes for walruses and ESA-listed polar bears before startup and after sighting); *Issuance of ITR for Oil and Gas Activity in Beaufort Sea*, 76 Fed. Reg. 47,010, 47,052 (Aug. 3, 2011) (same).

by the best available science and is not necessary to achieve the least practicable adverse impact.[12]

4. Vessel Strike Avoidance

In general, the pending IHA applications propose vessel strike avoidance measures that are more than adequate to effectively avoid vessel strikes. For example, the following measures are adopted in the majority of the pending IHA applications:

- Reducing speed to 10 knots or less when transiting across designated areas closed to active seismic operations for North Atlantic Right Whales ("NARW");
- Maintaining a 500 meter distance from any NARW and a 100 meter distance from any species listed under the Endangered Species Act ("ESA"); and
- Utilizing avoidance measures (e.g., vessel direction or speed alteration) if an ESA-listed species is seen within 100 m of the vessel.

The necessity of these proposed measures should be evaluated in the proper context. Seismic vessels are different than typical vessels due to the substantial amount of specialized equipment that they tow. Operationally, a seismic vessel must maintain forward motion to sustain the equipment spread. The consequence of immediately shifting the engine into neutral due to a marine mammal sighting could be significant equipment damage (potentially in the tens of millions of dollars), weeks of vessel downtime, and additional related safety risks to crew members. As a practical matter, a seismic vessel moving at 3 to 5 knots is very unlikely to strike an ESA-listed marine mammal. For instance, in the event of a sighting of an ESA-listed whale within 100 m of the vessel, the vessel could reasonably be expected to slow (to no less than 3 knots) and turn gently away from the animal, which would effectively avoid a collision and lessen the risk of damage to seismic equipment.[13]

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[12] Although a 60-minute "all clear" period is referenced in BOEM's Record of Decision, BOEM also indicated that "mitigation measures themselves will be reviewed as part of BOEM's commitment to adaptive management" in "subsequent environmental reviews of site-specific action." Record of Decision at 8. Moreover, BOEM's Record of Decision does not dictate the content of MMPA authorizations issued by NMFS, which must be based on the most rational conclusions that NMFS can draw from the best available science.

[13] *See, e.g., Issuance of IHA to SAExploration, Inc. for Seismic Survey in Cook Inlet*, 80 Fed. Reg. 29,162, 29,176 (May 20, 2015) ("NMFS neither anticipates nor authorizes takes of marine mammals from ship strikes."); PEIS at xiv ("It is unlikely that survey vessels would strike marine mammals because they would travel slowly during surveys (typically between 4.5-6 knots [kn]).").

We do not object to the IHA applicants proposing the above-listed vessel avoidance measures so long as they are practical and feasible for the operators. Indeed, some of the IHA applications reasonably provide that these measures will be implemented "when safety allows" or "to the extent practicable." This acknowledges the inherent limitations of fully operating seismic vessels and important safety concerns balanced against the very low strike risk posed by seismic vessels.

5. Protected species observers ("PSOs")

All four IHA applications commit to employing trained PSOs to maintain watch for marine mammals, including those protected under the ESA. The use of PSOs is a long-established, effective means of limiting the potential incidental take of cetaceans and pinnipeds.

More broadly, however, we recommend that NMFS not uniformly require implementation of the recommendations described in NOAA Technical Memorandum NMFS-OPR-49, *National Standards for a Protected Species Observer and Data Management Program: A Model Using Geological and Geophysical Surveys* (Nov. 2013) ("Observer Standards"). Although we appreciate the agencies' attempt to clarify and standardize observer guidelines and requirements, we believe the Observer Standards are flawed in a number of respects and have not yet been subject to public review and input. *See* May 7, 2014 comment letter of IAGC, API, and NOIA, Attachment A (included in Appendix). Among other things, the standards should encourage adaptive technology, remote monitoring, reduction of health, safety, and environmental risks, and use of an updated reporting form that provides substantive data from observations to inform the need (if any) for additional or revised mitigation measures. Although one of the IHA applicants has voluntarily proposed to adopt the Observer Standards, NMFS should not impose those standards on other current or future applicants.

6. Passive acoustic monitoring ("PAM")

Three of the four pending IHA applications commit to the use of PAM during all survey activities, whether or not visibility is compromised. The Associations recognize the utility of PAM during periods of low visibility. PAM is one of several monitoring techniques that complements (rather than replaces) traditional visual monitoring. Overall performance and capabilities of PAM are highly dependent on factors such as technical specification of equipment, operational setting, availability of experienced and trained personnel, and the species of marine mammals present in a given area. Use of PAM is therefore not always logistically possible. Moreover, mandatory use of PAM will increase survey cost and require the placement of more personnel on vessels (i.e., four dedicated PAM observers onboard). Accordingly, the Associations urge NMFS to either make the use of PAM optional, or require PAM only for operations at night and in periods of low visibility.

7. Special area avoidance and time-area closures

The four pending IHA applications present varied approaches to special area avoidance and time-area closures, all of which are reasonable means of minimizing and avoiding incidental take. NMFS should evaluate time-area closures on a case-by-case basis and should not require unsupported, blanket restrictions that may or may not apply to a given applicant's proposed program. Each application should be evaluated for the specific program proposed and the mitigation (time-area closures) should be narrowly tailored to only the activities proposed in a given IHA application.

D. Seismic surveys in the Atlantic OCS will not cause cumulatively significant impacts.

There has been no demonstration of population-level effects to marine life from seismic or other geophysical survey activity, individually or cumulatively. BOEM expressly recognizes this fact in its August 22, 2014 *Science Note*, in which it states that "[w]ithin the [Gulf of Mexico Central Planning Area] . . . 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."[14] BOEM similarly concluded in its March 9, 2015, *Science Note* that 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." http://www.boem.gov/BOEM-Science-Note-March-2015/. Moreover, BOEM has spent more than $50 million on protected species and noise-related research without finding evidence of adverse effects. The geophysical and oil and gas industries, the National Science Foundation, the U.S. Navy, and others have spent a comparable amount on researching impacts of seismic surveys on marine life and have found no evidence of cumulatively significant effects. In short, the best available data and information strongly support a conclusion that there will be no cumulatively significant impact from the surveys that have been proposed for the Mid- and South Atlantic OCS. *See* PEIS § 4.3.2.

[14] http://www.boem.gov/BOEM-Science-Note-August-2014/. Moreover, it is well documented that some marine mammal populations, such as the western Arctic bowhead whale population, have continued to grow in areas where seismic survey occurs. *See* Allen, B. M., and R. P. Angliss, 2013 Stock Assessment Reports, NOAA-TM-AFSC-277, available at: http://www.nmfs.noaa.gov/pr/sars/2013/ak2013_bowhead.pdf (from 1978 to 2001, Arctic bowhead whale abundance "doubled from approximately 5,000 to approximately 10,000 whales" is growing at a rate of over 3% per year).

III. CONCLUSION

The Associations appreciate NMFS's review of the IHA applications and consideration of these comments. Building on decades of industry experience, the four pending IHA applications set forth aggressive mitigation programs designed to effectively avoid and limit incidental take. Many of the proposed mitigation measures are more stringent than measures that have commonly been employed and, indeed, some of the proposed mitigation measures are unnecessary, based on the best available scientific information. With the use of proper mitigation measures, seismic survey operations can and do avoid Level A harassment and, therefore, the authorization of Level A harassment is not warranted or appropriate. The Associations support the issuance of IHAs for Level B harassment that prescribe mitigation measures that are effective and consistent with the best available data and information.

Sincerely,

Nikki Martin
International Association of Geophysical Contractors
President

Andy Radford
American Petroleum Institute
Sr. Policy Advisor – Offshore

Jeff Vorberger
National Ocean Industries Association
Vice President Policy and Government Affairs

APPENDIX

April 29, 2015

VIA Federal eRulemaking Portal

Mr. Gary D. Goeke
Chief, Environmental Assessment Section
Office of Environment (GM 623E)
Bureau of Ocean Energy Management
Gulf of Mexico OCS Region
1201 Elmwood Park Boulevard
New Orleans, LA 70123–2394

Re: Comments on Applications for G&G Permits in the Mid- and South Atlantic OCS

Dear Mr. Goeke:

This letter provides the comments of the International Association of Geophysical Contractors ("IAGC"), the American Petroleum Institute ("API"), and the National Ocean Industries Association ("NOIA") (collectively, the "Associations") in response to the Bureau of Ocean Energy Management's ("BOEM") request for comments on the pending Geological and Geophysical ("G&G") permit applications for the Mid- and South Atlantic Outer Continental Shelf ("OCS"). We appreciate BOEM's consideration of the comments set forth below.

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. IAGC members have expressed interest in conducting geophysical activities on the Atlantic OCS, and some IAGC members have already filed applications for authorizations relating to such activities.

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. OCS. The NOIA 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.

II. COMMENTS

A. Contextual Background

BOEM's plan to authorize exploratory activities on the Atlantic OCS is consistent with the Outer Continental Shelf Lands Act, which mandates the "expeditious and orderly development" of the OCS "subject to environmental safeguards." 43 U.S.C. § 1332(3). BOEM currently estimates that the Mid- and South Atlantic OCS holds at least 4.72 billion barrels of oil and 37.51 trillion cubic feet of natural gas.[1] Although these estimates are impressive, it is widely believed that modern seismic imaging using the latest technology will enable BOEM to more accurately evaluate the Atlantic OCS resource base. The industry's advancements in geophysical technology—including specifically and primarily seismic reflection technology, but also complimentary gravity, magnetics, and electromagnetic technology—will provide more realistic estimates of the potential resource. By utilizing these tools and by applying increasingly accurate and effective interpretation practices, industry operators can better locate and dissect prospective areas for exploration. In short, seismic and other geophysical surveys are the only feasible technologies available to accurately image the subsurface before a single well is drilled. Allowing the pending geophysical survey proposals to proceed, subject to appropriate "environmental safeguards," facilitates—indeed, makes possible—the orderly development of the Mid- and South Atlantic OCS.

For the energy industry, modern geophysical imaging reduces risk by increasing the likelihood that exploratory wells will successfully tap hydrocarbons and decreasing the number of wells that need to be drilled in a given area, which reduces the overall footprint for exploration. Because survey activities are temporary and transitory, they are the least

[1] *See* http://www.boem.gov/Assessment-of-Oil-and-Gas-Resources-2014-Update/.

intrusive and most cost-effective means to understanding where recoverable oil and gas resources likely exist in the Mid- and South Atlantic OCS.[2]

In addition, more than four decades of worldwide seismic surveying and scientific research indicate that the risk of direct physical injury to marine life as a result of seismic survey activities is extremely low, and currently there is no scientific evidence demonstrating biologically significant negative impacts to marine life. As BOEM stated in its August 22, 2014 *Science Note*:

> 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.

Moreover, IAGC, together with the oil and gas industry, funds independent research to further our understanding of the effects of seismic surveys on marine life. This is helping to reduce uncertainties about the possible effects of seismic surveys. Some of this research, in addition to other frequently cited references regarding the effects of sound on marine life, is reviewed in the annotated bibliography included as Attachment A to this letter.[3]

B. Seismic Survey Activities in the Mid- and South Atlantic OCS Will Have, at Most, a Negligible Impact on Marine Mammals

During the administrative process related to BOEM's issuance of its Final Programmatic Environmental Impact Statement for Proposed G&G Activities on the Mid- and South Atlantic OCS ("PEIS"),[4] the Associations provided comments that, among other things, explained why BOEM's assessment of marine mammal impacts was flawed and why

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[2] Although different surveys for different purposes may cover the same general area, these surveys are spread out in space and in time. If two or more surveys occur in the same place over a period of time, they are generating different information, designed to appeal to specific, unique customer needs not met by other surveys.

[3] Additional technical information regarding different types of seismic surveys is provided in Attachment B.

[4] BOEM, *Final Programmatic Environmental Impact Statement for Proposed G&G Activities on the Mid- and South Atlantic OCS* (Mar. 2014).

some of the mitigation measures proposed by BOEM were unnecessary and impractical. The Associations incorporate those comments by reference, and we have included a copy of IAGC's comment letter to the final PEIS as Attachment C. We also provide the following information, which is intended to supplement the information and positions presented in the PEIS comments.[5]

1. BOEM's site-specific environmental assessments should provide an accurate evaluation of expected marine mammal impacts

As explained in our PEIS comments, BOEM's evaluation of potential marine mammal impacts at the programmatic level is flawed because it is premised upon an unrealistic scenario in which exploration activities are projected to result in thousands of incidental takes of marine mammals, which BOEM has definitively stated will not actually occur. Indeed, in its response to comments in the Record of Decision associated with the PEIS ("ROD"), BOEM states very clearly that "the numbers estimated for incidental take are higher than BOEM expects would actually occur." ROD at 12; *see also id*. ("the take estimates are based on acoustic and impact models that are by design conservative, which results in an over-estimate of take"). The supposed effects of this "worst case" hypothetical scenario are then addressed in the PEIS with mitigation measures, many of which are similarly unrealistic because they mitigate inaccurately presumed effects.

Setting aside our continuing disagreement with BOEM's approach to the evaluation of marine mammal impacts in the PEIS, we respectfully request that BOEM perform a proper NEPA analysis in its site-specific environmental assessments and evaluate the actual environmental impacts that are expected to occur. For the reasons stated in our comments on the PEIS, such an approach would be consistent with both the law and the best available science. *See* IAGC PEIS Comment Letter § II.A (Attachment C).

2. A 40-km buffer between surveys is unnecessary and impractical

The PEIS recommends an expanded 40-km buffer zone between concurrent seismic surveys "to provide a corridor between vessels conducting simultaneous surveys where airgun noise is below Level B thresholds and approaching ambient levels." PEIS at 2-37. In the PEIS, BOEM acknowledges that there is "uncertainty about [the] effectiveness" of a 40-km buffer requirement and, in its ROD, BOEM states that it will "assess the value of this measure in site-specific environmental analyses . . . and decide whether to include it as a

[5] Consistent with BOEM's commitment "to adaptive management and the modification of mitigations if warranted by the facts at the site-specific level" (ROD at 11), we encourage BOEM to reconsider the data and information presented in the Associations' comments on the final PEIS as well as the information presented in this comment letter.

condition of a permit or other authorization." ROD at 10. We reiterate that a 40-km buffer is unnecessary and impractical for the reasons stated in the Associations' comments on the PEIS. *See* IAGC PEIS Comment Letter § II.B.2. We also provide the following additional points, and request that BOEM consider this information, in addition to our PEIS comments, as it conducts its site-specific analyses.

Although seismic operations can be detected at great distances under certain oceanographic conditions and locations, so can sound waves generated by earthquakes and baleen whale calls.[6] The deep sound channel in the Atlantic OCS, often cited for the notion that sound from seismic operations can be detected outside of a survey's established exclusion zone, does not extend onto the continental shelf off the mid-Atlantic region. Furthermore, this notion is only applicable if protected species and marine animals are present in the deep sound channel to receive the higher levels of sound. Few species dive that deep in the areas of the Atlantic Ocean under consideration. In particular, baleen whale species of greatest concern are not known to be present in waters at those depths.

The seismic sound source is engineered to direct its energy downward, rather than laterally, which the National Marine Fisheries Service ("NMFS") has admitted is itself a mitigation measure.[7] For any energy that is transmitted laterally, the signal strength decreases rapidly, well below the thresholds NMFS has established for Level B harassment and at such low frequency that it does not cause injury to marine mammals.[8] Consistent with this information, what evidence there is of potential behavioral disturbance from seismic operations suggests minor and transitory effects, such as temporarily leaving the survey area, and these effects have not been linked to negative or biologically significant impacts on marine mammal populations.

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[6] Nieukirk, S.L., Mellinger D.K., Moore S.E., Klinck K., Dziak R.P., and Goslin J. 2012. Sounds from airguns and fin whales recorded in the mid-Atlantic Ocean, 1999-2009. J. Acoust. Soc. Am. 131(2):1102-1112; Munk W., Worcester P., and Wunsch C. 1995. Ocean Acoustic Tomography. Cambridge U Press, Cambridge, UK.

[7] *See New Jersey v. National Science Foundation*, 3:14-cv-0429 (D. N.J.), Federal Defendants' Brief in Opposition to Plaintiffs' Motion for Declaratory and Injunctive Relief at 25 (July 7, 2014).

[8] Richardson W.J., Greene Jr. C.R., Malme C.I., and Thomson D.H. 1995. Marine Mammals and Noise. Academic Press, NY. *See also* Acoustic Ecology Institute, *Seismic Surveys at Sea: The contributions of airguns to ocean noise.* August 2005 (An air source array with a source level of 200 – 230 dB "drops quickly to under 180 dB (usually within 50- 500 m depending on source level and local conditions), and continues to drop more gradually over the next few kilometers, until leveling off at somewhere near 100 dB.").

Neither BOEM nor NMFS has yet to provide any scientifically supported rationale for the proposed 40-km buffer. Instead, the PEIS concluded the measure "would only potentially slightly reduce acoustic impacts on marine mammals, sea turtles, and other marine biota," but even then, the effectiveness of the measure is uncertain. ROD at 6. Accordingly, we respectfully request that BOEM decline to adopt the 40-km buffer zone in site-specific environmental assessments and, instead, recommend either no buffer zone or, alternatively, a 17.5-km buffer zone, consistent with standard practice and the best available science. *See* IAGC PEIS Comment Letter § II.B.2.

3. New research demonstrates that seismic impulses have insignificant effects on dolphins

The PEIS recommends a mitigation measure calling for the shutdown of operations if a dolphin enters the acoustic exclusion zone, unless the dolphin is determined by the observer to be voluntarily approaching the vessel. PEIS at 2-11. In our comments on the PEIS, we provided substantial information demonstrating that this proposed measure is contrary to the best available science, impractical, and otherwise unsupported. In those comments, we also directed BOEM to current research being conducted with the support of the E&P Sound and Marine Life Joint Industry Program to study the effects of multiple airgun pulses in odontocetes and, specifically, to study whether bottlenose dolphin exposure to airgun impulses results in temporary threshold shift ("TTS").[9] *See* IAGC PEIS Comment Letter § II.B.1. As the public abstract from the study states, "subjects participated in over 180 exposure sessions with no significant TTS observed at any test frequency, for any combinations of range, volume or pressure during behavioral tests."[10] This research will be published very soon in a peer-reviewed scientific journal.[11] We will provide the published paper to BOEM promptly upon its publication, and we request that it be included in the administrative record and considered by BOEM during the permitting process.

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[9] James J. Finneran et al., *Final Report* (2013). TTS in odontocetes in response to multiple airgun impulses. (The Associations understand that a copy of this Final Report was provided by the author to NMFS.)

[10] C.E. Schlundt et al., *Auditory Effects of Multiple Impulses from a Seismic Airgun on Bottlenose Dolphins*, presentation at the Effects of Noise on Aquatic Life Third International Conference, Budapest, Hungary (Aug. 11-16, 2013). The results of this study also are useful to support inclusion of frequency weighting in updated acoustic criteria.

[11] Finneran J.J., Schlundt C.E., Branstetter, B.K., Trickey, J.S., Bowman, V., and Jenkins, K. Effects of multiple impulses from a seismic air gun on bottlenose dolphin hearing and behavior. Submitted to J. Acoust. Soc. Am. (in review).

Additionally, PSO observation reports continue to indicate that there is no statistically significant difference between the frequency of dolphin sightings and acoustic detections during seismic operations, whether the source is active or silent. Enclosed with this letter as Attachment D is an updated version of an attachment to IAGC's PEIS comments, which includes additional data confirming this conclusion.

In sum, the proposed dolphin shutdown mitigation measure would broadly and substantially impact seismic operations without any corresponding environmental benefit and without any scientific support. For the reasons presented in this letter and in our comments on the PEIS, the Associations respectfully request that BOEM make an express finding that this recommended measure is unsupported and unnecessary.[12] In conjunction with this finding, we also request that BOEM clarify that shutdown is not required for dolphins within the exclusion zone in all circumstances, regardless of whether dolphins are exhibiting bow-riding behavior or any other behavior.

4. BOEM should modify the proposed 60-minute "all clear" requirement

The PEIS recommends that monitoring of the exclusion zone shall "begin no less than 60 min prior to start-up" and that restarting of equipment after a shutdown "may only occur following confirmation that the exclusion zone is clear of all marine mammals and sea turtles for 60 min." PEIS at C-29. As explained in our comments on the PEIS, this proposed measure is unprecedented and without factual or scientific support. Specifically, IAGC provided numerous examples confirming that the routine, and proven-to-be-effective, practice is to require 15- and 30-minute "all clear" periods—for marine mammals and for ESA-listed species. *See* IAGC PEIS Comment Letter § II.B.3. In its ROD, BOEM provides no substantive response to this indisputable information. Indeed, since the ROD was issued, additional MMPA incidental take authorizations that include 15- and 30-minute "all clear" periods have been proposed by NMFS.[13]

We sincerely hope that BOEM will reconsider this proposed requirement and work with NMFS to ensure that a reasonable 15- / 30-minute "all clear" requirement is included in the federal authorizations related to seismic activities in the Atlantic Ocean, consistent with

[12] Although BOEM notes that this and other measures were addressed in the draft PEIS, it still must consider comments on these measures as part of its site-specific analyses for the proposed surveys, and it may adjust mitigation requirements based upon those analyses.

[13] *See, e.g.*, 80 Fed. Reg. 9510, 9524 (Feb. 23, 2015) (proposed Cook Inlet incidental take authorization calling for a 15-minute "all clear" period for small odontocetes and pinnipeds and a 30-minute "all clear" period for large odontocetes); 80 Fed. Reg. 20,084, 20,097 (Apr. 14, 2015) (same provision for proposed Beaufort Sea incidental take authorization).

the well-supported current practice. Expanding the standard 15- / 30-minute "all clear" period to 60 minutes will substantially increase the duration and cost of seismic surveys, which, in turn, increases potential risks. *See* IAGC PEIS Comment Letter § II.B.3.[14]

5. There will be no cumulatively significant impact from the proposed surveys

As stated in our PEIS comments, there has been no demonstration of population level effects to marine life from seismic or other geophysical survey activity, individually or cumulatively. BOEM expressly recognizes this fact in its August 22, 2014 *Science Note*, in which it states that "[w]ithin the [Gulf of Mexico Central Planning Area] . . . 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 similarly concluded in its March 9, 2015 *Science Note* that 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." Moreover, BOEM has spent more than $50 million on protected species and noise-related research without finding evidence of adverse effects. The geophysical and oil and gas industries, the National Science Foundation, the U.S. Navy, and others have spent a comparable amount on researching impacts of seismic surveys on marine life and have found no evidence of cumulatively significant effects. In short, for the reasons stated in our comments on the PEIS, and as consistent with the well-established record and BOEM's public findings, there will be no cumulatively significant impact from the surveys that have been proposed for the Mid- and South Atlantic OCS.

C. Seismic Survey Activities in the Mid- and South Atlantic OCS Will Have, at Most, a Negligible Impact on Fish Populations and Fish Habitat

As part of the G&G permitting process in the Atlantic OCS, site-specific environmental assessments will include an Essential Fish Habitat ("EFH") assessment to determine whether the specific activity and location would cause a significant adverse effect

[14] The impact of this and other measures addressed by the Associations is magnified when coupled with the proposed expanded exclusion zones. The Associations reiterate their previous comments that exclusion zones should be based on the best available science, including when the science demonstrates that an exclusion zone of less than 500 m is appropriate. If the minimum 500 m exclusion zone requirement is not applied, IAGC would support the incorporation of power-down procedures to mitigate any potential effects, as described in IAGC's PEIS comments. *See* Attachment C, footnote 21; *see also, e.g.*, 80 Fed. Reg. at 9524 (Cook Inlet proposed incidental take regulations); 80 Fed. Reg. at 20,097 (Beaufort Sea proposed IHA); 80 Fed. Reg. 14,913, 14,928 (Mar. 20, 2015) (Cook Inlet Proposed IHA); 79 Fed. Reg. 36,730, 36,735 (June 30, 2014) (Notice of Issuance of Beaufort Sea IHA).

to fisheries and EFH. Because the sound output from a seismic survey is immediate and local, there is no contaminate residue or destruction of habitat, and therefore no significant adverse effect to EFH. For the reasons set forth below, seismic survey activities will also not result in any significant adverse effects to fish populations or to fisheries.

Marine seismic surveys have been conducted since the 1950s and experience demonstrates that fisheries and seismic activities can and do coexist. There has been no observation of direct physical injury or death to free-ranging fish caused by seismic survey activity, and there is no conclusive evidence showing long-term or permanent displacement of fish. Any impacts to fish from seismic surveys are short term, localized, and not expected to lead to significant impacts on a population scale.[15]

Seismic source vessels move along a survey tract in the water creating a line of seismic impulses. As the seismic source vessel is in motion, each signal is short in duration, local, and transient. Since seismic surveys are a moving sound source, any impacts to fish are inherently local and short term, potentially causing a localized reduction in fish abundance within close proximity to the seismic source.[16] There is no conclusive evidence,

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[15] *See* Attachment A; *see also* Science for Environment Policy, Future Brief: Underwater Noise, European Commission, June 2013: http://ec.europa.eu/environment/integration/research/newsalert/pdf/FB7.pdf; "Stocks at a Glance – Status of Stocks" 2011, U.S. Department of Commerce, NOAA: www.nmfs.noaa.gov/stories/2012/05/05_14; Boeger, W.A., Pie, M.R., Ostrensky, A., Cardoso, M.F., 2006. The Effect of Exposure to Seismic Prospecting on Coral Reef Fishes; Brazil. J. Oceanogr. 54, 235-239; 3D marine seismic survey, no measurable effects on species richness or abundance of a coral reef associated fish community. Mar. Pollut. Bull. (2013), http://dx.doi.org/10.1016/j.marpolbul.2013.10.031; Hassel, A., Knutsen, T., Dalen, J., Skaar, K., Lokkeborg, S., Misund, O.A., Osten, O., Fonn, M., Haugland, E.K., 2004. Influence of seismic shooting on the lesser sand eel. ICES J. Mar. Sci. 61, 1165-1173; Pena, H., Handegard, N.O. and Ona, E. 2013. Feeding herring schools do not react to seismic air gun surveys. ICES J. Mar. Sci, http://icesjms.oxfordjournals.org/content/70/6/1174.short?rss=1; Saetre, R. and E. Ona, 1996. Seismic investigations and damages on fish eggs and larvae; an evaluation of possible effects on stock level. Fisken og Havet 1996:1-17, 1-8.

[16] Although some studies have shown that various life stages of fish and invertebrate species can be physically affected by exposure to sound, in all of these cases, the subjects were very close to the seismic source or subjected to exposures that are virtually impossible to occur under natural conditions. For example, frequently cited experimental studies such as Skalski et al. (1992), Lokkeborg et al. (2010), Engas (1996), and Wardle (2001) employed artificially concentrated sound within hundreds of meters of the fish under observation and the fishing vessels. As Lokkeborg et al. (2012) noted in a recent review of the literature, "Seismic air gun emissions distributed over a large area may thus produce lower sound

(continued . . .)

however, showing long-term or permanent displacement of fish. Similar seismic surveys conducted for research in the Atlantic OCS in the past did not result in any detectable effects on commercial or recreational fish catch, based on a review of NMFS's data from months surveys were conducted, which noted that "there was absolutely no evidence of harm to marine species" (including fish).[17] Additionally, in the Gulf of Mexico, where G&G activities have routinely occurred for over 40 years, seafood harvested from the OCS is worth approximately $980 million annually and the fishing industry directly supports in excess of 120,000 jobs, suggesting that G&G activities can occur without negatively impacting commercial fisheries.

Finally, seismic and other geophysical surveys also do not result in closing areas to commercial or recreational fishing. During surveys, the survey crews work diligently to maintain a vessel exclusion zone around the survey vessel and its towed streamer arrays to avoid any interruption of fishing operations, including the setting of fishing gear. As with all combined uses of offshore waters, there must be a certain level of coordination by all parties. At sea, coordination is regulated by the U.S. Coast Guard under the International Regulations for Preventing Collisions at Sea, requiring a Local Notice to Mariners specifying survey dates and locations. BOEM has concluded that "there is only a limited potential for space-use conflicts between G&G activities and commercial fishing operations within the area of interest" and any impacts "would be intermittent, temporary, and short term." PEIS at 4-160, 4-161.

III. CONCLUSION

As explained above, the performance of seismic and other geophysical surveys is critical to the federally mandated "expeditious and orderly development" of the Mid- and South Atlantic OCS. A wealth of data and information demonstrates that these surveys will have no more than a temporary, localized, and negligible impact on marine life. The Associations respectfully encourage BOEM to proceed with approving the pending permit applications and to work with NMFS to ensure that only reasonable, well-supported, and effective mitigation measures are included as conditions of the permits and the related federal authorizations.

(. . . continued)
exposure levels and thus have less impact on commercial fisheries." As another example, Aguilar de Soto (2013) exposed scallop larvae to noise at loud volume for up to 90 hours at a distance of 9 centimeters, which is virtually impossible to occur outside of experimental settings.

[17] *New Jersey v. National Science Foundation*, No. 3:14-cv-0429 (D. N.J.), Federal Defendants' Brief in Opposition to Plaintiffs' Motion for Declaratory and Injunctive Relief at 25-26, citing Exhibit D, Higgins Decl. ¶ 21, Exhibit D, Mountain Decl. ¶ 8 (July 7, 2014).

We appreciate your consideration of our comments. Should you have any questions, please do not hesitate to contact Nikki Martin at (713) 957-8080.

Sincerely,

Nikki Martin
International Association of Geophysical Contractors
Vice President, Government and Legal Affairs

Andy Radford
American Petroleum Institute
Sr. Policy Advisor – Offshore

Jeff Vorberger
National Ocean Industries Association
Vice President Policy and Government Affairs

ATTACHMENT A

ANTHROPOGENIC SOUND AND IMPACTS TO MARINE LIFE:
An Annotated Bibliography of Selected & Frequently Cited References

IAGC, together with the oil and gas industry, funds independent research to further our understanding of the effects of seismic surveys on marine life. This is helping to remove uncertainties about the possible effects of seismic surveys. Some of this research, in addition to other frequently cited references regarding the effects of sound on marine life, is reviewed in the attached annotated bibliography.

More than four decades of worldwide seismic surveying and scientific research indicate that the risk of direct physical injury to marine life is extremely low, and currently there is no scientific evidence demonstrating biologically significant negative impacts to marine life. As BOEM stated in its August 22, 2014 *Science Note*, "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."

There has been no observation of direct physical injury or death to free-ranging fish caused by seismic survey activity, and there is no conclusive evidence showing long-term or permanent displacement of fish. Any impacts to fish from seismic surveys are short-term, localized and are not expected to lead to significant impacts on a population scale or to commercial and recreational fishing activities.

The seismic sound source is engineered to direct its energy downward, rather than laterally. For any energy that is transmitted laterally, the signal strength decreases rapidly and would not cause injury to marine mammals. Research indicates that in-water sounds received at 110-90 dB SPL are comparable to a whisper or soft speech, even if it travels hundreds or thousands of kilometers in water. In some areas, such as the busy ports of the Atlantic coast, ambient sound in the frequencies produced by seismic sources may be as high as 110-120 dB due to ship noise, thereby masking any additional contribution from distant seismic surveys. What evidence there is of potential behavioral disturbance from seismic operations suggests minor and transitory effects, such as temporarily leaving the survey area, and these effects have not been linked to negative biologically significant impacts on populations.

More information on our commitment to science can be found at www.soundandmarinelife.org.

ANTHROPOGENIC SOUND AND IMPACTS TO MARINE LIFE:
An Annotated Bibliography of Selected & Frequently Cited References

Aguilar de Soto N, Delorme N, Atkins J, Howard S, Williams J, Johnson M. 2013. Anthropogenic noise causes body malformations and delays development in marine larvae. Scientific Reports 3, 2831. DOI 10: 1038/srep02831. www.nature.com/scientificreports.

Purports to demonstrate that airgun sound affects development of scallop larvae at levels of 160 dB SPL or lower. But the work has many flaws; an unrealistically long sound, played at much shorter than normal intervals for as much as 90 hours continuous. The sound source used in the experiment was not able to accurately replicate the actual seismic sound and was placed only 9 cm from the test subjects, producing large particle displacement effects of 4-6mm/s velocity, comparable to an SPL of 195 dB SPL. The latter value translates to a distance of a few hundred meters from an actual source, not the hundreds of square kilometers postulated by the authors. The best laboratory culture methods typically yield some variation in survival and development, but this study reported perfect scores for all controls at all stages. The work needs to be replicated by an independent and expert experimentalist.

André M, Solé M, Lenoir M, Durfort M, Quero C, Mas A, Lombarte A, van der Schaar M, López-Bejar M, Morell M, Zaugg S, and Houégnignan L. 2011. Low-frequency sounds induce acoustic trauma in cephalopods. Front Ecol Environ 2011; doi: 10.1890/100124. www.frontiersinecology.org. The Ecological Society of America.

Another study where it is difficult to know what to make of the data because of the way the sound was presented and measured. The reported received level is 157 dB re 1 µPa, so one can presume that the measurement is of pressure, but whether this is averaged, spectrum level, total energy under the envelope is unclear. Levels up to 175 dB re 1 µPa are also reported but it is not clear if that is a single frequency peak or whether the received levels fluctuated around 157 dB to as high as 175 dB. Thus the actual exposure history as SEL for the two hours of exposure is unknown. The sound source is in air and its properties are not provided. Given the impedence mismatch of water the source would have had to be extremely loud to get as much as 157-175 dB SPL into the water. Squid do not have swim bladders or air spaces associated with the ears, so the appropriate value to report is actually particle velocity. This is especially true since the containers were so much smaller than the wavelengths of sound in water at those frequencies (4-30 meters). The sound field inside the containers is bound to be complex and should have been measured. What is most probable is that the squid experienced considerable vibratory motion for two hours, leading to the damage observed; damage that could have never occurred in an open water environment where pressure and particle velocity would never be experienced at those levels for that duration.

Bartol, S.M. and Bartol, I.K. 2011. Hearing Capabilities of Loggerhead Sea Turtles (Caretta caretta) throughout Ontogeny: An Integrative Approach involving Behavioral and Electrophysiological Techniques. Final Report, JIP Grant No.22 07-14. Available online at http://www.soundandmarinelife.org/research-categories/physical-and-physiological-effects-and-hearing/hearing-capabilities-of-loggerhead-sea-turtles-throughout-ontogeny.aspx

Bolle LJ, de Jong CAF, Bierman SM, van Beek PJG, van Keeken OA, Wessels PW, van Damme CJG, Winter HV, de Haan D, Dekeling RPA. 2012. Common Sole Larvae Survive High Levels of Pile-Driving Sound in Controlled Exposure Experiments. PLoS One 7(3): e33052. Doi 10:1371/journal.pone.0033052.

This is a well-designed and properly measured sound exposure experiment, although claims that recordings played from a speaker are able to replicate the impulse time amplitude signature should always be treated with skepticism. Exposures up to 206 dB SEL_{cum} did not produce mortality, with single strike SELs of 186 dB and zero to peak pressures of 32 kPa, erroneously reported as 210 dB re $1\mu Pa^2$ in the abstract.

Booman, C., Dalen, J., Leivestad, H, Levsen, A., van der Meeren, T. and Toklum, K. 1996. Effects from airgun shooting on eggs, larvae, and fry. Experiments at the Institute of Marine Research and Zoological Laboratorium, University of Bergen. (In Norwegian. English summary and figure legends). Fisken og havet No. 3. 83 pp. as reviewed in:

Dalen, J, Dragsund E, Næss A, and Sand O. 2007. Effects of seismic surveys on fish, fish catches and sea mammals. Report for the Cooperation group – Fishery Industry and Petroleum Industry, Report No. 2007-0512. Available at https://www.norskoljeoggass.no/PageFiles/6574/Effects%20of%20seismic%20surveys%20on%20fish,%20fish%20catches%20and%20sea%20mammals.pdf?epslanguage=no
Observed effects on eggs and larvae only extended 1 to 5 meters from a full seismic array, suggesting that powerful particle motion effects were responsible for damaging the microscopic eggs and larvae. The net effect would be a pencil line damage zone in the wake of the array that would conceivably account for some tiny fraction of 1% of pelagic eggs and larvae distributed in the larger region of interest. Considering that more than 99% of eggs and larvae typically never make it to adulthood, this is an inconsequential effect compared to predation, disease and many other natural density-dependent or density independent causes of mortality.

Castellote, M., Clark, C.W., and Lammers, M.O. 2012. Acoustic and behavioural changes by fin whales (*Balaenoptera physalus*) in response to shipping and airgun noise. Biological Conservation 147: 115–122.
The authors make a slim statistical case that calls were altered by the presence of shipping noise and in one case a seismic survey. Measured and modeled acoustic data in the Straits of Gibraltar, a very unusual acoustic environment, were extrapolated as a more general case to predict effects of seismic on fin and other related whales generally. This speculation should be supported with data. Inferences of whale displacement by sound were from reductions in numbers of vocalizations, not actual observed movement or changes in distribution.

Engås A, Løkkeborg S, Soldal AV, and Ona E. 1996a. Comparative fishing trials for cod and haddock using commercial trawl and longline at two different stock levels. J Northw Atl Fish Sci 19: 83-90. http://journal.nafo.int.

Commercial bottom trawl and longline vessels fished 7 days before, 5 days during, and 5 days after a seismic survey was conducted in the area. Acoustic surveys of fish populations were also conducted, along with a sampling bottom trawl of different dimensions and mesh size than the commercial trawl. Only before and after data were analysed in this paper; "during" data were omitted but are reported in Engås et al (1996b). Because multiple fishing methods were employed on two species of fish, the matrix of data are somewhat complicated: generally, catches declined, smaller fish were caught after the seismic survey, and the ratio of haddock to cod increased after survey. It is difficult to know what to make of the results given the number of uncontrolled and possibly contributing variables that could have confounded the results, including the unusual prolonged proximity of survey vessels to fishing, and the amount of continuous fishing in one place that may have contributed to reduced catches and smaller size fish being caught over time.

Engås A, Løkkeborg S, Ona E, and Soldal AV. 1996b. Effects of seismic shooting on local abundance and catch rates of cod (Gadus morhua) and haddock (Melanogrammus aeglifinus). Can J Fish Aquat Sci 53:2238-2249.

Same study as above but includes data during the survey and more spatial information showing the effects described above tended to be greatest near the seismic survey and less out to the borders of the study area. An independent re-analysis of the data (JRHGeo, unpublished) suggest a different interpretation of declining catches during the before-exposure period suggestive of depletion of stocks within the unusually heavy, concentrated fishing effort within the test area, followed by clearly decreased catches within 1 km of the survey but smooth

decline through pre- and during exposure periods, suggesting little to no effect beyond 1 km. In the 5 days following seismic survey there is a rebound of catch at both the < 1 km and 1-3 km ranges, which suggests that there may have indeed been an effect from the seismic sound on catches, but catches recovered immediately afterward, confounded by the ongoing 10-15 days of continuous intensive fishing in the area. The re-analysis suggests that the data may have been confounded by variables other than sound, and that the original clearcut conclusions in Engas et al 1996a,b are perhaps not quite as pronounced as initially stated.

Finneran J.J., Schlundt C.E., Branstetter, B.K., Trickey, J.S., Bowman, V., and Jenkins, K. (2013). Temporary threshold shift (TTS) in odontocetes in response to multiple air gun impulses. Final Report for JIP Project 2.1.1., 51 pp. Available online at http://www.soundandmarinelifejip.org/index.php?doc=docmeta&id=3695

Finneran J.J., Schlundt C.E., Branstetter, B.K., Trickey, J.S., Bowman, V., and Jenkins, K. (in review). Effects of multiple impulses from a seismic air gun on bottlenose dolphin hearing and behavior. Submitted to J. Acoust. Soc. Am.

Gross JA, Irvine KM, Wilmoth S, Wagner TL, Shields PA and Fox JR . 2013. The Effects of Pulse Pressure from Seismic Water Gun Technology on Northern Pike. *Trans Am Fish Soc* 142: 1335-1346. ISSN: 0002-8487 print / 1548-8659 online DOI: 10.1080/00028487.2013.802252.

The study assessed the probability of mortality of pike (freshwater) when exposed to two pulses at 3, 6 and 9 meters distance from either a 343 cu in water gun or a 120 cu in water gun, both pressurized at 2000 psi. Measures of peak and peak to peak pressure were made as well as $SEL_{cum.}$ SEL_{cum} was used as the metric for effects in most of the results and discussion since it seemed to correlate best with levels of injury and mortality. Mortality within 72-168 hours was correlated with SELs in excess of 195 dB. Gas bladder rupture was observed at 199 dB SEL; 100% of fish at 3-6 meters and 87% of fish at 9 meters. Given the history of water guns producing greater injury and mortality than airguns, these results with two pulses from good sized single guns, indicate that fish would need to be within a few meters of a single airgun or full array to achieve comparable effects.

Harrington JJ, McAllister J, and Semmens JM. 2010. Assessing the short term impact of seismic surveys on adult commercial scallops (Pecten fumatus) in Bass Strait: Final Report. Tasmanian Aquaculture and Fisheries Institute, U. of Tasmania

Scallops were sampled from control and exposure sites before and after an extensive 2-D seismic survey. No statistical differences were found between control and exposed populations, neither in survival nor body condition. Exposure levels were not recorded. The paper also reviews several prior studies of seismic effects on scallops in Ireland and other sites, all also with no effect. One cited paper reported that one of three scallops experienced a split in its shell at distance of 2 meters from an airgun.

Higgins SM. 2014. Declaration; State of New Jersey, Dept of Environmental Protection vs National Science Foundation, et al. United States Federal District Court, District of New Jersey. Case 3:14-cv-04249-PGS-LHG, Document 6-7, filed 07/07/14, pageID 1520-1527

Contains a comparison of annual commercial and recreational fishery catches for years and months in which seismic surveys were conducted off the New Jersey coast, relative to the same months in other years, between 1990-2004. No discernable differences were found between periods with seismic survey and without. (Fishery statistical data from NMFS 2014, http://www.st.nmfs.noaa.gov/).

Lavender, A.L., Bartol, S.M., and Bartol, I.K. (2014). Ontogenetic investigation of underwater hearing capabilities in loggerhead sea turtles (*Caretta caretta)* using a dual testing approach. J. Exp. Biol., 2014, 217(14):2580-2589.

Løkkeborg S, Ona E, Vold A, and Salthaug A. 2012. Effects of sounds from seismic air guns on fish behaviour and catch rates. In A.N. Popper and A. Hawkins (eds.), *The Effects of Noise on Aquatic Life*, Advances in Experimental Medicine and Biology 730, DOI 10.1007/978-1-4419-7311-5_95, pp. 415-419. Springer, NY NY.

This paper provides a good review of prior behavioral studies. They also report recent data from what is arguably the most realistic and thorough study to date; monitoring of two fisheries (gillnet and longline) for four species of fish; a halibut, two gadids (pollack and haddock) and a seabass (Sebastes marinus), along with acoustic (HF sonar) surveys of the fish populations. Gillnet catches of halibut and seabass increased during and after survey, possibly due to increased swimming activity, while longline catches of halibut and pollack decreased. Acoustic surveys revealed decreases in pollack abundance, but not other species, consistent with prior study by Engås et al (1996a,b).

McCauley RD, Kent CS, Archer M. 2008. Impacts of seismic survey pass-bys on fish and zooplankton, Scott Reef Lagoon, Western Australia: Full report of Curtin University findings. Center for Marine Science and Technology, Curtin University, Perth WA. 92 pp. CMST Report 2008-32.

An extensive research effort involving a real seismic survey over a thoroughly monitored reef lagoon. Caged snapper and damselfish were exposed to seismic passes as close as 45-74 meters with 1% loss of hearing hair cells, later fully recovered. Behavioral reaction was observed at 155-165 dB SPL sound exposure levels but avoidance only occurred out to 200 meters on either side of survey. There was no effect on normal fish sound choruses.

McCauley RD, Fewtrell J and Popper AN. 2003. High intensity anthropogenic sound damages fish ears. J Acoust Soc Am 113(1):638-642 DOI: 10.1121/1.1527962

The authors were able to produce considerable unrecovered damage to the sensory structures of a typical fish ear (Pink snapper) after seven close passes (5-15 meters) by a towed 20 cubic inch seismic air source in the span of four hours. Although no cumulative Sound Exposure Level (SEL) or peak pressure or particle velocity measures were reported, the graphical display of the passes indicates multiple exposures over short periods of time at levels in excess of 180 dB SPL $rms_{0.95}$. The fish were caged and the authors noted that their movements indicated that the fish would have moved away from the sound source if possible, thus preventing the artificially high levels of exposure experienced.

Miller I. and Cripps E. 2013. Three dimensional marine seismic survey has no measurable effect on species richness or abundance of a coral reef associated fish community. Mar Pol Bull. Elsevier Press. http://dx.doi.org/10.1016/j.marpolbul.2013.10.031

No change in abundance or species composition was found in a natural reef community of resident reef fishes (emphasis on damselfishes) and mobile demersal fishes (emphasis on snappers of the Family Lutjanidae). Multiple passes by a full working seismic array were separated by about 6 hours between pass. Minimum stand-off distances from the reef were 400 meters on the outside and 800 meters inside the reef lagoon. Estimated exposures were generally around 187 dB SEL with some exposures as high as 200 dB SEL. Instantaneous peak or average SPL or particle velocity/acceleration were not measured.

Moein, S. E., Musick, J. A., Keinath, J. A., Barnard, D. E., Lenhardt, M. L. & George, R. 1995. Evaluation of seismic sources for repelling sea turtles from hopper dredges. In *Sea Turtle Research Program: Summary Report*. (Ed. Hales, L. Z.) pp 90-93. Technical

Report CERC-95.

National Research Council (NRC). 2005. Marine Mammal Populations and Ocean Noise: Determining When Noise Causes Biologically Significant Effects. National Academy Press, Washington DC. www.nap.edu.

This NRC report lays out a framework for estimating long term, cumulative population consequences from behavioral disturbance by sound, and by extension, any source of behavioral perturbation, individually or cumulatively. While developed for marine mammals, the principles of the Population Consequences of Acoustic Disturbance (PCAD) model are appropriate to any biological population.

Parry GD and Gason A. 2006. The effect of seismic surveys on catch rates of rock lobsters in western Victoria, Australia. Fisheries Research 79 (2006): 272-284.

A statistical comparison of changes in commercial catch rates (Catch Per Unit Effort, CPUE) coincident with seismic survey effort. No correlation was found in a two way analysis of variance, although the authors do note that most survey effort was in deep water away from the shallow water fishery, and that one survey in shallow water was in an area of low lobster abundance.

Peña H, Handegard NO, and Ona E. 2013. Feeding herring schools do not react to seismic air gun surveys. ICES J Marine Science, doi:10.1093/icesjms/fst079. 7 pp. http://icesjms.oxfordjournals.org/

A full 3-D seismic survey array was used to assess responses of herring monitored by an omnidirectional fisheries sonar. The source vessel approached the fish school from a distance of 26 km to a close approach at 2 km without any effect on the swimming and schooling behavior of the fish.

Popper AN, Smith ME, Cott PA, Hanna BW, MacGillivray AO, Austin ME and Mann DA. 2005. Effects of exposure to seismic airgun use on hearing of three fish species. J Acoust Soc Am 117:3958-3971.

Whitefish and juvenile pike did not show any TTS after exposure to five seismic playbacks of about 209 dB SPL_{peak} or 180 dB SEL, and particle displacements of 139 db SVL re 1nm/s (it is not possible to determine which physical property was responsible for any TTS observed in any of the tests). Adult pike under similar exposure conditions showed a TTS of about 20 dB at 400 Hz, which was recovered within 18 hours. Chub, also under similar exposure levels, showed slightly higher levels of TTS, about 25 dB at 200 Hz and 35 dB at 400 Hz, similar for 5 playbacks or 20 playbacks, and fully recovered within 18 hours. Chub are members of a hearing specialist family of freshwater fishes with no marine species.

Santulli A, Modica A, Messina C, Ceffa L, Curatolo A, Rivas G, Fabi G, D'Amelio V. 1999. Biochemical Responses of European Sea Bass (Dicentrarchus labrax L.) to the Stress Induced by Off Shore Experimental Seismic Prospecting. *Marine Pollution Bulletin*, Volume 38, Issue 12, December 1999, Pages 1105-1114.

This study involved exposure of caged fish to very close and very prolonged seismic air source in order to obtain physiological responses typical of stress. The fish returned to baseline levels within 72 hours, with no injury and no apparent lasting effect, despite the unusually high and prolonged sound exposures.

Song, J., D.A. Mann, P.A. Cott, B.W. Hanna, and A.N. Popper. 2008. The inner ears of Northern Canadian freshwater fishes following exposure to seismic air gun sounds**. J. Acoust. Soc. Am.** 124(2):1360-1366.

No damage was found to any of the ears of the test fish from Popper et al (2005), despite findings of Temporary Threshold Shift in two cases where peak pressure exceeded 205-209 dB re 1µPa SPL (peak) or 176-180 dB re 1 μPa^2-s single impulse (shot) SEL.

United States Navy. 2013. Atlantic Fleet Training and Testing Final Environmental Impact Statement / Overseas Environmental Impact Statement. Available online at http://aftteis.com/DocumentsandReferences/AFTTDocuments/FinalEISOEIS.aspx

Wardle CS, Carter TJ, Urquhart GG, Johnstone ADF, Ziolkowski AM, Hampson G, Mackie D (2001) Effects of seismic air guns on marine fish. Cont Shelf Res 21:1005–1027.

A study of free swimming cod, pollack and hake on a reef, using a fixed seismic source. C-start but no movement away from the source was observed at exposure levels up to 195 dB SPL at a distance of 109 meters. The authors speculate on possible reasons for the lack of response, including site fidelity to the unique reef environment at which the study was performed.

ATTACHMENT B

ATTACHMENT B

Currently, three types of surveys are proposed in the Atlantic OCS: 2D seismic surveys, a 3D seismic survey, and an airborne gravity and magnetic survey. These surveys are described in more detail below.

A. Seismic Surveys – Towed Streamers

For the energy industry, modern seismic imaging reduces risk by increasing the likelihood that exploratory wells will successfully tap hydrocarbons and decreasing the number of wells that need to be drilled in a given area, reducing associated safety and environmental risks and the overall footprint for exploration. The use of modern seismic technology is similar to ultrasound technology—a non-invasive mapping technique built upon the simple properties of sound waves. Because survey activities are temporary and transitory, it is the least intrusive and most cost-effective means to understanding where recoverable oil and gas resources likely exist in the Mid- and South Atlantic OCS.

To carry out these surveys, marine vessels use acoustic arrays, most commonly as a set of compressed air chambers, to create seismic pulses. A predominantly low-frequency sound pulse is generated by releasing compressed air into the water as the vessel is moving. The pulses are bounced off the layers of rock beneath the ocean floor. The returning sound waves are detected and recorded by hydrophones that are spaced along a series of cables that are towed behind the survey ship. Seismologists then analyze the information with computers to visualize the features that make up the underground structure of the ocean floor. Geophysical contractors often have proprietary methods of data acquisition that may vary depending on their seismic target and data-processing capabilities, making each contractor's dataset unique. Once the data is processed, geophysicists interpret it and integrate other geoscientific information to make assessments of where oil and gas reservoirs may be accumulated. Based largely on this information, exploration companies will decide where, or if, to conduct further exploration for oil and gas.

2D Seismic Surveys

Two-dimensional surveys are so-called because they only provide a 2D cross-sectional image of the Earth's structure. These surveys are typically used for geologic research, initial exploration of a new region, and to determine data quality in an area before investing in a 3D survey. 2D towed-streamer surveys are acquired with a single vessel usually towing a single air source array and a single streamer cable. The streamer is a polyurethane-jacketed cable containing several hundred to several thousand sensors, most commonly hydrophones. The air source array directs energy downward towards the ocean floor. An integrated navigational system is used to keep track of where the air sources are activated, the positions of the streamer cable, and the depth of the streamer cable. The end of the cable is tracked with global positioning system (GPS) satellites, and tail buoys are attached at the end. Radar reflectors are routinely placed on tail buoys for detection by other vessels, and automatic identification system (AIS) devices are also routinely integrated into the tail buoys.

Ships conducting 2D surveys are typically 30-90 m (100-300 ft) long and tow a single-source array 200-300 m (656-984 ft) behind them approximately 5-10 m (16-33 ft) below the sea surface. The source array often consists of three subarrays, with six to twelve air source elements each, and measures approximately 12.5-18 m (41-60 ft) long and 16-36 m (52-118 ft) wide. Following behind the source array by 100-200 m (328-656 ft) is a single streamer approximately 5 to 12 or more km (3.1-7.5 mi) long. The ship tows this apparatus at a speed of approximately 3 to 5 knots. Approximately every 10-15 seconds (i.e., a distance of 23-35 m [75-115 ft] for a vessel traveling at 4.5 kn [8.3 km/hr]), the air source array is activated. The actual time between activations varies depending on ship speed and the desired spacing.

Typical spacing between ship-track lines for 2D surveys, which is also the spacing between adjacent streamer line positions, is greater than a kilometer. Lines can transect each other and can be parallel, oblique or perpendicular to each other. 2D towed-streamer surveys are normally regional, covering a large area of ocean so that activity is not always limited to a particular area. 2D surveys can provide high resolution imaging with tight line spacing intervals in shallow areas.

2D surveys can cover a larger area with less data density in less detail, resulting in a lower cost per area covered. While surveying, and after a prescribed ramp-up of the output of the array to full-operation intensity, a vessel will travel along a linear track for a period of time until a full line of data is acquired. Upon reaching the end of the track, the ship takes typically 2 - 6 hours to turn around and start along another track, varying depending on the spacing between track lines, the length of track lines, and the objectives of a specific survey. Some 2D surveys might include only a single long line. Others may have numerous lines, with line spacings of 2 km in some cases, and 10 km in other cases. Data acquisition generally takes place day and night and may continue for days, weeks, or months, depending on the size of the survey area. Data acquisition is not, however, continuous. A typical seismic survey experiences approximately 20 to 30 percent of non-operational downtime due to a variety of factors, including technical requirements or mechanical maintenance, standby for weather or other interferences, and performance of mitigation measures (e.g., ramp-up, pre-survey visual observation periods, and shutdowns).

3D Seismic Surveys

3D towed-streamer seismic surveys enable industry to image the subsurface geology with much greater clarity than 2D data because of the much denser data coverage. The quality is such that 3D data can often indicate hydrocarbon-bearing zones from water-bearing zones. Because 3D seismic data has been continuously and rapidly improving since its introduction in the 1970s, areas covered by 3D data shot only a few years ago may be reshot with current, improved technology, offering greater clarity than previous surveys. In addition, areas already covered using 2D techniques may be resurveyed with 3D. Further, 3D surveys may be repeated over producing fields at successive calendar times (at 6-month to several-year intervals) to better characterize and record changes over producing reservoirs. These 4D, or time-lapse 3D, surveys are used predominantly as a reservoir monitoring tool to detect and evaluate reservoir changes over time. Conventional, single-vessel 3D surveys are referred to as narrow azimuth 3D surveys.

The current state-of-the-art ships conducting 3D surveys are purpose-built vessels with much greater towing capability than the vessels conducting 2D surveys. While these vessels are generally 60 - 120 m long, with the largest vessels over 120 m (ft) in length and greater than 65 m (230 ft) wide at the back deck. These seismic ships typically tow two parallel source arrays 200-300 m (656-984 ft) behind them. The two source arrays are identical to each other and are the same as those used in the 2D surveys described previously. Following 100-200 m (328-656 ft) behind the dual source arrays are the streamers.

Most 3D ships can tow eight or more streamers at a time, with the total length of streamers (number of streamers multiplied by the length of each one) exceeding 80 km (50 mi). The theoretical towing maximum today is 24 streamers, each of which can be up to 12 km (7.5 mi) long, for a total of 288 km (179 mi). Towing 8-14 streamers that are each 3-8 km (1.9-5 mi) long is normal practice. Towing 10 streamers that are separated by 75-150 m (246-492 ft) means that a swath 675-1,350 m (2,215-4,429 ft) wide is covered on the sea surface in one pass of the ship along its track line. Other streamer configurations (number of streamers and their separation distance) can produce narrower or wider swaths.

The survey ship tows the apparatus at a speed of 3 to 5.5 kn during production. Approximately every 11 - 15 s (i.e., a distance of 25 m [82 ft] for a vessel traveling at 4.5 kn [8.3 km/hr]), one of the dual air source arrays is fired. The other array is fired 11 - 15 s later. To achieve the desired spacing, the time between firings depends on the ship speed. While surveying, a ship travels along a track for 12-20 hours (i.e., a distance of 100-167 km [62 - 104 mi] at 4.5 kn [8.3 km/hr]), depending on the size of the survey area. Upon reaching the end of the track, the ship takes 3 to 5 hours to turn around and start along another track. This procedure takes place day and night, and may continue for days, weeks, or months, depending on the size of the survey area. Data acquisition is not, however, continuous. A typical seismic survey experiences approximately 20-to-30 percent of non-operational downtime due to a variety of factors, including technical or mechanical problems, standby for weather or other interferences, and performance of mitigation measures (e.g., ramp-up, pre-survey visual observation periods, and shutdowns).

B. Non-Seismic Gravity and Magnetic Surveys

Both conventional gravity surveys and gravity gradiometry surveys are conducted today, most often by fixed-wing aircraft, or where necessary, by marine vessel deployment. There is no sound source associated with gravity or magnetic surveys. The dimensions of the gravity instruments and stand are approximately 1 m by 1 m by 1.5 m high (3 ft by 3 ft by 5 ft) and the total weight is approximately 150 kg (330 lb). The survey acquisition grid is similar to ship-based seismic surveys, generally with flight-line spacing of 0.5-3 km (0.3-2 mi). Surveys of 500 sq. km (180 sq. mi) can be completed in a few hours, with the aircraft flying at an altitude of 70-300 m (230-1,000 ft). The objectives of the survey will determine the flight-line spacing (distance between flight lines) and the altitude at which the survey will be conducted.

Measurements of the earth's magnetic field are useful in helping to determine geologic structures and stratigraphy in the subsurface in frontier exploration areas, such as the Atlantic OCS, and as a complement to existing seismic data. There are at least five types of magnetometers, three of which are commonly used in airborne magnetic surveying. In addition to the different types of magnetometers, there are also several different configurations that can be used on the aircraft. These configurations include: (1) a single sensor, typically a tail installation; (2) two horizontally separated magnetometers, usually wingtip pod sensors; (3) two vertically separated sensors, usually tail-mounted; and (4) a total magnetic intensity configuration, typically involving three, but potentially four, magnetic sensors. The sensor pods are cylindrical in shape, and typically 1-2 m (3.3-6.6 ft) long and several centimeters (several inches) in diameter.

The objectives of the survey (such as the amount of area to be covered, the desired detail to be obtained, etc.) and the cost determine three of the most important factors to be specified for any given survey: (1) the altitude at which the survey will be conducted; (2) the flight-line separation; and (3) the flight-line orientation, or direction. Recent surveys done in the Gulf of Mexico have been flown at altitudes of 60-300 m (200-1,000 ft), at speeds of 110 knots (250 km/hr), and with line spacings of 0.5-2 km (0.3-1.3 mi). Similar surveys were recently completed offshore Greenland and offshore Honduras.

ATTACHMENT C

May 7, 2014

Via Federal eRulemaking Portal

Mr. Gary D. Goeke
Chief, Environmental Assessment Section
Office of Environment (GM 623E)
Bureau of Ocean Energy Management
Gulf of Mexico OCS Region
1201 Elmwood Park Boulevard
New Orleans, LA 70123–2394

Re: Comments on Final Programmatic Environmental Impact Statement for Proposed G&G Activities on the Mid- and South Atlantic OCS

Dear Mr. Goeke:

This letter provides the comments of the International Association of Geophysical Contractors ("IAGC") in response to the Bureau of Ocean Energy Management's ("BOEM") Notice and Request for Comments on its Final Programmatic Environmental Impact Statement for Proposed G&G Activities on the Mid- and South Atlantic OCS ("PEIS"). *See* 79 Fed. Reg. 13,074 (Mar. 7, 2014). We appreciate BOEM's consideration of the comments set forth below.

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. IAGC members have expressed interest in conducting geophysical activities on the Atlantic OCS, and some IAGC members have already filed applications for authorizations relating to such activities.[1]

[1] In a joint letter with the American Petroleum Institute ("API") and the National Ocean Industries Association ("NOIA"), IAGC earlier commented on the draft PEIS ("DPEIS"). *See* Letter from Andy Radford, Sarah Tsoflias, and Luke Johnson to Gary D. Goeke (July 2, 2012) ("DPEIS Comment Letter"). API, NOIA, and IAGC have also submitted a comment letter dated

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Seismic surveys are the only feasible technology available to accurately image the subsurface before a single well is drilled. BOEM currently estimates that the Mid- and South Atlantic OCS holds at least 3.3 billion barrels of oil and 31.3 trillion cubic feet of natural gas. Although these estimates are impressive, it is widely believed that modern seismic imaging using the latest technology will enable BOEM to more accurately evaluate the Atlantic OCS resource base. The industry's advancements in geophysical technology – including seismic reflection and refraction, gravity, magnetics, and electromagnetic – will provide more realistic estimates of the potential resource. By utilizing these tools and by applying increasingly accurate and effective interpretation practices, IAGC's members can better locate and dissect prospective areas for exploration.

For the energy industry, modern seismic imaging reduces risk by increasing the likelihood that exploratory wells will successfully tap hydrocarbons and decreasing the number of wells that need to be drilled in a given area, reducing associated safety and environmental risks and the overall footprint for exploration. Because survey activities are temporary and transitory, it is the least intrusive and most cost-effective means to understanding where recoverable oil and gas resources likely exist in the Mid- and South Atlantic OCS.

I. OVERVIEW

IAGC supports BOEM's plan to authorize exploratory activities on the Atlantic OCS consistent with the Outer Continental Shelf Lands Act ("OCSLA"), which calls for the "expeditious and orderly development" of the OCS "subject to environmental safeguards." 43 U.S.C. § 1332(3). However, the PEIS undermines OCSLA's mandate, as well as the requirements of other applicable laws, such as the Marine Mammal Protection Act ("MMPA"), in a number of ways. In general, a fundamental flaw with the PEIS is its establishment of an unrealistic scenario in which exploration activities are projected to result in thousands of incidental takes of marine mammals, which BOEM admits will not actually occur. The supposed effects of this "worst case" hypothetical scenario are then addressed in the PEIS with mitigation measures, many of which are similarly unrealistic because they mitigate inaccurately presumed effects. This approach is contrary to both the best available scientific information and applicable law.

Many of the mitigation measures recommended in the PEIS are infeasible, will impose serious burdens on industry, may discourage exploration of the Atlantic, and will result in no benefits to protected species (because they address unrealistic effects). IAGC can and will support mitigation measures that are well supported by the best available science, consistent with existing practices that are proven to be effective and operationally feasible. However, we cannot

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May 7, 2014 (the "Joint Trades Letter"), in response to the PEIS, which IAGC incorporates by reference.

support mitigation measures with no basis in fact or science, which are intended to address effects that will not occur, and which will result in less exploration of the OCS, contrary to OCSLA.

Accordingly, we strongly urge BOEM to include in its Record of Decision ("ROD") the modifications suggested in the comments set forth below. With respect to the alternatives presented in the PEIS, Alternative A presents the option that is most supported by the best available science and applicable law. However, IAGC would support BOEM's adoption of Alternative B only so long as all of the modifications suggested below are incorporated into the ROD. All of these suggested modifications are within the scope of the analyses contained in the PEIS. *See Great Old Broads for Wilderness v. Kimbell*, 709 F.3d 836, 854-55 (9th Cir. 2013) (modified alternative in ROD upheld because all relevant impacts analyzed in NEPA document); *see also W. Watersheds Project v. BLM*, 721 F.3d 1264, 1277-78 (10th Cir. 2013) (same).

II. DETAILED COMMENTS

A. The PEIS's Marine Mammal Impact Analyses Are Factually and Legally Flawed

The PEIS's analysis of marine mammal impacts is, by BOEM's admission, an unrealistic assessment of the potential impacts of geophysical surveys on marine mammals that is purposefully constructed to overestimate levels of incidental take. The PEIS explains:

> The acoustic and impact modeling conducted to develop these [incidental take] estimates is by its very nature complex and demands numerous specific details be identified and used during calculations[.] However, it must be emphasized that each of these assumptions are purposely developed to be conservative and accumulate throughout the analysis (e.g., representative sound source is modeled at highest sound levels and always at maximum power and operation, sound levels received by an animal are calculated at highest levels, marine mammal density values used likely exceed actual densities, and models do not include the effect of all mitigations in reducing take estimates). Therefore, the results of the modeling predictions will overestimate take.

PEIS at 1-5 (emphases added); *see also* PEIS at 4-62 ("BOEM emphasizes that these estimates should be seen as highly conservative of potential take without the consideration of most mitigation with the exception of the time-area closure described in Alternative A."). The results of this hypothetical "worst case" scenario analysis are strikingly divergent from the record of actual observed marine mammal impacts related to offshore exploration activities. *See* DPEIS Comment Letter §§ I, II & Appx. 1. For example, the PEIS implausibly concludes that thousands of marine mammals will experience Level A incidental take, and that hundreds of thousands of marine mammals will experience Level B incidental take, as a result of seismic activities. PEIS at Tables 4-9, 4-10, 4-11, 4-12. These take estimates would result in tens of thousands of shutdown events per year, in contrast to the average 55 shutdowns that are required per year in the Gulf of Mexico under existing operations, monitoring, and mitigation.[2] *See* DPEIS Comment Letter, Appx. 1.

We are aware of no federal agency assessment of the effects of seismic activities on marine mammals that results in incidental take estimates that are remotely similar to those stated in the PEIS. Moreover, the history of incidental take authorizations for offshore seismic activities demonstrates that levels of actual incidental take are far smaller than even the most balanced pre-operation estimates of incidental take. *See* DEIS at E-69.[3] The PEIS's flawed

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[2] Aggregating the estimated takes presented in Table 43 of the PEIS yields a total of 26,000 estimated takes.

[3] *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."); *id*. at 2-23 (with respect to sea turtles, "no significant cumulative impacts to sea turtles would be expected as a result of the proposed exploration activities when added to the impacts of past, present, or reasonably foreseeable oil and gas development in the area, as well as other ongoing activities in the area"); 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, 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."); *id*. at 4-235, 4-741 ("[T]here are no data to suggest that routine activities from the preexisting OCS Program are significantly impacting sea turtle populations."); 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

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approach to assessing the impacts of seismic activities on marine mammals results in a number of significant legal and factual errors, as set forth below.

1. The PEIS unlawfully analyzes a worst case scenario

Prior to 1986, NEPA regulations required a lead agency to prepare a "worst case analysis" of impacts for which there is incomplete or unavailable information. *See* 51 Fed. Reg. 15,618 (Apr. 25, 1986).[4] However, this requirement was expressly rescinded decades ago because it was found to be "an unproductive and ineffective method of achieving [NEPA's] goals; one which can breed endless hypothesis and speculation." *Id.*; *see Robertson v. Methow Valley Citizens Council*, 490 U.S. 332, 354-56 (1989) (U.S. Supreme Court confirming that worst case analysis is no longer applicable). In place of the worst case analysis requirement, the federal Council on Environmental Quality ("CEQ") promulgated "a wiser and more manageable approach to the evaluation of reasonably foreseeable significant adverse impacts in the face of incomplete or unavailable information in an EIS." 51 Fed. Reg. at 15,620. The new (and current) approach, codified in 40 C.F.R. § 1502.22, requires federal lead agencies to disclose such impacts and perform a "carefully conducted" evaluation based upon "credible scientific evidence." *Id.*; 40 C.F.R. § 1502.22(b)(1). In developing this requirement, CEQ explained that "credible" means "capable of being believed" and stated that "[i]nformation which is unworthy of belief should not be included in an EIS." 51 Fed. Reg. at 15,622-23 (responses to comments) (emphasis added).

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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 ").

[4] In the PEIS, BOEM determines that there is incomplete or unavailable information for a full assessment of the impacts of the proposed activities on marine mammals. *See* PEIS at 4-6, 4-47.

By performing an analysis of marine mammal impacts that is "purposely developed to be conservative," based on the "highest" sound levels and erroneously high marine mammal densities, and intended to "overestimate take," BOEM has performed precisely the same type of "worst case analysis" that was rejected by both CEQ and the U.S. Supreme Court many years ago. By its terms, and as expressly stated in the PEIS, the analysis of marine mammal impacts is purposely designed to be inaccurate and to evaluate the worst possible consequences that could hypothetically result from unmitigated seismic surveying. Indeed, it is hard to imagine an analysis that presents a scenario worse than the hundreds of thousands of incidental takings that are erroneously predicted by the PEIS. The PEIS's analysis of marine mammal effects is plainly not credible, it evaluates effects that, by BOEM's admission, will not occur, and, therefore, it is "unworthy of belief." The PEIS's assessment of marine mammal impacts unlawfully applies a "worst case" analysis and does not comply with NEPA or currently applicable CEQ regulations (40 C.F.R. § 1502.22).

2. The PEIS does not present an accurate scientific analysis

An EIS must rely upon "high quality" information and "accurate scientific analysis." 40 C.F.R. § 1500.1(b); *Conservation Nw. v. Rey*, 674 F. Supp. 2d 1232, 1249 (W.D. Wash. 2009); *Envtl. Def. v. U.S. Army Corps of Eng'rs*, 515 F. Supp. 2d 69, 78 (D.D.C. 2007) ("Accurate scientific analysis [is] essential to implementing NEPA."). It also must have "professional integrity, including scientific integrity" and may not rely on "incorrect assumptions or data" or "highly speculative harms" that "distort[] the decisionmaking process." *See Theodore Roosevelt Conservation P'ship v. Salazar*, 616 F.3d 497, 511 (D.C. Cir. 2010); 40 C.F.R. § 1502.24; 73 Fed. Reg. 61,292, 61,299 (Oct. 15, 2008) (CEQ regulations require "high quality" information and "scientific integrity"); *Native Ecosystems Council v. U.S. Forest Serv.*, 418 F.3d 953, 964 (9th Cir. 2005); *City of Shoreacres v. Waterworth*, 420 F.3d 440, 453 (5th Cir. 2005) (internal citations omitted).[5] To be sure, courts have invalided EISs that did not meet these standards, that were based on "stale scientific evidence . . . and false assumptions," or that failed to disclose the "potential weakness" of relied-upon modeling. *See, e.g., Seattle Audubon Soc'y v. Espy*, 998 F.2d 699, 704 (9th Cir. 1998); *Or. Natural Res. Council Fund v. Goodman*, 505 F.3d 884, 897 (9th Cir. 2007) (citations omitted).

Respectfully, the PEIS fails to satisfy any of these important NEPA principles. An analysis that, by the agency's admission, overestimates take and relies upon incorrect assumptions, is, by definition, "inaccurate." Moreover, the PEIS's analysis of marine mammal impacts is, at best, "highly speculative" because it is based on scenarios and assumptions that will not occur.

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[5] *See also CBD v. BLM*, 937 F. Supp. 2d 1140, 1155 (N.D. Cal. 2013) (principle that reasonably foreseeable environmental effects may not include "highly speculative harms" is equally applicable to direct and indirect effects).

3. The conclusions of the PEIS fail to consider, and are contrary to, the MMPA

The PEIS's assessment of marine mammal impacts is directly contrary to the MMPA. BOEM has defined the proposed action to include only those activities that have first received incidental take authorizations under the MMPA. *See* PEIS at 1-14, 1-25. As a prerequisite to incidental take authorization, the MMPA requires the permitting agency to find that the authorized take will have a "negligible impact" on marine mammals. 16 U.S.C. § 1371(a)(5)(A), (D). Accordingly, by definition, the proposed action analyzed in the PEIS should include only those seismic activities causing incidental take at levels that NMFS has expressly determined result in a "negligible impact" to marine mammal stocks. However, in sharp contrast, the PEIS concludes that the impacts of airguns on marine mammals under the proposed action are "moderate." PEIS at Table 2-4. By concluding that "moderate" impacts will result from seismic operations, BOEM has incorrectly analyzed the proposed action that is defined in the PEIS. Moreover, this discrepancy highlights the significant flaws that result from the PEIS's erroneous analysis of marine mammal impacts.[6] BOEM must analyze the effects of the action it has proposed, which includes offshore seismic operations that will receive incidental take authorizations under the MMPA and, by definition, will have no more than a negligible impact on marine mammal stocks. Based on 40 years of experience and recent scientific research and observational data, BOEM should find in the ROD that the impacts of seismic exploration are indeed negligible.

B. Certain Mitigation Measures Recommended in the PEIS Are Unsupported and Unreasonable

The record demonstrates that the scope of mitigation measures applied to offshore operations in the Gulf of Mexico is already more than adequate to protect marine mammals and sea turtles in a manner consistent with federal laws.[7] Despite this record, the PEIS recommends

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[6] The PEIS's "moderate" impact finding is also factually inconsistent. "Moderate" impacts are defined in the PEIS as "detectable, short-term, extensive, and severe; or … detectable, short-term or long-lasting, localized, and severe; or … detectable, long-lasting, extensive or localized, but less than severe." PEIS at x. Accordingly, a "moderate" seismic impact must be either "long-lasting" or "severe." However, insofar as we are aware, no seismic activities that have received MMPA incidental take authorizations have caused impacts amounting to anything more than temporary changes in behavior, without any known injury, mortality, or other adverse consequence to any marine mammal species or stocks. *See supra* note 3.

[7] *See supra* note 3; *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) ("[I]t is unlikely that the

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certain mitigation measures that have never been required for offshore exploratory operations, and that are more stringent (and less supported) than the measures that have already been successfully implemented. The unprecedented measures recommended in the PEIS are a direct result of BOEM's flawed impact assessments. For example, as described above, the PEIS creates a hypothetical worst case scenario for marine mammal impacts, determines that the projected adverse effects in that scenario will be substantial, and then recommends mitigation measures to address those supposed effects. However, because the adverse effects identified in the PEIS are inaccurate and unrealistic, the mitigation measures intended to address those effects are similarly flawed and without any factual or scientific support.

The mitigation measures that particularly concern IAGC are addressed in detail below. Without question, these measures, if implemented, will have substantial adverse effects on offshore geophysical operations. These measures will result in increased survey duration, which, in turn, can increase the potential exposure of marine mammals to seismic-related effects.[8] We strongly urge BOEM to reconsider these mitigation measures as it prepares the ROD.[9]

1. Dolphin shutdowns

The PEIS recommends a mitigation measure calling for the shutdown of operations if a dolphin enters the acoustic exclusion zone unless the dolphin is determined by the observer to be

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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, 14789 (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.").

[8] The mitigation measures also increase the amount of time the vessel spends surveying because shutdowns and delays necessarily result in overall increased surveying time to preserve data quality and integrity.

[9] The effects analysis contained in NMFS's associated biological opinion suffers from the same flaws as the PEIS's effects analysis. In addition, the terms and conditions stated in the biological opinion (which mitigate the inaccurate effects conclusions) lack a rational basis for the reasons stated in this letter with respect to the PEIS's corresponding mitigation measures. IAGC requests that BOEM work with NMFS to similarly reconsider and modify the biological opinion's terms and conditions.

voluntarily approaching the vessel. PEIS at 2-11.[10] This proposed measure is contrary to the best available science, impractical, arbitrary, and unsupported for at least the following reasons.

First, dolphins are mid- to high-frequency specialists and, therefore, insensitive to the low frequency impulse sounds emitted by seismic operations. The E&P Sound and Marine Life Joint Industry Program has supported research to study the effects of multiple airgun pulses in odontocetes and, specifically, to study whether bottlenose dolphin exposure to airgun impulses results in temporary threshold shift ("TTS").[11] As the public abstract from the study explains, "subjects participated in over 180 exposure sessions with no significant TTS observed at any test frequency, for any combinations of range, volume or pressure during behavioral tests."[12] Even at ranges as close as 3.9 m and with the air gun operating at 150 in^3 and 2000 psi, resulting in cumulative Sound Exposure Levels of 189-195 dB re 1μPa^2s, the impulses did not result in detectable TTS in any dolphin tested. As a result of the relative low-frequency content of airgun impulses compared to the relative high-frequency hearing ability of dolphins, no injuries or significant behavioral responses were observed in this study.[13] Industry observations corroborate this scientific evidence. For example, dolphins are frequently observed by personnel on seismic vessels to approach the vessels during operations to bow-ride and chase towed equipment – a direct indication of insensitivity to seismic sound generation. PSO observation reports indicate that there is no statistically significant difference between the frequency of dolphin sightings and

[10] "Voluntary approach" is defined as "a clear and purposeful approach toward the vessel by delphinid(s) with a speed and vector that indicates that the delphinid(s) is approaching the vessel and remains near the vessel or towed equipment." PEIS at 2-11.

[11] James J. Finneran et al., *Final Report* (2013). TTS in odontocetes in response to multiple airgun impulses. (IAGC understands that a copy of this Final Report has been furnished by the author to NMFS).

[12] C.E. Schlundt et al., *Auditory Effects of Multiple Impulses from a Seismic Airgun on Bottlenose Dolphins*, presentation at the Effects of Noise on Aquatic Life Third International Conference, Budapest, Hungary (Aug. 11-16, 2013) (emphasis added). The results of this study also are useful to support inclusion of frequency weighting in updated acoustic criteria.

[13] In a 2011 Programmatic EIS, the National Science Foundation recognized that "[t]here has been no specific documentation that TTS occurs for marine mammals exposed to sequences of air-gun pulses during operational seismic surveys." Programmatic EIS/OEIS for NSF-Funded & USGS Marine Seismic Research, at 3-133 (June 2011), http://www.nsf.gov/geo/oce/envcomp/usgs-nsf-marine-seismic-research/nsf-usgs-final-eis-oeis_3june2011.pdf (recognizing 180 dB re 1 uPa (rms) criterion for cetaceans "is actually probably quite precautionary, i.e., lower than necessary to avoid TTS at least for delphinids, belugas and similar species").

acoustic detections during seismic operations when the source is active or silent. *See* Attachment A.[14]

Second, even if there were scientific justification for the proposed dolphin shutdown mitigation measure (which there is not), implementation of the measure is impractical. We are aware of no mitigation measures applicable to offshore exploration activities in which an observer is required to subjectively determine the intent of a marine mammal. Determining marine mammal intent from great distances is very difficult for experienced marine mammal biologists in staged scientific experiments, let alone for observers who will be attempting to determine dolphin intent over vast distances in the ocean environment. Based on observation reports, PSOs will be unable to confidently assess animal behavior or "intentions" because they cannot accurately determine species within the expanded exclusion zone.[15] The result is that observers will likely, out of caution, call for shutdowns in almost all instances where dolphins are observed within the exclusion zone.

Third, in areas of high-density dolphin populations, such as the Atlantic Ocean and the Gulf of Mexico, shutdown requirements for a species that enjoys bow-riding and approaching vessels could effectively bring all seismic activity to a halt. Implementation of this proposed measure will substantially increase the number of shutdowns and delays in ramp-ups, which will result in much longer surveys and significantly increased costs with no environmental benefit. *See* Barkaszi, *supra*, note 7, at 1 (75% of delays in ramp-ups due to presence of protected species in exclusion zone during 30 minutes prior to ramp-up were due to dolphins).

Fourth, the proposed measure is without precedent. Under Joint NTL No. 2012-G02 (and previously NTL No. 2007-G02), BOEM required seismic operators in the Gulf of Mexico to shut down for any whale observed in the exclusion zone. BOEM defined "whales" as all marine mammals except dolphins and manatees. In the June 2013 settlement of litigation challenging BOEM's permitting of seismic activity in the Gulf of Mexico, the U.S. District Court for the Eastern District of Louisiana extended the shutdown requirements to manatees. In short, no

[14] *See also* A. MacGillivray et al., *Marine Mammal Audibility of Selected Shallow-Water Survey Sources*, J. Acoustical Soc'y of Am. 135(1) (Jan. 2014).

[15] *See* Attachment A. It is well known that different species will exhibit different behaviors. For example, Risso's dolphins generally avoid vessels and rarely bow-ride, rough-toothed dolphins generally avoid vessels but do bow-ride, and common dolphins are avid bow-riders. *See* K. Wynn & M. Schwartz, *Guide to Marine Mammals and Turtles of the U.S. Atlantic and Gulf of Mexico* (2009).

dolphin shutdown provision, as recommended in the PEIS, has ever been required by any federal agency.[16]

Finally, there is no legal basis for the proposed dolphin shutdown measure. Under the MMPA, mitigation measures attached to incidental take authorizations must address the reduction of incidental take. *See* 16 U.S.C. §§ 1371(a)(5)(A), (a)(5)(D); 50 C.F.R. § 216.104(a)(13). However, as set forth above, there is no scientific evidence demonstrating that active acoustic seismic surveys result in any incidental takes of dolphins. Accordingly, there is no statutory basis for recommending the dolphin shutdown mitigation measure.

In sum, the proposed dolphin shutdown mitigation measure would broadly and substantially impact seismic operations without any corresponding environmental benefit and without any scientific support. IAGC respectfully requests that BOEM, in its ROD, expressly find that this recommended measure is unsupported and unnecessary, and exclude the measure from the ROD's recommended mitigation measures. The ROD should also affirmatively clarify that shutdown is not required for dolphins within the exclusion zone in all circumstances, regardless of whether dolphins are exhibiting bow-riding behavior or any other behavior.

2. 40 km buffer zone between concurrent surveys[17]

In Alternative B, BOEM recommends an expanded 40 km buffer zone between concurrent seismic surveys. The rationale for this expanded buffer is "to provide a corridor between vessels conducting simultaneous surveys where airgun noise is below Level B thresholds and approaching ambient levels." PEIS at 2-37. The agency's stated scientific basis for this proposed measure is, at best, ambiguous: "New information suggests that, in some circumstances, airgun noise can be detected at great distances from the sound source, such as across ocean basins (Nieukirk et al., 2012), yet it is unknown if detection of sound at these distances has any effect on marine mammals or other marine species." PEIS at 2-38. No other scientific evidence, no published studies, and no other rationale are provided for this proposed measure, which is given a half-page explanation in Appendix C. In addition, this proposed

[16] For example, in the Gulf of Mexico, the average shutdown lasts for 58 minutes, *see, e.g.*, Barkaszi, *supra*, note 7, which the PEIS would extend by at least 30 minutes by increasing the visual monitoring period following a shutdown from 30 to 60 minutes. Multiplying a rough 1.5-hour average shutdown by 26,000 shutdowns would yield roughly 39,000 hours of shutdowns or approximately 1625 days. Because the typical seismic survey operation costs roughly $1.5 million per day, the total potential costs arising from the PEIS's assumptions equal a staggering $2.5 billion.

[17] This measure, as well as the 60-minute "all clear" period addressed below, were not addressed anywhere in the DPEIS. This is the first opportunity the regulatory community has had to comment on these measures.

measure is not mentioned at all in the biological opinion.

In contrast, the best available scientific information supports a buffer zone, if any, of 17.5 km, which is the standard separation distance maintained by seismic operators. The modeling performed by JASCO (*see* PEIS at Appx. D) demonstrates that the typical exposure radius for the 160 dB threshold is 10 km. The largest observed exposure radius was 15 km, but this occurred in less than 10% of the modeled cases. The lowest observed exposure radius was 5 km. Current technology has enabled many operators to decrease typical exposure radii to 7 to 9 km.

A buffer zone that more than doubles the highest possible exposure radii is clearly not reasonable or scientifically supportable – i.e., it is arbitrary. Moreover, the PEIS's reference to airgun noise detections at "great distances" does not support the proposed buffer zone because those detections occur (if at all) at very low levels that are well below the thresholds NMFS has established for Level B harassment.

The recommendations and analyses in an EIS must be "accurate," not speculative, and grounded in "high quality" scientific information. *See supra* Section II.A.2. The recommended 40 km buffer zone fails all of these standards. There is literally no scientific information that supports this measure, and, as explained above, the best available information contradicts it. To our knowledge, no buffer zones even approaching this magnitude have ever been required as a condition of offshore seismic authorizations.[18] To make matters worse, BOEM admits in the PEIS that implementation of the 40 km buffer would result in no additional benefits to protected species. PEIS at xxiv (40 km buffer "would not be expected to change any impact ratings"). Consequently, BOEM must decline to adopt the 40 km buffer zone mitigation measure in the ROD and, instead, recommend either no buffer zone, as recommended in Alternative A, or, alternatively, a 17.5 km buffer zone, consistent with standard practice.

3. 60-minute "all clear" period

The PEIS recommends that monitoring of the exclusion zone shall "begin no less than 60 min prior to start-up" and that restarting of equipment after a shutdown "may only occur following confirmation that the exclusion zone is clear of all marine mammals and sea turtles for 60 min." PEIS at C-29. However, again, BOEM has provided no factual or scientific support for this measure, nor is any meaningful supporting information provided in the biological opinion. To our knowledge, a 60-minute "all clear" period has never been required as a condition of any offshore seismic authorization in the United States. In fact, the routine and proven-to-be-effective practice is to require a 30-minute "all clear" period – for marine mammals

[18] *See, e.g.*, 78 Fed. Reg. 35,364, 35,423 (June 12, 2014) (vessel spacing of 24 km required to avoid any effects of multiple surveys on migrating or foraging walruses).

generally and for ESA-listed species.[19] There is no available information suggesting that the standard practice has not been effective and, to the contrary, all available information demonstrates that the standard practice has been very successful in protecting marine mammals.

Expanding the standard 30-minute "all clear" period to 60 minutes will substantially increase the duration and cost of seismic surveys, which, in turn, increases safety and environmental risks. Extrapolated over all surveys that will be performed over a five-year period, the increased time and expenses resulting from this mitigation measure alone will be dramatic. Increased survey time will also increase the amount of time that protected species are exposed to the potential effects associated with the presence of vessels. The PEIS contains no analysis of the increased operational or environmental effects associated with the 60-minute "all clear" period, compared to the standard 30-minute period (and sometimes 15-minute period) that has successfully been implemented in all offshore seismic operations to date.[20] Accordingly, in the ROD, BOEM should decline to adopt the 60-minute period as unsupported and unprecedented and, instead, adopt the standard 30-minute period.

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[19] *See Issuance of IHA to Apache Alaska Corp. for Seismic Survey in Cook Inlet*, 79 Fed. Reg. 13,626, 13,636-37 (Mar. 11, 2014) (requiring 30-minute observation period before startup and after sightings of killer and ESA-listed beluga whales and large odontocetes, but only 15-minute period after sightings of pinnipeds and small odontocetes); *Issuance of IHA to Apache Alaska Corp. for Seismic Survey in Cook Inlet*, 78 Fed. Reg. 12,720, 12,732-33 (Feb. 25, 2013) (providing same requirements, and specifying that the shorter 15-minute clearance period applies to harbor porpoises); *Issuance of IHA to TGS-Nopec for Seismic Survey in Chukchi Sea*, 78 Fed. Reg. 51,147, 51,154, 51,160 (Aug. 20, 2013) (same); *Issuance of IHA to Shell and WesternGeco for Seismic Surveys in the Beaufort and Chukchi Seas*, 73 Fed. Reg. 66,106, 66,135-36 (Nov. 6, 2008) (requiring 30-minute observation period before ramp-up and 15- or 30-minute delay of ramp-up for sightings of small odontocetes and pinnipeds, or baleen whales and large odontocetes, including ESA-listed species, respectively); *Issuance of ITR for Oil and Gas Activity in Chukchi Sea*, 78 Fed. Reg. 35,364, 35,424, 35,425 (June 12, 2013) (requiring monitoring period of 30 minutes for walruses and ESA-listed polar bears before startup and after sighting); *Issuance of ITR for Oil and Gas Activity in Beaufort Sea*, 76 Fed. Reg. 47,010, 47,052 (Aug. 3, 2011) (same).

[20] Pre-ramp-up and post-shutdown, the vessel is still moving and likely would move 8-9 km at 3-5 knots in a 60-minute period, bypassing any established exclusion zone several times. *See* 79 Fed. Reg. at 14,797 (NMFS stating that ramp-up is unnecessary "[b]ecause the vessel has transited away from the vicinity of the original sighting during the 8-minute period, implementing ramp-up procedures for the full array after an extended power-down (i.e. transiting for an additional 35 minutes from the location of initial sighting) would not meaningfully increase the effectiveness of observing marine mammals approaching or entering the exclusion zone for the full source level and would not further minimize the potential for take").

4. Exclusion zones greater than 500 meters

The PEIS explains that exclusion zones "shall be calculated independently and shall be based on the configuration of the array and the ambient acoustic environment, but shall not have a radius of less than 500 m…." PEIS at 2-10. BOEM's suggested approach for exclusion zones will require substantial modeling effort and will result in exclusion zones that are many times greater than those that have typically been implemented (with success) in the Gulf of Mexico. *See supra* note 3. The expanded exclusion zones are especially concerning because they will ultimately be dictated by the hearing group with the largest modeled radii once new group-specific acoustic criteria are implemented. High-frequency cetaceans, particularly delphinids, will therefore determine the size of the exclusion zone in most instances. Since BOEM is applying shutdown requirements to delphinids, and, as described above, because the exception to those requirements will rarely be applied in practice, this will result in numerous shutdowns due to the observation of delphinids within the large exclusion zone.

Moreover, these shutdowns will serve no environmental benefit because, as explained above, the best available science and information demonstrates that delphinids are unaffected by the lower frequency sounds produced by seismic operations. Exclusion zones should be based on the best available science and modeling and, if that modeling demonstrates that exclusions zones of less than 500 meters are warranted, then there is no basis for arbitrarily requiring a minimum exclusion zone of 500 m. If the minimum 500 m exclusion zone requirement is not applied, IAGC would support the incorporation of power-down procedures to mitigate any potential effects. Power-down procedures acceptable to IAGC are a modified version of the procedures described at 79 Fed. Reg. 14,780, 14,797 (Mar. 17, 2014) ("Langseth IHA").[21]

5. Turtle shutdowns

The PEIS applies exclusion zone shutdown criteria equally to marine mammals and sea turtles. However, the PEIS does not meaningfully address the fact that sea turtles are much more difficult to observe than marine mammals. Sea turtles can be reasonably observed at distances of 100 m to 300 m from a vessel, but it is very unlikely that sea turtles can be reliably observed at greater distances. *See* Attachment A (most turtle observations within 100 m). In addition, if a sea turtle is observed within the exclusion zone (triggering a shutdown of airguns), assuming the vessel is moving at 3 to 5 knots, the observed turtle will be outside of the exclusion zone within approximately 15 minutes because sea turtles swim very slowly compared to marine mammals.

[21] Specifically, IAGC would support power-down procedures similar to those in the Langseth IHA provided that: (1) power-down would be implemented only if a marine mammal is observed in or entering (not "likely" to enter) the exclusion zone; (2) power-down procedures may involve a reduction in the volume and/or pressure of the array; and (3) if a marine mammal is observed within the 500 m exclusion zone, then the reduced array would be shut down and shutdown procedures would apply.

In such circumstances, a 60-minute "all clear" requirement would plainly be unnecessary (setting aside the fact that it is unnecessary in all circumstances).

Because turtles are difficult to observe at distances greater than 300 m, application of the exclusion zone shutdown to sea turtles is infeasible and will very likely result in unwarranted shutdowns because observers, acting out of precaution, will call for shutdowns when anything resembling a sea turtle is observed. There is also no existing scientific basis for the proposed turtle shutdown requirement, and none is provided in the PEIS. *See supra* note 3. The ROD should therefore recommend a reduced exclusion zone for sea turtles that is feasible and practical. Such a reduction is also consistent with the best available science, which indicates that sea turtles are not as sensitive to sound as marine mammal species. *See* PEIS, Appx. I. IAGC recommends a 300 m exclusion zone for all sea turtle species.

6. Expanded NARW time-area closure and DMAs[22]

As part of Alternative B, BOEM recommends an expansion of the time-area closure applicable to North Atlantic Right Whales ("NARW") to a continuous 37 km-wide zone extending from Delaware Bay to the southern limit of the programmatic area. PEIS at C-32. It appears that BOEM intends this closure to be applied to <u>any</u> sound produced by seismic vessels such that no portion of a vessel's ensonification zone may enter the closed area. The result is that the proposed NARW time-area closure will be much larger than what is described in the PEIS. Because NARWs are primarily threatened by ship strikes and fishing entanglement – not seismic sound – BOEM should clarify in its ROD that the NARW time-area closure applies to the <u>presence of vessels, not a vessel's ensonified zone.</u> BOEM should also clarify in its ROD that vessels may transit through the closure area when seismic equipment is not active.

In addition, the PEIS includes time-area closure measures in areas designated as Dynamic Management Areas ("DMAs") under NMFS's ship-strike reduction regulations. *See* PEIS at C-16. These measures are very problematic, and unwarranted, for at least the following reasons:

- DMAs were created to address ship strike situations, which involve vessels traveling at high rates of speed (12-20 knots). Indeed, NMFS has indicated that vessel speeds of less than 10 knots are sufficiently protective. *See* 78 Fed. Reg. 73,726 (Dec. 9, 2013). BOEM's proposed application of DMAs to seismic operations is therefore contrary to both the original purpose of DMAs (to address ship strikes, not potential acoustic impacts) and NMFS's recent finding. Moreover, the proposed application to seismic vessels is particularly arbitrary because BOEM intends to broadly apply it to the vessel's 160 dB ensonified zone.

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[22] The DMA-related measures were also not included for public review in the DPEIS.

- Nowhere has either BOEM or NMFS evaluated the operational practicability or effectiveness of applying DMAs to seismic operations.

- Unlike NMFS's approach to DMAs, BOEM appears to propose to make seismic industry compliance with DMAs mandatory. There is no basis for such a measure, especially given that NMFS has taken no such step for the vessels that DMAs were intended to address.

- DMAs are unpredictable and the identification of DMAs on short notice will compromise the implementation of seismic survey operations that have been carefully planned over a substantial period of time, with no corresponding benefit.

7. Vessel strike avoidance

The PEIS's recommended vessel strike avoidance measures for ESA-listed whales present serious operational and safety problems, and must be modified. Specifically, the PEIS recommends that if a vessel comes within 100 m of an ESA-listed whale species, it "must reduce speed and shift the engine to neutral, and must not engage the engines until the whale(s) has moved outside of the vessel's path and the minimum separation distance has been established." PEIS at C-9. Respectfully, this measure fails to consider that seismic vessels are significantly different than typical vessels due to the substantial amount of highly specialized equipment that is towed behind a seismic vessel. Operationally, a seismic vessel must maintain forward motion to sustain the equipment spread or the whole system will collapse. The consequence of immediately shifting the engine into neutral could be significant equipment damage in the tens of millions of dollars, and weeks of vessel downtime. As a practical matter, a seismic vessel moving at 3 to 5 knots is very unlikely to strike an ESA-listed marine mammal. In the event of a sighting of an ESA-listed whale within 100 m of the vessel, the vessel could slow (to no less than 3 knots) and turn gently away from the animal, which would both avoid a collision and lessen the risk of damage to seismic equipment. In its ROD, BOEM must decline to adopt the vessel strike avoidance mitigation measure.

8. Passive acoustic monitoring

Under Alternative B, BOEM would require the use of Passive Acoustic Monitoring ("PAM") as part of the Seismic Airgun Survey Protocol. IAGC encourages consideration of PAM during periods of low visibility in its 2011 best practices guidelines. PAM is one of several monitoring techniques that compliments (rather than replaces) traditional visual monitoring. However, commercially available PAM systems can be highly variable, the equipment is unreliable, and PAM's utility as a secondary monitoring source during daylight observations has not been proven. Overall performance and capabilities of PAM are highly dependent on factors such as technical specification of equipment, operational setting, availability of experienced and trained personnel, and the species of marine mammals present in a given area. Mandatory use of PAM will increase survey cost, require the placement of more personnel on vessels (i.e., four dedicated PAM observers onboard), and increase entanglement risk due to more gear being towed in the water.

IAGC therefore urges BOEM to either make the use of PAM optional, as recommended in Alternative A, or require PAM only for operations at night and in periods of low visibility.[23] This is reasonable given BOEM's admission that "it is difficult to quantify any difference in impact level [of Alternative B] relative to Alternative A." PEIS at 2-40; *see also* PEIS at xxiv ("The degree of improvement [due to making PAM mandatory] has not been estimated but would not be expected to change any impact ratings."). IAGC encourages BOEM to use risk-based mitigation and monitoring measures based on the best available information and promote development of technologies that can best accomplish effective detection and monitoring of marine mammals.

9. National standards for protected species observers

The PEIS and biological opinion purport to adopt the recommendations described in NOAA Technical Memorandum NMFS-OPR-49, *National Standards for a Protected Species Observer and Data Management Program: A Model Using Geological and Geophysical Surveys* (Nov. 2013) ("Observer Standards"). However, this document was never released for public review and comment and was not referenced in the PEIS. Although we appreciate the agencies' attempt to clarify and standardize observer guidelines and requirements, the Observer Standards are flawed in a number of respects. It is imperative that the agencies consider public input on the Observer Standards and make the revisions necessary to ensure that the standards are workable, accurate, and appropriate. The standards should encourage adaptive technology, remote monitoring, reduction of health, safety, and environmental risks, and use of an updated reporting form that provides substantive data from observations to inform the need (if any) for additional or revised mitigation measures. The letter by IAGC, API, and NOIA, dated May 2, 2014, addressing the Observer Standards (attached) more specifically addresses our concerns with the Observer Standards and offers constructive solutions. We appreciate BOEM's consideration of our concerns.

C. The Adaptive Management Provisions Must Be Clarified and Improved

Although the PEIS states that BOEM will consider future data regarding the efficacy of mitigation measures and will adjust requirements for individual surveys, the PEIS appears to establish minimum standards that can only become more stringent through adaptive management. *See* PEIS at 2-39 (adaptive management at the site-specific level "would analyze the best available information and apply additional mitigation, depending on the site-specific proposed action" (emphasis added)); *see also* PEIS at 1-27 to 1-28 (examples largely focus on

[23] NMFS's biological opinion (page 308) only requires PAM for ramp-up at night or in periods of low visibility.

"additional" measures). As just one example, BOEM has established 500 m as a minimum exclusion zone and indicates that it will not set exclusion zones less than 500 m even if a smaller zone is supported by data and modeling.

The ROD must clarify that BOEM will implement "adaptive management" in the true sense of the term – i.e., site-specific requirements may be adjusted to be either less restrictive or more restrictive based on the project-specific information, the species present in the project area, the assessment of relevant risks, and the best available information.

III. CONCLUSION

IAGC appreciates this opportunity to comment on the PEIS. Although we support BOEM's plan to authorize exploratory activities on the Atlantic OCS, there are several aspects of the PEIS that are not supported by science or by law, or are otherwise infeasible. Of the alternatives presented in the PEIS, Alternative A presents the option that is most supported by the best available science and applicable law. However, IAGC would support BOEM's adoption of Alternative B only so long as all of the modifications suggested in these comments are incorporated into the ROD. We appreciate your consideration of our comments and sincerely hope that BOEM will prepare a ROD that addresses the concerns set forth above. Should you have any questions, please do not hesitate to contact me.

Sincerely,

Karen St. John
Group Vice President - Environment

International Association of Geophysical Contractors

cc: Mr. Walter Cruickshank (Walter.Cruickshank@boem.gov)
Ms. Jill Lewandowski (Jill.Lewandowski@boem.gov)

ATTACHMENT D

ATTACHMENT D

PSO Data 2013 - March 2015: Dolphin Sightings ***Provided by CGG based on MMO reports submitted to BOEM during this period representing approximately 33% of total vessel activity days in the GOM since 2013.[1] Data prior to 2013 is not included in this analysis because PAM was not used consistently until this point.***		
Species Identification		
% of Unidentified Dolphin	85%	In many reports, PSOs contribute sea state, distance, or the sun's glare as a key factor for not being able to identify species. The significant number of acoustic detections without confirmed species identification is also a main contributor.
% of Identified Dolphin	15%	
PAM		
% of PAM Detections	78%	PAM detections accounted for a majority of the total dolphin sightings and detection reports. However, only 1% of the acoustic detections successfully identified a specific dolphin species. Visual corroboration was necessary to identity the species about 25% of the time.
Source Activity Comparison		
% of sightings and/or acoustic detections – source active	55%	The frequency of sightings and acoustic detections are almost proportional when the source is active or silent.
% of sightings and/or acoustic detections –source silent	45%	
Animal Behavior		
% of sightings when bow-riding was observed (active or silent)	6%	The data indicates an estimated 2% variance in observed bow-riding when the source was active versus when the source was silent. Fewer PSO observations when the source is silent could account for some variance. The values are close enough to conclude the frequency of animal engagement with the vessel is not specific to source status.
Average Distance of Animal at Initial Sighting	570m	Initial sightings and detections are made most often at a distance between 500m and 800m.

PSO Data 2013 - March 2015: Turtle Sightings ***Provided by CGG based on MMO reports submitted to BOEM during this period representing approximately 33% of total vessel activity days in the GOM since 2013.[2] Data is taken from 2013 to be consistent with Dolphin sighting period.***		
Total Sightings	410	410 sea turtles were observed overall.
Average Distance of Animal at Initial Sighting	53m	Analysis of turtle sightings indicates observations are typically within 100m. It is often difficult to ascertain if an object in the water is a turtle or floating debris at further ranges.

[1] Estimated calculation based on level of activity from January 2013 to March 2015 from IHS SeismicBase Vessel Search Database.

[2] *Id.*

May 7, 2014

Mr. Gary D. Goeke
Chief, Environmental Assessment Section
Office of Environment (GM623E)
Bureau of Ocean Energy Management
Gulf of Mexico OCS Region
1201 Elmwood Park Boulevard
New Orleans, Louisiana 70123-2394

Submitted via email: *ggeis@boem.gov*

Re: Comments on the Final Programmatic EIS for the Mid- and South Atlantic

Dear Mr. Goeke:

This letter provides the comments of the American Petroleum Institute ("API"), the International Association of Geophysical Contractors ("IAGC"), and the National Ocean Industries Association ("NOIA"), in response to the Bureau of Ocean Energy Management's ("BOEM") Notice of Availability and Request for Comments on its Final Programmatic Environmental Impact Statement (EIS) for proposed Geological and Geophysical ("G&G") Activities on the Mid- and South Atlantic Outer Continental Shelf ("OCS"). *See* 79 Fed. Reg. 13,074 (March 7, 2014). We appreciate BOEM's consideration of the comments set forth below.

API is a national trade association representing over 600 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. API is a longstanding supporter of allowing new exploration in the Atlantic OCS and the Final Programmatic Environmental Impact Statement ("FPEIS") is the first step toward the much needed collection of new and improved data on potential oil and natural gas resources in the Mid-and South Atlantic OCS Planning Areas.

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.

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. Outer Continental Shelf ("OCS"). The NOIA membership comprises more than 275 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.

The Associations support BOEM's plan to authorize exploratory activities on the Atlantic OCS consistent with the Outer Continental Shelf Lands Act ("OCSLA"); however, the FPEIS undermines OCSLA's mandate to expeditiously and orderly develop the natural resources of the OCS, and the requirements of other applicable laws such as the Marine Mammal Protection Act, in a number of ways. We feel that the FPEIS establishes an unsupported, unobserved, and unrealistic scenario where G&G activities are projected unrealistically to result in thousands of incidental takes of marine mammals – incidental takes that, in fact, BOEM admits will not actually occur. From this fundamentally flawed and inaccurate approach, the FPEIS develops and analyzes unrealistic mitigation measures to address the effects of a "worst case" hypothetical scenario. This approach is contrary to both the best available scientific information and applicable law. The Associations respectfully recommend that BOEM's Record of Decision (ROD) reflect a revised agency judgment on these issues.

Because G&G activities have little documented impact on marine mammals, the mitigation measures endorsed by Alternative B employ speculation to impose potentially substantial operational and economic burdens on future G&G activities that undermine Congress's clear policy mandate that the Department of Interior facilitate expeditious development of the OCS.

The results of our detailed review of the FPEIS are presented in Appendix 1 attached to this letter, but we have included an overview of the key points contained in the appendix:

1. The FPEIS and future permitting decisions must consider the statutory and environmental context of G&G activities, including the OCSLA. Geological and geophysical activities are critical to the expedited development of OCS resources and the national economic and energy policy goals mandated by OCSLA. The FPEIS omits and undermines much of the critical substantive context and plain congressional directives for the G&G activities analyzed, and it also fails to adequately consider the critical importance of G&G data to OCS development and to the reduction of risks. The ROD that will be prepared based on the FPEIS must consider all relevant factors in balancing the importance of the activities to be permitted, which are critical to the essential purpose of OCSLA.

2. The FPEIS does not incorporate all of the best available science. BOEM discounts observational data that contradict its modeled quantification of G&G impacts and instead relies on unrealistic assumptions regarding sound exposure that are not supported by the best science currently available.

3. Alternative B encourages BOEM to impose unnecessary, arbitrary, and impracticable mitigation measures lacking scientific justification, including the following:

- The FPEIS's expansion of the exclusion zone – compounded by the extension of the shutdown requirement to delphinids – will significantly increase the number of array shutdowns required during a seismic survey, and thereby substantially impact the economics and operations of conducting a seismic survey in the Atlantic. The establishment of a 500-meter minimum is an arbitrary departure from BOEM's rationale for amending the exclusion zone provision. Because BOEM justifies the new exclusion zone provision on the modeled footprint of the individual array's characteristics and site-specific ambient noise conditions, the exclusion zone should always be based upon the modeled output of the array, even if the modeled output results in an exclusion zone of less than 500 meters.
- The FPEIS extends the visual monitoring period for ramp-up of the airgun array – both prior to beginning the survey and after a shutdown – from 30 minutes to 60 minutes. The extension of the visual monitoring period compounds the other operational difficulties Alternative B imposes on seismic surveys. The FPEIS itself offers no justification for the extension of the visual monitoring period.
- The FPEIS extends shutdown requirement to include delphinids. Both the Associations' 2012 DPEIS comments, and BOEM's approval of past seismic survey applications illustrate that extending the shutdown requirement to delphinids is not scientifically justified because delphinids are mid-frequency hearing specialists, with an effective hearing range largely outside of the low frequency range characteristic of airgun arrays. Implementation of this proposed measure will substantially increase the number of shutdowns with no proven environmental benefit.
- The proposed geographic separation between simultaneous seismic airgun surveys is scientifically unsupported. Because the separation distance rests on NMFS's exposure criteria for Level B takes, it suffers from the same flaws as NMFS's thresholds (most notably that the thresholds do not represent the best available science). In addition, this measure is not included in the NMFS Biological Opinion and BOEM offers no evidence to support its underlying assumption that marine mammals would utilize the "corridor" that the separation requirement is designed to create.

4. The Expanded Time-Area Closure provisions for North Atlantic Right Whales lack sufficient basis in existing data, and are otherwise unsupported and unjustified. Similarly, the addition of an acoustic buffer zone around closure zones and the inclusion of Dynamic Management Areas ("DMAs") in the FPEIS are unsupported by the science. The fact that DMAs and acoustic buffer zone mitigations were not included in the Draft EIS has precluded the opportunity for public evaluation and comment.

5. The FPEIS proposes unprecedented observation and shutdown requirements for High Resolution Geophysical (HRG) activities that mimic closely those required of seismic surveys, despite the fact they are significantly different in many ways.

In addition, we note that the FPEIS incorporates the recently published NMFS-OPR-49, *National Standards for Protected Species Observers and Data Management: A Model Using Geological and Geophysical Surveys* ("Observer Standards"). The Associations recently sent a letter to agency staff regarding changes that we would like to see incorporated into the Observer Standards and we have included that letter as Attachment A in our comments on the FPEIS.

The Associations feel that BOEM has failed to provide a reasoned justification for choosing Alternative B as the preferred alternative. While BOEM justifies Alternative B as providing the "highest practicable" level of mitigation measures, it is not required to make its selection based on this standard at the expense of other valid concerns necessary for achieving balance as required under OCSLA. Moreover, many of the mitigation measures recommended in the FPEIS are infeasible, will impose serious burdens on industry, may discourage exploration of the Atlantic, and will result in no benefits to protected species because they address unreal and unsupported effects. The Associations support mitigation measures that are based on the best available science, consistent with existing practices that are proven to be effective, and are operationally feasible. However, we cannot support mitigation measures with no basis in fact or science, that address effects that have not been observed, and will result in less exploration of the OCS.

The Associations appreciate the opportunity to comment on the FPEIS. Although we support BOEM's plan to authorize exploratory activities on the Atlantic OCS, there are a number of aspects of the PEIS that are not supported by science or by law, or are otherwise infeasible. Of the Alternatives presented in the FPEIS, Alternative A presents the option that is most supported by the best available science and applicable law. However, the Associations would support BOEM's adoption of Alternative B so long as <u>all</u> of the modifications suggested in separate comments to the FPEIS submitted by the IAGC (see Attachment B) are incorporated into the ROD. All of these suggested modifications are within the scope of the analyses contained in the PEIS. *See Great Old Broads for Wilderness v. Kimbell*, 709 F.3d 836, 854-55 (9th Cir. 2013) (modified alternative in ROD upheld because all relevant impacts analyzed in NEPA document); *see also W. Watersheds Project v. BLM*, 721 F.3d 1264, 1277-78 (10th Cir. 2013) (same).

We appreciate your consideration of our comments and sincerely hope that BOEM will prepare a ROD that addresses our concerns. Further, we hope that the ROD will be issued as soon as possible so that much needed seismic surveys in the Atlantic can be initiated. Should you have any questions please contact Andy Radford at (202)682-8584 or radforda@api.org.

Sincerely,

Andy Radford
American Petroleum Institute

Karen St. John

International Association of Geophysical Contractors

Jeffrey Vorberger
National Ocean Industries Association

Appendix 1

Comments of the American Petroleum Institute, International Association of Geophysical Contractors, and National Ocean Industries Association

API, IAGC, and NOIA (collectively, "the Associations") respectfully request that BOEM revise the FPEIS to effectuate the purposes of the Outer Continental Shelf Lands Act (OCSLA) and the agency's obligations under the National Environmental Policy Act (NEPA). For the reasons set forth below, in the accompanying documents, and in prior comments to BOEM, the Associations believe the FPEIS's selection of Alternative B as the preferred alternative violates BOEM's obligations under NEPA and OCSLA. Because G&G activities have little documented impact on marine mammals, the mitigation measures endorsed by Alternative B employ speculation to impose potentially prohibitive operational and economic burdens on future G&G activities that undermine Congress's clear policy mandate that the Department of Interior facilitate expeditious development of the OCS.

Of the Alternatives presented in the FPEIS, Alternative A presents the option that is most supported by the best available science and applicable law. However, the Associations would support BOEM's adoption of Alternative B so long as <u>all</u> of the modifications suggested in separate comments to the FPEIS submitted by the IAGC (see Attachment B) are incorporated into the ROD. All of these suggested modifications are within the scope of the analyses contained in the PEIS. *See Great Old Broads for Wilderness v. Kimbell*, 709 F.3d 836, 854-55 (9th Cir. 2013) (modified alternative in ROD upheld because all relevant impacts analyzed in NEPA document); *see also W. Watersheds Project v. BLM*, 721 F.3d 1264, 1277-78 (10th Cir. 2013) (same).

I. The FPEIS Must Consider the Statutory and Environmental Context of G&G Activities.

NEPA is a purely procedural statute that "does not mandate particular results, but simply prescribes the necessary process." *Robertson v. Methow Valley Citizens Council*, 490 U.S. 332, 350 (1989). "If the adverse environmental effects of the proposed action are adequately identified and evaluated, the agency is not constrained by NEPA from deciding that other values outweigh the environmental costs." *Id*. *See also Utahns for Better Transportation v. U.S. Dep't of Transportation*, 305 F.3d 1152, 1162–63 (10th Cir. 2002) ("[A]gencies are not required to elevate environmental concerns over other valid concerns"). Because NEPA itself provides no substantive guide for consideration of the underlying action—here, the conduct of G&G activities—the "statutory context" of the underlying action must inform the analysis of costs and benefits in an EIS. *See, e.g., League of Wilderness Defenders—Blue Mountains Biodiversity Project v. U.S. Forest Serv*., 689 F.3d 1060, 1070 (9th Cir. 2012).

Consideration of the statutory context informs an entire EIS. For example, "the goals of an action delimit the universe of the action's reasonable alternatives." *City of Alexandria, Va. v. Slater*, 198 F.3d 862, 867 (D.C. Cir. 1999) (quotation omitted). *See also, e.g., Kootenai Tribe of Idaho v. Veneman*, 313 F.3d 1094, 1121 (9th Cir. 2002) (Forest Service "not required under NEPA to consider alternatives . . . that were inconsistent with its basic policy objectives").

Indeed, an agency may eliminate both alternatives and mitigation measures that do not meet the purposes and needs of a project. *See Biodiversity Conservation Alliance v. BLM*, 608 F.3d 709, 715 (10th Cir. 2010). And the goals must be "heavily influenced by the agency's consideration of the views of Congress, expressed, to the extent the agency can determine them, in the agency's statutory authorization act, as well as in other congressional directives." *Natural Resources Defense Council, Inc. v. Pena*, 972 F. Supp. 9, 18 (D.D.C. 1997) (quotation omitted).

As set forth below, the FPEIS omits and undermines much of the critical substantive context and plain congressional directives for the G&G activities analyzed.

A. G&G Activities Are Critical to the Expedited Development of OCS Resources Mandated by OCSLA.

"Where an action is taken pursuant to a specific statute, the statutory objectives of the project serve as a guide by which to determine the reasonableness of objectives outlined in an EIS." *Westlands Water District v. U.S. Dep't of the Interior*, 376 F.3d 853, 866 (9th Cir. 2004). Here, OCSLA provides the specific statutory authorization of G&G activities. *See* 43 U.S.C. § 1340. While Chapter 1.4.2 of the FPEIS defines the purpose and need of G&G activities with reference to development of "oil and gas reserves," BOEM's generalized discussion of purpose neglects the strong statutory objectives Congress identified in OCSLA. *See* FPEIS at 1-9. That omission is critical.

Congress enacted OCSLA to promote and ensure the "expedited exploration and development of the [OCS] in order 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); *see also id*. § 1332(3) (the OCS "should be made available for expeditious and orderly development, subject to environmental safeguards, in a manner which is consistent with the maintenance of competition and other national needs"). Indeed, Congress specified that it wished to "make [OCS] resources available to meet the Nation's energy needs as rapidly as possible." *Id*. § 1802(2)(A). OCSLA accordingly "has an objective—the expeditious development of OCS resources . . . ." *California v. Watt*, 668 F.2d 1290, 1316 (D.C. Cir. 1981). Because "[t]he first stated purpose of the Act . . . is to establish procedures to expedite exploration and development of the OCS," OCSLA's remaining purposes primarily concern measures to eliminate or minimize the risks attendant to that exploration and development. Several of the purposes, in fact, candidly recognize that some degree of adverse impact is inevitable." *Id.*[1] *Cf.* Executive Order 13212 (May 18, 2001) (directing that "executive departments and agencies . . . shall take appropriate actions, to the extent consistent with applicable law, to expedite projects that will increase the production, transmission, or conservation of energy").

While the FPEIS concedes that G&G activities generate data that contribute to "informed" and "orderly" development decisions by industry and Government, *see* FPEIS at 1-8–1-9; *see also* FPEIS at 3-3 (noting importance of G&G data), BOEM's choice of Alternative B undercuts the

[1] The FPEIS concludes that the majority of impacts from the proposed G&G activities will be "negligible" or "minor." *See* FPEIS at x–xiv (summarizing anticipated impacts from Alternative A).

critical importance of G&G activities to expeditious OCS development and, thus, to OCSLA's animating purpose. And Alternative B endorses restrictive mitigation measures despite the generally "minor" impacts of G&G activities. As further explained *infra*, the operational and practical limitations imposed by the FPEIS threaten the viability of critical G&G activities and thereby directly undermine Congress's stated purpose to "promote the swift, orderly and efficient exploration" of OCS oil and gas resources.[2]

B. The FPEIS Fails To Adequately Consider the Critical Importance of G&G Activities to Development of OCS Oil and Gas Resources, and To the Reduction of Risks to Environmental Resources from OCS Development.

The FPEIS candidly acknowledges that "[t]he G&G surveys acquired during the period when Atlantic oil and gas leasing took place in the 1970's and 1980's have been eclipsed by newer instrumentation, technology, and data processing that make seismic data of that time period inferior," FPEIS at 1-9, and existing estimates of energy reserves in the Atlantic woefully out-of-date. Rather, "[n]ew surveys conducted with current technology would significantly improve the ability of both industry and Government predict where, and in what quantity, fossil fuel hydrocarbons are more likely to be found," and "allow the Government to place a fair and appropriate value on these resources for the Nation." FPEIS at 2-58.

Moreover, as the FPEIS concedes, "using . . . vintage surveys to optimally site an exploratory well or a well field, or to interpret the nature of formation fluids or gases, is generally not reasonable." FPEIS at 2-57. Having the most accurate and state-of-the-art seismic data for use in drilling and production activities reduces the environmental impact of exploration and production, by significantly reducing the number of unsuccessful wells and, thus, reducing the potential environmental impact of each well so avoided. As technology continues to advance, the seismic 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 the most modern technology to make improved seismic evaluations.

Indeed, vast improvements in geophysical imaging technologies in recent years now afford the oil and gas industry significant precision in subsurface imaging, which reduces environmental risks during drilling operations. For example, subsurface imaging provides a key input to help predict hazardous over-pressurized zones in a reservoir and thus allows an operator to better design a well to minimize its associated types and levels of risk.

G&G activities thus provide environmental benefits in the conduct of the expeditious OCS oil and gas development activities mandated by OCSLA.[3] The FPEIS, however, fails to consider the environmental benefits of improved G&G activities. Rather, BOEM disregards such benefits

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2 H.R. Rep. No. 95-590, at 8, *reprinted in* 1978 U.S.C.C.A.N. 1450, 1460.

3 *Cf.* Executive Order 12866, § 1(b)(6) (Sept. 30, 1993) ("Each agency shall assess both the costs and the benefits of the intended regulation and . . . propose or adopt a regulation only upon a reasoned determination that the benefits of the intended regulation justify its costs.").

as "outside of the scope of the NEPA document." FPEIS Vol. III, Table L-6 at L-116 (response to comments of API, IAGC, and NOIA).

Contrary to BOEM's narrow view of G&G activities, "[t]he purpose of NEPA is to require agencies to consider ***environmentally significant aspects*** of a proposed action." *Utahns for Better Transportation v. U.S. Dep't of Transportation*, 305 F.3d 1152, 1162 (10th Cir. 2002) (emphasis added). *Cf. Utahns*, 305 F.3d. at 1174 ("An EIS must analyze not only the direct impacts of a proposed action, but also the indirect impacts of past, present, and reasonably foreseeable future actions . . . ."). By ignoring the environmental benefits of G&G activities to anticipated oil and gas development activities, the FPEIS fails to "adequately set[] forth sufficient information to allow the decisionmaker to consider alternatives and make a reasoned decision after balancing the risks of harm to the environment against the ***benefits*** of the proposed action." *Friends of the Boundary Waters Wilderness v. Dombeck*, 164 F.3d 1115, 1128 (8th Cir. 1999) (emphasis added). *See also Coal. for a Livable Westside v. U.S. Postal Serv.*, No. 99-cv-10873, 2000 WL 1264256, at *3 (S.D.N.Y. Sept. 7, 2000) (explaining that an EIS must "assess[]the environmental benefits and detriments of the proposed action").

II. The FPEIS Does Not Incorporate the Best Available Science.

As explained in the Associations' comments on the DPEIS ("2012 DPEIS Comments"), BOEM's scientific analysis must be based upon the best available science. *See* 2012 DPEIS Comments, Appendix 1 at 1 (identifying requirements of NEPA and Executive Order 13563). *See also* 40 C.F.R. § 1502.24 (requiring agency to "insure the professional integrity, including scientific integrity, of the discussions and analyses in the environmental impact statements"); *id.* § 1500.1(b). For the reasons identified in the Associations' 2012 DPEIS Comments, and as further set forth below, the FPEIS does not satisfy BOEM's obligation to use the best available science.

A. BOEM Discounts Marine Mammal Field Observational Data that Undermines its Modeled Quantification of G&G Impacts.

Data accumulated from Marine Mammal Observers demonstrate the absence of documented effects—in particular, injury or death to an animal—of seismic surveys on marine mammals. Nevertheless, the FPEIS estimates an enormous number of Level A and Level B takes from G&G activities in the Atlantic. Relying on the sound exposure criteria developed by the National Marine Fisheries Service (NMFS), the FPEIS predicts, for example, up to nearly 12,000 Level A takes of bottlenose dolphins per year from seismic survey operations, and over 1.1 million Level B takes. *See, e.g.*, FPEIS at xi. Because such estimates bear no relation to the minimal impacts actually observed from seismic survey activities, BOEM has apparently ignored the existing data on actual, observed impacts in derogation of its obligation to utilize the best available science. *Cf. San Juan Citizens Alliance v. Stiles*, No. 08-cv-144, 2010 WL 1780816, at *16 (D. Colo. May 3, 2010) (noting that Forest Service regulation requiring use of "best available science" means agency "cannot ignore existing data" (quotation omitted)); *Turtle Island Restoration Network v. U.S. Dep't of Commerce*, No. 12-cv-594, 2013 WL 4511314, at *22 (D. Hawai'i Aug. 23, 2013) (Under the ESA, "the 'best available data' requirement keeps agencies from ignoring available information."); *The Ecology Ctr., Inc. v. U.S. Forest Serv.*, 451

F.3d 1183, 1194 n.4 (10th Cir. 2006) (looking to meaning of "best available science" under other statutory regimes to inform meaning of requirement in National Forest Management Act).

Rather than rely on observational data, BOEM estimated impacts with a predictive computer model of sound propagation and exposure. *See* FPEIS at 2-17 & Appendices D, E. The FPEIS explains that Acoustic Integration Model (AIM), which is used to estimate takes, as "a 4D, individual-based, Monte Carlo statistical model" that "is by its very nature complex and requires numerous assumptions to predict results . . . ." FPEIS at 4-58. Even with that complexity, AIM does not incorporate animal behaviors, such as avoidance, which likely occur and would likely reduce the estimated number of exposures.

Notably, the D.C. Circuit has cautioned that "although computer modeling is a useful and often essential tool for performing the Herculean labors Congress imposes on administrative agencies, such models, despite their complex design and aura of scientific validity, are at best imperfect and subject to manipulation." *Gas Appliance Mfrs. Ass'n v. Dep't of Energy*, 998 F.2d 1041, 1045 (D.C. Cir. 1993) (quotation and alteration omitted). "Since the accuracy of any computer model hinges on whether the underlying assumptions reflect reality . . . [t]he agency's burden [to demonstrate the reasonableness of a model] becomes heavier when a method of prediction is being relied on to overcome adverse actual test data." *Id*. (quotations and alteration omitted).

Here, BOEM's modeling predicts levels of take that vastly exceed, *see infra*, the observational impact data accumulated by Marine Mammal Observers on survey vessels.[4] Far from supporting the FPEIS, the observed data conflicts with the enormous number of takes predicted by the models. *Cf. Conservation Congress v. U.S. Forest Serv.*, No. 10-17298, 489 F. App'x 151, 153 (9th Cir. June 4, 2012) (recognizing that agency's scientific support may be insufficient where scientific studies indicate the agency's "analysis is outdated or flawed or indicate any scientific information directly undermining" the agency's conclusion (quotation omitted)); *Native Ecosystems Council v. U.S. Forest Serv.*, 418 F.3d 953, 964 (9th Cir. 2005) ("To take the required 'hard look' at a proposed project's effects, an agency may not rely on incorrect assumptions or data in an EIS."). Thus, while a model fails to satisfy NEPA requirements if it "is so oversimplified that the agency's conclusions from it are unreasonable," *Small Refiner Lead Phase-Down Task Force v. U.S. EPA*, 705 F.2d 506, 535 (D.C. Cir. 1983), the FPEIS employs a model with the opposite, but equally fatal, flaw: complication that is not grounded in, and deviates significantly from, existing data.

Given the FPEIS's deviation from observed impact data, BOEM's defense of the FPEIS as providing "a detailed description for each step in the impact assessment process," FPEIS Vol II, Table L-6 at L-109, is non-responsive to the Association's concerns, *compare Montana Wilderness Ass'n v. McAllister*, No. 09-36051, 460 F. App'x 667, 670 (9th Cir. Dec. 1, 2011) (finding agency met its duty to respond to comments where is "adequately responded to the ***substance*** of . . . comments" (emphasis added)), or the agency's NEPA obligations.

[4] One BOEM review of Marine Mammal Observer data, for example, logged a total of 194,273 visual survey hours, with only 125.74 hours of down time attributed to protected species shutdowns. *See* BOEM, *Seismic Survey Mitigation Measures and Marine Mammal Observer Reports*, at 1 (June 2012).

In addition to its deviation from observed impacts, the FPEIS's underlying AIM model suffers from documented weaknesses. In 2006, NMFS initiated an independent peer review of the AIM model. *See* Summary Report: Review of Acoustic Integration Model (AIM), University of Miami Independent System for Peer Review at 1 (Dec. 11, 2006), *available at* http://www.nmfs.noaa.gov/pr/pdfs/permits/lfa_aim_review.pdf. The peer review did not reach a consensus on whether AIM meets the Council for Regulatory Monitoring (CREM) guidelines "since [AIM] is not an application model (but a tool for developing such models)." *Id*. (noting "there was some diversity of opinion"). Rather, the peer review noted "[t]he need for expertise in the use of AIM" as well as "the absence of appropriate uncertainty and sensitivity tests in the current applications of AIM." *Id*. While the peer review agreed that "the use of AIM ***can*** lead to models which will meet CREM guidelines . . ., such models, at this stage, would need to be evaluated on a case-by-case basis (i.e., merely using AIM is not sufficient . . .)." *Id*. (emphasis added). The FPEIS provides no verification that such a case-by-case analysis was undertaken of the use of AIM here. That lack of verification is particularly significant in this case because the peer review further identified the absence of data on "real" animal behavior as a fundamental limitation of AIM, *see id*. at 7–11 (noting "knowledge of marine mammals was identified as the weakest component"), and, as explained above, observed impact data undermines the model's predictions of G&G impacts.

Finally, BOEM's explanation that the sound "propagation models" employed by the FPEIS "have been extensively tested against field measurements," FPEIS Vol. III, Table L-6 at L-111–L-112, is likewise non-responsive to the Associations' concerns. The absence of observed impacts from seismic surveys relates to the sound exposure modeling conducted by BOEM, not the propagation modeling that is limited to determining the ways that sound moves through the ocean (and is an input in the exposure model). The fact that BOEM believes the propagation models are "appropriate" and "considered" acceptable, *see, e.g.*, FPEIS Vol. III, Table L-6 at L-109, fails to respond to the Association's showing that the sound exposure models are scientifically or practically flawed.

B. BOEM Relies on Assumptions Regarding Sound Exposure that Are Not Supported by the Best Available Science.

As explained in the Associations' 2012 DPEIS comments, BOEM's impact analysis improperly equates received sound levels to takes. *See, e.g.,* 2012 DPEIS Comments, Appendix 2 at 10–15. The FPEIS responds simply that the impact analysis is justified because it is (1) "conservative" and (2) based upon exposure criteria developed by NMFS that is beyond BOEM's control. *See, e.g.*, FPEIS Vol. III, Table L-6 at L-113; *id*. at L-111 (stating BOEM "cannot use the Southall criteria as the basis for take estimates because they have not been adopted by NMFS"); *id*. at L-112 (explaining that sound exposure criteria used to estimate take "are based on their acceptance by NMFS"); *id*. at L-114 ("[T]he choice of metric to use to determine takes was made by NMFS."); *id*. at L-118. The former explanation merely demonstrates BOEM's failure to adopt clear or consistent standards, and the latter abdication to NMFS violates BOEM's independent NEPA obligations.

<u>First</u>, the FPEIS simply states that its take estimates are "conservative" and the result of conservative—or "very conservative"—assumptions, "and this conservatism accumulates throughout the analysis." FPEIS at xii, xiii. The bare identification of an accumulated conservatism does not itself justify BOEM's decision to employ such a conservative bias. Indeed, the FPEIS compounds its conservative bias by classifying the impacts of G&G activities on the majority of species as "negligible," but nonetheless choosing the more conservative Alternative B. *See* FPEIS at x–xxv. Yet the FPEIS offers no data as justification; rather, Marine Mammal Observer data indicates little seismic survey impact on marine mammals and provides no support for the FPEIS' conservatism. As the Associations' 2012 DPEIS comments make clear, BOEM's overly conservative impact analysis is exacerbated by BOEM's failure to use consistent or objective standards for assessing the severity of impacts on species, which often conflates "minor," "moderate," and "severe" impacts. *See* 2012 DPEIS Comments, Appendix 2 at 6.[5]

Second, the FPEIS's repeated invocations of NMFS's decisions to justify BOEM's impact analysis runs counter to the best available science on sound exposure impacts and improperly abdicates BOEM's NEPA obligations. As the Associations' demonstrated in their 2012 DPEIS Comments, NMFS's sound exposure criteria for Level A and Level B takes—180 dB re: 1µPA (rms) SPL for the former, 160 dB re: 1µPA (rms) SPL for the latter—improperly rest upon outdated data, *see, e.g.*, *N. Plains Res. Council, Inc. v. Surface Transp. Bd.*, 668 F.3d 1067, 1086–87 (9th Cir. 2011) ("Reliance on data that is too stale to carry the weight assigned to it may be arbitrary and capricious."), and fail to incorporate the more current science on this question developed by the Marine Mammal Noise Exposure Criteria Work Group ("Southall Work Group"), *see, e.g.*, 2012 DPEIS Comments, Appendix 2 at 10.[6]

In contrast to the FPEIS, the Southall Work Group does not subjectively label animal responses to sound as "minor," "moderate," or "severe," but rather uses a nine-point continuum and thirty-four separate types of behavioral responses, and emphasizes "extreme degree of group, species, and individual variability in behavioral responses in various contexts and conditions . . .," (Southall et al. 2007) at 449. With respect to Level A takes, the Southall Work Group recommended an increase in the sound threshold to 230 dB re: 1µPA (rms) SPL, *see id.*, at 442, and supports a more contextual approach to Level B takes, that is wholly absent from the FPEIS. Indeed, the Southall Work Group's analysis of what constitutes a Level B take is substantially more nuanced than the FPEIS's practice equating certain received levels of sound with takes. *See id*. at 447 (noting one must "differentiat[e] brief, minor, biologically unimportant reactions from profound, sustained, and/or biologically meaningful responses related to growth, survival, and reproduction").

While the FPEIS purports to provide analysis based on the Southall Work Group, *see* FPEIS Vol. III, Table L-6 at L-112, that analysis is, at best, incomplete because it is limited to Level A takes, *see, e.g.*, FPEIS at xi. Moreover, BOEM's principal response is that the FPEIS "cannot use the Southall criteria as the basis for take estimates because they have not been adopted by NMFS."

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5 BOEM's lack of objective standards for categorizing effects will also foster arbitrary, and potentially conflicting, decisionmaking in assessing the vague boundaries between "minor," "moderate," and "severe" impacts. *See* 2012 DPEIS Comments, Appendix 2 at 6–7.

6 Other reports on marine sound impacts released after the Southall Work Group, such as J.J. Finneran & A.K. Jenkins, *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis* (2012), do not consider "[t]he criteria and thresholds for . . . airguns," *id*. at 2.

FPEIS Vol. III, Table L-6 at L-111; *see also, e.g.*, *id*. at L-112 (explaining that sound exposure criteria used to estimate take "are based on their acceptance by NMFS").[7] Such abdication to NMFS on an issue central to assessing the impacts of G&G activities falls short of BOEM's obligation to take a "hard look" at the environmental consequences of the proposed activities. "One agency cannot rely on another's examination of environmental effects under NEPA." *S. Or. Citizens Against Toxic Sprays, Inc. v. Clark*, 720 F.2d 1475, 1480 (9th Cir. 1983) (rejecting Interior Department's reliance on EPA decision with respect to herbicide) (quotation omitted). Rather, BOEM must "assess independently," *id*., the environmental effects of the proposed actions it considers.

C. BOEM's Impact Analysis Rests on Speculation.

Because the FPEIS ignores existing data demonstrating the absence of significant impacts—in particular, a lack of injuries—from G&G activities, and relies on thinly supported or outdated sound exposure assumptions, *see supra*, the FPEIS's impact analysis ultimately provides little more than speculation about potential adverse effects of seismic surveys without regard to the probabilities of either occurrence or scope of such effects. Even with its flawed assumptions, moreover, the FPEIS concedes that the impact analysis—and the resulting choices regarding required mitigation—rests on predicted "possibility" of harm. *See, e.g.*, FPEIS at 2-20 (explaining that models predicted "possibility" of Level A takes, but did not take into account proposed mitigation measures); *id*. at 2-41 (explaining choice of Alternative B's Brevard County time-area closure to "reduce the possibility of temporarily displacing breeding and nesting"). Yet BOEM has no obligation to assess such mere possibilities of harm. *See, e.g., S. Fork Band Council of W. Shoshone of Nevada v. Dep't of Interior*, 588 F.3d 718, 727 (9th Cir. 2009); *Wyoming v. U.S. Dep't of Agriculture*, 661 F.3d 1209, 1253 (10th Cir. 2011) (explaining that an agency is "not required to consider 'speculative' impacts"); *Sierra Club v. Hodel*, 544 F.2d 1036, 1039 (9th Cir. 1976).

III. Alternative B Encourages BOEM to Impose Unnecessary, Vague, and Impracticable Mitigation Measures.

The overarching errors in the FPEIS identified *supra* greatly overstate the impacts of G&G activities and, as a consequence, greatly overstate the alleged necessity for mitigation measures generally, and for the additional mitigation measures in BOEM's preferred Alternative B in particular. By comparison, the FPEIS concludes that "the impacts associated with Alternative A would result in a ***minor*** incremental increase in underwater noise and a ***minor*** increase [in] impacts to marine mammals under the cumulative scenario." FPEIS at 4-75 (emphases added). In light of the FPEIS's overstatement of G&G impacts and the admittedly "minor" effect of G&G activities under Alternative A, BOEM's choice of Alternative B is unjustified.

Moreover, viewed individually, the mitigation measures proposed in Alternative B are likewise unnecessary in light of the best available science, vaguely phrased in a manner that encourages arbitrary enforcement, and/or impose impractical operational burdens that threaten to

[7] The FPEIS similarly attributes BOEM's failure to consider the frequency weighting advocated by recent studies, *see* (Southall et al., 2007), to NMFS's policy. *See* FPEIS Vol. III, Table L-6 at L119.

significantly limit seismic surveying that is necessary to meet OCSLA's goals, and may even threaten the overall viability of G&G activities in the Atlantic. Further, The PEIS incorporates significant new mitigation measures including dynamic management areas, acoustic buffer zones around closure areas, and a doubling of the time period required for observation of the exclusion zone before start-up is authorized. There has been insufficient justification and no opportunity for public comment; therefore, these mitigations should not be adopted.

A. The Proposed Seismic Survey Protocol.

Joint NTL 2012-G02 currently defines the current standard, "Seismic Survey Mitigation Measures and Protected Species Observer Program," in the Gulf of Mexico where the bulk of seismic surveys are conducted in U.S. waters.[8] It has proven effective, and is therefore the best baseline for assessing proposed mitigation for G&G activities. Among other things, Joint NTL 2012-G02 (1) establishes a 500 meter exclusion zone surrounding the center of an airgun array; (2) permits the array to recommence operations only following a 30-minute visual clearance of the exclusion zone; and (3) requires the array to shut down if visual monitoring reveals a marine mammal (excluding dolphins) or sea turtle within the exclusion zone. The monitoring is conducted by a visual observer who has successfully completed a protected species observer training course.

The FPEIS proposes unjustified and unjustifiable changes to the baseline provisions of Joint NTL 2012-G02.

First, the FPEIS provides that the exclusion zone "shall be calculated independently and shall be based on the configuration of the array and the ambient acoustic environment, but shall not have a radius of less than 500 m . . . ." FPEIS at 2-10. In contrast to the current, fixed 500 meter exclusion zone, the FPEIS's proposal would result in enormously expanded exclusion zones. Indeed, the FPEIS calculates the exclusion zone—based upon NMFS's 180 dB re: 1µPA (rms) SPL criteria for Level A takes—that would be required in particular scenarios based on the size of the airgun array, resulting in exclusion zone radii ranging from 800 to over 2100 meters. *See* FPEIS Vol. II, Table D-22. The latter results in a spatial area more than 17 times larger than required by Joint NTL. 2012-G02. More recent scientific research, however, undercuts this expansion; using the Southall Work Group's Level A sound threshold of 230 dB re: 1µPA (rms) SPL, (Southall et al. 2007) at 449, would in many instances result in an exclusion zone less than 500 meters.

The FPEIS's expansion of the exclusion zone—compounded by the extension of the shutdown requirement to delphinids, *see infra*—will significantly increase the number of array shutdowns required during a seismic survey, and thereby threaten the economic and operational feasibility of conducting a seismic survey in the Atlantic. Among other things, survey vessels continue to move along their tracklines even after the airgun array is shutdown. Once the exclusion zone has been visually cleared of marine mammals for, under the FPEIS, at least 60 minutes, the array can resume operations. To acquire seismic data for the region between the shutdown and start-up

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8 U.S. Dep't of the Interior, Joint NTL No. 2012-G02, "Implementation of Seismic Survey Mitigation Measures and Protected Species Observer Program, *available at* http://www.boem.gov/Regulations/Notices-To-Lessees/2012/2012-JOINT-G02-pdf.aspx.

positions of the array requires maneuvering the seismic survey vessels (and miles of trailing streamers) back to the shutdown position. An increase in the number of shutdowns thus increases downtime and wasteful maneuvering. Because a survey's data quality is also tied to acquiring data along specific tracklines, by breaking acquisition along a trackline, a shutdown potentially impairs data quality and prolongs the length of the survey, increasing exposure of human health, safety and environmental risks. *See, e.g.*, Site-Specific Environmental Assessment of G&G Survey Application No. L11-020 (Jan. 23, 2012), at 7–8.

The FPEIS estimates that over 26,000 Level A takes would occur—thus indicating the number of shutdown events that would be necessary assuming perfect observation of species in the exclusion zone—in 2016 alone. *See* FPEIS at Tables-43.[9] That figure dwarfs the 55 shutdowns that are typically caused by whale sightings in the Gulf of Mexico (baseline) in a year. Yet the FPEIS threatens the level or viability of seismic surveying in the Atlantic based solely on its scientifically flawed assessment of impacts, *see supra*, and expansion of the shutdown requirement to include delphinids, *see infra*. For example, in the Gulf of Mexico, the average shutdown lasts for 58 minutes, *see, e.g.*, BOEM, *Seismic Survey Mitigation Measures and Marine Mammal Observer Reports*, at 1 (June 2012), which the FPEIS would extend by at least 30 minutes by increasing the visual monitoring period following a shutdown from 30 to 60 minutes. *See infra*. Multiplying a rough 1.5-hour average shutdown by 26,000 shutdowns would yield roughly 39,000 hours of shutdowns, or approximately 1625 days. Because the typical seismic survey operation costs roughly $1.5 million per day, the total potential costs arising from the FPEIS's assumptions equal a staggering $2.5 billion.

BOEM's revision of the exclusion zone is, moreover, incomplete. While the FPEIS requires a survey operator to model its array in order to calculate the proper exclusion zone, the FPEIS also mandates that the zone "shall not have a radius of less than 500 m." FPEIS at 2-10. The establishment of a 500-meter floor is an arbitrary departure from BOEM's rationale for amending the exclusion zone provision. Because BOEM justifies the new exclusion zone provision on the modeled footprint of the individual array's sound, the exclusion zone should always be based upon the modeled output of the array, even if the modeled output results in an exclusion zone of less than 500 meters. *See also supra*. In other words, the FPEIS must be consistent in its reliance on calculations of the exclusion zone and follow BOEM's own justification to its logical conclusion.

Notably, in response to the Associations' 2012 DPEIS Comments, BOEM acknowledged the need for logical consistency in calculating the size of the exclusion zone by revising the FPEIS to acknowledge that "the modeling could increase or decrease the size of the exclusion zone." FPEIS Vol. III, Table L-6 at L021. While the revision properly acknowledges the logic of decreasing an exclusion zone on the basis of the array's modeling, BOEM has not provided a justification for its failure to extend this logic below a 500-meter exclusion zone radius. *See, e.g.*, *Business Roundtable v. SEC*, 647 F.3d 1144, 1153 (D.C. Cir. 2011) (holding agency action arbitrary where discussion of issue was "internally inconsistent").

Second, the FPEIS extends the visual monitoring period for ramp-up of the airgun array—both prior to beginning the survey and after a shutdown—from 30 minutes to 60 minutes. *See* FPEIS

[9] The Associations aggregated the estimated takes presented in the FPEIS at Tables-43.

at 2-10–2-11. The extension of the visual monitoring period compounds the other operational difficulties Alternative B imposes on seismic surveys.

The FPEIS itself offers no justification for the extension of the visual monitoring period. *See, e.g.*, FPEIS at 2-9–2-12; FPEIS Vol. III, Appendix C. Rather, BOEM's revision appears to be based on a comment from the Georgia Department of Natural Resources that existing "visual detection mitigation techniques for right whales are inadequate due to the animal's ability to lie just under the surface and remain undetected." FPEIS Vol. III, Table L-6 at L-71–L-72. Despite the specific context of the question—related to right whales—BOEM nevertheless created a broad 60 minute monitoring period "to assist visual observers locate marine mammals during their normal dive (or subsurface rest) frequency." *Id.* BOEM did not provide any evidence demonstrating (or even indicating) that the existing 30-minute period is inadequate to identify any marine mammal.[10] Indeed, in response to the Georgia Department of Natural Resources, BOEM stated generally "[t]hough right whales may lie below the surface for periods of time, it is expected that trained PSOs would spot exhalation plumes and surface disturbances." FPEIS Vol. III, Table L-6 at L-71–L-72.

Third, the FPEIS extends NTL 2012-G02's shutdown requirement, which presently applies only to whales, to include delphinids. *See* FPEIS at 2-11. Both the Associations' 2012 DPEIS Comments, *see, e.g.*, 2012 DPEIS Comments, Appendix 2 at 20–21, and BOEM's approval of past seismic survey applications, *see, e.g.*, *id.*, Appendix 1 at 6 n.9, illustrate that extending the shutdown requirement to delphinids is not scientifically justified because delphinids are mid-frequency specialists, with an effective hearing range largely outside of the low frequency range characteristic of airgun arrays. *E.g.* (Southall, et. al 2007) at 430–31. In response to the Associations' 2012 DPEIS Comments, BOEM again explained this provision based on NMFS's outdated sound exposure criteria. *See* FPEIS Vol. III, Table L-6 at L-122. Further, the illogical contradiction that dolphins that do not happen to bow ride require a different mitigation strategy makes no sense scientifically. Despite the lack of scientific justification, the FPEIS's extension of the shutdown requirement will vastly increase the likely number of shutdowns, with tens of thousands of shutdown events predicted for dolphins alone. *See* FPEIS, Table 4-10 at Tables 43.

Moreover, bow-riding of seismic survey vessels—a normal behavior seen with dolphins—further demonstrates the lack of injurious impact (or take) from seismic airguns. The FPEIS fails to analyze recent research into harbor porpoise (Linnenschmidt et al, 2012) and the bottlenose dolphin (Li et al, 2011, 2012) that suggest hearing control may apply to a number of different species of delphinids and cetaceans and that the animals have the ability to reduce their hearing sensitivity. The Associations appreciate BOEM's attempts, through creation of a bow-riding exception to shutdown requirements, to recognize the commonality of bow-riding and ameliorate the danger of unnecessary shutdowns brought-on by a dolphin's affirmative approach of a survey in order to bow-ride. *See, e.g.*, FPEIS Vol. III, Table L-6 at L-122. The proposed exception, however, offers little protection from unnecessary shutdowns. That exception provides:

[10] Likewise, NMFS's Biological Opinion for Programmatic G&G Activities in the Mid- and South Atlantic Planning Areas from 2013 to 2020 (July 19, 2013) simply recites the mitigation included in the FPEIS, *see* FPEIS, Appendix A, without justification for lengthening the visual monitoring period.

> Shutdown would not be required for delphinids approaching the vessel (or vessel's towed equipment) that indicates a "voluntary approach" on behalf of the delphinid. A "voluntary approach" is defined as a clear and ***purposeful*** approach toward the vessel by the delphinid(s) with a speed and vector that indicates that the delphinid(s) is approaching the vessels and remains near the vessel or towed equipment. ***The intent of the delphinid(s) would be subject to the determination of the PSO***. If the PSO determines that the delphinid(s) is actively trying to avoid the vessel or the towed equipment, the acoustic sources must be immediately [shutdown] as per his/her instruction.

FPEIS Vol. III, Appendix C at C-21 (emphases added). Even if implemented to preclude shutdowns for all purposefully approaching delphinids, BOEM has estimated that only roughly one-third of dolphins within 500 meters of a survey vessel exhibit bow-riding behavior, which still leaves many thousands of potential (and scientifically unjustified) shutdowns on account of delphinids.

However, the Associations doubt that the bow-riding exception could be implemented appropriately to preclude all purposeful approaches. Because a shutdown must occur upon a delphinid's entry into the exclusion zone, the determination as to the delphinid's "intent" must be made at a great distance—a distance the FPEIS now potentially extends up to more than 2000 meters. *See supra*. The decision as to the delphinid's intent, moreover, is left wholly to the subjective discretion of PSOs who (1) are likely to err on the side of precaution and order a shutdown when it does not prove necessary, and (2) are subject to training on NMFS's 2013 National Standards for a Protected Species Observer and Data Management Program, *see* FPEIS at 2-10, which may not be consistent with the best available science and technology, clearly written, transparently implemented, or fully informed by the public, *see* Attachment A.[11]

B. The Proposed Geographic Separation Between Simultaneous Seismic Airgun Surveys.

BOEM's choice of Alternative B "may establish a 40-km (25-mi) geographic separation between the sources of simultaneously operating seismic airgun surveys." FPEIS at 2-37. The FPEIS explains the creation of this separation requirement "to provide a corridor between vessels conducting simultaneous surveys where airgun noise is ***below Level B thresholds*** and approaching ambient levels such that animals ***may pass through*** rather than traveling larger distances to go around the survey vessels." *Id*. (emphases added). The FPEIS's justification, however, is scientifically unsupported. First, because the separation distance rests on NMFS's 160 dB re: 1µPA (rms) SPL exposure criteria for Level B takes,[12] it suffers from the same flaws

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[11] Additionally, because the exception rests upon a PSO's discretionary assessment of a delphinid's subjective "intent" around and within the exclusion zone—as a proxy for the absence of harm to the animal—the PSO should have similar discretion to assess the intent of—and prevent a shutdown upon the purposeful approach of—other marine mammals.

[12] The absence of this measure from the Biological Opinion, *see* FPEIS, Appendix A, further undermines BOEM's reliance on NMFS to support a 40-km separation.

as NMFS's thresholds. *See supra*.[13] In addition, BOEM offers no evidence to support its underlying assumption that marine mammals would utilize the "corridor" that the separation requirement is designed to create.

The proposed 40-km separation is also inconsistent with BOEM practice in the Arctic. The 2006 Final Programmatic Environmental Assessment for Arctic Ocean OCS Seismic Surveys provided for a 24 kilometer separation between the seismic source vessels of simultaneous surveys.[14] Thus, the FPEIS imposes nearly twice the separation distance even though the physical environment of the Arctic—with its relatively shallow depth, rocky bottoms, and prevalent sea ice—results in greater sound propagation.

BOEM acknowledges that, even if seismic sound can theoretically propagate great distances, "it is unknown if detection of sound at these distances has any effect on marine mammals or other marine species." FPEIS at 2-38. Rather than question the propriety of its proposed 40-km separation distance, however, BOEM's sole concession to this scientific uncertainty is to claim the agency "will consider the value of this measure at the site-specific NEPA and environmental analyses level, as well as any new information available at that time. BOEM ***may not*** apply this specific mitigation measure programmatically." *Id*. (emphasis added). Setting aside the possibility that BOEM "may" actually employ the separation measure programmatically, the FPEIS does not explain how the uncertainty as to whether impacts occur at great distances can be resolved on site-specific information.

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[13] BOEM's secondary reliance on the assertion that "in some circumstances, airgun noise can be detected at great distances from the sound source, such as across ocean basins (Neukirk et al., 2012)," FPEIS at 2-38, is no more availing. The FPEIS does not identify any sections of the Mid- or South Atlantic planning areas that meet the specific "circumstances" of the cited study; nor does the FPEIS account for the rate of energy loss (*i.e.*, transmission loss) in specific propagating conditions in the Atlantic.

[14] Mineral Management Service, Final Programmatic Environmental Assessment for Arctic Ocean OCS Seismic Surveys - 2006 (OCS EIS/EA MMS 2006-038), at p. 235.

C. The Expanded Time-Area Closure for North Atlantic Right Whales (NARW).

Alternative B prohibits seismic airgun surveys in (1) the Mid-Atlantic and South Atlantic Seasonal Management Areas (SMAs) from November 1 to April 30, *see* FPEIS, Appendix C at C-16; (2) the NARW critical habitat area from November 15 to April 15, *see id*.; and (3) in a continuous 37 km-wide zone extending from Delaware Bay to the southern limit of the programmatic area, *see id*. at C-32.[15] In addition, "G&G surveys using airguns would not be allowed in [an] active" Dynamic Management Area" (DMA) created by NMFS based on "a reliable sighting of a NARW." FPEIS, Appendix C at C-36. And surveys conducted outside of the closure areas "would be required to remain such distance that received levels at those boundaries do not exceed" 160 db re: 1µPA (rms). *Id*. The time-area closure provisions lack sufficient basis in existing data, and are otherwise unsupported and unjustified.

First, according to the FPEIS, "[t]he purpose of the expanded time-area closure," through implementation of a 37 km-wide zone extending south from Delaware Bay, "is to prevent impacts to NARWs along their entire migration route and calving and nursery grounds." *Id*. at C-32. While the Associations share BOEM's concern for the health of the NARW population, as the Associations' 2012 DPEIS Comments demonstrate, there are no documented injuries, deaths, or significant disturbances to NARWs from airguns (even though the NARW is among the most studied species of whale). *See* 2012 DPEIS Comments at 5; *id*., Appendix 2 at 3, 17–18. Rather, the primary documented risks to the NARW population are vessel strikes and fishing gear entanglement. *See id*. at 5 & nn. 4, 5. Yet, while the NARW is particularly susceptible to lethal strikes from vessels exceeding 10 knots, seismic survey vessels—operating to carefully gather data—travel only at 4 to 5 knots (or half the mandatory speed limit under the NARW ship strike reduction rule (50 C.F.R. § 224.105)), and would have visual observers on board. *See* 2012 DPEIS Comments, Appendix 2 at 18. No closure for the NARW is therefore warranted.[16]

Although the Associations raised these concerns in their 2012 DPEIS Comments, BOEM's subsequent explanation missed the point of the Associations' comments and was therefore non-responsive. BOEM stated that "the potential for vessel strikes was not the main reason for proposing the closures . . . ." FPEIS Vol. III, Table L-6 at L-109–L-110. The Associations did not contend that BOEM based the closures on vessel strikes or the applicability of the NARW vessel strike rule. Instead, the Associations have shown that vessel strikes—not the sound from airguns[17]—is the primary, known danger to the NARW, and that this primary danger is largely inapplicable to seismic surveys that operate at reduced speeds, *cf. Utahns*, 305 F.3d at 1180 (finding that agency improperly "ignored the primary concern" of commenters on the project),

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15 Alternative A includes only the closures in the SMAs and critical habitat areas. *See* FPEIS, Appendix C at C-16.

16 BOEM's response that it "would not be prudent based on the endangered status of these whales," *see* FPEIS Vol. III, Table L-6 at L-107, to issue an FPEIS without a time-area closure for the NARW is improperly conclusory in light of other species that do not similarly trigger a closure.

17 To the extent the closure is "based" on impacts from acoustic sources, FPEIS Vol. III, Table L-6 at L-109–L-110, there is no documented evidence of any such impact.

and only "represent a small percentage (i.e., 1.5–2.9%) of the projected vessel activity" in the area of interest, FPEIS at 3-52.

Moreover, Alternative B's expansion nearly doubles the size of the closure area proposed in Alternative A. *See* FPEIS, Appendix C at C-16, C-32. Yet the FPEIS predicts, at most, only a 13 percentage point reduction in incidental takes of NARWs. *See*, *e.g.*, FPEIS at 2-66. Not only does BOEM fail to explain the differential between the expanded closure area and the predicted benefit, the FPEIS concedes that "incidental take was not modeled for Alternative B," and that the alleged benefit of doubling the time-area closures was only "estimated." FPEIS at 4-229.

<u>Second</u>, the FPEIS prohibits airgun surveys in DMAs without explaining the process by which a DMA is established. Rather, the FPEIS simply recites that the "locations vary as designated by NMFS," FPEIS, Appendix C at C-17, based on "a reliable sighting of a NARW," *id*. at C-36. The 15-day duration, *see id*. at C-23, of such vaguely established DMAs threatens severe disruption and significantly increased costs to surveys, *see supra* (describing data quality and economic burdens of survey interruption). The vague and discretionary DMA standard both lacks the requisite specificity necessary for BOEM to make a reasonable decision on implementation of the measure, and significantly hampers G&G activities despite the minimal danger G&G activities pose to the NARW. *See supra*. The unnecessary burdens also extend to HRG surveys, which must be "discontinued within 24 hr" of the establishment of a DMA in the survey area. *See* FPEIS, Appendix C at C-23.

<u>Third</u>, these problems with Alternative B's expanded time-area closures is exacerbated by the creation of a further buffer at "such distance that received levels at those boundaries do not exceed" 160 db re: 1μPA (rms). FPEIS, Appendix C at C-36. The buffer effectively extends the extends of the (already unjustified) time-area closures. This further extension is likewise unjustified given that (a) available evidence indicates that vessel strikes—rather than such sound levels—pose the primary danger to the NARW, *see supra*, (b) BOEM offers no evidence that any adverse effects are probable from such sound levels at the boundaries of the closure areas, *see supra*, and (c) the buffer assumes that NARW distribution along the closure area boundaries without actual PSO confirmation.

D. The High Resolution Geophysical (HRG) Protocol Requirements.

In addition to the new limitations placed on seismic airgun surveys, the FPEIS proposes unprecedented observation and shutdown requirements for HRG activities. *See* FPEIS at 2-12–2-15.

Survey Protocols for HRG activities mimic closely those required of deep penetration seismic surveys, despite the fact they are significantly different in many ways. Airgun seismic sources are almost exclusively deployed from surface, where sounds are propagated through the water column to image the subsurface. Imaging targets can be at great depths, requiring complementary frequencies and volumes that propagate throughout the water column.

By contrast, HRG surveys are frequently conducted subsea from autonomous underwater vehicles (AUVs) pre-programmed at surface to survey along set transects. The frequency of the sources is typically mid- to high-frequency, with the associated high transmission loss of those wavelengths. Multibeam systems commonly employed on AUVs operate in the 200-400 kHz range (Reson 7125 or Kongsberg EM 2040). Sidescan systems operate in the same range or at even higher frequencies. Sub-bottom CHIRP profilers typically operate in the 1 – 12 kHz range (and use a 10-50 ms swept frequency pulse). AUV surveys are conducted 20 meters above seabed (maximum 40 meters) to maintain high resolution. At these depths, sound is refracted along the seabed, with minimal loss upward into the water column.

A survey protocol based on surface deployment does not consider activities conducted close to the sea floor, with little to no sound propagation into the water column. Employing Protected Species Observers and deploying passive acoustic monitoring from surface vessels during these types of HRG surveys is impractical and unwarranted. Additional protocols of ramp-up and shut-down for these surveys cannot be adopted for surveys that are pre-set prior to subsea deployment as direct communication with these vehicles is not always possible. Regardless, surface or near-surface activity of cetaceans would not be expected to be impacted by the activity of an autonomous vehicle deployed at depth and maneuvering at long distances from the deployment vessel along pre-programmed transects.

High-resolution AUV surveys are a key tool for identifying culturally sensitive areas, such as marine archaeological sites, environmentally sensitive areas, such as cold water corals, and complex seafloor topography that could pose a hazard for future seafloor installation or drilling operations. The ability to accurately identify these types of features is not always possible with surface based seismic or multibeam bathymetry surveys, especially in deeper water environments, so AUV surveys are an efficient, low power, method of collecting regulatory and safety-critical data beneficial to a wide range of regulatory agencies and future operations. In addition, AUV platforms can carry a wide payload of sensors, which all tend to benefit from integration with the acoustic bathymetry, backscatter, and sidescan data. The benefits of these payload systems, such as still cameras, turbidity sensors, ADCP's, methane sensors, and other environmental sensors would be reduced by restrictions placed on acoustic surveys.

Industry recommends that BOEM amend the Atlantic PEIS to exclude all AUV Surveys conducted at depth from the described HRG Survey Protocol.

E. BOEM's Commitment to Adaptive Management.

The Associations appreciate and encourage BOEM's general commitment to adaptive management. In particular, the Associations agree that "its use can ensure mitigation measures effectively match existing conditions and knowledge," FPEIS Vol. III, Table L-6 at L-120, and we feel it is very important to establish that adaptive management may be used to remove mitigation measures (in addition to adding them) where the circumstances do not warrant the measures. *See*, *e.g.*, 2012 DPEIS Comments, Appendix 2 at 17.

The FPEIS's discussion of adaptive management raises two further concerns. First, that the FPEIS uses the term as justification for the proposed imposition of mitigation measures, such as the 40-km separation distance between simultaneous surveys, *see* FPEIS Vol. III, Table L-6 at L-121–L-122, that otherwise lack scientific or practical justification. Second, it is not clear how BOEM intends to implement its planned adaptive management. While the FPEIS includes a general discussion of adaptive management from programmatic NEPA documents to site-specific analyses, *see* FPEIS at 1-26–1-28, it is unclear how this process fits into BOEM's (or BSEE's) governing regulations. For example, would the agencies be required to implement adaptive management through a new rulemaking to ensure that the applicants' and Government's respective rights and obligations are clearly defined?

The Associations look forward to further discussions with BOEM regarding the effective, and balanced, implementation of adaptive management.

F. Imposing the Proposed Mitigation Measures Would Violate the Administrative Procedure Act.

In addition to the scientific and practical failings with the mitigation measures endorsed by Alternative B, because the FPEIS's measures would plainly "supplement existing law and . . . impose additional duties and requirements," their imposition may only be accomplished pursuant to Administrative Procedure Act (APA) notice-and-comment procedures. *See, e.g.*, *Ensco Offshore Co. v. Salazar*, 10-cv-1941, 2010 WL 4116892, at *5 (E.D. La. Oct. 19, 2010) (vacating NTL for failure to comply with notice and comment requirements).

That BOEM intends ultimately to apply the measures through site-specific NEPA analyses cannot evade the APA requirements because the notice and comment requirement "turns on an agency's intention to bind itself to a particular legal policy position." *U.S. Telephone Ass'n v. FCC*, 28 F.3d 1232, 1234 (D.C. Cir. 1994).[18] Consistent imposition of the measures through site-specific analyses represents the precise intent to be bound that triggers the notice-and-comment requirement. *See id*. at 1234–36 (FCC violated APA by issuing schedule for fines and consistently applying the schedule with limited departures). And the FPEIS fails to indicate any circumstances under which BOEM believes the measures may not be applied.

Similarly, that the FPEIS has been subject to comment does not cure this procedural defect. *Cf. In re Polar Bear Endangered Species Act Listing & § 4(D) Rule Litig*., 818 F. Supp. 2d 214, 236 (D.D.C. 2011) (rejecting argument that following APA notice-and-comment procedures satisfied NEPA comment procedures).

IV. BOEM Failed to Provide a Reasoned Justification for Choosing Alternative B as the Preferred Alternative.

Although the FPEIS justifies the choice of Alternative B as providing "the highest practicable level of mitigation measures . . .," FPEIS at 2-68, NEPA requires only "a discussion of 'all practicable means to avoid or minimize environmental harms,'" *The Protect Our Communities Foundation*, No. 12-cv-2211, 2013 WL 5947137, at *10 (S.D. Cal. Nov. 6, 2013) (quoting 40 C.F.R. § 1505.2(c)). By grafting "highest" onto its obligation to consider practicable mitigation, BOEM appears improperly "to elevate environmental concerns over other valid concerns." *Utahns for Better Transportation*, 305 F.3d at 1162–63.

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[18] Nor are certain measures—such as the separation requirement and NARW time-area closure—clearly amenable to site-specific reevaluation. These measures apply on their face from the FPEIS *ab initio*.

Indeed, with respect to several additional mitigation measures proposed by Alternative B, BOEM failed properly to support the selection of mitigation beyond Alternative A. Rather, the FPEIS simply assumes that additional or expanded mitigations would necessarily achieve significant environmental benefits. For example, while Alternative B added a 40-km separation zone between surveys, "[t]he degree of improvement has not been estimated . . . ." *See* PEIS at xxiv. Because BOEM did not calculate any improvement, it did not conduct any balancing against the additional burdens placed upon applicants' operations, *see supra*, applicants' interests, *see infra*, or OCSLA's purpose to expedite development of the OCS, *see supra*.

V. The FPEIS Fails to Take into Account the Context and Economic Consequences of Alternative B's Proposed Mitigation Measures.

"Where the action subject to NEPA review is triggered by a proposal or application from a private party, it is appropriate to give substantial weight to the goals and objectives of that private actor." *Citizens' Committee to Save Our Canyons*, 297 F.3d at 1030. *See also*, *e.g.*, *Sylvester v. U.S. Army Corps of Eng'rs*, 882 F.2d 407, 409 (9th Cir. 1989) (explaining that agency has a duty to take into account objectives of applicant's project). An alternative considered in an EIS is not reasonable where it renders the applicant's proposed project "impractical," or not "technologically or economically feasible." *Citizens' Committee to Save Our Canyons*, 297 F.3d at 1031–32. *See also Sylvester*, 882 F.2d at 409 (explaining that agency must consider whether alternative is "economically advantageous" to applicant's objective). As demonstrated above, and in the Associations' 2012 DPEIS Comments, the mitigation measures imposed by the FPEIS's Alternative B threaten the operational and economic viability of G&G activities in the Mid- and South Atlantic.

BOEM concedes that "technical feasibility and economic viability" are necessary for an alternative to satisfy NEPA's reasonableness requirement. *See* FPEIS Vol. III, Table L-6 at L-115. Yet the FPEIS's only response to the Associations' showing that one of the many mitigation measures imposed by Alternative B is likely to render seismic surveys impractical is simply:

> BOEM and NMFS appreciate the comment and are committed to ensuring that mitigation requirements are feasible. The Programmatic EIS has been revised to clarify the shutdown requirement for delphinids.

FPEIS Vol. III, Table L-6, at L-110. It is not, however, a lack of clarity in the mitigation measures, but rather their substantive requirements, that threaten the viability of G&G activities. To take only the delphinid shutdown example; even the allegedly clarified provision is—by BOEM's own estimation—likely to result in tens of thousands of shutdowns. *See supra*. Under the operational conditions created by Alternative B, G&G surveys may no longer be practicable in exchange for little or no perceived environmental benefits. And the FPEIS both ignores this impracticability and fails to balance such cost against the alleged environmental benefits of Alternative B. *See Cape May Greene, Inc. v. Warren*, 698 F.2d 179, 187 (3rd Cir. 1993) (noting

NEPA "requires a balancing between environmental costs and economic and technical benefits").[19]

Indeed, BOEM's failure to fully consider the impact of mitigation measures on G&G activities compounds a second error in the FPEIS's analysis of impacts. As the Associations' 2012 DPEIS Comments illustrate, the FPEIS overstates the level of reasonably anticipated G&G activities because industry interest has decreased following exclusion of the Atlantic planning areas from the 2012-2017 OCS Leasing Program. *See* 2012 DPEIS Comments, Appendix 2 at 1–2 (noting "it is unrealistic to expect significant, if any, geophysical activity within this timeframe"). The significant operational limitations (and resulting economic costs) arising from Alternative B's required mitigation measures will further depress the number of G&G activities that will actually be conducted in the Atlantic. Accordingly, the FPEIS's estimate of anticipated industry activity—and resulting estimates of anticipated environmental impacts—is doubly overstated.

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19 BOEM's observation that "[t]here is no NEPA requirement for a cost-benefit analysis," FPEIS Vol. III, Table L-6 at L-116, is irrelevant because (1) BOEM acknowledges its obligation to consider non-environmental factors relevant to a proposed project, *see, e.g., id.*, and (2) the observation ignores BOEM's independent obligations under Executive Order 13563, *see* 2012 DPEIS Comments, Appendix 1 at 4. Moreover, the Associations comments provide a general discussion on economic burdens. *See* FPEIS Vol. III, Table L-6 at L-116 (stating that cost benefit analysis not conducted "because of the proprietary nature of cost information").

Via Electronic Mail

May 2, 2014

Kyle Baker
NOAA Fisheries Service
Southeast Regional Office
263 13th Avenue South
St. Petersburg, FL 33701
kyle.baker@noaa.gov

Subject: Comments of the American Petroleum Institute, the International Association of Geophysical Contractors, and the National Ocean Industries Association on NOAA Technical Memorandum NMFS-OPR-49, *National Standards for a Protected Species Observer and Data Management Program: A Model Using Geological and Geophysical Surveys*

Mr. Baker,

This letter provides the comments of the American Petroleum Institute ("API"), the International Association of Geophysical Contractors ("IAGC"), and the National Ocean Industries Association ("NOIA") (collectively, the "Associations") on the National Oceanic and Atmospheric Administration ("NOAA") Technical Memorandum NMFS-OPR-49, *National Standards for a Protected Species Observer and Data Management Program: A Model Using Geological and Geophysical Surveys* ("Observer Standards"). We appreciate your consideration of the comments set forth below.

API is a national trade association representing over 600 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. API is a longstanding supporter of the Marine Mammal Protection Act ("MMPA") regulatory process as an effective means of balancing and rationalizing responsible oil and gas activities with the conservation of marine mammals. We continue to support issuance of incidental take authorizations under the MMPA because, for example, it has been demonstrably effective in the Arctic in protecting marine mammal species without unduly and unnecessarily burdening industry.

IAGC is the international trade association representing the industry that provides geophysical services (geophysical data acquisition, processing and interpretation, geophysical information ownership and licensing, 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.

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. Outer Continental Shelf ("OCS"). The NOIA membership comprises more than 275 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.

General Comments

The Associations commend NOAA's National Marine Fisheries Service ("NMFS"), together with the Bureau of Ocean Energy Management ("BOEM") and the Bureau of Safety and Environmental Enforcement ("BSEE"), (collectively "the agencies") for providing recommendations for a Protected Species Observer and Data Management Program ("PSO program"). We understand that a technical memorandum is used for timely documentation and communication of preliminary results, interim reports, or more localized or special purpose information that may not have received formal outside peer reviews or detailed editing and that there is not a formal comment process. It is evident, however, that the agencies intend the recommendations in this technical memorandum to be immediately implemented for G&G surveys in the US OCS, and have incorporated the Observer Standards in the Atlantic OCS Proposed Geological and Geophysical Activities Mid-Atlantic and South Atlantic Planning Areas Final Programmatic Environmental Impact Statement ("Atlantic PEIS"). The Atlantic PEIS "Seismic Airgun Survey Protocol" requires that protected species observers complete a PSO training program "in accordance with the recommendations described in [the Observer Standards]."

In general, we are supportive of a process to standardize PSO eligibility requirements, training courses, data collection and reporting requirements. After carefully reviewing the Observer Standards, however, we have identified a number of concerns and opportunities for improvement, which are briefly summarized below and described in more detail in the following sections of this letter. Although we appreciate the agencies' attempt to clarify and standardize observer guidelines and requirements, it is imperative that the agencies consider public input on the Observer Standards and make the revisions necessary to ensure that the standards are workable, accurate, and appropriate. The standards should encourage adaptive technology, such as remote visual and acoustic monitoring and infrared technology, reduction of health and safety risks, and also the use of an updated reporting form that would be able to provide substantive data from observations to substantiate the implementation of appropriate mitigation measures.

The Associations' comments are intended to be constructive and further the goal of improving the PSO Program for G&G surveys consistent with the best available science and technology, clearly written, transparently implemented, and fully informed by the public.

Role of the US Fish and Wildlife Service

With jurisdiction over several marine mammals, the US Fish and Wildlife Service (USFWS) is an important stakeholder to the PSO process; however, it does not appear that USFWS was a part the Protected Species Working Group or that USFWS provided any input into the development of the Observer Standards. While the Observer Standards provide recommendations of report requirements for PSO sightings of polar bear and walrus (*see* p.31), the Observer Standards specifically exclude these species and all other species under USFWS jurisdiction from the purview of the standards (*see* p.v). A comprehensive national PSO program necessitates the review and input of the USFWS in addition to NMFS.

Establishment of a PSO Standardized Training Program

The Associations generally support the establishment of a standardized training program for PSOs and are interested in working with the agencies to ensure that appropriate standards are set for the "approved" vendors. We are concerned, however, that some of the recommendations for the program are based on unsupported assertions that current PSO training and reporting is inconsistent. The agencies should provide context to these assertions so that stakeholders can better understand the improvement the recommendations seek to achieve.

The Observer Standards recommend that any standardized training program should not only provide training in mitigation and monitoring requirements, but also provide health and safety considerations. The Associations agree. All PSOs should be trained to ensure complete compliance with all applicable safety procedures. A standardized training program should cover knowledge of the heightened risks working offshore on a vessel in remote locations with no or limited shore side infrastructure, and should teach personnel how to minimize risks. Training should also include information on safe travel, logistics, onboard medical infrastructure, and security including International Ship and Port Facility Security (ISPS) information.

As the Observer Standards acknowledge, many geophysical companies will also have specific requirements related to health and safety risks associated with their operations. The PSO is required to adhere to those requirements as well as any PSO provider or agency requirements. The Observer Standards should note, and any PSO training program should advise, that industry standards often exceed those of the federal agencies. Most oil and gas companies and geophysical companies require contractors to provide evidence of safety programs and requirements that meet those defined through company management systems. This should be acknowledged in any discussion of health and safety, and the agencies should also clarify whether the program intends to include medical and helicopter underwater egress training (HUET) typically required of PSOs by the industry.

The Observer Standards recommend that as part of "health and safety training," a vessel owner should "allow a PSO to briefly walk through the vessel to ensure no hazardous conditions exist according to a safety checklist, and to visually examine any safety item, upon request." PSOs are not, however, safety professionals qualified to conduct safety walkthroughs or inspections on every vessel to which they are assigned. The agencies should provide additional information on what information will be included on the safety checklist to clarify what the PSO would be looking for during this initial walkthrough to prevent misunderstandings and unnecessary effort.

The Associations suggest that a standardized training program for PSOs should include a course in effective communications. It is vital that PSOs establish direct communications with the instrument room on a seismic vessel to prevent problems and delays in the event of sightings that trigger shutdown requirements and to ensure the visual observation timeframes are adhered to before ramp up and after shutdown. All parties must work effectively together to ensure compliance: PSO, Seismic Technicians, Vessel Captain, and crew.

In addition, as the use of Passive Acoustic Monitoring ("PAM") to identify marine mammals increases in geophysical operations, the PSO Program should also include a course specific to PAM operations. PAM is a highly specialized skill and it is not appropriate to expect PSOs to possess those skills. If PAM is included in the program, training should also include rigging, mobilization and demobilization of equipment.

Finally, while the Observer Standards provide opportunity for PSO candidates who do not successfully pass an approved training course to reapply, there should be a limit on the number of times a potential PSO candidate can reapply for training.

Recommendations for BOEM/BSEE

The Observer Standards provide a list of recommendations for BOEM and BSEE to satisfy the objectives of the national standards. The Associations respectfully request that as BOEM and BSEE act on these recommendations, they solicit input from industry stakeholders and consider the following comments.

The Observer Standards recommend that BOEM and BSEE "develop permits or agreements detailing expectations and data collection and reporting of third-party PSO provider companies, including performance standards, conflicts of interest, and standards of conduct." The Associations respectfully request the agencies provide additional information and opportunity for stakeholder input regarding any proposed permitting program for PSO provider companies, including the requirements, process times, reporting requirements, and any penalties for alleged permit violations. Without well-defined boundaries, an open-ended PSO provider permitting program will provide little utility.

In addition, the Observer Standards recommend that BOEM and BSEE "develop a mechanism, procedure, or regulation to ensure that selected PSO providers are being compensated prior to deployment of approved observers." The Observer Standards do not, however, provide sufficient explanation of the need for PSO provider compensation prior to deployment of observers. More information would need to be provided to support the development of any requirement for prior compensation.

Development of Permit Fees

The Observer Standards recommend that BOEM and BSEE "consider assessing permit fees to financially support the PSO program needed for industry activities." It is unclear how the agencies would determine the amount of the fees or how the fees would be assessed. The Associations recommend that all monies generated from any such permit fees be developed solely for, and directly benefit, the PSO program and not be used for any other, non-related federal activities. Because other industries conduct similar activities requiring PSOs, the agencies should also ensure that any permitting fees are equitable to supporting the PSO program.

Recommended PSO Eligibility Requirements

In addition to a national PSO training course and PSO eligibility standards, the Observer Standards recommend the development of a policy for national PSO qualifications and eligibility. The difference between these two objectives is not immediately apparent. Qualifications, including education and competency, should be satisfied with completion of the training program. An additional policy on qualifications and eligibility is unnecessary and the Associations are concerned that limiting qualified PSO candidates to those who possess a science degree would result in a shortage of personnel.

In the recommended PSO training and provider services model, *NMFS-Approved Private Sector PSO Trainers and PSO Providers*, the Observer Standards explain that "PSO providers and PSO eligibility requirements would be defined by NMFS." While the Associations agree that the recommended mechanism for PSO training would provide more flexibility and less concern of the availability of PSO staff than the other mechanisms analyzed (*see* p.10), the agencies should clarify that NMFS' definition of PSO providers would only entail identification of those providers that meet eligibility requirements.

In the recommended waiver of education and experience requirements for PSOs, PSO candidates can provide proof of previous work experience as a PSO overseas. Some additional detail or information should be required for eligibility based on overseas work as programs and processes in other countries can vary substantially from what is expected/required for US programs. The Observer Standards also provide that the approving federal agency official has the sole discretion to waive eligibility requirements on a case-by-case basis after reviewing a waiver request and written justification. The Associations are concerned that the agency can waive "some or all of the education/experience requirements on a case-by-case basis if a lack of qualified PSOs is demonstrated." It would not be in the best interests of the regulators or the geophysical industry to employ PSOs who lack some critical or all necessary qualifications or experience. The Associations respectfully request that the waiver request, supporting justification and agency decision be made available to the PSO provider to ensure that a complete record of a PSO's experience is on file should issues arise.

The Associations agree that PSO candidates should also be in good health and have no physical impairments that would prevent them from performing their assigned tasks. The agencies should clarify, however, whether documentation or medical certification would be required similar to the *National Minimum Eligibility Standards for Marine Fisheries Observers*.

PSO Demand & Cost Estimates

The Observer Standards estimate that currently 30 PSOs are needed on a daily basis for G&G surveys in the Gulf of Mexico, with an average of 15 PSOs at sea on any given day. Based on 2009 data in the GOM, the total estimated annual costs are $2,116,547. BOEM and BSEE indicate, however, that future demand for PSOs is likely to "significantly increase over the next 5 years, and many G&G surveys are expected to occur in federal water of the Atlantic EEZ." Accordingly, the Observer Standards severely underestimate the costs and level of PSO demand. Assuming daily rates of $700.00 for each PSO, a reasonable estimate of 30 PSOs would cost $21,000 per day or $3.8M for 6 months. Travel, reporting, and health insurance would likely entail additional costs. The Associations request that the agencies update the cost and level of demand estimates with more recent data.

In addition, the Observer Standards estimate the training for each PSO in the Gulf of Mexico to cost $3,000.00. The agencies should provide a description of the various training costs detailed in this estimate, as described in Table 3, recognizing the uncertainties/unknowns associated with each estimate. For example, the estimated costs of safety training and medical examination appear lower than the industry standard.

PSO Evaluation During Permit/Authorization Approval

The Observer Standards specify that the recommended time to evaluate PSO coverage required for all G&G projects is during BOEM's permit application review or when applications for incidental take authorizations are submitted to NMFS. When weighing factors to determine the number of PSOs required for each survey, in addition to vessel size, the agencies should consider the number of bunks available on board the survey vessel.

Once the number of required PSOs is determined, the agencies assert that a single entity responsible for scheduling and deploying PSOs would result in "a greater level of consistency in many aspects of the PSO program…including maintaining an appropriate number of PSOs to meet scheduling and deployment needs." The Associations are concerned, however, that the selection of a single entity, whether a third-party provider or federal agency, to meet PSO scheduling demand would be inefficient and would result in a strain on the ability to timely contract with and obtain the number of PSOs required for each geophysical survey.

In addition, the Associations are concerned that requiring a senior-level (or lead) PSO who has specific experience observing protected species in the proposed survey geographic area will drastically limit the number of available senior-level PSOs, potentially resulting in unnecessary project delays.

During monitoring, the Observer Standards recommend that in order to reduce bias, observation periods should be limited to "favorable viewing conditions." It is unclear what is meant by unfavorable viewing conditions. During periods of "low visibility" PAM is currently required in water depths greater than 100 meters (328 feet) in the Gulf of Mexico. The agencies should be careful not to define unfavorable conditions as anything different than low visibility or nighttime to ensure there is no gap in monitoring coverage.

Conflicts of Interest

Throughout the Observer Standards, the agencies reference "inherent conflicts of interests" between PSO providers and industry, allegedly influencing accurate reporting of data. There are several unsupported assertions of inappropriate influence and pressure by industry. These assertions are unsubstantiated, and in the absence of supporting statements or examples provided by the agencies, should be deleted. If a statement denying conflict of interest is required from the PSOs prior to deployment as recommended, the statement should also include language to the effect that the PSO will conduct all their activities and report all data in full compliance with all applicable laws and regulations.

The Observer Standards defines "a direct financial interest" as payment or compensation received directly from the owner of the seismic survey's vessel, the G&G surveying company, or associated shore-based facility. The definition should also include any entity or leaseholder who employs or contracts with the survey company.

Standardized Data Collection

The Associations agree with and reaffirm the recommendation of the agencies to implement "standardization including data collection methods, standardized electronic forms, and software used in collaboration with NMFS and non-federal stakeholders." Collaboration with NMFS should result in a form that produces data the agency can use and rely on to assess population numbers, stock assessments, and effects on marine species. The Associations note that Industry best practices already recommend the use of a standard reporting form, *the Marine Mammal Recording Form*, developed under a project funded by the Exploration and Production (E&P) Sound and Marine Life Joint Industry Programme.[1] The Associations would be interesting in working with the agencies to update current reporting forms to enable the reporting of substantive data from observations that could substantiate the implementation of appropriate mitigation measures.

Creation of PSO Database

The Associations support the creation and maintenance of a database to manage PSO data for geological and geophysical surveys. This information is already supplied to NMFS and BSEE, but it would be useful for interested stakeholders to have full and timely access to such a database as a means to assess PSO activities and monitor their effectiveness.

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[1] *See* Barton, Carolyn J.S., Jaques, Robert, and Mason, Mike. 2008. Identification of Potential Utility of Collation of Existing Marine Mammal Observer Data. RSK Environmental Ltd., Cheshire, UK. The Marine Mammal Recording Form can be accessed at: http://www.iagc.org/files/3193/.

Conclusion

We appreciate the effort that the agencies have devoted to the development of PSO and data management programs for geological and geophysical surveys. We support this effort generally but, as detailed above, we have a number of concerns about the implementation of the recommendations. We respectfully request that the agencies engage with stakeholders prior to taking action on many of the recommendations, including the development of a PSO provider permit program, and system for permitting fees. We also encourage the agencies to pursue a program that encourages technology and remote monitoring, reducing health and safety risks. In addition, any program established should provide opportunity for feedback not only from PSOs, but also industry stakeholders. The Associations look forward to working with the agencies towards implementation of a PSO Program for geophysical surveys that is consistent with the best available science and technology, clearly written, transparently implemented, and fully informed by interested stakeholders.

Should you have any questions, please contact the undersigned at 202.682.8584, or via e-mail at radforda@api.org. Thank you for considering and responding to these comments.

Sincerely,

Andy Radford
American Petroleum Institute

Karen St. John
International Association of Geophysical Contractors

Jeffrey Vorberger
National Ocean Industries Association

cc: Deborah Epperson, BSEE Environmental Enforcement Division
Gregg Gitschlag, NMFS Southeast Fisheries Science Center
Howard Goldstein, NMFS Office of Protected Resources

Jill Lewandowski, BOEM Environmental Assessment Division
Kimberly Skrupky, BOEM Environmental Assessment Division
Brad Smith, NMFS Alaska Region Office
Teresa Turk, NMFS Office of Science and Technology

July 2, 2012

Mr. Gary D. Goeke
Chief, Environmental Assessment Section
Leasing and Environment (MS 5410)
Bureau of Ocean Energy Management
Gulf of Mexico OCS Region
1201 Elmwood Park Boulevard
New Orleans, Louisiana 70123-2394

Comments on the Draft PEIS for Atlantic G&G Activities
Via E-mail to GGEIS@BOEM.gov

Dear Mr. Goeke:

The American Petroleum Institute (API), the International Association of Geophysical Contractors (IAGC), and the National Ocean Industries Association (NOIA) offer the following comments on the U.S. Department of Interior Bureau of Ocean Energy Management's (BOEM's) Draft Programmatic Environmental Impact Statement (DPEIS) for Geological and Geophysical (G&G) Exploration on the Atlantic Outer Continental Shelf (OCS). On March 30, 2012, BOEM published the *Notice of Availability* in the *Federal Register* announcing publication of the DPEIS and requesting comments on or before May 30, 2012, a deadline subsequently extended to July 2, 2012. These comments are submitted as a supplement to comments provided during the public hearings held in April 2012.

The API is a national trade association that represents over 490 members involved in all aspects of the oil and natural gas industry, including exploring for and developing oil and natural gas resources in the GOM– a vital part of our nation's economy. The industry supports millions of American jobs and delivers billions of dollars in annual revenue to our government. Last year, it directly contributed more than $470 billion to the U.S. economy in spending, wages and dividends, and it is one of the few industries creating jobs throughout the recession and the ongoing national economic downturn.

The IAGC is the international trade association representing the industry that provides geophysical services (geophysical data acquisition, processing and interpretation, geophysical information ownership and licensing, 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.

The NOIA, founded in 1972, represents more than 270 companies among all segments of the offshore industry with an interest in the exploration and production of both traditional and renewable energy resources on the nation's outer continental shelf. NOIA's mission is to secure reliable access and a fair regulatory and economic environment for the companies that develop the nation's valuable offshore energy resources in an environmentally responsible manner.

BOEM's DPEIS addresses potential environmental effects of multiple Geological and Geophysical (G&G) activities in the Mid- and South Atlantic Planning Areas of the OCS. These activities include, but are not limited to, seismic surveys, sidescan-sonar surveys, electromagnetic surveys, geological and geochemical sampling, and remote sensing. These activities are critically important and are needed to provide information that will be used to update existing oil and natural gas resource assessments, and should a lease sale be scheduled for the Atlantic OCS, to inform company decisions on areas of interest for future exploration. Therefore, IAGC member companies that actually perform the activities noted above and API member companies that use the data collected during these activities are keenly interested in the DPEIS and the timely completion of the Final PEIS.

Industry has been supportive of the need for oil and gas exploration on the Atlantic OCS. However, it is critical to note that anticipated industry G&G activity will be significantly related to future leasing opportunities. At present no lease sale is scheduled for the Atlantic OCS under the proposed 2012-2017 5-year Leasing Program. It is important to remember that the government does not generate this necessary data; geophysical companies do. And they generally do this on a speculative basis, hoping to sell the data to operators who plan to purchase leases in an area. With no lease sale scheduled in the Atlantic, and thus no potential customers, companies have little incentive to gather new G&G data.

Comment Overview and Structure

In recent months, the Associations have reviewed and provided comment on separate environmental documents/regulatory actions that considered the acoustic effects of seismic surveys and other industry activities. These actions include the BOEM Petition for Incidental Take Authorization for the Gulf of Mexico [*Federal Register, Vol. 76, No. 114. p.34656*] and the DEIS for Effects of Oil and Gas Activities in the Arctic Ocean [*Federal Register, Vol. 77, No. 11, pp. 2513-14*]. Our review of this DPEIS is taken in the context of our comments filed on the Federal Register notices mentioned above. We recognize that while there are unique aspects associated with the Atlantic OCS, there are both technical and policy issues that should be consistent from region to region. The industry has used the following principles to evaluate the documents issued by the BOEM and National Marine Fisheries Service (NMFS):

- The U.S. needs to encourage energy resource development to meet its national economic security interests.
- Development should proceed with reasonable and balanced environmental protection.
- Industry has acknowledged subsistence use, has supported reasonable balance of competing uses and reasonable requirements to satisfy the Marine Mammal Protection Act (MMPA) requirement for no "unmitigable adverse effects" on the subsistence harvests of these species.
- The nature and scope of the conventional energy industry's activities must be accurately described and regulated using the same criteria as applied to other ocean users.
- Assessment of the environmental consequences must use scientifically accepted information and risk characterization/assessment methodologies and identify reasonable probabilities of risk and uncertainty.

- Agency decisions regarding U.S. Atlantic development should be made using clearly stated, legally supported criteria yielding results that can be scientifically replicated.

This transmittal letter provides an overview of our technical comments and comments dealing with the legal aspects of the DPEIS that we feel need to be addressed by BOEM before the issuance of the Final PEIS. Detailed legal comments are included as Appendix 1 and technical comments are included as Appendix 2 to this letter. In addition, we provide a brief examination of the practical impacts of one of the proposed mitigation measures, shutdown requirements, one of several measures that we believe are based on flawed analysis that do not take into account the best available science.

I. Summary of Industry Positions and Technical Comments

A. Geographic Scope:

The DPEIS specifies that the Area of Interest (AOI) includes the Mid- and South Atlantic OCS Planning Areas, as well as adjacent State waters (outside of estuaries) and waters beyond the Exclusive Economic Zone (EEZ) extending to 350 nautical miles (nmi) (648 kilometers [km]) from shore (Figure 1-1). [Page 1-5]. As recommended in our previous comments on the scope of the DPEIS, we believe that the AOI should be expanded to include the North Atlantic Planning Area. Undertaking an environmental assessment of this area now would remove a potential impediment to future exploration and lease sales in an area adjacent to Canadian OCS waters that have yielded successful oil and gas exploration, development and production.

B. Action alternatives

We recommend that BOEM provide another alternative without closure areas prior to issuance of the final PEIS. We strongly encourage that both the range of alternatives analyzed and their evaluation reflect the nature and extent of the known causes of injury and mortality faced by various protected species. In addition, for the reasons explained further in these comments, we oppose as unwarranted several of the mitigation measures proposed as part of Alternative A. Further, we believe that Alternative B is unwarranted for a number of reasons including the finding in the DPEIS that doubling the size of the closure area does not provide additional protection for right whales or marine life generally.

If BOEM does not provide a new alternative that provides no closure areas and reasonable mitigation measures, the Associations believe that Alternative A is the least objectionable of the three alternatives presented in the current DPEIS.

C. Equivalent Use Principal for High Resolution Geophysical (HRG)

The approach to High Resolution Geophysical activities would be improved if the DPEIS recognized that this type of survey equipment is also used by many other sectors not identified in the DPEIS. The DPEIS should explain why a wide range of sectors can use these technologies during certain times and in locations where the oil and natural gas E&P industry could not. Since the environmental consequences of a survey tool's use do not vary by who is using it, there is no apparent basis for this discriminatory treatment, particularly if it shows lack of effect. Industry would note that a wide range of marine users, including scientific researchers, routinely apply one or more of these or similar tools.

The DPEIS also proposes to require unprecedented observation and shut-down zone requirements for HRG but does not provide necessary environmental impact information that would indicate adverse effects of a nature to warrant requiring such zones. The shut-down requirements are in industry's opinion, not warranted, scientifically substantiated nor feasible in many circumstances, including but not limited to, HRG activities conducted by Autonomous Underwater Vehicles (AUVs) that collect data only a few feet above the sea bed.

D. Assessment of Seismic Survey Environmental Effects

Industry appreciates the agency's acknowledgment of the difficulty of assessing acoustic impacts on various species. The DPEIS's selection of sound characterization and propagation model components is more geared toward a portrayal of the size of the sound field rather than the actual impact of that sound. Industry has pointed out in recent months a variety of methodological flaws where the agency's choices in acoustic propagation models, the use of frequency weighting, and acoustic thresholds can result in individual differences in take estimates that vary by several orders of magnitude.

Improving models to better portray 3-D sound fields and animal exposures is a step in the right direction, but nevertheless, these model efforts as utilized in the DPEIS predict unrealistic Level A takes and proportionally greater numbers of Level B takes, using the simplistic 20 dB decrease from 180 dB to 160 dB. Marine Mammal Observer data does not support these model predictions, and in fact, provide no verification of takes the model predicts. Based on both field observations and recent studies, injury or death of marine mammals exposed to airguns seems increasingly unlikely (Richardson et al 2010).

The DPEIS draws conclusions based on model predictions, notably a finding of "moderate" impact, yet fails to provide any basis for an apparent confidence in model results in the face of contradictory observations. The size of the gap between presented estimates of incidental takes and observed few-to-no mortalities/injuries or population level effects undermines the credibility of the assessment. The gap between predictions and the observations provided in IHA observer reports is substantial. [1]This PEIS further highlights the gap between the estimated take numbers and the observational data by presenting large numbers of estimated dolphin takes despite extensive observations of dolphins choosing to bow ride seismic vessels.

The size of the gap between presented estimates of incidental takes based only on exposure and no observation of mortalities/injuries or population level effects undermine the credibility of the assessment. This PEIS notes that injury or death is not an expected or likely outcome yet uses contradictory Level A predictions to support conclusions of impact. The gap between predictions and the observations provided in IHA observer reports is substantial.

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[1] http://www.data.boem.gov/PI/PDFImages/ESPIS/5/5177.pdf

A good example of where exposure does not equal take as defined under the MMPA is in the estimation of dolphin takes. There is extensive documentation of dolphins choosing to bow ride seismic vessels.[2,3] This is a seemingly normal behavioral pattern frequently observed regardless of vessel type, where the animal displays a behavioral response that is not consistent with a response to harassment.

The DPEIS does a better job than some other recent NEPA documents in discussing acoustic impact analysis. However, the PEIS should contain agency explanations of all the steps, choices and assumptions that were made in impact determinations. The effects of these choices are not adequately disclosed nor discussed in the environmental consequences assessment. In the end, industry believes that the DPEIS 1) does not employ the best available science, 2) grossly overestimates the number of Level A and Level B takes, and 3) that these overestimations lead to incorrect choices in the Alternatives presented and the mitigation measures proposed.

These are not new requests. Industry has long requested transparent guidance, for example, on acoustic threshold criteria that uses widely accepted science. The industry's confidence is further eroded by repeated requests from both industry and environmental conservation organizations for clear guidance on how the agencies apply judgment to these estimates of takes to arrive at their "small number of takes" and "negligible impact" determinations. Inconsistencies in agency methods, model components, and inputs from one regulatory action to another do not instill confidence. It appears that the absence of such guidance, for example, allows various agency contractors developing NEPA documents to make choices on behalf of the agency. Variations in methods evaluation criteria, modeling components and data inputs from one agency assessment to the next naturally leads to questions about whether decisions exceed agency discretion.

Technical input on various factors in the calculation of take estimates is offered in Appendix 2.

E. North Atlantic right whale Risk Assessment & Closure Areas.

Industry shares the stated concern regarding the health of the North Atlantic right whale population. The DPEIS properly identifies the long recognized and documented major risks to this species – vessel strikes[4,5] and fishing gear entanglement. In contrast, there are no documented injuries, deaths, or significant disturbances from airguns for one of the most studied

[2] Moulton, V.D. and Miller, G.W. 2005. Marine mammal monitoring of a seismic survey on the Scotian Slope, 2003. In *Acoustic Monitoring and Marine Mammal Surveys in The Gully and Outer Scotian Shelf before and during Active Seismic Programs*, ed. Lee, K., H. Bain, and G.V. Hurley. Environmental Studies Research Funds Report No. 151, pp. 29-39.
[3] Weir et al. 2011. Cetacean encounters around the island of Montserrat (Caribbean Sea) during 2007 and 2010, including new species state records. *Marine Biodiversity Records*, 4:e42
[4] Knowlton, A.R. and S.D. Kraus. 2001. Mortality and serious injury of northern right whales (Eubalaena glacialis) in the western North Atlantic Ocean. J. Cetacean Res. Manage. (Special Issue) 2:193-208.
[5] 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(1):35-75.

whale populations in the world. In the absence of such observed impacts, the basis for the proposed closure areas disappears. The DPEIS would be improved by placing hypothetical seismic survey risks in a context relative to the significant known risks. So doing, for example, would note that the speeds of working seismic survey vessels are less than half of the current regulatory limit of 10 knots. Industry believes that the evaluation of the need for closure areas would be different if this analysis were conducted.

Moreover, the size of the proposed closure areas is premised upon defining areas of habitat critical for life function that includes not only breeding and foraging, but also migration pathways. These three components comprise the totality of activities for these animals rather than critical habitat. The critical habitat designation for North Atlantic right whales determined in 1994 considered but rejected migration routes as inconsistent with the ESA approach to critical habitat. Although there is a petition to revise critical habitat, no decision has been made. BOEM should clearly state on what basis and under what authority it proposes to regulate using migration pathways. Industry does not agree that such regulation is permissible.

Required levels of protection and mitigation standards should be risk based, practicable in implementation and equally applied to all ocean users.

II. Implications of proposed shutdown requirements

If we consider one specific proposed mitigation, the shutdown requirement, to demonstrate just how impactful the incorrect analysis and selection of alternatives and mitigation measures can be, we believe it to be so great as to cast into doubt the very feasibility of conducting seismic activities.

The proposed mitigation measures are designed to respond to and mitigate projected Type A and Type B takes. But because the DPEIS greatly overstates the number of Type A and Type B takes and exclusion zones for potential takes it greatly overstates the risk and extent for reasonable mitigation measures. This is of critical importance, because, based on predictions, some of the proposed mitigation measures would impose potentially high costs, greatly impede or altogether preclude the conduct of seismic surveys and geohazard and cultural resource identification, and deeply frustrate the achievement of the goals of the OCS Lands Act.

The outcome of decision making in the absence of sound science is manifested in the proposed mitigation measure that would: (a) greatly expand the size of the vessel exclusion zone, (b) extend it to include dolphins, and (c) apply discriminately to high resolution geophysical surveys conducted for oil and gas operators only.

Both Alternatives set forth in the DPEIS would substantially expand, by an enormous amount, the spatial area covered by the exclusion zone. This is clearly shown in Table D-21, set forth on p. D-51 of Volume II, which lists the various scenarios examined by BOEM and the resultant exclusion zone. These scenarios establish different exclusion zone radii, based upon the size of the airgun array, the water depth, the bottom type, and the time of year. **Every single scenario would materially expand the exclusion zone beyond the currently allowed 500 meter radius, whenever a large airgun array is being employed.** In some scenarios, the exclusion zone radius would be **over 2,100 meters,** meaning that **the spatial area covered by the**

exclusion zone would be 17 times larger than the current exclusion zone under Joint NTL 2012-G02. New findings of acoustic impacts or a scientific basis for such an increase in regulatory requirements is absent. What recent research does indicate is that thresholds for possible hearing damage (PTS) from an airgun source are above the antiquated 180 dB standard.[6]

That change, plus the expansion of shutdown requirements to include not only whales, as is provided by Joint NTL 2012-G02, but also dolphins, could greatly increase the number of mandatory shutdowns over that experienced under Joint NTL 2012-G02 (and previously under NTL 2007-G02).

The practical consequences of the proposed changes for the conduct of seismic surveying are enormous. We are highly doubtful that seismic survey operations could even be attempted were shutdowns to be required with anything approaching the frequency estimated in the DPEIS.

A more detailed discussion of this topic is found in Appendix 1.

In conclusion, industry has offered specific comments on the DPEIS. However, this input should not distract from higher level issues. Do seismic surveys significantly and adversely affect the marine environment relative to other well known risks? The industry does not believe they do, based on the absence of observed effects and recently released BOEM marine mammal observer data.

To build its case that seismic does have significant adverse effects, BOEM relies on models that have not been validated against field data to create unrealistic estimates of incidental takes. Further, the estimate of the number of takes is only achievable by using acoustic threshold criteria based on 15-year old obsolete data that does not meet the NEPA requirement to use the best available science. In addition, in the face of no observable injury/mortality data and no population level behavioral effect, the DPEIS demands more and more unreasonable mitigation measures, including six-month area closures and the addition of dolphins (who at times intentionally approach seismic vessels) to the list of animals that require operations to shut down. Not only is there little to no basis for these demands, the DPEIS will require the conventional energy industry to comply with operational mitigations that industries having known causes of cetacean mortality do not. In so doing, the agency decision-making is not only impossible to justify but also discriminatory.

We appreciate the work done by BOEM in developing this DPEIS. We request that BOEM review the DPEIS in light of the comments made herein and revise the DPEIS as appropriate prior to issuance of the final PEIS. If you should have any questions on these comments, please contact Andy Radford at 202-682-8584 or radforda@api.org.

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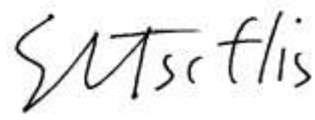
[6] Finneran, US Navy Marine Mammal Program at the Acoustical Society of America meeting, October 2011

Sincerely,

Andy Radford, API



Sarah L. Tsoflias, IAGC

Luke Johnson, NOIA

Appendix 1

Legal and Economic Issues

Several key legal principles and economic considerations must guide the preparation of this PEIS.

I. Legal Aspects

A. The DPEIS must be based on best available science

The scientific analysis set forth in the DPEIS, and upon which alternatives and recommendations set forth in the DPEIS are developed, must be based upon the best available science. This obligation stems from two separate legal mandates.

First, NEPA itself requires that an agency "utilize 'high quality' science in preparing EISs." *Sierra Club v. Marita*, 46 F.3d 606, 621 (7th Cir. 1995), citing 40 C.F.R. § 1500.1(b). "Accurate scientific analysis [is] essential to implementing NEPA." *Environmental Defense v. U.S. Army Corps of Engineers*, 515 F. Supp. 2d 69, 78 (D.D.C. 2007).

Second, the use of the best available science is mandated by Presidential Executive Order 13563 (Jan. 18, 2011). Section 1(a) of that Order provides that "[o]ur regulatory system must protect public health, welfare, safety, and our environment while promoting economic growth, innovation, competitiveness, and job creation. It must be based on the best available science."

Accordingly, as one example, BOEM must apply the best available evidence in assessing the sound levels at which Level A or Level B harassments may occur. It is entirely inappropriate for BOEM instead to rely upon historical practice at DOI or any other federal agency. "Accurate scientific evidence remains essential to an Environmental Impact Statement, and…an agency [can]not rely on 'stale' scientific evidence." *City of Carmel–By–the–Sea v. U.S. Dep't of Transportation*, 123 F.3d 1142, 1151 (9th Cir. 1997). BOEM therefore must assess the currently available science, and reach sound conclusions based upon the best available scientific evidence.

As discussed in detail in these comments, the DPEIS does not utilize the best available scientific evidence, and the conclusions reached on critical issues are therefore simply wrong. Specifically, the DPEIS errs when it concludes that exposure to sound levels in excess of 180 dB re: 1 µPa (rms) results in Level A harassment, and that exposure to sound levels in excess of 160 dB re: 1 µPa (rms) results in Level B harassment. Nor is an adequate scientific basis provided for the proposed expansion of shutdown requirements to include delphinids, the proposed expansion of the shutdown zones, or the proposed separation requirement for seismic vessels conducting simultaneous operations.

Further to this, industry does not believe the principle of equating received sound levels to takes has been subjected to public comment or peer review as is required for rulemaking. In addition, this interpretive application of exposure as a proxy for incidental take is not supported by the MMPA, which requires that harassment must take place. 16 U.S.C. 1362(18)(A). In the case of Level B Harassment, the disturbance must be related to a disruption in behavioral patterns, not just behavioral change. 16 U.S.C. 1362(18)(A)(ii), 1362(18)(D). Bow-riding by dolphins is an excellent example of a normal behavioral pattern and should not therefore be assessed as a take based on received sound levels, using any metric. Finally, there is no jurisdictional precedent defining whether sound occurring at a certain level constitutes take. It is simply not enough for an animal to be exposed to a sound. For there to be a "take" based on harassment, there must be "disruption" of a "pattern" of behavior and it must be caused by an act of pursuit, torment or annoyance. 16 U.S.C. 1362(18)(A).

B. The DPEIS must reflect programmatic needs and goals

Congress has been quite explicit in its programmatic goals under the OCS Lands Act. The OCS Lands Act's organizing principle is the "*expedited exploration* and development of the Outer Continental Shelf in order 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) (emphasis added); *see also* 43 U.S.C. § 1332(3) (the OCS "should be made available for *expeditious and orderly development*, subject to environmental safeguards, in a manner which is consistent with the maintenance of competition and other national needs" (emphasis added)).

Congress mandated these programmatic goals when it substantially amended the OCS Lands Act in 1978 for the express purpose of "promot[ing] the *swift*, *orderly and efficient* exploitation of our almost untapped domestic oil and gas resources in the Outer Continental Shelf." H.R. Rep. No. 95-590, at 8 (1977), *reprinted in* 1978 U.S.C.C.A.N. 1450, 1460 (emphasis added). As the D.C. Circuit observed soon thereafter, "the Act has an objective — the expeditious development of OCS resources." *California v. Watt*, 668 F.2d 1290, 1316 (D.C. Cir. 1981).

Despite these clear statements of Congressional intentions and programmatic goals, the PEIS lacks any analysis of the Congressional purpose enshrined in the OCS Lands Act; the manner in which the seismic surveying at issue in the DPEIS advances those goals; and the question whether Alternative A versus Alternative B, or the proposed mitigation measures contained in both Alternative A and Alternative B would have a materially negative impact upon the accomplishment of those goals. This is a fundamental flaw in the DPEIS, and one that leads to the inclusion of inappropriate proposed mitigation measures.

"NEPA itself does not mandate particular results, but simply prescribes the necessary process." *Robertson v. Methow Valley Citizens Council*, 490 U.S. 332, 350 (1989); accord *Winter v. Natural Resources Defense Council, Inc*., 555 U.S. 7, 48 (2008); *accord Robertson v. Methow Valley Citizens Council*, 490 U.S. 332, 350 (1989). Furthermore, while an agency must consider mitigation measures as part of its assessment of alternatives, NEPA neither "require[s] agencies to discuss any particular mitigation plans that they might put in place," nor … require[s] agencies—or third parties—to effect any." *Theodore Roosevelt Conservation Partnership v. Salazar*, 616 F.3d 497, 503 (D.C. Cir. 2010). Moreover, "[i]f the adverse environmental effects of the proposed action are adequately identified and evaluated, the agency is not constrained by NEPA from deciding that other values outweigh the environmental costs." *Robertson*, 490 U.S. at 350.

In conducting a NEPA environmental evaluation, an agency is *not* required to consider alternatives "inconsistent with the [government's] policy objective" in undertaking the program that is under NEPA review. *Kootenai Tribe of Idaho v. Veneman*, 313 F.3d 1094, 1121 (9th Cir. 2002) (Forest Service "not required under NEPA to consider alternatives in the DEIS and FEIS that were inconsistent with its basic policy objectives.").

The courts have been adamant on this point: an agency's only NEPA obligation is to evaluate "reasonable alternatives," 40 C.F.R. 1502.14(a), and a '"proposed alternative is reasonable only if it will bring about the ends of the federal action' measured by whether it achieves the goals the agency sets out to achieve." *National Resources Defense Council, Inc. v. Pena*, 972 F. Supp. 9, 17 (D.D.C. 1997), *quoting Citizens Against Burlington, Inc. v. Busey,* 938 F.2d 190, 195 (D.C. Cir. 1991).

What is a "reasonable alternative" is evaluated in light of the "purpose and need of the project." *Audubon Naturalist Society of the Central Atlantic States, Inc. v. U.S. Dep't of Transportation*, 524 F. Supp. 2d 642, 671 (D. Md. 2007), citing *City of Alexandria v. Slater,* 198 F.3d 862, 868 (D.C. Cir. 1999). "Alternatives addressing different purposes and goals are inherently unreasonable or infeasible." *Id.* at 671 n. 26. A federal agency may therefore eliminate alternatives and mitigation measures that do not meet the purposes and needs of the project. *Biodiversity Conservation Alliance v. BLM*, 608 F.3d 709, 715 (10th Cir. 2010); *accord City of Richfield, Minn. v. F.A.A.,* 152 F.3d 905, 907 (8th Cir.1998) ("Under NEPA, an EIS must examine 'reasonable alternatives' to a project.... An alternative is unreasonable if it does not fulfill the purpose of the project.").

Furthermore, in determining programmatic goals, and hence what proposed alternatives are "reasonable," an "agency's *evaluation of its objectives is heavily influenced by the agency's consideration of "the views of Congress, expressed*, to the extent that the agency can determine them, in the agency's statutory authorization to act, as well as in other congressional directives." *Pena*, 972 F. Supp. at 18 (emphasis added).

C. The DPEIS must focus upon reasonably likely effects, not merely potential effects

BOEM's only obligation is to assess reasonably likely environmental impacts, *South Fork Band Council of Western Shoshone of Nevada v. DOI*, 588 F.3d 718, 727 (9th Cir. 2009), not impacts that are simply a mere possibility. "An EIS need not discuss…conjectural consequences," *Sierra Club v. Hodel*, 544 F.2d 1036, 1039 (9th Cir. 1976), and alternatives and mitigation measures therefore cannot be imposed to counteract purported effects for which there exists no credible scientific proof. The Draft PEIS violates these precepts in, for example, its establishment of exclusions zones based upon conjectural impacts of exposure to arbitrarily selected sound thresholds.

III. Economic Considerations

A. The DPEIS must assess economic effects

An associated but separate requirement is that an agency appropriately "consider alternatives in a manner that is consistent with the economic goals of a project's sponsor." *Weiss v. Kempthorne*, 683 F. Supp. 2d 549, 567 (W.D. Mich. 2010) *aff'd in part and vacated in part on other grounds*, 2012 WL 204494 (6th Cir. Jan. 25, 2012). Indeed, "the consideration of alternatives may accord substantial weight to the preferences of the applicant and/or sponsor in the . . .design of the project." *Id*. at 568 (citations omitted); *see also Citizens' Comm. to Save Our Canyons v. U.S. Forest Serv.*, 297 F.3d 1012, 1030 (10th Cir. 2002) (where a private party's proposal triggers a project, the agency may "give substantial weight to the goals and objectives of that private actor").

Thus, in considering alternatives and possible mitigation measures, the agency "may legitimately consider such facts as cost to the applicant and logistics." *Sylvester v. U.S. Army Corps of Engineers*, 882 F.2d 407 (9th Cir. 1989). Indeed, the agency "has a *duty* to take into account the objectives of the applicant's project," and the effect of proposed alternatives on the achievement of those objectives. *Id*., quoting *Louisiana Wildlife Fed'n, Inc. v. York,* 761 F.2d 1044, 1048 (5th Cir.1985) (per curiam). This includes consideration whether possible alternatives would allow the project to remain "economically advantageous." *Id*.

Here, private parties are proposing to engage in seismic surveying in order to determine the presence of commercially recoverable hydrocarbons, with the intent that the leasing, exploration and production of such hydrocarbons may be fostered. "[I]t is appropriate for the agency to give substantial weight to the goals and objectives of [such] private actor[s]" when considering which alternatives are to be evaluated in the EIS and conducting that evaluation. *Fuel Safe Washington v. FERC,* 389 F.3d 1313, 1324 (10th Cir. 2004). Yet the DPEIS contains no discussion of the effect of the proposed alternatives and mitigation measures upon project economics.

B. The DPEIS must contain a cost-benefit analysis

The required consideration of economic costs must include a cost-benefit analysis. Section 1(b) of Executive Order 13563 explicitly mandates that "to the extent permitted by law, each agency must…propose or adopt a regulation only upon a reasoned determination that its benefits justify its costs…" Section 1(c) of the Order further dictates that the agency "use the best available techniques to quantify anticipated present and future benefits and costs as accurately as possible."

Nothing in the law prohibits BOEM's inclusion of a cost benefit analysis in the DPEIS, *see also Cape May Greene, Inc. v. Warren*, 698 F.2d 179, 187 (3rd Cir. 1993) ("[T]he National Environmental Policy Act requires a balancing between environmental costs and economic and technical benefits."). Thus, under the Executive Order, the PEIS should contain a cost-benefit analysis but the DPEIS does not.

C. Operational and economic implications of the proposed shutdown requirements

Industry discusses the proposed mitigations in detail in the attached technical analysis. We focus here on one specific proposed mitigation, the shutdown requirement, to demonstrate just how impactful the incorrect analysis and selection of alternatives and mitigation measures can be.

These impacts can be so great as to cast into doubt the very feasibility of conducting seismic activities.

The proposed mitigation measures are designed to respond to and mitigate projected Type A and Type B takes. But because the DPEIS greatly overstates the number of Type A and Type B takes using a flawed sound exposure equals take argument it greatly overstates the need for mitigation measures. Put another way, because the environmental impact of G&G activities is based on inaccurate science (for example, does not utilize Southall et al. 2007) and greatly overstated, the need for mitigation measures is also greatly overstated.

This is of critical importance, because some of the proposed mitigation measures would impose potentially high costs, greatly impede or altogether preclude the conduct of seismic surveys and geohazard and cultural resource identification, and deeply frustrate the achievement of the goals of the OCS Lands Act. This is antithetical to core legal principles discussed in II.B above, which require that the DPEIS, and the alternatives and proposed mitigation measures, be consonant with the programmatic goals established by Congress under the OCSLA. Fostering the expedited exploration and development of OCS resources is at the core of those goals.

The outcome of decision making in the absence of sound science that leads to decisions that are not aligned with the intent of the law is manifested in the proposed mitigation measure that would: (a) greatly expand the size of the vessel exclusion zone, (b) extend it to include dolphins, and (c) discriminately include high resolution geophysical surveys for oil and gas operators only.

Under current Joint Notice to Lessees ("NTL") 2012-G02, as well as under its predecessor notice, NTL 2007-G02, a seismic survey operator must shut down seismic operations whenever a marine mammal (except delphinids and manatees) or a sea turtle sighted within a 500 meter radius "exclusion zone," measured from the center of the airgun array and the area within the immediate vicinity of the survey vessel. The operator cannot recommence seismic operations for 30 minutes, or until the animal is no longer sighted within the 500 meter radius, whichever takes longer.

Both Alternatives set forth in the DPEIS would substantially expand, by an enormous amount, the spatial area covered by the exclusion zone. This is clearly shown in Table D-21, set forth on p. D-51 of Volume II, which lists the various scenarios examined by BOEM and the resultant exclusion zone. Additional details about these scenarios are set forth in Table D-19 on p. D-42 of Volume II, and in Vol. II, pp. D-58 through D-67.

These scenarios establish different exclusion zone radii, based upon the size of the airgun array, the water depth, the bottom type, and the time of year. **Every single scenario would materially expand the exclusion zone beyond the currently allowed 500 meter radius, whenever a large airgun array is being employed.** In some scenarios, the exclusion zone radius would be **over 2,100 meters,** meaning that **the spatial area covered by the exclusion zone would be 17 times larger** than the current exclusion zone under Joint NTL 2012-G02.

That change, plus the expansion of shutdown requirements to include not only whales, as is provided by Joint NTL 2012-G02, but also dolphins, could lead to at least a 450-fold increase in the number of mandatory shutdowns over what that experienced under Joint NTL 2012-G02 (and previously under NTL 2007-G02).[7] **The differences are not supported by the evidence.** In recent Supplemental Environmental Assessments associated with the permitting of seismic surveying in the Gulf of Mexico, BOEM has stated that there have been a total of approximately **55** required shutdowns in a typical year, due to a whale being within the 500 meter radius shutdown zone.

By contrast, BOEM in the DPEIS has estimated that there will be literally thousands of occasions a year in which a marine mammal will come within the proposed expanded exclusion zone radius surrounding an active seismic vessel and its arrays, thus triggering a shutdown of at least 30 minutes and possibly longer. Specifically, Table 4-10, found in Volume II, page Tables-32, shows that there would be **over 26,000** such shutdown events in 2016, as contrasted with the roughly **55** such events per year under current NTL requirements.[8] These estimates reflect the Government's modeling of Atlantic survey activities and marine mammal movement patterns, and are likely overstated based on the assumptions that went into that modeling. Nonetheless, the estimates do indicate the enormous effect of the proposed changes to the size of the exclusion zone and the hypothetical number of marine mammals subject to the shutdown requirement.

The fact that dolphins engaged in bow riding would not trigger a shutdown requirement, as stated in the draft PEIS at, *e.g.*, Volume II, p. C-11, does not significantly ameliorate the problem. BOEM has stated in its recent SEAs that approximately 33% of dolphins within 500 meters of a survey vessel were exhibiting bow-riding behavior.[9] Thus, the bow riding exception would at most apply approximately one-third of the time, and probably less, given that the ability to determine that a dolphin is exhibiting bow riding behavior is arguably diminished if the exclusion zone is expanded beyond the 500 meters radius, and that determination must therefore be made when the dolphin is at a considerably greater distance from the survey vessel. Further, the illogical contradiction that dolphins that do not happen to bow ride require a different mitigation strategy makes no sense. The fact that they do bow ride during seismic surveys has been observed for decades as a behavioral indicator that the surveys do not in fact cause them harm. There is no empirical evidence to the contrary.

The practical consequences of the proposed changes for the conduct of seismic surveying are enormous. We are highly doubtful that seismic survey operations could even be attempted were shutdowns to be required with anything approaching the frequency estimated in the DPEIS.

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[7] Table 4-10, Volume II, page Tables-32, indicates over 26,000 such shutdown events in 2016 versus approximately 55 such events reported under current BOEM NTL requirements.

[8] Table 4-10 sets forth "Annual Level A Take Estimates," using (incorrectly for the reasons stated in these comments) an exposure to sound at a decibel level greater than 180 dB as establishing a Level A take. And, the proposed exclusion zone radius is set at the distance from the array at which sound levels are thought to drop to 180 dB. Therefore, in setting forth the projected number of Level A Takes (using the 180 dB criteria), Table 4-10 is simultaneously setting forth the number of projected occasions a year on which a marine mammal is expected to come within the exclusion zone, and trigger a shutdown requirement.

[9] *Site-Specific Environmental Assessment of G&G Survey Application No. L11-023* (Jan. 26, 2012), at 7.

Appendix 2
Detailed Technical Discussion

I. Industry activity

Given the lack of active leases and planned lease sales, the DPEIS greatly overstates the anticipated level of industry seismic activity. The projected activity estimates submitted in May 2010 are no longer endorsed by the geophysical industry and should not be used in the development of the DPEIS.

Table 3-3 projects the acquisition of 321,600 line-kilometers and 141,700 line-kilometers of 2D seismic for the mid and south Atlantic planning areas respectively, for the first 5 years of the 9-year period covering 2012-2020. By comparison, submitted industry (IAGC) estimates were significantly less – by 36% (298,200 line-kilometers).

Accordingly, regardless of when seismic acquisition begins, the DPEIS has overstated the amount of 2D seismic that will be acquired. Although E&P companies continue to have interest in exploring the Atlantic OCS, their level of interest will likely not manifest itself into supporting and licensing (buying) non-exclusive seismic data since these areas are not included in the proposed 5-year leasing plan (2012-2017).

The industry estimates also assumed that the DPEIS would be completed in a timely manner (April 2012) with G&G permits approved in 2012 and each of the subsequent years through 2016 in support of future lease sales. Notwithstanding Secretary Salazar's statement at the time the PEIS was released that the DPEIS would be final by the end of this year, it is highly unlikely that the MMPA rulemaking and ESA section 7 consultation will be completed – pushing back the start of any geophysical activity (if any) well beyond 2012. At best, assuming Atlantic acreage is included in a 5-year leasing plan for 2017-2022, geophysical activity may commence in 2015 or 2016.

The proposed 5-year leasing plan (2012-2017) does not propose any lease sales in the mid- and south Atlantic OCS. Additionally, although the oil and gas industry believes that there are hydrocarbon resources underlying these areas and that new geophysical seismic data will illuminate those resources, lacking a commitment from the Federal Government to hold lease sales in the future (2017 and beyond), as well as support from the coastal states (Delaware, Maryland, Virginia, North Carolina, South Carolina) for lease sales and exploration and production, it is unrealistic to expect significant, if any, geophysical activity within this timeframe.

Several geophysical companies have submitted G&G permit applications to the former MMS in response to the then (2010) high level of interest expressed by E&P companies. The permit applications remain in the "queue" at BOEM. However, BOEM should not interpret this to mean that because none of the permit applications have been withdrawn that there remains a high level of interest in acquiring seismic data in the mid- and south Atlantic OCS planning areas. If a geophysical company with a permit application covering these areas were asked if they want to withdraw their application, the response would be "no". The geophysical company has already paid the cost of submitting a permit application to BOEM and there is no additional cost for it to remain with BOEM pending review. Furthermore, unless and until E&P companies clearly indicate an interest in licensing seismic data, survey activities allowed under any issued the permit would not be conducted.

II. Environmental Benefits of Geological & Geophysical Technologies

The accuracy of the DPEIS would be enhanced by more fully characterizing the important role that geophysical imaging technologies offer E&P operations toward increasing safety and reducing environmental risks in E&P operations, particularly during drilling operations. At present, there are no commercially available and viable alternatives to current geophysical imaging technologies, which have been employed and continuously refined over the last six decades to be more efficient and emit less sound energy.

Geophysical imaging technologies such as 2D and 3D seismic surveys, near surface / shallow hazard surveys and electromagnetic surveys help reduce the safety and environmental risks of future exploration activities. Vast improvements in these technologies in recent years now afford the E&P industry significant precision in subsurface imaging, resulting in significant environmental benefits. Over the E&P lifecycle, these benefits include: siting wells, facilities and pipelines at safe locations on the seafloor; the need for fewer wells and fewer facilities due to improved drilling success; the ability to predict hazardous over-pressurized zones, and thus to be able to better design wells that manage the associated risks; and improved overall safety of operations.

As a result, wells are drilled at safe locations, platforms and other facilities are placed in safe locations, and operators can route pipelines safely and around archeologically sensitive areas.

Today, conventional oil and gas companies are able to predict the pore pressures of rocks through which a well is drilled, and the predictions are improved when able to combine attributes provided by geophysical imaging technologies with subsurface information.

III. The Alternatives

The DPEIS notes the requirement for reasonable alternatives:

> *These regulations (40 CFR 1500-1508) provide for the use of the NEPA process to identify and assess reasonable alternatives to a proposed action that avoid or mitigate adverse effects of a given action upon the quality of the human environment. [Page 1-11]*

The range of alternatives should include one without the closure areas for the North Atlantic right whales. This would address the agency's NEPA requirements to include a reasonable range of alternatives. In addition, for the reasons explained in Industry's cover letter and in these comments, the proposed mitigation measures should not expand the seismic airgun survey protocol beyond what already appears in NTL 2012-G02.

Of the alternatives presented, industry favors Alternative A as the most reasonable but would note that the Alternative proposes a range of protective measures that, in some cases, exceed those required for the Gulf of Mexico. [Page 2-3].

> *Alternative A includes the following mitigation measures developed specifically for this Draft Programmatic EIS (Table 2-1):*
>
> - *a time-area closure for North Atlantic right whales;*
> - *a seismic airgun survey protocol;*
> - *an HRG survey protocol (for renewable energy and marine minerals sites);*
> - *guidance for vessel strike avoidance;*
> - *guidance for marine debris awareness;*
> - *avoidance and reporting of historic and prehistoric sites;*
> - *avoidance of sensitive benthic communities;*
> - *guidance for activities in or near National Marine Sanctuaries (NMSs);*
> - *guidance for military and National Aeronautics and Space Administration (NASA) coordination.*

BOEM notes in the DPEIS that the range of alternatives and their evaluation was significantly influenced by concern over protected species particularly the North Atlantic right whale. Industry supports this sensitivity but would encourage the BOEM to ensure a comparative risk assessment reflecting the nature and extent of the known causes of injury and mortality faced and placing the risk of industry activities within this context. The primary reason for establishing the North Atlantic right whale Seasonal Management Areas was to reduce ship strikes on this highly endangered species. The conditions that make them highly vulnerable to ships traveling greater than 10 knots – i.e., slow movements, time spent at the surface, and time spent near the coast – makes it easier for an observer to see these whales and avoid them during seismic operations. Based upon the DPEIS evaluation of the relative risks, industry does not believe that Alternative B is warranted. Industry comments will address the proposal for closure areas in greater detail later in this Appendix.

IV. DPEIS Scope, Utility and Regulatory Consistency

It is a fundamental tenet of NEPA law that an EIS is not a decisional document – such that it requires an agency to take a specific action. NEPA analyses are intended to look at the consequences of proposed actions and suggest a reasonable range of feasible alternatives. NEPA analyses are intended to inform subsequent agency decisions. The DPEIS scoping must reflect the range of decisions that may be brought forward and the DPEIS itself must be informed by and consistent with regulatory standards and the requirements of <u>all</u> Federal statutes under which the agencies make their decisions. The Atlantic DPEIS does identify and reference the Outer Continental Shelf Lands Act, the Marine Mammal Protection Act and the Endangered Species Act but industry suggests a more clear statement of the requirement to balance the three statutes and guidance to resource managers on how to achieve that balance.

There are no regulations defining the term "potential effects". The DPEIS analysis provides extensive attention to potential effects, many of which are questionable due the lack of scientific certainty, and in some critical areas – the virtual absence of knowledge. Furthermore, the DPEIS in several key respects fails to utilize the best scientific evidence that does exist. Moreover, it gives too little attention to the probability of impact. Next, the DPEIS provides little attention to the potential severity of effects. The DPEIS provides an improvement over other recent seismic survey evaluations such as the Arctic DEIS. However, more work needs to be done to avoid a situation in which the DPEIS presents an extensive list of "potential effects" as if they are likelihoods or even certainties and then demands they be mitigated. This makes it impossible for the DPEIS to inform, guide or instruct agency managers on how to differentiate between activities that have no effect, minor or major effect to a few animals, or to an entire population.

The different purposes and considerations of MMPA/ESA/OCSLA require balancing judgments by multi-agency decision-makers. The accuracy of the underlying environmental consequences analysis is critical to this proper balancing. The DPEIS provides extensive information regarding potential impacts of industry activities on marine life. Industry would continue to encourage much greater and appropriate attention in the DPEIS to the impacts the alternatives and mitigation measures would have on development of OCS resources and whether they are warranted. This should include information on lost opportunity costs and the effect of time and area closures, which under various alternatives could amount to six months per year of important areas in the AOI. The same analysis is needed with respect to mitigation measures.

V. The Environmental Consequences Methodology

A. Overview: The characterization of risk is highly subjective and is not based on sound science. This results in overstatement of impact from the industry operations and proposed mitigation measures that inappropriately allocate resources and are in conflict with the historical reality of no meaningful effect.

The comments in the cover letter identified a number of shortcomings in the gap between the assessments of environmental consequences, including the estimate of takes. These problems to a significant degree are not merely disputes over specific data issues or modeling approaches, although this itself is certainly an area in which improvements are needed. Rather they are related to a flawed environmental consequence analysis.

The DPEIS itself validates industry's concern over the modeled overestimate of takes, the ability to create representative model information and the inconsistency with actual observable effects.

> "Ultimately, the accuracy of the task relies less on the accuracy of the models and more on the accuracy of the modeler's ability to estimate these representative or average conditions. To date, probably the best measurement of this need to estimate representative or average conditions is the annually reported level of impacts for any given year of operations; as compared to that year's take authorization number. To the best of our knowledge, this has not been done officially, but anecdotal information and experience with years of annual reports has shown that typically the number of animals observed at sea is less than predicted, and their potential impacts appear lower since they are seldom observed near the sources." (DPEIS at E-69)

The DPEIS presents an environmental consequences analysis that incorrectly assesses the environmental effects of seismic operations on both an absolute basis and equally importantly on a comparative basis with other known sources of risk to individual animals and populations.

The analysis appears to give equivalent weight to potential risks, which are not equivalent – Level A (mortality/injury) and Level B (behavioral effect many of which are likely short-term and transitory). These low behavioral effect levels are then labeled as a greater risk ("Moderate") than non-industry activities such as vessel strikes and fishing gear entanglements involving mortality to marine mammals of concern, which are labeled as "Minor" environmental effects.[10]

Conflicting standards in the environmental consequence yields an internally contradicted DPEIS assessment of risks regarding a multitude of activities. Minor and short-term behavioral effects associated with seismic surveys appear to be judged more consequential than known causes of animal mortality, such as ship strikes.

B. Methodology

The DPEIS does properly concede the difficulty in evaluating acoustic risk to marine mammals and thus should require the agency to be especially vigilant and attentive in characterizing and calculating risk. The methodology outlined is inadequate and suffers from multiple problems. Industry would encourage BOEM risk assessors to consider other ecological risk assessment experiences and approaches conducted by NOAA, EPA, OMB and other agencies that are able to inform development of an improved methodology.

An improved DPEIS would better explain in the Environmental Consequence analysis how the inaccurate proxy of the incidental takes estimates progresses from assertions of single-animal effects to the population-level effect. It is not clear how this

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[10] NMFS reported 272 vessel strikes from 1972-2002, with recognition that total number of vessel strikes is unknown and only a small fraction of ship strikes reported and verified. Jensen, A.S. and G.K. Silber. 2003. Large Whale Ship Strike Database. NOAA Technical Memorandum, U.S. Department of Commerce. NMFS-OPR. 37 pp.

determination is made, (e.g., whether the analysis is premised on a deterministic approach, a probabilistic approach, or some other method).

i. The Mechanics of Assessment

The EIS describes "potential" impacts of the alternatives. Definitions of Individual Effect Criteria – the "criteria" for characterizing impact are not clear and do not adequately differentiate between "minor" and "moderate" and "moderate and "severe". To some degree they appear to be distinctions without a difference.

Moreover, the criteria used to assess acoustic effects vary from NEPA document to NEPA document, creating additional confusion. See the table below for a comparison of the criteria used on the 2012 Arctic DEIS and the 2012 Atlantic G&G DPEIS.

	Atlantic DPEIS (3/30/12)	Arctic DEIS (12/22/11)
Negligible	Little or no measurable / detectable impact	Impacts are generally extremely low in intensity (often they cannot be measured or observed), are temporary, localized, and do not affect unique resources.
Minor	Impacts are detectable, short-term, extensive or localized, but less than severe	Impacts tend to be low in intensity, of short duration, and limited extent, although common resources may experience more intense, longer-term impacts
Moderate	Impacts are detectable, short-term, extensive and severe; or impacts are detectable, short-term or long-lasting, localized and severe; or impacts are detectable, long-lasting, extensive or localized, but less than severe	Impacts can be of any intensity or duration, although common resources may be affected by higher intensity, longer-term, or broader extent impacts while unique resources may be affected by medium or low intensity, shorter duration, local or regional impacts.
Severe	Impacts are detectable, long-lasting, extensive, and severe	Impacts are generally medium or high intensity, long-term, or permanent in duration, a regional or state-wide extent, and affect important or unique resources

Thus, there is no objective or reproducible scientific basis for agency personnel to make decisions. The DPEIS process would inherently require agency decision makers to make **arbitrary** decisions not based upon objective boundaries. There needs to be consistency between the BOEM regions and NMFS on how the effects criteria are defined and how the impacts are analyzed.

ii. Characterization of Aggregated Effect

The second step in the assessment process provides for a relative judgment about Intensity versus Duration versus Extent versus Context. The same problem outlined above becomes an order of magnitude worse since there is no reproducible scientific process.

The net result is an assessment with a wide potential range of outcomes. Based upon this system, the DPEIS asserts that the effect of industry seismic activity is "Moderate" on marine mammals. If the effect of seismic is moderate, what is the assessment of risks from vessel strikes or a host of other activities? Industry would like to see the comparative assessment. These identified problems in the risk assessment make it virtually impossible to meet the NEPA requirements and guidelines requiring objective decision-making procedures. More importantly, it would yield inconsistent assessments from reviewer to reviewer. No matter how conscientious a decision maker is, there are no objective boundaries for making determinations. A minimum test is whether decisions are 1) internally consistent and 2) consistent from decision to decision. On both counts this decision making process would exceed agency discretion – in violation of both NEPA and the Administrative Procedures Act requirements.

The characterizations of risk are highly subjective and appear to be dependent upon the selection of the evaluator, who would be authorized to use his/her own, individual scientific understanding, views and biases. Thus, the assessments do not appear able to be replicated.

The DPEIS itself seems to acknowledge the inconsistency from assessment to assessment. This creates a situation in which the DPEIS determines that otherwise minor effects from industry operations (ranging from non-detectable to short-term behavioral effects with no demonstrated population-level effects) are judged to be a higher-rated risk to the species than known causes of mortality such as vessel strikes and entanglements. Thus, the projection of acoustic risk is inconsistent with reality of effect.

iii. Use of data that is not best available science.

The DPEIS acknowledges the requirement to utilize best available science and assert the agencies have met this requirement. Industry does not share that assessment.

With respect to marine mammal noise exposure criteria, industry and many scientists believe that the best available science is Southall et al. 2007, which proposes thresholds above the 160/180 dB levels for assessing Level B and Level A takes, for pulsed-sound sources such as airguns. The NMFS-initiated expert panel likewise substantially argued that the 190/180/160 dB re: 1 μPa (rms) criteria are inadequate and improved criteria are needed. Additional new information since 2007 further shows the inadequacy of the present thresholds and should contribute to a revised acoustic criteria. Historical precedent is an entirely inadequate justification for continuing to apply these thresholds because they fail to reflect the best available science. Industry is pleased that the DPEIS did include one table for estimated takes based upon Southall et al. 2007. However, other approaches are also reflected and it is not clear which approach BOEM and NMFS will ultimately utilize. The mitigation measures incorporated into the proposed alternatives do not reflect the Southall et al. 2007 conclusions.

The NMFS acoustic threshold of 180 dB re: 1 μPa (rms) for Level A takes is a dated initial criterion long overdue for revision. Again, the expert panel created by NMFS clearly provides more recent science on acoustic criteria (Southall et al 2007) and recommends a Level A sound pressure level threshold of 230 dB re: 1 μPa (peak) (flat) (or sound exposure level of 198 dB re: 1 μPa2-s) for a pulsed sound source. However, the question of sound pressure level or sound energy level as the more accurate predictor of potential injury is also discussed. The use of 160 dB re: 1 μPa (rms) as a threshold for Level B takes is a NMFS guideline. For potential disruption of behavioral patterns, the question of a dose-response versus a context-response is very much in question.

More important to the concept of take and marine mammal well being, is the question, "What responses actually represent a biologically significant impact?" Richardson et al. (2011) provides a review of potential impacts on marine mammals that concludes injury (i.e., permanent hearing damage) from airguns is extremely unlikely and behavioral responses are both highly variable and short-term. In a NMFS October 5, 2006 notice to Lamont-Doherty Earth Observatory (LDEO), the agency stated in the Disturbance Reactions section that "Simple exposure to sound, or brief reactions that do not disrupt behavioral patterns in a potentially significant manner, do not constitute harassment or "taking". By potentially significant, we mean "in a manner that might have deleterious effects to the well-being of individual marine mammals or their populations".[11] The DPEIS reverts to dated acoustic thresholds and ignores significant and more recent recommendations on improving criteria. The agency should not use outdated criteria, but should in the final PEIS utilize this more recent and far more reliable information.

iv. <u>Probabilities of Effect Ignored</u>

[11] 71 Fed. Reg. at 58790

The environmental consequences analyses are burdened by increasing attention given to more and more speculative "potential" effects without adequate consideration to probability of occurrence or applying the required "reasonable likelihood" standard or utilizing standard "weight of the evidence" tests.

v. Uncertainty & Use of Conservative Factors

The discussion of acoustics and acoustic effects suggests – but does not explicitly say --that "precautionary factors" were injected at various points in its consideration of noise criteria and acoustic effects to offset the absence of adequate information.

The Associations urge NMFS/BOEM to examine this process to handle uncertainty and to include in a revised DPEIS the assumptions and precautionary factors applied that are associated with each step of this process such as: 1) estimates of seismic activity, 2) source sizes and characterizations, 3) underwater sound propagation, 4) population estimates and densities of marine mammals, 5) noise exposure criteria, 6) marine mammal behavior, including the context of a behavioral reaction. Until the agencies document and communicate these underlying decisions in a transparent fashion neither the industry nor agency resource managers can know and understand how such decisions are made and therefore the range and rate of error. The DPEIS as presently written presents an "on the one hand; on the other" approach which does not inform the issue for agency resource managers.

The use of precautionary principles that are not reflected in actual scientific knowledge is particularly inappropriate given their fundamental inconsistency with the programmatic goals of encouraging the expeditious exploration of the OCS.

vi. Socio-Economic Considerations

The Environmental Consequences analysis must more fully consider essential economic factors, to properly evaluate and to give appropriate consideration to socio-economic impacts as required by NEPA and necessary for subsequent regulatory decisions under OCSLA. The DPEIS should, for example, discuss economic effects that would result from leasing and successful exploration that leads to production. The positive economic experiences in more mature areas such as the Gulf of Mexico should be included.

The environmental consequences analysis as noted earlier does not properly address the relative evaluation of effects (biological, physical, socio-economic). For example, the evaluation system suggests that a "Minor" biological effect and a "Minor" Socio-Economic effect would be equivalent. Industry would assert that the analysis not only does not appear to arrive at this conclusion but the DPEIS analysis does not provide a basis for assessing the relative costs and benefits of the alternatives.

As Industry observed in its cover letter to these comments, under Executive Order 13563 and controlling case law, the PEIS should include a cost benefit analysis, must take into account programmatic goals, and must also take into account the goals sought to be achieved by the private parties that will be conducting the seismic surveys. None of these are reflected in the draft PEIS.

VI. Acoustics

A. Acoustic Issues Overview

After increasing public attention to the potential impact of marine sound, the Marine Mammal Noise Exposure Criteria Work Group (the Southall Work Group) (Southall et al. 2007) was formed in the early 2000's to review the body of scientific evidence and recommend thresholds that regulators could employ. The Southall Work Group examined the prior work by the High Energy Seismic Survey (HESS) team, (HESS, 1999) and determined that those levels were "precautionary estimates" below which physical injury was considered unlikely (Southall et al. 2007). After reviewing all the available research, the Southall Work Group proposed a sound pressure level threshold for Level A injury of 230 dB re: 1 µPa (peak) (flat) (or 198 dB re 1 μPa^2-s, sound exposure level). The Southall Work Group also repeatedly stated that precaution factors had also been applied in creating its own new proposed criteria.

This represents the best scientific evidence on this question, and it should form the starting point for assessing alternatives and mitigation measures.

As previously noted the issue of acoustic-related incidental takes has suffered from the absence of a clear risk characterization and assessment. At a minimum, it is necessary for the DPEIS to clearly define what constitutes a take and why and what thresholds will be utilized in the rulemaking. If for example, there is a reason for differing thresholds (e.g. for commercial or military vessels versus seismic survey vessels), those differences should be clearly communicated and their rationale thoroughly explained.

B. Industry recommends that the DPEIS:

- Clearly differentiate the difference between the sound field, the animals exposed to sound and injury or behavioral exposure.

- Adopt the Southall Criteria (Southall, et al. 2007), which would establish the following thresholds: Level A at 198 dB re: 1 µPa2-s with M-weighting embedded in calculated RL's SEL (Sound Exposure Level); Level B at the lowest level of TTS-onset as a proxy until better data is developed.

The DPEIS does not clearly establish and communicate this information. In fact NMFS has been unable to clearly communicate that sound exposure does not equal a take although that position is often inferred. Instead, the DPEIS often uses, in our opinion, significantly inflated model predictions of takes to justify concern. This has been an issue for more than a decade. NMFS has also been unable to make a decision about utilizing Southall et al. (2007) – which has been published in a peer reviewed journal, peer reviewed by other panels and under consideration by BOEM and NMFS officials for four years. Industry believes that these are the first necessary steps in addressing the acoustics/incidental take issue. We encourage BOEM to use Southall et al. 2007 in estimating takes in the Atlantic DPEIS as it represents the best available science and not rely on the outdated, historically used 180 and 160 dB re: 1 µPa (rms) criteria.

C. Estimates of Potential Level A and B "Takes"

i. Level A: The growing scientific consensus is that seismic sources pose little risk of Level A takes (Southall et al., 2010; Richardson et al. 2011)[12]. Southall et al. (2010) and Richardson et al. (2011) recommend a Level A threshold, 230 dB re: 1 µPa (peak) (flat) (or 198 dB re 1 µPa2-s, sound exposure level). The National Research Council's expert panel assessment (NRC 2005) and further review, as discussed by Richardson et al., (2011) also support the Associations' position that this be the level adopted.

Utilizing the Southall approach greatly reduces the estimated number of Level A takes, as shown in Table 4-9 of Volume II of the DPEIS. This correction properly eliminates the proposed revisions to the established and effective shutdown requirements now set forth in NTL 2012-G02.

ii. Level B: The level of sound exposure that will induce behavioral responses may not directly equate to biologically significant disturbance; therefore additional consideration must be directed at response and significance (NRC 2005; Richardson et al. 2011; Ellison et al., 2011) [13]. To further complicate a determination of an acoustic Level B take, the animals' surroundings and/or the activity (feeding, migrating, etc.) being conducted at the time they receive the sound rather than solely intensity levels may be as important for behavioral responses (Richardson et al., 2011).

The Southall Work Group also questioned the relevance of the 160 dB re: 1 µPa disturbance criterion noting that thresholds for odontocetes and pinnipeds exposed to pulsed sounds is not at all well-established …" (Southall et al. 2007, Page 417).

Further, the Southall Work Group recognized that a difference existed between "a significant behavioral response from [and] an insignificant, momentary alteration in behavior." (See also Richardson et al., 2011). The Southall Work Group went on to

[12] Southall 2010 is a further extension of the work undertaken by Southall 2007

[13] W.T. Ellison, B.L. Southall, C.W. Clark, and A.S. Frankel. 2011. A new context-based approach to assess marine mammal behavioral response to anthropogenic sounds. *Conservation Biology*.

propose that "[c]onsequently, upon exposure to a single pulse, the onset of significant behavioral disturbance is proposed to occur at the lowest level of noise exposure that has a measurable transient effect on hearing (i.e., TTS-onset). We recognize that this is not a behavioral effect per se, but we use this auditory effect as a de facto behavioral threshold until better measures are identified."

D. Factors Impacting Thresholds

Other considerations should be recognized in establishing thresholds:

The biological significance of sound may also depend more so on how long the sound persists (Richardson et al. 2011). The DPEIS fails to allow for the fact that 3D seismic surveys are typically acquired in a racetrack pattern resulting in lower chances of an individual animal being exposed to loud sounds for extended periods of time. In other words, given that the seismic vessel is moving in and out of a localized area and the fact that animals are believed to avoid vessel traffic and seismic sounds, cumulative sound exposure is again likely being overestimated in the DPEIS. The acoustic integration model (AIM©) further does not address avoidance and, for purposes of the model limitations, does not allow animals (animats) to move out of the area. Seismic operations are most often in timescales of weeks to months which reduces the possibility of significant displacement since they do not persist in an area for an extended period of time. However, little evidence of area-wide displacement exists or has been demonstrated. For typical scales of habitat displacement studies, seismic surveys are short-term and impacts are localized.

The DPEIS analysis does not adequately consider the fact that many animals avoid vessels regardless of whether they are emitting loud sounds and may increase that avoidance distance during seismic operations (Richardson et al. 2011). Therefore, it should be a reasonable assumption that natural avoidance serves to provide another level of protection to the animals.

While crude dive profiles are included in AIM© exposure modeling, the profiling does not incorporate any animal response to the 3D sound field predicted exposures. Yet observations clearly indicate that another likely behavioral response (if the animal does not simply depart) is a change in diving behavior. Changing water depth, orientation to the source, and even an increasingly better understood mechanism of "built-in ear plugs" (stapedial reflex) all amount to significant loud noise responses (or reflexes) that reduce exposure risks.

As previously noted, the DPEIS is unclear about what constitutes an incidental taking. The MMPA defines Level B takes in the context of behavioral change, not in the context of sound exposure levels, or RMS Sound Pressure Levels. It is debatable whether behavioral changes are dose-responses or context-responses. There are also indications that some animals change their behavior in the presence of RMS Sound Pressure Levels of 160 dB re: 1 μPa (rms) or lower. In other cases of exposure to sounds of 160 dB (and higher), there is no evidence of behavioral change. It is neither logical nor reasonable to assume that every exposure to 160 dB re: 1 μPa (rms) or higher results in a behavioral change of biologically significant impact equating to a Level B take (Southall et al., 2007; Ellison et al, 2012).

There is also mounting scientific evidence that behavioral reactions are dependent upon the species and often the individual animal (Stone and Tasker, 2006) and can vary due to biological and environmental context (Wartzok et al., 2004; Frost et al., 1984; Finley et al., 1990; Richardson et al., 2011; Miller et al., 2005; Richardson et al., 1999). Most behavioral studies conducted to date have not recorded the received sound pressure levels nor is it clear that sound pressure level (rms) is the best measurement to use for behavioral studies (Southall et al. 2007). In other words, there is not enough scientific evidence to provide a convincing argument that 160 dB re: 1 μPa (rms) should be used as behavioral "take" criteria. In the base case, it is highly likely, just as the case where 180 dB re: 1 μPa (rms) was previously used, that 160 dB re: 1 μPa (rms) is overly cautious and results in an exceedingly high number of "takes".

In other rulemakings, NMFS has asserted that animals within calculated isopleths of sound above 160 dB re: 1 μPA (rms) are considered a take[14]. This basic rationale (independent of uncertainties in numbers) also likely overestimates actual take numbers and therefore should be rejected (exposure of an animal to a sound is not necessarily equivalent to the animal being taken).

Southall et al (2007) went to great effort to define functional groups in terms of sound sources and the specialized hearing characteristics of marine mammal species. Industry remains concerned with the use of the antiquated 160 dB re: 1 μPa (rms) guideline for Level B take estimation and, to a great extent, the inability to define a more reasoned criterion, which rests with an inability to document and quantify marine mammal responses to known sound levels and, more so, what response constitutes a biologically significant effect (NRC 2005). The Associations strongly encourage BOEM in the DPEIS analysis to consider the frequency component, nature of the sound source, cetacean hearing sensitivities, and biological significance when determining what constitutes a Level B incidental take.

E. Using and Explaining The Appropriate Acoustic Units of Measure

To foster meaningful dialogue and avoid confusion and poor decisions regarding industry acoustics issues, the DPEIS should adequately and accurately describe acoustic source levels.

Evaluation of acoustic effects should include both the cumulative energy criterion in Southall et al., (2007) as well as proposed cumulative energy criterion. Southall et al. indicates that, for impulse sounds, any cetacean exposed to either a peak pressure ≥230 dB re 1 μPa or a cumulative sound exposure level (energy) of 198 dB re 1 μPa2 -sec might incur auditory injury. The DPEIS should explicitly note the SEL criteria, which

[14] Federal Register/Vol. 75, No. 95/Tuesday, May 18, 2010 at Page 27712

is the one that will almost always (if not always) be the determining factor. The document in several places relies on Root Mean Square (RMS) Sound Pressure Level criteria for acoustic impacts. The most recent research has questioned the adequacy of these criteria.[15] Instead, they should be replaced by a combination of Sound Exposure Level limits and Peak (not RMS) Sound Pressure Levels or other metric being considered.

Seismic source levels are regularly quoted but they require explanation in order for the reader to have a clear understanding of what the numbers mean. Failure to do so can lead many unfamiliar with acoustics to make inaccurate judgments about the effect of seismic surveys (for example by taking 255 dB minus 180 dB as an indicator of the risk). That approach is flawed but left unexplained, the DPEIS would contribute to presentation of inaccurate information and discussion. The emitted sound pressure level close to the source array is lower than that calculated using the 'far field' calculation.

These source levels are the back-calculated, modeled sound pressure values and are not actually realized at any point in the water column. In virtually all cases they are derived from modeling and are an over-estimate of the true source sound level (sound output from a seismic source array at 1 meter distance from the array). This is an extremely significant point and we suggest BOEM add the following text or similar and a graphic to further expand upon this important point:

> *"It is difficult to measure the actual sound pressure level close to a full source array that is being activated, due to the physical environment surrounding an active seismic array. Therefore assumptions are made that enable the response of a given source array to be modeled."*

The far field assumption suggests that at some distance away from a source array, which is much greater than the dimensions of the source array, the peak energy pulses from the various individual source elements (near field signature) arrive at the same time and add together constructively to form the far field response of the source. This response is corrected or back-projected to one meter from the source array to produce the far field signature of the source at one meter, which is a standard modeled measure of a source array output. It is well known that the peak energy pulses from individual source elements no longer align at locations close to the seismic source array (in the near field) as a seismic source array is a distributed, rather than a point source.

F. Frequency Weighting

The PEIS should incorporate frequency weighting in development of incidental take estimates. Hearing (frequency) varies from species to species and among the cetaceans discussed in the DPEIS. Not all the frequencies used by industry fall within an animal's functional hearing range. In assessing the effects of noise, the M-weighted curve is

[15] Ellison et. al 2011; Madsen, P.T., Marine Mammals and Noise: Problems with Root Mean Square Sound Pressure Levels for Transients; Acoustical Society of America, 2005

applied to correct the sound-level measurement for the frequency-dependent hearing function. (Southall et al., 2007)

Without these frequency-weighted hearing curves, “extremely low- and high-frequency sound sources that are detected poorly, if at all, might be subject to unrealistic criteria.” (Southall et al., 2007, pg.413)

The primary application of the M-weighting curve is for predicting auditory damage or a dose-response situation. It should be noted that the M-weighting functions are “quite precautionary” but nevertheless are superior to flat weighting to estimate dose-response exposure.

The DPEIS should make clear whether frequency weighting to account for the hearing ranges of the species was applied in the Environmental Consequences analysis. We understand NMFS has not yet publicly accepted that M- (or similar) weighting should be applied when estimating takes during seismic surveys. At an absolute minimum, BOEM should provide examples of the potential effects of M-weighting on dose response (Level A) takes and a rationale for excluding this significant factor should have been provided.

Aggregating all frequencies for the purpose of calculating exclusion zone (safety radii) for baleen whales which are believed to have hearing sensitivity in the lower frequencies is scientifically supportable. However, it is not supportable with respect to dolphins and other odontocetes known to be mid-frequency hearing specialists. If BOEM ignores this and proceeds to require shutdowns for these species, no more than a 500 meter exclusion zone should be used which is conservative and precautionary.

VII. Biology Issues

Dolphins

The Atlantic DPEIS highlights the issue of dolphins and exclusions zones. From a biology standpoint dolphins are important in the discussion of Mid and South Atlantic G&G activities. The DPEIS notes that there are several strategic stocks of dolphins. There are large numbers of these animals. They comprise a very large percentage of the modeled estimate of Level A and Level B incidental takes.

The biology of dolphin hearing mechanisms should be considered in the DPEIS. It is well known in the Gulf of Mexico and other regions that dolphins frequently enter the seismic exclusion zone to bow ride seismic vessels.

It has also been long recognized that often cetaceans emit sounds as they echolocate that are well above the regulatory protective levels of 180/160 dB re: 1 µPa (rms). Repeated dolphin clicks have been measured up to 230dB (Au, et al 1978).

Alexander Supin and Paul Nachtigall developed a way of measuring the hearing of cetaceans during echolocation by examining the brain wave patterns of the animals to both the outgoing echolocation signal and the echo that returned from that signal (Supin et al, 2003; Nachtigall and Supin 2008).

Research into harbor porpoise (Linnenschmidt et al, 2012), and the bottlenose dolphin (Li et al, 2011, 2012) suggest hearing control may apply to a number of different species of echo locating whales and dolphins.

The DPEIS should consider this new research regarding animal sound tolerance. An example of this is the recent work conducted by Jim Finneran and his colleagues that investigated the auditory effects of multiple underwater impulses produced by a seismic airgun. The pre- and post-exposure hearing thresholds in exposed dolphins were compared to determine the amount of temporary hearing loss, called a temporary threshold shift (TTS), as a function of exposure level and the number of impulses. The research shows that dolphins exposed to airguns up to 186 dB dB re 1 μPa2 -sec (SEL) show ZERO temporary threshold shift (Finneran, US Navy Marine Mammal Program at the Acoustical Society of America meeting, October 2011). The DPEIS would be improved by a discussion of research specifically exploring the hearing control of dolphins and cetaceans. There are indications that animals naturally reduce their hearing sensitivity and therefore the estimates of incidental takes should be reduced. These results would further explain why dolphins may bow ride seismic vessels with no injury.

As mentioned previously, the PEIS should incorporate frequency weighting. It is well documented that dolphins are mid-frequency hearing specialists. Failure to incorporate frequency weighting likely results in overestimating dolphin incidental takes by at least a factor of two.

VIII. Mitigation Measures

The DPEIS proposes to require standard mitigation measures for all action alternatives. It also then proposes consideration of future optional mitigation measures.

Consideration of mitigating measures cannot be disassociated from the risks they are intended to mitigate and requirements that they be effective. In fact, a Council on Environmental Quality memorandum notes that if agencies cannot determine if mitigation was implemented or effective, mitigation requirements fail to advance NEPA objectives of informed and transparent decision-making. [CEQ 2011] Decisions regarding mitigation come through a variety of channels as the DPEIS notes and decisions about mitigation measures should be respectful of the procedures and jurisdictions that have historically evaluated and implemented mitigation requirements.

A. Considering Mitigation Effects

The DPEIS spends considerable time talking about the need for mitigation and the effects of observation zones and shut-down requirements. The DPEIS explicitly notes that it does not do so. Industry requests that BOEM consider the effects of standard required mitigation measures and reduce its takes estimates accordingly.

The Level A incidental takes predicted by the AIM© modeling do not take into account the operational mitigation measures included in the seismic airgun survey protocol to ensure that marine mammals are not present within the 180-dB exclusion zone. Although these measures are not expected to be 100 percent effective, they are expected to significantly reduce the risk of Level A harassment to marine mammals. The exclusion zone could extend up to 2.1 km (1.3 mi) from a large airgun array (5,400 in3) and up to 186 m (610 ft) from a small airgun array (90 in3). If the operational mitigation measures were100 percent successful, then all Level A harassment of marine mammals would be avoided. [Page 2-13]

B. Adaptive Management Considerations

The DPEIS mentions adaptive management on page ES-34 and elsewhere. The implication is that mitigation requirements could be altered over time. Industry has supported the application of adaptive management in a number of contexts. However, in the DPEIS the term is positioned toward the use of adaptive management to further restrict activities and it does not leave room for adaptive management to reduce restrictions. Adaptive management should also be applied to the need for corrections, if new science alters existing understandings. If monitoring shows undetectable or limited impacts, an adaptive management strategy should allow for decreased restrictions on oil and gas exploration. The conditions under which decreased restrictions will occur should be plainly stated in the discussion of adaptive management.

C. Right Whale Closure Area Proposal

The DPEIS proposes a six-month right whale closure area for Alternative A. An expanded closure area is proposed for Alternative B. The proposals are shown below:

Alternative A:

> *The total closure area under Alternative A would be 7,589,594 acres (ac) (30,714 square kilometers [km2]), or approximately 4 percent of the AOI. No G&G surveys using airguns would be authorized within the right whale critical habitat area from November 15 through April 15 nor within the Mid-Atlantic and Southeast U.S. Seasonal Management Areas (SMAs) during the times when vessel speed restrictions are in effect under the Right Whale Ship Strike Reduction Rule (50 CFR 224.105).* [Page 2-4]

Alternative B:

> *Alternative B includes one additional mitigation measure developed specifically to reduce impacts on marine mammals: an expanded time-area closure for North Atlantic right whales. The time-area closure would be expanded to a continuous 37-km (20-nmi) wide zone extending from Delaware Bay to the southern limit of*

the AOI (Figure 2-3). No G&G surveys using airguns would be authorized within the designated Right Whale critical habitat area from November 15 through to April 15, nor within the Mid-Atlantic and Southeast U.S. SMAs or the additional 37-km (20-nmi) closure areas during the times when vessel speed restrictions are in effect under 50 CFR 224.105.

The DPEIS explains the rationale for the Alternative A closure below. This logic would of course also extend to the expanded closure of Alternative B.

Alternative A includes a time-area closure intended to avoid most impacts from vessel strikes ***or ensonification of the water column [emphasis added]*** *on North Atlantic right whales. It is estimated that this closure would avoid about two-thirds of the incidental takes of North Atlantic right whales by active acoustic sound sources over the period of the Draft Programmatic EIS.* [Page 2-4]

The DPEIS observes that seismic vessels travel at notionally 5 knots/hour (or half the mandatory vessel speed limit under the right whale ship strike reduction rule) and the seismic vessels are required to have onboard dedicated marine mammal observers. The proposal raises the obvious questions:

- Does BOEM believe that seismic vessels should be held to a standard even more restrictive than one that is twice as restrictive as every other vessel operating in these management zones along the Mid- and South Atlantic?

- If the proposal is based not on vessel strike risks but rather acoustic effects, should the agencies revise the many risk assessments that include vessel strikes and fishing gear entanglements to include acoustic noise as an equivalent level threat before applying a six-month no-activity requirement?

The proposal to establish a six-month no-seismic activity zone is a significant step. BOEM should initiate rulemaking to enable sufficient study and public comment before requiring it. Such a proposal would need to consider other sound producers. Assuming that such a proposal is warranted, would such a restriction apply for example to all NOAA vessels or do the agencies propose selectively enforcing such a requirement only on one set of vessels?

The Alternative B proposal is largely based upon attention to migration routes. At present, the Critical Habitat Designation for North Atlantic Right Whales does not include these areas. Establishing migration pathways as opposed to aggregation areas for critical life functions of feeding, calving, etc. is a significant step. What basis does BOEM have in proposing such a step and has it considered rulemaking to ensure there is adequate consideration of all the factors before implementing such a regime?

D. <u>Seismic Airgun Survey Protocol</u>

The DPEIS proposes a Seismic Airgun Survey Protocol [Page 2-5 and Appendix C]. This and associated proposals in the DPEIS would require important changes in the historic operation of observation and shut-down zones including (a) shut-down zone dimensions larger than 500-meters; (b) use of the zones in waters under 200-meters, and (c) extending the use of zones from whales to dolphins. Each warrants discussion and further consideration regarding the need for such protective measures and their practicability.

E. Exclusion zone size

The DPEIS proposes that: *The radius of the exclusion zone would be calculated on a survey-specific basis but would not be less than 500 m (1,640 ft). Based on calculations in Appendix D, the 180-dB zone for a large airgun array (5,400 in3) ranges from 799 to 2,109 m (2,622 to 6,920 ft), with a mean of 1,086 m (3,563 ft).* [Page 2-5 and Appendix C]

If sound source modeling is to be required and be used to increase the size of the exclusion zone – then it should also be available to reduce the size of the exclusion zone. The DPEIS should also be more specific as to how sound measurements are to be conducted. In addition, the proposal does not explain how long such a requirement would be in place. Experience in other areas including the U.S. Arctic have shown that after a few such field source verification tests the size of such zones are well established and there is adequate knowledge of them. Requiring verification tests after such a point brings no new knowledge and is not warranted.

Finally, the DPEIS notes the size of the zone, particularly for Level B effects, may be large and impractical to visually monitor.

F. Separation between simultaneous airgun surveys

Alternative B would establish a 40-km (25-mi) separation distance between simultaneously operating seismic airgun surveys. This is in contrast to Alternative A, which does not require any geographic separation of concurrent seismic surveys. However, in practice, operators typically maintain a separation of about 17.5 km (9.5 nmi) between concurrent surveys to avoid interference (i.e., overlapping reflections received from multiple source arrays). The separation distance under Alternative B was created by rounding up this typical "operational" separation distance to 20 km (10.8 nmi), then doubling it. The purpose of this measure is to limit ensonification of large areas of the AOI at the same time by specifying a conservative separation distance between simultaneous surveys. The largest exposure radii estimated for the 160-dB threshold for a large airgun array is approximately 15 km …" [Page 29]

The need for such a requirement and the manner in which it was calculated are questionable. A separation requirement for seismic surveys should therefore not be established at this time.

NMFS has noted that "[i]n general, NMFS expects the masking effects of seismic pulses to be minor, given the normally intermittent nature of seismic pulses." 76 Fed. Reg. at 6438

The DPEIS notes that seismic survey vessels already maintain separation distances of more than 15 kilometers, which limit overlapping ensonified areas. It is noted there is a desire is to establish a "conservative separation distance". Beyond whether there is a need are questions about what standards are used to establish the need for that additional distance. The DPEIS acoustics risk assessments do not adequately address the issue of overlapping sound fields. The stated procedure of "rounding up to 20 and then doubling" does not convey a well thought out approach.

By comparison, the Final Programmatic Environmental Assessment for Arctic Ocean Outer Continental Shelf Seismic Surveys resulted in standard seismic-survey G&G stipulations providing that "operators must maintain a minimum spacing of 15 miles [24 kilometers] between the seismic-source vessels for separate operations." [16]. Thus, the DPEIS proposes a separation distance two and one-half times greater than that required in the Arctic – even though conditions in the Atlantic OCS would be expected to result in shorter sound propagation distances.

G. Dolphins Shut-down Factors

Use of observation/shut-down zones have historically been applied to cetaceans, excluding dolphins. BOEM's existing requirements are documented in NTL 2012-G02 and were premised upon a 2002 NMFS Biological Opinion.

BOEM has itself previously recognized that extending the shutdown requirement to delphinids is unwarranted. In its recent Supplemental Environmental Assessment for a specific seismic survey permit in the Gulf of Mexico, BOEM concluded that "From a biological standpoint, the best available information suggests that delphinids are considered mid-frequency specialists (i.e., auditory bandwith of 150 Hz to 160 kHz) (Southall et al., 2007). Low frequency seismic arrays, such as the ones considered for use under this proposed action, generally operate in the frequency range of 20 Hz to 20 kHz (Goold and Fish, 1998) and may extend well into the ultrasonic range up to 50 kHz (Sodal, 1999). Therefore, while the majority of the seismic noise occurs at frequencies below that of delphinids, there are some components that may enter into the hearing range of delphinids (Goold and Fish, 1998). These higher frequency components would be at lower intensity levels (i.e., not as loud). It is unclear, though, from a scientific standpoint whether any of the seismic noise that might be heard by delphinids is in fact disruptive."

[16] Minerals Management Service, Final Programmatic Environmental Assessment, Arctic Ocean Outer Continental Shelf Seismic Surveys - 2006 (OCS EIS/EA MMS 2006-038), at p. 235.

BOEM also noted the disruptive effect of a shutdown requirement on seismic operations: "Unlike other sound producing activities (e.g., sonar), seismic surveys occur on specified tracklines that need to be followed in order to meet the data quality objectives of the survey. In other words, seismic vessels in operation cannot simply divert away from nearby marine mammals without a loss in data quality." Site-Specific Environmental Assessment of G&G Survey Application No. L11-020 (Jan. 23, 2012), at 7-8. *See also* Site-Specific Environmental Assessment of G&G Survey Application No. L11-023 (Jan. 26, 2012), at 6-7; Site-Specific Environmental Assessment of G&G Survey Application No. L11-007 (Sept. 16, 2011), at 7-8; Site-Specific Environmental Assessment of G&G Survey Application No. L10-048 (Sept. 16, 2011), at 7-8.

While BOEM in these Supplemental SEISs left open the possibility of examining the issue further in a PEIS, the fact is that none of the scientific data presented in the draft PEIS for the Atlantic OCS calls into question the conclusions reached in these Supplemental SEISs.

The DPEIS nonetheless proposes adding dolphins to the shut-down requirement. It is not clear on what basis BOEM proposes such a change. The DPEIS should include a biological assessment indicating that the acoustic risks to dolphins warrant such a change.

It has been commonly observed, in fact, that dolphins seek to "bow ride" seismic and other vessels, challenging assertions of harm to the animals. The fact that various marine mammals want to approach and enter the ensonified area raises serious questions about the basic validity of a regulatory approach that rigidly established proximity to sound as its basis.

As discussed more fully in the Biology Factors section of this Appendix, recent science on the stapedial reflex is providing insight into why various animals in ensonified zones may not be adversely affected.

The DPEIS recognizes this issue of forcing shut-downs for animals that want to be in the exclusion zone: *However, shutdown would not be required for dolphins approaching the vessel or towed equipment at a speed and vector that indicates voluntary approach to bow-ride or chase towed equipment. If a dolphin voluntarily moves into the exclusion zone after the airguns are operating, it is reasoned that the sound pressure level is not negatively affecting that particular animal*. [Appendix C-11]

Industry suggests that rather than adding dolphins to the survey protocol, BOEM should provide similar provisions to not shut-down when cetaceans are voluntarily in the observation zone.

H. Whale Shut Down Factors

On page C-16, section 3.4, *Guidance for Vessel Strike Avoidance,* key element number 6 states:

> *"Whales may surface in unpredictable locations or approach slowly moving vessels. When animals are sighted in the vessel's path or in close proximity to a moving vessel, the vessel must reduce speed and shift the engine to neutral. The engines must not be engaged until the animals are clear of the area"*

As the motion of the vessel is required to provide the hydrodynamic force to keep the streamer cables in position, putting the engines in neutral for more than a moment, will result in a potential streamer cable tangle. These are very serious incidents. One recent one in French Guiana resulted in about one month of downtime and dozens of small boat sorties to untangle. The Association recommends the wording should be changed to "steer the vessel away from the whale." With the streamers in the water, a seismic vessel is traveling at 4 to 5 knots and is not at high speed.

I. Passive Acoustic Monitoring (PAM) and Protected Species Observers

Though there are limitations to current PAM technology, there are also limitations to visual observations. PAM offers another tool, in addition to visual observers, for use in monitoring. We support the use of PAM as a monitoring tool during certain conditions.

The capability of any PAM system to detect vocalizing marine mammals is dependent on various factors including level of background noise levels, animal vocalization source levels relative to background noise conditions and the experience of personnel operating the PAM system. PAM is useful under certain conditions and for certain species which have somewhat regular vocalization patterns. However, at this time, standard PAM systems are not able to reliably and accurately determine the location of the vocalizing animal automatically. In addition, the species identification capability of the PAM systems varies. The PAM system may not correctly differentiate between species of concern and other marine mammals. Current PAM systems are not able to determine if the vocalizing animal is a calf. A significant amount of research and effort is underway to improve the localization and species classification capabilities of PAM systems.

We recommend that basic training criteria, such as that specified by many countries for marine mammal observers (MMOs), be developed and required for PAM operators. In addition, minimum requirements for PAM equipment (including capabilities of software and hardware) should also be considered.

A period of confidence in the current PAM capabilities, understanding of limitations, and experienced operator capacity-building is needed before government agencies consider requiring PAM as a mandatory monitoring tool during seismic operations.

On page C-11 of Appendix C, it suggests there would be up to three PSOs plus a PAM operator on a shift together. With a typical limit of four hours per shift with a two hour break, this implies a large number of PSOs on board. With bunk space limits on vessels – usually stipulated by the USCG or other regulatory agencies – this may be an infeasible requirement.

ATTACHMENT B

to July 21, 2017
Letter of IAGC/API/NOIA

International Association of Geophysical Contractors

THE VOICE OF THE GEOPHYSICAL INDUSTRY SINCE 1971

Duplicative Surveys: Fact vs. Fiction

There has long been a misunderstanding that seismic surveys covering the same geographical areas in some way are "duplicative" or overlapping, suggesting that they are not necessary or can be "reduced" in some form by sharing data. On the contrary, there is no such thing as a duplicative survey.

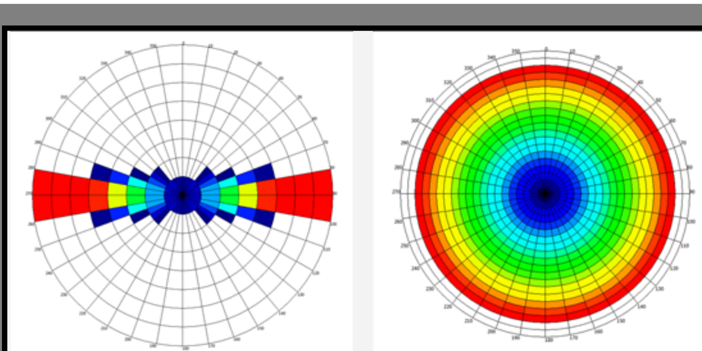
While it may appear that the sound sources, listening hardware, and vessel operations are similar, the configuration of the survey and data acquisition and processing options are numerous. Companies use proprietary, patented survey acquisition and data processing methods that make their data and each survey distinctive. The largest conference and trade show for the seismic industry, the Society for Exploration Geophysics (SEG), promotes the richness of technological acquisition and data processing options in this highly competitive industry. These value-difference options are significant to companies choosing a contractor for exploration projects, and ultimately to the energy consumer.

The bottom line is that it is commercially impractical for any geological & geophysical company to conduct a duplicative survey. Survey data, even if acquired in overlapping geographical areas or periods of time, contain different information about the subsurface and what lies beneath it. Geophysical customers find these data differences substantial enough that they often pay for multiple sets of information for the same geographical area in order to have the confidence to invest billions of dollars in the effort to bring those resources to the consumer.

Surveys are Unique

Acquisition of seismic data varies greatly depending upon the design and objectives of the survey. Diversity of acquisition can include one or a combination of the following (list is not all-inclusive):

- *2-D, 3-D, 3D wide-azimuth survey geometry* – Each survey geometry will provide a different image or image-quality of the underlying geology.
- *Orientation of the survey* – The orientation of a survey is based on the direction of the survey, for example from southwest to northeast versus southeast to northwest or west to east. Different orientations will image the underlying geology differently.
- *Towed streamer versus autonomous nodes* – A seismic streamer has multiple hydrophones encased within the streamer pulled behind a seismic vessel at 5 to 10 meters below the waterline. An autonomous node is placed on the seabed that allows full azimuth acquisition and enhanced imaging of the subsurface.
- *Streamer length* – The longer the steamer cable the better the image produced and the deeper the geology can be imaged.
- *Streamer spacing* – Spacing distances between streamers produce different qualities of data. The tighter the spacing between streamers, the better the data produced leading to better understanding of the underlying geology.
- *Acquisition azimuth* – Narrow azimuth, wide-azimuth, full azimuth are different types of acquisition techniques that provide different imaging qualities.



Narrow Azimuth **Full Azimuth**

Images depict the differences in the range of data acquired.

Images courtesy of FairfieldNodal

While different surveys for different purposes may cover the same general area, these are likely to be done over extended amounts of time, not in rapid succession.

If surveys had ceased in the U.S. Gulf of Mexico twenty to thirty years ago, before technological advancements to see geological structures beneath dense structures like salt domes that previously hid oil and gas deposits, the latest GOM discoveries would not have occurred, and U.S. production offshore would have stalled. The world's known oil reserve discoveries have doubled thanks to advancements in geophysical technology resulting in better seismic imaging to pinpoint reservoirs, especially in deep water and below thick salt formations and within shale formations.

Mitigation Measures

IAGC-recommended mitigation practices reduce potential effects of surveys on marine life. Those mitigation measures include but are not limited to exclusion zones, observers who keep a lookout for marine mammals as well as startup and shutdown procedures to further protect these animals. The stable, healthy status of marine populations where we operate, both on land and in the marine environment for as long as six decades in many locations, demonstrates that seismic surveys are environmentally safe, even over prolonged periods of continuous activity. Further, the recommended mitigation measures are accepted by governments around the world.

Flaws in the Duplicative Survey Assumptions Framework

While environmental advocacy groups may believe there are unnecessary "duplicative" surveys, they do not consider the influence of competition which leads to better seismic surveys, better decisions about when and where to drill and cost savings that are passed on to the consumer.

Each proposal for a new survey is based upon fundamental shortcomings in existing data. For example, the technical specifications for existing data may be inadequate for the target of the new survey or new technologies and survey designs may render existing data wholly obsolete.

Oil and gas exploration and production (E&P) companies pay for new G&G data, and by extension, new surveys to identify new resources. Confidence in the identification of new resource potential in turn reduces the risks and uncertainties associated with finding and developing new resources. Because the reduction in risk is valuable to the industry, technological advancements in seismic surveys and data processing, along with years of research and development and funding they require are encouraged and protected as valuable intellectual property.

As new technical and technological advancements are made, there is an incentive to conduct new surveys, which promise better geophysical data quality and stronger assurances of continuous, reliable yields of offshore oil and gas.

An example of the impracticality of the "duplicative" survey notion can be readily seen when applying the same concept to something more familiar to all of us, the automobile. One could argue that a car is simply four wheels and an enclosed space for the driver and passengers, so who needs different makes and models of cars? Because drivers have different needs (fuel economy, space, etc.), there are a variety of vehicles offered and purchased. One vehicle does not fit everyone's needs. Similarly, E&P companies purchase a variety of data sets from G&G companies to meet their varying needs.

Given the effort and expenditure involved in conducting a survey, G&G companies and oil and gas industry customers who hire them have no financial incentive to conduct a new survey if existing data is adequate (as-is or through re-processing). In reality, a duplicative survey would be economically and commercially impractical and as such, would not be conducted.

IAGC Vision Statement

The International Association of Geophysical Contractors is the most credible and effective voice for promoting and ensuring a safe, environmentally responsible and competitive geophysical industry.

ATTACHMENT C

to July 21, 2017
Letter of IAGC/API/NOIA

JNCC Report
No. 463a

Marine mammal observations during seismic surveys from 1994-2010

Carolyn J. Stone

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For further information please contact:

Joint Nature Conservation Committee
Monkstone House
City Road
Peterborough PE1 1JY
www.jncc.defra.gov.uk

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Peer review was undertaken throughout this project both internally by JNCC and externally by Department of Energy and Climate Change (DECC).

Summary

Data from 1,196 seismic surveys in UK and adjacent waters between 1994 and 2010 were examined to assess the effects of seismic operations on marine mammals and overall trends in compliance with the JNCC Guidelines for Minimising the Risk of Injury and Disturbance to Marine Mammals from Seismic Surveys (JNCC 2010 and earlier versions). Over 190,000 hours were recorded as monitoring for marine mammals (over 181,000 hours visual monitoring and over 9,000 hours acoustic monitoring), with airguns firing for 38.8% of this time.

There were 9,073 sightings or acoustic detections of marine mammals, comprising 124,024 individuals. The most frequently encountered species (where identified) was the white-beaked dolphin, although due to larger pod sizes Atlantic white-sided dolphins were the most numerous in terms of total number of individuals. Minke whales, sperm whales, harbour porpoises and long-finned pilot whales were also encountered frequently, with fin whales, killer whales and short-beaked common dolphins seen moderately often. Changes in occurrence of fin whales and harbour porpoises in 2006-2010 compared to 1994-2005 were not adequately explained by survey effort.

When 'large arrays' of airguns (500 cubic inches [cu. in.] or more) were firing a significant response (lateral displacement, more localised avoidance or a change in behaviour) was evident for all small and medium-sized odontocetes (including beaked whales) where sample sizes permitted testing, with the exception of Risso's dolphin. The minke whale and the fin whale were the only individual species of baleen whale where a significant response to 'large arrays' was found. Lateral displacement, where found, sometimes extended beyond the visual range of the observer. Behavioural responses observed when 'large arrays' were firing included changes in swimming or surfacing behaviour and there were indications that cetaceans remained near the water surface at these times. Cetaceans were recorded as feeding significantly less often when 'large arrays' were active. In addition to the responses found in cetaceans, grey seals were also displaced when 'large arrays' were firing.

When 'small arrays' (less than 500 cu. in.) were firing, fewer effects on marine mammals were noted. However, significant lateral displacement was found for sperm whales and harbour porpoises when 'small arrays' were firing and localised avoidance was apparent for some species groups. Furthermore, there were indications that initial tolerance of 'small arrays' by delphinids and small odontocetes might have decreased as surveys progressed. While with 'large arrays' cetaceans sometimes remained near the surface when the airguns were firing, with 'small arrays' there were indications that cetaceans may remain submerged more during periods of firing. Other effects on swimming or surfacing behaviour were not evident with 'small arrays'.

There was some evidence that the soft start may be an effective mitigation measure. Detection rates of cetaceans during the soft start were significantly lower than when the airguns were not firing and on surveys with 'large arrays' more cetaceans were observed avoiding or travelling away from the survey vessel during the soft start than at any other time. These results were found for all of the few species or species groups that were able to be tested. Further studies on the effectiveness of the soft start, particularly for other species, would be valuable.

Long term trends in compliance with the JNCC guidelines were examined. Standards of pre-shooting searches have remained stable over the years, while standards of soft starts and the implementation of delays in firing when required have improved. However, although there has been improvement, of particular concern was the number of occasions when

delays in firing were not properly implemented following a detection of marine mammals in the mitigation zone as compliance with this aspect of the guidelines still lags behind that of pre-shooting searches and soft starts. Incorrect procedures in a delay situation were sometimes due to the subsequent soft start being too short, but more often due to the delay not being long enough.

This report represents one of the longest term analyses of MMO data to date and provides a valuable resource for investigating the potential impacts of industrial activities on marine mammals and the effectiveness of the guidelines and compliance therewith. Continued collection and analysis of MMO data will continue to improve mutual understanding of these issues and benefit both the conservation of these species and appropriate mitigation measures.

Contents

Glossary

2D survey Two dimensional exploration where a single streamer (containing hydrophones for detection of reflected sound) is used and the reflections from the subsurface are assumed to lie directly below the sail line that the survey vessel traverses. For regional surveys, sail lines are typically widely spaced (typically several kilometres apart) over a large area; a two dimensional image is obtained and is generally used for wide-scale surveys.

3D survey Three dimensional exploration where multiple streamers (containing hydrophones for detection of reflected sound) are used and sail lines are closely spaced (typically a few hundred metres apart). The use of multiple streamers results in the acquisition of many closely spaced sub-surface 2D lines, typically 25-50m apart, and the data are processed into a three dimensional image of the subsurface.

4D survey 3D seismic survey repeated at an interval of months or years, to identify changes to the hydrocarbon reservoir over time due to production in order to maximise hydrocarbon recovery from the field.

Airgun Device into which air is pumped into chambers at high pressure and then released through ports to form an oscillating bubble, thereby producing sound waves.

Baleen whale Cetaceans belonging to the suborder Mysticeti, which lack teeth and have two external blowholes; baleen whales in north-west European waters include the blue whale, fin whale, sei whale, humpback whale and minke whale.

Bottling Behaviour where a seal assumes a vertical position with its head out of the water, allowing it to breathe while resting or sleeping.

Breaching Behaviour where a cetacean launches itself into the air head-first and falls back into the water with a splash.

Cetacean The group of marine mammals comprising the whales, dolphins and porpoises.

Dedicated MMO Person dedicated to the role of MMO and not any other job on board.

Delphinid Cetaceans of the family Delphinidae, a subdivision of the odontocetes which in north-west European waters includes the dolphins, long-finned pilot whales and killer whales.

Effort Number of hours of visual or acoustic monitoring.

Full power Firing the airguns at their full operational level, reached at the end of a soft start.

JNCC Joint Nature Conservation Committee; the public body that advises the UK Government and devolved administrations on UK-wide and international nature conservation.

Line change The activity of turning the vessel at the end of one survey line prior to commencement of the next line.

Logging Behaviour where cetaceans float motionless at the water surface.

Lunging A method of feeding used by some baleen whales where they lunge forwards with mouths open engulfing a large volume of water and any prey species contained therein are sieved from the water using the baleen plates.

Marine European Protected Species Marine species in Annex IV(a) of the Habitats Directive that occur naturally in the waters of the United Kingdom; these consist of several species of cetaceans (whales, dolphins and porpoises), turtles and the Atlantic sturgeon.

Milling Behaviour where cetaceans continue to surface in the same general vicinity.

Mitigation zone The area where an MMO or PAM operator keeps watch for marine mammals (and delays the start of activity should any marine mammals be detected); currently the area within 500m of the centre of the airgun array.

MMO Marine mammal observer; person who will monitor for the presence of marine mammals visually and will provide advice to enable compliance with the JNCC guidelines.

Mysticete The suborder of cetaceans including the baleen whales, which lack teeth and have two external blowholes; mysticetes in north-west European waters include the blue whale, fin whale, sei whale, humpback whale and minke whale.

Non-dedicated MMO Person undertaking the role of MMO who may also do another job on board.

Non-parametric statistical test A statistical test that is appropriate where the underlying data are not normally distributed.

OBC survey Ocean Bottom Cable survey, where the streamers or cables (containing both hydrophones and geophones) are laid on the sea bed and a separate source vessel is utilised.

Odontocete The suborder of cetaceans including the toothed whales and dolphins, which possess teeth and have a single external blowhole; odontocetes in north-west European waters include the sperm whale, beaked whales, killer whale, long-finned pilot whale, dolphins and harbour porpoise.

PAM Passive acoustic monitoring; listening for marine mammal vocalisations using hydrophones deployed in the water linked to specialist software.

PAM operator Person who operates PAM equipment to monitor for the presence of marine mammals acoustically and will provide advice to enable compliance with the JNCC guidelines.

Pinniped The group of marine mammals comprising the seals, sea lions and the walrus.

Porpoising Swimming behaviour where cetaceans leap clear of the water whilst moving forwards.

Pre-shooting search Search for marine mammals prior to commencing firing of the airguns.

Rorqual whale Baleen whale of the family Balaenopteridae, all possessing many longitudinal throat grooves that allow expansion of the mouth cavity when feeding.

Seismic survey Survey where sound waves are generated (by using airguns) and sent into the seabed and the reflected energy is recorded (with hydrophones) and processed to produce images of the geological strata below the seabed.

Site survey Survey over a specific site in order to identify seabed and shallow subsurface hazards (e.g. shallow pockets of gas) prior to the location of infrastructure or a drilling rig. The technique is that of a 2D survey but typically utilises smaller volumes of airguns, commonly around 160 cu. in. Other equipment may also be used, including side scan sonar and sub-bottom profilers such as boomers, pingers and sparkers.

Soft start (or ramp up) Process whereby the power of an airgun array is built up slowly from a low energy start-up, gradually and systematically increasing the output until full power is achieved.

Source The source of the noise, i.e. for a seismic survey the airguns.

Spy-hopping Behaviour where a cetacean positions itself vertically with its head poking above the water surface.

Tail-slapping Behaviour where a cetacean forcefully slaps its tail flukes on the water surface.

Time-sharing When vessels engaged on adjacent surveys take turns to run survey lines to avoid interference from the noise of each other's airguns. This is becoming less necessary with improvements in software and increases in computer processing power.

UKCS UK continental shelf.

VSP Vertical seismic profiling; undertaken during drilling operations where the geophone is lowered into the borehole and the airguns are lowered over the side of the drilling rig (zero offset VSP) or from a vessel at a fixed location (offset VSP) or from a vessel traversing lines away from the platform (walkaway VSP).

1 Introduction

Over the past few decades concern has developed over potential negative impacts of anthropogenic noise on marine mammals. Amongst the activities of concern are marine seismic surveys, used to explore the sea floor in the search for oil and gas reserves. This exploration is achieved by directing sound, produced by airguns, at the seabed and analysing the resultant reflections of that sound to map the geological structures below the sea floor. The airguns produce high levels of impulsive low frequency sound with an inherent risk of disturbance and possibly acoustic trauma (e.g. auditory injury) to marine mammals.

In 1992, the Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas (ASCOBANS; now the Agreement on the Conservation of Small Cetaceans of the Baltic, North east Atlantic, Irish and North Seas) introduced a requirement to work towards the prevention of significant disturbance, especially of an acoustic nature, to small cetaceans. In 1995, the UK government adopted a set of guidelines developed by the Joint Nature Conservation Committee (JNCC) to minimise disturbance to small cetaceans from seismic surveys in particular, partly as a response to the ASCOBANS requirement. Amongst the provisions of these guidelines was the requirement to monitor for the presence of cetaceans prior to commencing firing the airguns; this was the origin of the role of the marine mammal observer (MMO) on seismic surveys. The guidelines have been revised on a number of occasions and since 1998 have included all marine mammals. The relevant regulator is the Department of Energy and Climate Change (DECC) and the latest revision of the guidelines, the JNCC Guidelines for Minimising the Risk of Injury and Disturbance to Marine Mammals from Seismic Surveys, was published in August 2010 (JNCC 2010). The guidelines also aim to reduce the risk of causing deliberate injury or deliberate disturbance to European Protected Species (EPS, including cetaceans) as required by Article 12 of the EC Habitats Directive (92/43/EEC) and the Directive's transposition into UK legislation. All applications to conduct seismic surveys for oil and gas exploration within the UKCS require consent from DECC. JNCC are consulted on all such applications, as one of DECC's statutory consultees, with the JNCC guidelines informing the consent conditions for such surveys.

Monitoring for the presence of marine mammals prior to commencing firing of the airguns is a key component of the JNCC guidelines. This is primarily achieved by visual means (i.e. MMOs), however there is provision for passive acoustic monitoring (PAM) to be used at times when conditions are not conducive to effective visual monitoring (e.g. darkness, poor visibility and increased sea states). If marine mammals are detected within a defined mitigation zone, then the start of airgun firing must be delayed. When it is clear to start, the level of firing must increase gradually by using a soft start/ ramp up procedure. The primary role of the MMO is to provide advice to enable the crew to comply with the guidelines and hence mitigate potential negative impacts of seismic operations on marine mammals. This work involves collecting data on the seismic operations, the watches and any marine mammals observed. Marine mammal recording forms are available for this purpose (JNCC 2012a) and all data from seismic surveys in UK waters are returned to JNCC where, after appropriate quality checks, they are included in a database. Although MMOs only need to observe prior to firing commencing, most continue to observe at other times, including during soft starts and full power firing, hence the database includes a large amount of data providing a valuable resource for analysis.

This report presents the results of an analysis of that database, including all data from 1994, just prior to the introduction of the guidelines, until the end of 2010. Previous analyses have analysed subsets (one to four years) of these data (Stone 1997, 1998, 2000, 2001, 2003a, b, 2006; Stone and Tasker 2006), but analysis of the larger dataset increased sample sizes and permitted further statistical testing of more individual species. The aim of the analysis was to identify any effects of seismic operations on marine mammals and any long term trends in compliance with the JNCC guidelines. More specifically, the following comparisons were made:

- general trends in survey effort and species distribution;
- detection rates (firing versus not firing);
- detection rate trends throughout the duration of surveys;
- closest distance of approach to the airguns (firing versus not firing);
- behaviour;
- effectiveness of the soft start;
- general trends in compliance with the JNCC guidelines.

2 Methods

2.1 Marine mammal observations and effort

Marine mammal observations were undertaken from seismic surveys operating in UK waters. Some MMOs also voluntarily submitted their records from surveys operating in the waters of neighbouring countries (Norway, Ireland, Faroes, the Netherlands, Denmark, Germany and France), although these formed a minority of records. Data from 1994 until 2010 were recorded; although the JNCC guidelines were not introduced until 1995 some companies started recording their observations while the guidelines were in preparation.

Visual watches for marine mammals were carried out during daylight hours. Observers ranged from biologists experienced in marine mammal surveys to non-scientific personnel who in many cases had undergone JNCC-recognised MMO training (http://jncc.defra.gov.uk/page-4703); the proportion of surveys with trained MMOs steadily increased over time. In addition PAM was utilised on some surveys during night-time operations and sometimes also during the day. In 1994 and 1995 sightings were recorded using a non-standard format. Since 1996, MMOs and PAM operators have completed standard marine mammal recording forms that also require that effort (number of hours of visual or acoustic monitoring) is recorded. A number of versions of these forms have been issued over the years (latest version JNCC 2012a), but all versions are compatible and allowed data to be included in the database. There are currently four forms: ‘Cover Page’ (general information about the survey), ‘Operations’ (times of all airgun operations and associated mitigation), ‘Effort’ (details of visual and acoustic monitoring, including time, position, source activity and weather conditions) and ‘Sightings’ (details of any marine mammals encountered). When marine mammals were encountered observers recorded the species (with a supporting description), number of animals, behaviour, closest distance of approach to the airguns and the airgun activity at the time of the encounter. Photographs were sometimes taken to aid in confirmation of species identification. Observers used different methods to estimate the range to animals, but the use of a rangefinder stick (Heinemann 1981) was the most common. Observers recorded any behaviours that were apparent rather than selecting from a set list, but the Guide to Using Marine Mammal Recording Forms (latest version JNCC 2012b) gave examples of behaviours that may be seen. Feeding can be difficult to be sure of, but MMOs are taught that behaviours indicative of feeding might include cetaceans being observed with a fish; lunge-feeding in baleen whales; and in dolphins erratic, fast swimming with frequent changes of course and birds diving alongside etc.

2.2 Airgun arrays

The observations encompassed a range of types of seismic survey with widely varying sizes of airgun array. The smallest airgun array volume was 6 cu. in. (on some site surveys), while the largest was 10,170 cu. in. (on a 2D survey). Very large volumes of airguns were rare, with only nine surveys using volumes exceeding 5,500 cu. in. Where appropriate, surveys with airguns of small volumes were analysed separately from those with larger airgun volumes with the split occurring at 500 cu. in. (following the threshold used in the JNCC guidelines to determine action during a line change). Therefore, in the context of this report, ‘large arrays’ refers to arrays with a volume of 500 cu. in. or more and ‘small arrays’ refers to arrays with a volume of less than 500 cu. in. Surveys were assigned to each category based on the reported airgun volume, but where airgun volume was not recorded for individual surveys 2D, 3D, 4D and OBC surveys were assigned to the ‘large arrays’ category and site surveys were assigned to the ‘small arrays’ category, as the vast majority of these types of surveys consistently used airgun volumes in the respective category. VSP operations used airgun volumes ranging from 150 cu. in. to 1,200 cu. in. with a substantial proportion in each of the 'large arrays' and 'small arrays' categories. Therefore where airgun volume was not recorded, individual VSP operations were not included in any analysis where 'large arrays' and 'small arrays' were distinguished. The 'small arrays' category included 678 surveys, while 500 surveys used 'large arrays'.

The frequency and source level of the airguns were often not recorded as this information was not requested on recording forms in earlier years. However, from available information 'large arrays' typically produce frequencies predominantly up to around 200Hz, with a peak-to-peak energy output from the source of around 130-140 bar metres, equating to a peak source level of around 256dB re. 1μPa @ 1m. 'Small arrays' (e.g. as used on site surveys) typically produce frequencies predominantly up to around 250Hz, with a peak-to-peak energy output from the source of around 10 bar metres, equating to a peak source level of around 235dB re. 1μPa @ 1m.

2.3 Data quality control

Only data of acceptable quality were entered into the database and were subject to analysis. Data checks were applied consistently following a standard list of over 60 checks (Barton 2012). Examples included: checking that source activity was accurately recorded during observation effort; that airgun array characteristics corresponded with information within the MMO report; that consecutive positions were credible given the time interval and speed of the vessel; that species identity corresponded with the description and/ or photograph; and that there was reasonable confidence that behaviour had been recorded accurately (e.g. not an unusually high proportion of sightings by one observer exhibiting the same behaviour). Any errors found were corrected where possible. If data were accurate or minor inaccuracies were able to be corrected then the data were entered into the database. Data with key information missing or errors that were not able to be corrected were discarded; approximately 15% of surveys had at least part of the associated data discarded, although this happened slightly less often (11%) on surveys with 'large arrays' where dedicated MMOs were more often used. The recording forms have evolved over the years so it is not possible to make a meaningful comparison between years of the amount of data discarded.

After following the quality control process, data from a total of 1,196 surveys were entered into the database and were available for analysis, spanning the period from 1994 to 2010. Of the surveys included in the database, 91% were entirely in UK waters, 3% spanned both UK and adjacent waters and 6% were only in adjacent waters of neighbouring countries.

2.4 Analysis and statistical tests

For some analyses it was not appropriate to use all of the data in the database. For example, some sightings or acoustic detections had no accompanying effort data so could not be used where detection rates per unit effort were calculated; for some other aspects of analysis, effort data was not necessary and all sightings and acoustic detections were used. When considering biological responses of marine mammals to airgun activity, it was appropriate to include the minority of records from waters of neighbouring countries, as these animals belong to the same stocks as those occupying UK waters, but when considering compliance with JNCC guidelines records from outside the UKCS were excluded.

Where airgun volume was likely to influence the results, surveys with 'large arrays' were analysed separately from surveys with 'small arrays' where possible. For some analyses other variables had the potential to influence the results. Weather conditions influence the ability of observers to detect marine mammals (e.g. Hammond *et al* 2013; Northridge *et al* 1995). If weather was likely to bias the results, periods with the same weather conditions were compared where possible, or otherwise only periods of good observation conditions (i.e. 'glassy' or 'slight' sea states, swell < 2m and visibility > 5km) were used. Location, season, observer ability and monitoring method (visual or acoustic) also needed to be considered as potential influences for some analyses. The following sections indicate how data was treated in order to reduce bias from these influences.

Non-parametric statistical tests were used throughout (Siegel and Castellan 1988); these tests make fewer assumptions about the nature of the populations from which the data are drawn and do not require that the data are normally distributed. The following sections describe the tests that were used for each aspect of the analysis.

Results are presented for individual species where sample size permitted. Sometimes sample sizes were too small to be able to run the statistical test for individual species (this varied depending on the test being used), so groups of combined species were used, e.g. all seals, all cetaceans, all baleen whales, all beaked whales, all delphinids or all small odontocetes. These combined species groups comprised all identified and unidentified animals within that taxonomic grouping (Table 2.1), e.g. the baleen whale group included both fin whales and unidentified fin/ sei whales, amongst other species. The group of all small odontocetes included all the dolphin species (identified or unidentified) and the harbour porpoise. Combined species groups were more often used for surveys with 'small arrays' than those with 'large arrays', as surveys with 'small arrays' were often of short duration so sample sizes were lower. For surveys with 'large arrays' sample sizes were mostly greater, but beaked whales were combined due to low numbers of detections of individual species.

Table 2.1 Division of cetacean species into combined species groups for analysis (combined species groups also included unidentified animals within that group).

Baleen whales	Beaked whales	Delphinids	Small odontocetes
Northern right whale	Northern bottlenose whale	Long-finned pilot whale	Risso's dolphin
Humpback whale	Sowerby's beaked whale	Killer whale	Bottlenose dolphin
Blue whale		False killer whale	White-beaked dolphin
Fin whale		Risso's dolphin	Atlantic white-sided dolphin
Sei whale		Bottlenose dolphin	Short-beaked common dolphin
Minke whale		White-beaked dolphin	Striped dolphin
		Atlantic white-sided dolphin	Harbour porpoise
		Short-beaked common dolphin	
		Striped dolphin	

2.4.1 General trends in survey effort and species distribution

Maps of effort and species distribution were plotted using DMAP for Windows with the geographic areas referred to throughout the text shown in Figure 2.1. Effort maps were plotted using data since 1996, when effort was first recorded. As the early effort data did not always record positions in sufficient detail to calculate effort per block, effort maps were plotted after summing the amount of effort in each quadrant (1^{o} latitude and longitude rectangle, comprising 30 licensing blocks) where the watch started. Individual species maps are included in Appendix 1. For rarer species (northern right whale, blue whale, Sowerby's beaked whale and false killer whale) locations of sightings were plotted. All other species maps were plotted after summing the number of individuals of each species in each offshore oil and gas licensing block (10' latitude x 12' longitude). All sightings and acoustic detections were included on species maps, but where shifts in distribution over time were apparent, sighting rates in different areas over five year periods were calculated using only sightings that had accompanying effort data. To reduce bias, sighting rates for each five year period were calculated using only visual data from months of peak occurrence of animals (June to September) and periods when the airguns were not firing during good observation conditions.

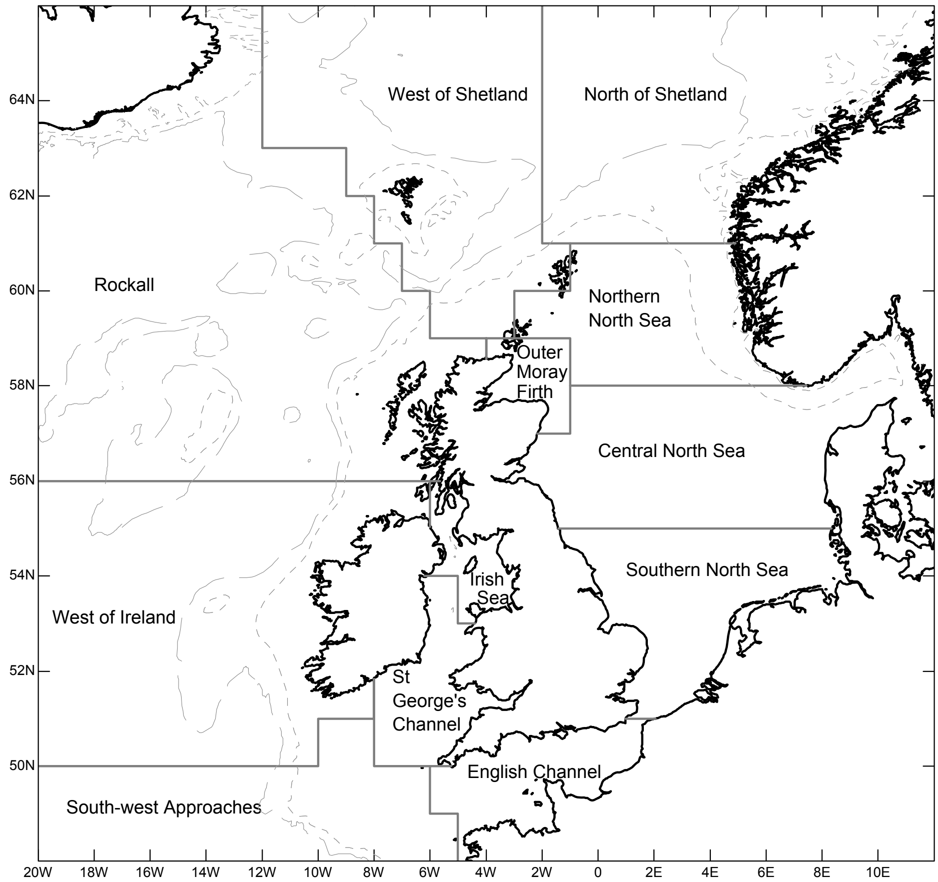

Figure 2.1. Geographic areas used in data analysis (short dashed line = 200m isobath; long dashed line = 1,000m isobath).

2.4.2 Detection rates (firing versus not firing)

Only sightings or acoustic detections that had accompanying effort data were used to calculate detection rates. As there was no distinction between effort during the soft start and that at full power prior to 2009, the airguns were regarded as firing whether they were firing at full power, undertaking a soft start, or firing at reduced power for some reason other than a soft start. Most effort when firing would have been at full power, as the soft start and other reduced power firing is of relatively short duration. As airgun volume was likely to influence the results, surveys with 'large arrays' were analysed separately from those with 'small arrays'.

Detection rates may be influenced by other variables, e.g. location, season, weather, monitoring method and observer ability. Therefore matched pairs (firing versus not firing) were used where for each pair the survey, ship, date, observer, monitoring method (visual or acoustic) and weather conditions (sea state, swell and visibility) were the same, so the only remaining variable was the source activity (the combination of ship and date controlled for location within the range that could be travelled by a ship during the course of one day). The resulting matched pairs (firing versus not firing) were tested using the Wilcoxon signed ranks test, a non-parametric test appropriate for two related or matched samples that ranks the differences between each pair. It compares both the direction of the difference in each pair (i.e. which is greater) and also the magnitude of the difference (i.e. by how much is it greater). The Wilcoxon signed ranks test can be performed on small samples, with significant results being able to be detected with sample sizes as low as five matched pairs (Siegel and Castellan 1988). For larger samples the test statistic T^+ is approximately normally distributed so in these cases z was calculated and its associated probability was determined by reference to tables for the normal distribution.

2.4.3 Detection rate trends throughout the duration of surveys

Data from surveys lasting three weeks or longer were examined to see if there was any evidence of a decline in numbers of marine mammals after the survey commenced if activity was prolonged. Only surveys with ‘small arrays’ were considered as these were mostly site surveys where firing occurred within a small area (surveys with ‘large arrays’ often covered a wide area with temporal variation in the precise location of firing throughout the survey). Only surveys where the airguns became active during the first week were used.

The Wilcoxon signed ranks test was used to compare sighting rates between the first and later weeks of each survey. Comparing within each survey controlled for the influence of location and, to some extent, observer. Due to the nature of the question, seasonal variations may have had an influence, as numbers of animals may have naturally increased or decreased throughout the duration of each survey. The influence of monitoring method and weather were controlled by using only visual sightings during good observation conditions.

2.4.4 Closest distance of approach to the airguns (firing versus not firing)

The airguns were regarded as firing whether they were firing at full power, undertaking a soft start or firing at reduced power for some reason other than a soft start. As airgun volume was likely to influence the results, surveys with ‘large arrays’ were analysed separately from those with ‘small arrays’. Distance estimation with PAM was not as accurate as with visual monitoring (Stone 2015), so only visual detections (with or without accompanying effort data) were used to compare the closest distance of approach to the airguns. Airguns were less likely to be firing in rough weather conditions and in such conditions animals would be harder to detect at distance; this could result in bias towards closer distances at times when the airguns were not firing. This potential bias was controlled by using only sightings during good observation conditions. Similarly, the experience of the observer could have introduced bias, as less experienced observers (e.g. non-dedicated MMOs) would be less likely to detect animals at greater distances and such observers were more likely only to observe during the required pre-shooting search (i.e. only when airguns were not firing); this could also result in bias towards closer distances when the airguns were not firing. To reduce this potential bias only sightings by observers with good detection skills were used. An initial examination of data from a small sample of known experienced observers found that a minimum of 20% of detections were more than 1km away. This was applied as a criterion for selecting observers with good detection skills throughout the database; in order to determine which observers met this standard, only those who had at least 20 sightings were considered.

The closest distance of approach of animals to the airguns was compared (firing versus not firing) using the Wilcoxon-Mann-Whitney test. Scores were ranked and W_x was determined by summing the ranks in the smallest group. The Wilcoxon-Mann-Whitney test can be performed on small samples, with significant results being able to be detected with sample sizes as low as three in each group (Siegel and Castellan 1988). For larger samples the distribution of W_x approaches that of the normal distribution and therefore z was calculated in these cases and its associated probability was determined by reference to tables for the normal distribution.

2.4.5 Behaviour

Only visual sightings were used to examine behaviour of marine mammals. All sightings were used, including those without associated effort and during any weather conditions. The airguns were regarded as firing whether they were firing at full power, undertaking a soft start or firing at reduced power for some reason other than a soft start. As airgun volume was likely to influence the results, surveys with ‘large arrays’ were analysed separately from those with ‘small arrays’.

The frequency of occurrence of each recorded behaviour was compared between periods of firing and not firing. Similar behaviours (e.g. breaching, jumping, somersaulting) were grouped together to avoid any bias due to inter-observer variation in terminology. The chi-squared test was used to compare the observed frequency with the expected frequency had there been no difference between groups (firing versus not firing), for all behaviours and species where the expected frequency in both groups was at least five (Siegel and Castellan 1988). For some behaviours where non-significant trends were found for individual species, combined species groups were used to increase the sample size, thereby increasing the power of the statistical test (Siegel and Castellan 1988).

2.4.6 Effectiveness of the soft start

The data were examined to look for responses of marine mammals to the soft start that might indicate whether it is an effective mitigation measure. Detection rates, the closest distance of approach to the airguns, and behaviour were compared for periods when the airguns were not firing, periods when they were firing at full power and periods when they were firing during the soft start. As the soft start is of relatively short duration sample sizes were often low; only three individual species (minke whale, white-beaked dolphin and Atlantic white-sided dolphin) could be examined, otherwise combinations of species were used.

Matched samples were used to compare detection rates at each source activity level during each day of each survey on each ship when monitoring method (visual or PAM) and weather conditions (sea state, swell, visibility and sun glare) were the same, thereby controlling for any influence of location, season, weather, type of survey, monitoring method and, to some extent, observer. Only surveys where effort during the soft start had been differentiated from effort at full power were used (July 2009 onwards). As this limited sample sizes all available data were used regardless of total airgun volume (both visual sightings and acoustic detections). The results were tested using the Friedman two-way analysis of variance by ranks, a non-parametric equivalent of the analysis of variance. Scores for each matched sample were ranked (1, 2 or 3) and a value for F_r was calculated with the associated probability determined with reference to the χ^2 distribution. For significant results, multiple comparisons of pairs of treatments were tested using the Wilcoxon signed ranks test to determine where the significant differences lay.

The closest distance that marine mammals approached the airguns was compared using only visual sightings during good observation conditions, as PAM did not give accurate range estimation and weather may affect visual detection at distance. As sightings did not need to be effort-related records from all years could be used; this meant that sample sizes were sufficient to analyse surveys with ‘large arrays’ separately from those with ‘small arrays’. The Kruskal-Wallis one-way analysis of variance by ranks was used; for larger samples the Kruskal-Wallis statistic KW is well approximated by the χ^2 distribution thus the associated probability was determined. Where results were significant, multiple comparisons of pairs of treatments were used to determine where the significant differences lay.

Behaviour was compared using visual sightings where source activity did not change during the course of the encounter. All sightings from all years were used, regardless of observer or weather conditions (as weather was unlikely to influence the ability of the observer to record behaviour) or whether there was accompanying effort data. ‘Large arrays’ were analysed separately from ‘small arrays’. The frequency with which different behaviours were exhibited was compared using the chi-squared test, for all behaviours and species where the expected frequency in all groups was at least five.

The chi-squared test was also used to examine behaviour at the commencement of the soft start by comparing encounters where the airguns were not firing throughout, those where the soft start commenced during the course of the encounter or those where the airguns were performing a soft start throughout. The chi-squared test can only be performed if the expected frequency in all groups is at least five; diving was the only behaviour where this condition was met (only when surveys with arrays of any size were included).

2.4.7 General trends in compliance with the JNCC guidelines

Three key areas of compliance with the guidelines were compared over time. These were the number of visual pre-shooting searches during daylight hours that were at least 30 minutes long (or 60 minutes in deep waters since June 2009), the number of soft starts that were at least 20 minutes long and the proportion of delays that were correctly implemented (delay of at least 20 minutes plus subsequent soft start of at least 20 minutes). Compliance was compared as far back as records would allow; pre-shooting searches and soft starts were compared for all years since 1998 (when operations data were first recorded), while delays were compared since the introduction of the guidelines in 1995. Only data from the UKCS were analysed when assessing compliance with the JNCC guidelines.

3 Results

3.1 General trends in survey effort and species distribution

Observations for marine mammals were undertaken on seismic surveys throughout the UK and some adjacent waters, covering 199 quadrants (1^{o} rectangles = 30 licensing blocks), some of which were passed in transit to or from the survey location when airguns were not firing but sightings were still recorded. A total of 190,727 hours 54 minutes were recorded as monitoring for marine mammals between 1996 and 2010 (effort was not recorded prior to 1996); of this, 181,119 hours 19 minutes were recorded for visual monitoring and 9,608 hours 35 minutes for acoustic monitoring. The airguns were firing for 38.8% of the total time spent monitoring.

All areas had at least some survey effort, with the exception of the English Channel (Figure 3.1). The majority of effort was in the central and northern North Sea, but there was also substantial effort in the southern North Sea and to the west and north of Shetland. There was comparatively little PAM effort, with PAM being adopted as a mitigation measure gradually over the years. PAM effort was concentrated in the northern and central North Sea (particularly the Outer Moray Firth and adjacent waters) and waters to the west and north of Shetland, reflecting its use in sensitive areas. The maps only show effort where it was correctly recorded on the ‘Effort’ form (or ‘Location and Effort’ form in earlier years), hence a small number of surveys with PAM in the central North Sea, the Rockall Trough and St George’s Channel are not illustrated on the map.

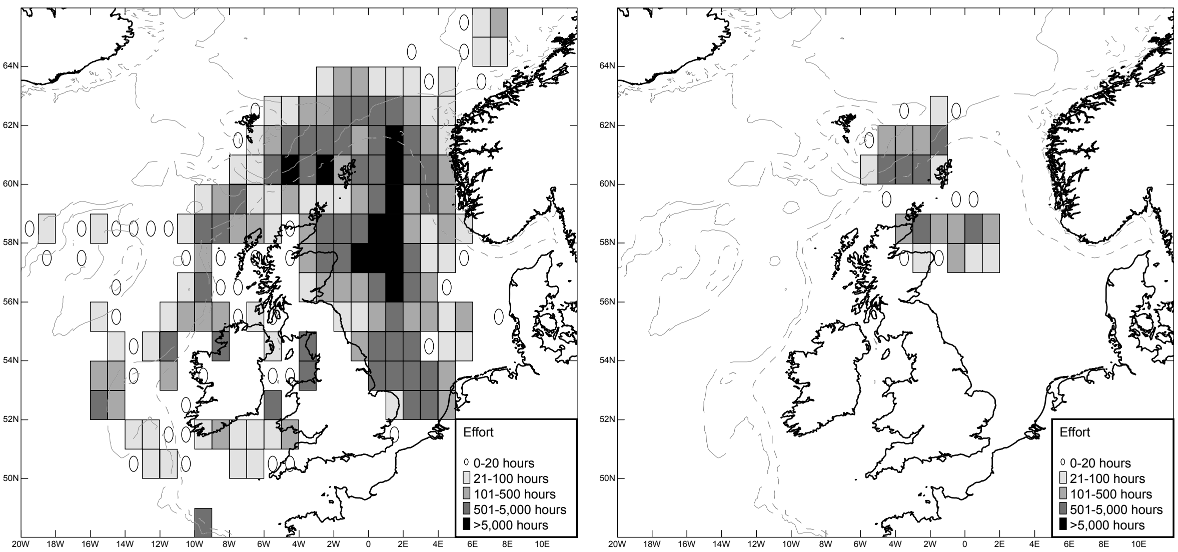

Figure 3.1. Visual effort (left) and PAM effort (right) during seismic surveys from 1996-2010 (short dashed line = 200m isobath; long dashed line = 1,000m isobath).

PAM effort was seldom recorded prior to 2006, so visual effort and PAM effort were combined together when considering the variation in effort over time. In the period from 1996-2000 there were many surveys in deep water areas to the west of Shetland and in the Rockall Trough (corresponding with the 16th and 17th rounds of offshore oil and gas licensing) and extending down through deep waters to the west of Britain and Ireland (Figure 3.2). Between 2001 and 2005 there were fewer surveys in these deep water areas, while between 2006 and 2010 there was another increase in effort to the west of Shetland and lower effort extending out over banks to the west of the Rockall Trough.

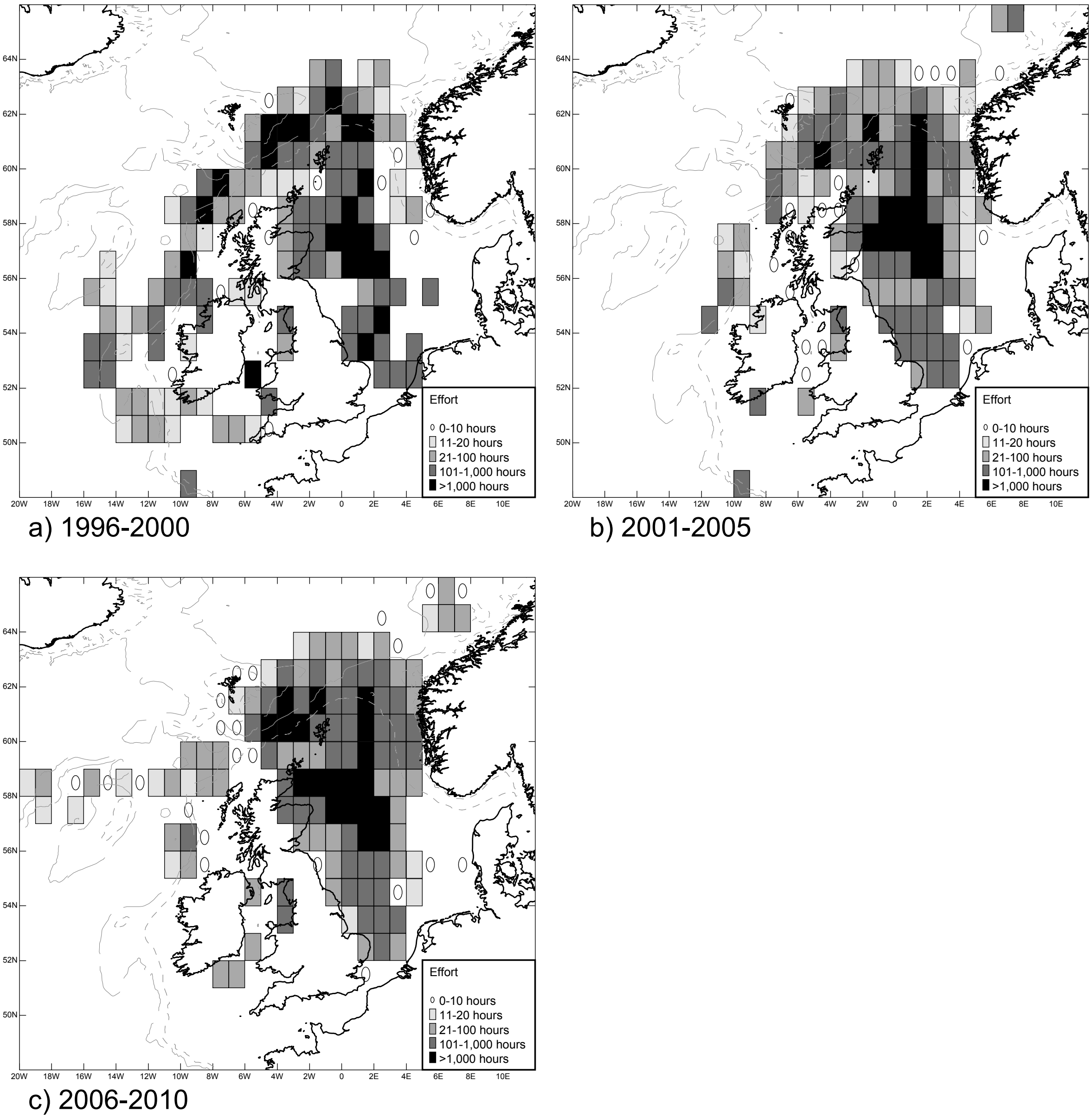

Figure 3.2. Effort (visual and PAM) during seismic surveys at different time periods (short dashed line = 200m isobath; long dashed line = 1,000m isobath).

Visual effort was mainly confined to the North Sea during the first quarter of the year, but was much more widespread during the second and third quarters, when weather conditions are usually more favourable for surveying in exposed areas such as deep waters to the west of Britain and Ireland (Figure 3.3). In the last quarter of the year conditions would again be expected to be suboptimal, but although surveying in deep water areas was limited, it was not as restricted as during the first quarter, probably due to unfinished surveys continuing from the summer months. Although visual effort was extensive in the North Sea throughout the year, the amount of effort was greatest in the second and third quarters and least in the first quarter. Visual effort is strongly influenced by available daylight, which is greatest in the second and third quarters and is particularly limited in northern areas during winter months, as well as by the extent of surveying.

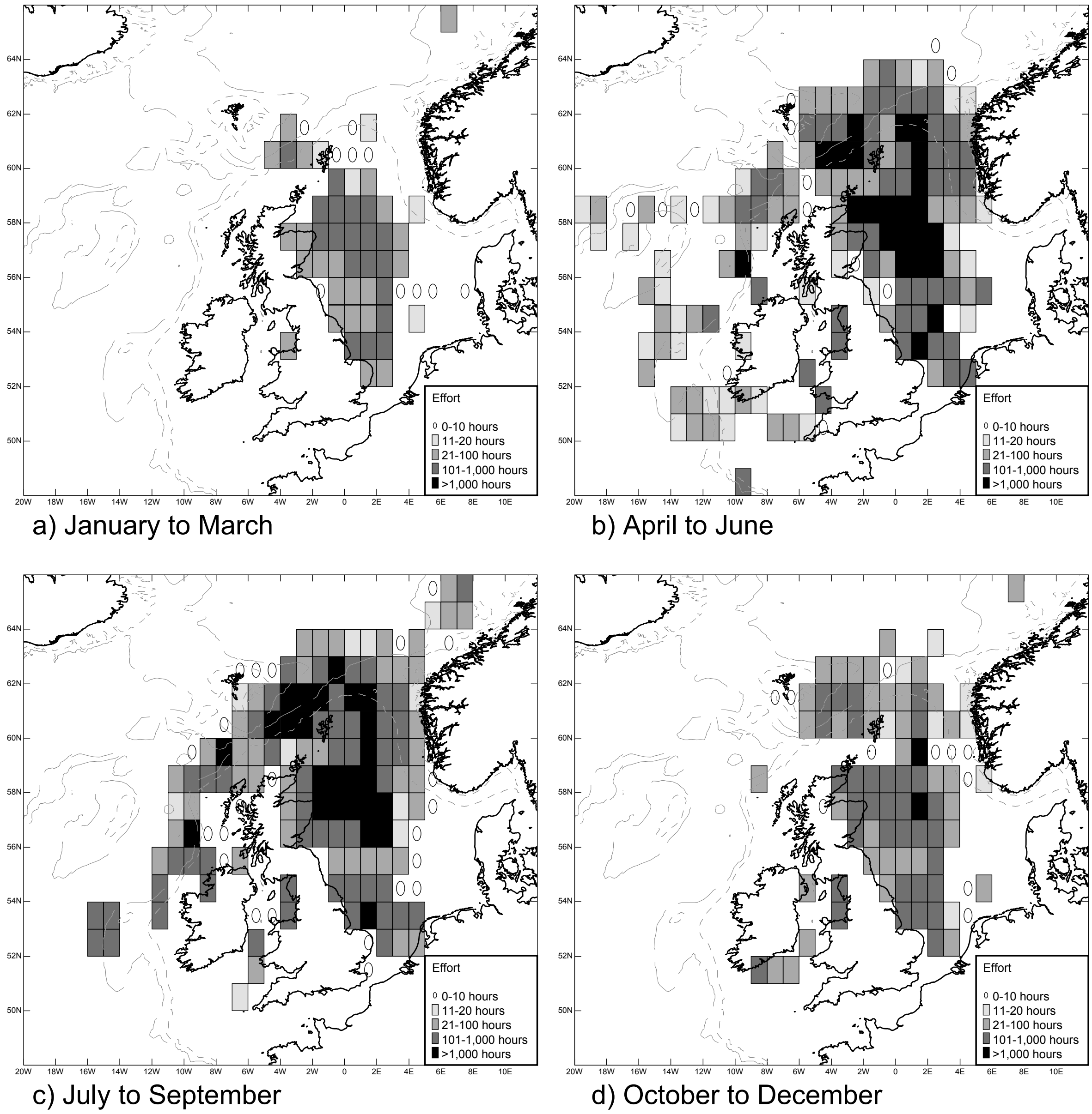

Figure 3.3. Visual effort by season during seismic surveys from 1996-2010 (short dashed line = 200m isobath; long dashed line = 1,000m isobath).

PAM effort continued to the west of Shetland throughout the year, although again it was reduced during the first quarter and greatest during the second and third quarters (Figure 3.4). PAM effort in the Outer Moray Firth and adjacent waters peaked during the third quarter.

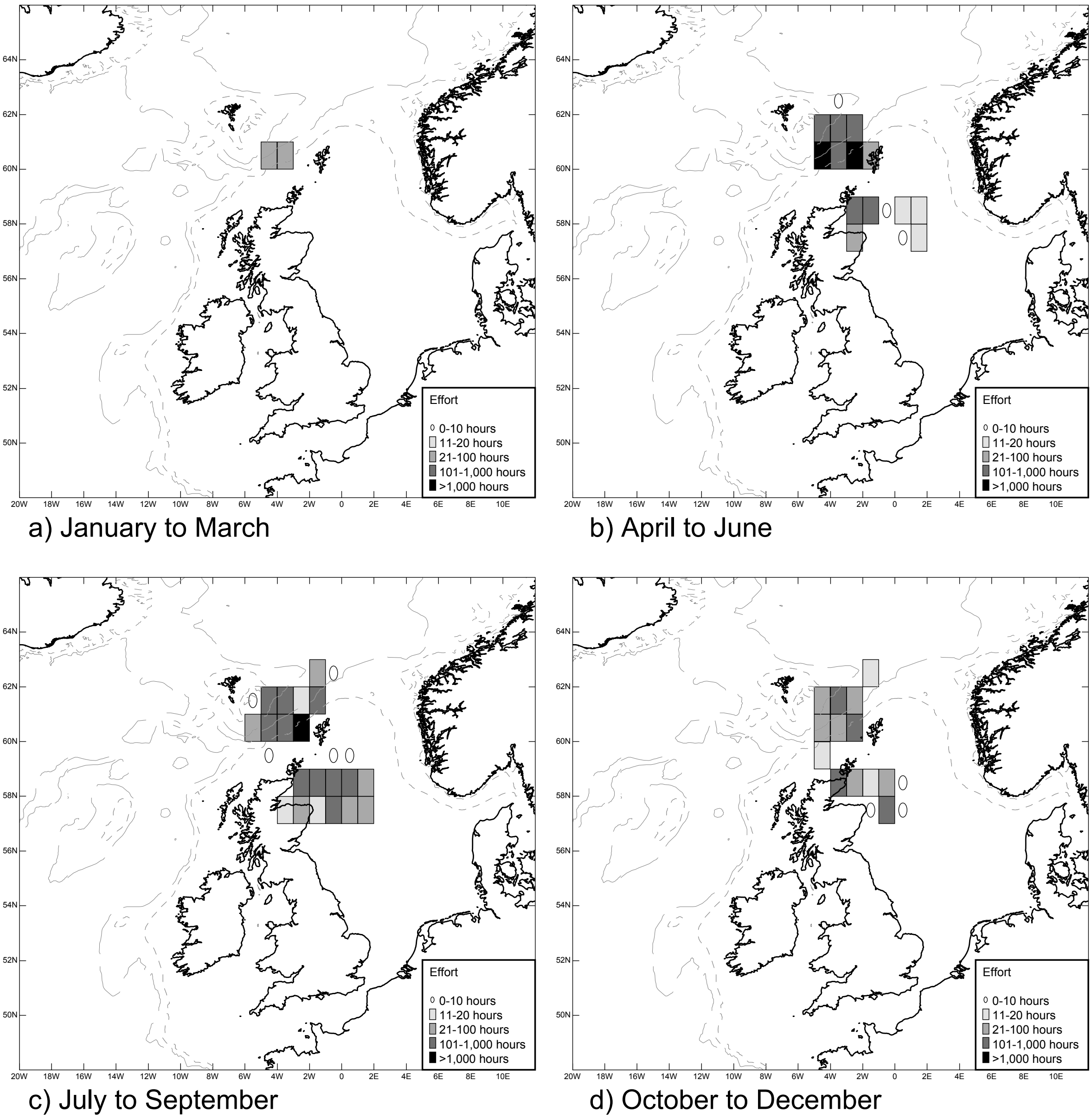

Figure 3.4. PAM effort by season during seismic surveys from 1996-2010 (short dashed line = 200m isobath; long dashed line = 1,000m isobath).

There were 9,073 sightings or acoustic detections of marine mammals during seismic surveys in UK and adjacent waters, comprising 124,024 individuals (Table 3.1). The most frequently encountered species of marine mammal identified was the white-beaked dolphin (an encounter being one or more animals occurring together). Atlantic white-sided dolphins, minke whales, sperm whales, harbour porpoises and long-finned pilot whales were also seen frequently, with fin whales, killer whales and short-beaked common dolphins seen moderately often. The most numerous species (number of individuals seen) was the Atlantic white-sided dolphin, followed by the white-beaked dolphin and then the long-finned pilot whale and short-beaked common dolphin, reflecting the often large number of animals in each pod of these species.

Multi-species associations were sometimes observed; 163 of the 9,073 sightings comprised more than one species. The species most commonly occurring in association with other species was the long-finned pilot whale (80 associations) followed by the Atlantic white-sided dolphin (66 associations). These two species together represented the most common combination of species found in association (35 associations). Long-finned pilot whales were also seen with unidentified dolphins on 28 occasions while Atlantic white-sided dolphins were also recorded as associating with fin whales (8 associations) and white-beaked dolphins (7 associations). Other combinations of species were recorded infrequently.

Table 3.1. Species of marine mammal encountered during seismic surveys in UK and adjacent waters from 1994-2010.

Species	No. sightings/ acoustic detections	No. individuals
Seal sp.	92	122
Grey seal	108	113
Harbour seal	23	24
Cetacean sp.	541	4,181
Whale sp.	301	681
Large whale sp.	207	425
Northern right whale (probable)	1	1
Humpback whale	22	48
Blue whale	13	14
Fin whale	342	789
Sei whale	23	34
Humpback/ sperm whale	21	26
Blue/ fin/ sei whale	18	30
Fin/ sei whale	127	252
Fin/ sei/ humpback whale	55	109
Fin/ sei/ blue/ humpback whale	169	370
Fin/ blue whale	42	83
Sperm whale	547	758
Medium whale sp.	81	133
Minke whale	724	854
Beaked whale sp.	9	21
Northern bottlenose whale	10	44
Minke/ northern bottlenose whale	1	1
Sowerby's beaked whale	6	14
Long-finned pilot whale	485	9,321
Killer whale	332	2,229
False killer whale	1	7
Long-finned pilot/ false killer whale	2	7
False killer whale/ killer whale/ Risso's dolphin	1	2
Delphinid sp. (dolphin, long-finned pilot, killer, false killer whale)	9	9
Dolphin sp.	1,614	20,451
Dolphin sp. (not porpoise)	65	550
Unpatterned dolphin (Risso's/ bottlenose)	5	28
Risso's dolphin	81	716
Bottlenose dolphin	101	1,349
Patterned dolphin (common/ striped/ white-beaked/ Atlantic white-sided)	108	2,328
White-beaked dolphin	1,166	16,169
Atlantic white-sided dolphin	727	45,926
Lagenorhynchus sp.	175	5,740
Short-beaked common dolphin	315	8,205
Striped dolphin	10	427
Short-beaked common/ striped dolphin	5	39
Short-beaked common/ striped/ Atlantic white-sided dolphin	1	4
Short-beaked common/ Atlantic white-sided dolphin	20	267
Harbour porpoise	539	1,123
Total	**9,073***	**124,024**

*includes some mixed species sightings

The distribution of sightings and acoustic detections of marine mammals to a large extent reflected the location of surveys and the amount of effort spent in observing or acoustic monitoring (Figure 3.5). There were many sightings and acoustic detections in the areas where there was most effort (the central and northern North Sea and to the west of Shetland). However, there were high numbers of sightings/ acoustic detections relative to effort throughout shelf edge and deep waters to the north and west of Britain and Ireland, in the Outer Moray Firth and in St George's Channel. Conversely, there were low numbers of sightings relative to effort in the southern North Sea, most likely to be correlated with variation in the distribution of species.

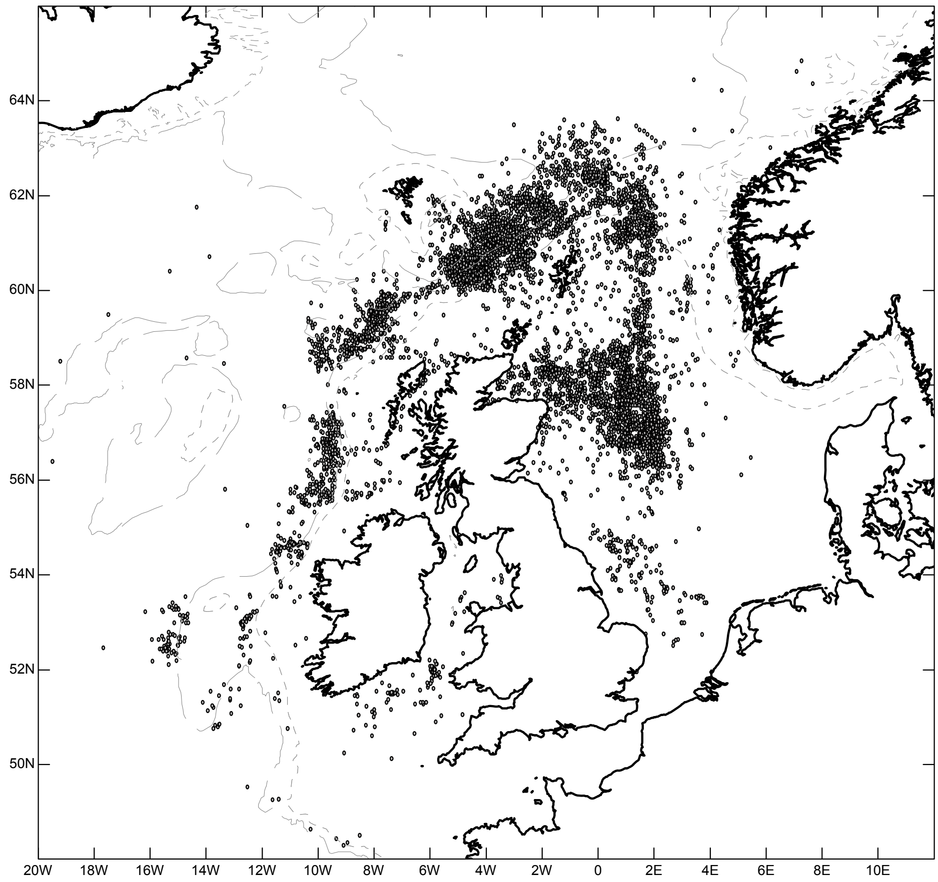

Figure 3.5. Sightings and acoustic detections of marine mammals during seismic surveys, 1994-2010 (short dashed line = 200m isobath; long dashed line = 1,000m isobath).

Individual species maps are included in Appendix 1 (Figure 8.1-Figure 8.19). The large rorqual whales (humpback, blue, fin and sei) were recorded mainly in deep waters and waters over the continental shelf edge (i.e. depths greater than 200m) to the north and west of Britain, with fin whales being the most commonly recorded of the large rorquals (Figure 8.1). In addition to sightings in deeper waters, there was also a sighting of a single fin whale in shelf waters to the east of Shetland and a single animal in the central North Sea (Figure 8.1). Also, a humpback whale was recorded close inshore on the east coast of Shetland (Figure 8.2) and there were occasional sightings of sei whales over the shelf to the north-east of Shetland (Figure 8.3). Few blue whales were seen, with all sightings being in shelf edge and deep waters (Figure 8.4) and there was a single sighting of a probable northern right whale near deep waters to the north of Shetland in 2000 (Figure 8.4). Minke whales, a medium-sized rorqual, were widespread throughout the central and northern North Sea and to the west of Shetland, in waters of all depths (Figure 8.5). Their distribution extended into the southern North Sea and to the west and south-west of Ireland.

Like the large rorqual whales, sperm whales were found in deep waters and waters over the continental shelf edge to the north and west of Britain, particularly in the Shetland-Faroes channel (Figure 8.6). There were also single sperm whales recorded in the central North Sea and in St George’s Channel. Similarly northern bottlenose whales (Figure 8.7) and Sowerby’s beaked whales (Figure 8.8) were also found in deep water and shelf edge areas in low numbers, although a single northern bottlenose whale was also seen close inshore off Aberdeen (Figure 8.7). Long-finned pilot whales were also distributed in deep waters and along the shelf edge, ranging throughout the area studied from the South-west Approaches to the north of Norway, although more were recorded in the northern half of the area (Figure 8.9). Their distribution also extended into the northern North Sea along the western edge of the Rinne (a channel between 200m and 500m depth that lies parallel to the south-western coast of Norway) and there were scattered occasional sightings over the shelf in the northern North Sea.

With the exception of some recorded in deep waters to the west of Ireland, all killer whales were recorded in the northern half of the area studied (Figure 8.10). There was a cluster of sightings of killer whales over the outer shelf and shelf edge to the north-east of Shetland and also a number of sightings over the shelf edge and deep waters to the west and north of Shetland. They were also recorded in lower numbers throughout the northern North Sea and Outer Moray Firth and extending into the central North Sea. There was one sighting of false killer whales to the west of Ireland (Figure 8.8).

Of the dolphin species, Risso's dolphins, Atlantic white-sided dolphins and striped dolphins seemed to prefer shelf edge and deep waters. Risso's dolphins were mainly recorded over the continental shelf edge to the west and north of Shetland, with some extending into deep waters and low numbers seen over the shelf edge to the west of Ireland (Figure 8.11). Scattered sightings also occurred in the Outer Moray Firth, the central and northern North Sea and St George's Channel. Atlantic white-sided dolphins were recorded frequently over the shelf edge and in deep waters to the west and north of Shetland and also over the outer shelf to the east of Shetland and along the western edge of the Rinne (Figure 8.12). A cluster of sightings also occurred in an area in the central and northern North Sea and Outer Moray Firth. Their distribution also extended along the shelf edge and deep waters to the west of Scotland and Ireland, with some sightings in shelf waters around the Hebrides. Striped dolphins were recorded infrequently, but mainly in deep waters to the west and north of Britain and Ireland (Figure 8.13). There was also one sighting in the central North Sea and one in the southern North Sea.

White-beaked dolphins were frequently recorded, with an apparent preference for shelf waters. Their distribution centred on an area in the central and northern North Sea and Outer Moray Firth, extending northwards from there over the outer shelf, shelf edge and deep waters to the west and north of Shetland and to a lesser extent southwards into the southern North Sea (Figure 8.14). There were scattered sightings in shelf waters to the west of Scotland and one sighting in St George's Channel. Harbour porpoises also seemed to prefer shelf waters, being widespread throughout the North Sea, including extending into the Moray Firth, but also extending in lower numbers into shelf edge and deeper waters to the north and west of Shetland (Figure 8.15). Low numbers were also recorded over the shelf to the west of Scotland, in the Irish Sea and St George's Channel.

Bottlenose dolphins and short-beaked common dolphins were more evenly split between shelf waters and the deeper waters over the shelf edge and beyond. Bottlenose dolphins mainly occurred in an area encompassing the central and northern North Sea and Outer Moray Firth and along the shelf edge and deep waters to the north and west of Britain and Ireland (Figure 8.16). Some were also recorded in St George's Channel and there were low numbers in the southern North Sea. Short-beaked common dolphins were frequently recorded in St George's Channel, but were also found over the outer shelf, shelf edge and deep waters from the west of Ireland to the north of Shetland (Figure 8.17). They were also recorded in the Outer Moray Firth and central and northern North Sea.

Grey seals were recorded throughout the Outer Moray Firth and the central and northern North Sea, often relatively close to land but also extending further offshore (Figure 8.18).

Low numbers were also recorded around Shetland, the southern North Sea and the Irish Sea. Harbour seals were recorded in the Outer Moray Firth and the northern North Sea, with low numbers in the southern and central North Sea (Figure 8.19).

Shifts in distribution over time were observed for fin whales and harbour porpoises. Fin whales were mainly encountered to the west of Shetland, but sighting rates were low in that area between 2006 and 2010 compared to previous years (Table 3.2). More harbour porpoises were recorded in the southern North Sea and fewer in the northern North Sea between 2006 and 2010 compared to previous years (Table 3.2). The distribution of other species both between years and throughout the year largely corresponded with the distribution of effort.

Table 3.2. Sighting rates of fin whales and harbour porpoises per 1,000 hours survey effort.

Species	Area	1996-2000	2001-2005	2006-2010
Fin whale				
	West of Shetland	25.97	17.63	1.77
Harbour porpoise				
	Northern North Sea	5.77	5.09	2.55
	Outer Moray Firth	0.00	7.57	6.71
	Central North Sea	1.69	6.01	4.54
	Southern North Sea	0.00	1.83	28.56

3.2 Effects of seismic operations on marine mammals

3.2.1 Detection rates (firing versus not firing)

On surveys with ‘large arrays’ detection rates were significantly higher when the airguns were not firing for the grey seal, minke whale, all beaked whales combined, killer whale, white-beaked dolphin, Atlantic white-sided dolphin and harbour porpoise (Figure 3.6Figure 3.6 and Table 3.3). On surveys with ‘small arrays’ detection rates were mostly similar whether the airguns were firing or not (Figure 3.7), but sperm whales and harbour porpoises were seen significantly less often when the airguns were firing (Table 3.3).

Table 3.3. Statistical significance of difference in detection rate of marine mammals in relation to airgun activity, using Wilcoxon signed ranks test (z = Wilcoxon statistic; for small samples T^+ = sum of ranks of pairs where detection rate when not firing exceeded detection rate when firing; n = sample size; P = probability; n.s. = not significant).

Species	z	T^+	N	P
'Large arrays'				
Grey seal	2.956	-	36	< 0.01
Harbour seal	-	25	9	n.s.
Humpback whale	-	16	7	n.s.
Fin whale	-0.444	-	103	n.s.
Sei whale	-	39	12	n.s.
Minke whale	3.093	-	281	< 0.001
All beaked whales	-	27	7	< 0.05
Sperm whale	1.528	-	116	n.s.
Long-finned pilot whale	0.639	-	127	n.s.
Killer whale	2.808	-	103	< 0.01
Risso's dolphin	1.039	-	31	n.s.
Bottlenose dolphin	1.176	-	31	n.s.
White-beaked dolphin	7.061	-	403	< 0.001
Atlantic white-sided dolphin	3.208	-	295	< 0.001
Short-beaked common dolphin	1.312	-	39	n.s.
Harbour porpoise	8.330	-	92	< 0.001
'Small arrays'				
All seals combined	-	5	7	n.s.
Grey seal	-	4	5	n.s.
All cetaceans combined	0.817	-	171	n.s.
All baleen whales combined	-1.272	-	32	n.s.
Minke whale	0.322	-	19	n.s.
Sperm whale	-	114	15	< 0.001
All delphinids combined	1.419	-	116	n.s.
Long-finned pilot whale	-	29	9	n.s.
Killer whale	-	7	5	n.s.
All small odontocetes combined	0.971	-	118	n.s.
Risso's dolphin	-	8	5	n.s.
White-beaked dolphin	0.327	-	18	n.s.
Atlantic white-sided dolphin	0.687	-	37	n.s.
Short-beaked common dolphin	-	10	6	n.s.
Harbour porpoise	-	21	6	< 0.05

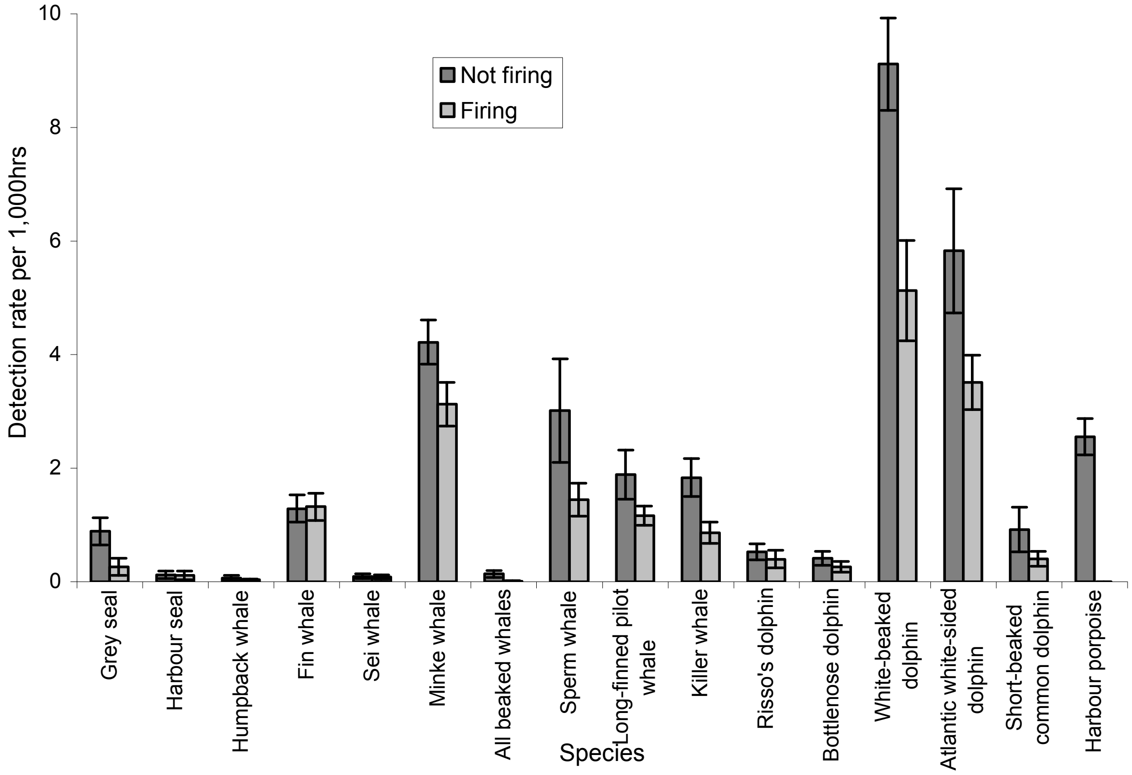

Figure 3.6. Mean detection rates (and standard error) of marine mammals in relation to airgun activity on surveys with ‘large arrays’.

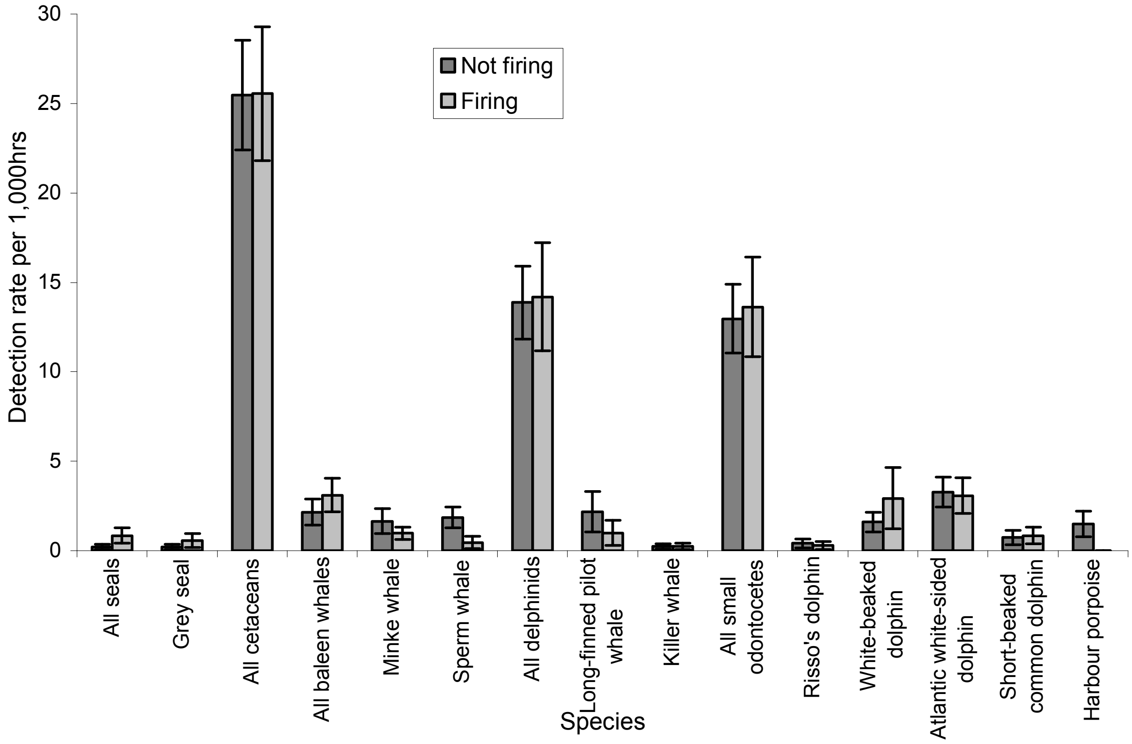

Figure 3.7. Mean detection rates (and standard error) of marine mammals in relation to airgun activity on surveys with ‘small arrays’.

3.2.2 Detection rate trends throughout the duration of surveys

Surveys with 'small arrays', where firing tended to be concentrated within a small area, were generally of short duration, with 46% lasting less than one week and only 17% lasting three or more weeks. The amount of time spent firing in each week of each survey was hugely variable, depending on factors such as weather, technical problems and time-sharing etc. For those surveys with 'small arrays' that did last at least three weeks, sighting rates of delphinids and small odontocetes decreased significantly after the first week of the survey (Table 3.4, Figure 3.8).

Table 3.4. Statistical significance of difference in sighting rate of marine mammals between the first and later weeks of surveys with 'small arrays', using the Wilcoxon signed ranks test (for small samples T^+ = sum of ranks of pairs where sighting rate during week one exceeded that of later weeks; n = sample size; P = probability; n.s. = not significant).

Species	T^+	n	P
All cetaceans combined	39	10	n.s.
All baleen whales combined	11	5	n.s.
All delphinids combined	31	8	< 0.05
All small odontocetes combined	32	8	< 0.05
Harbour porpoise	11	5	n.s.

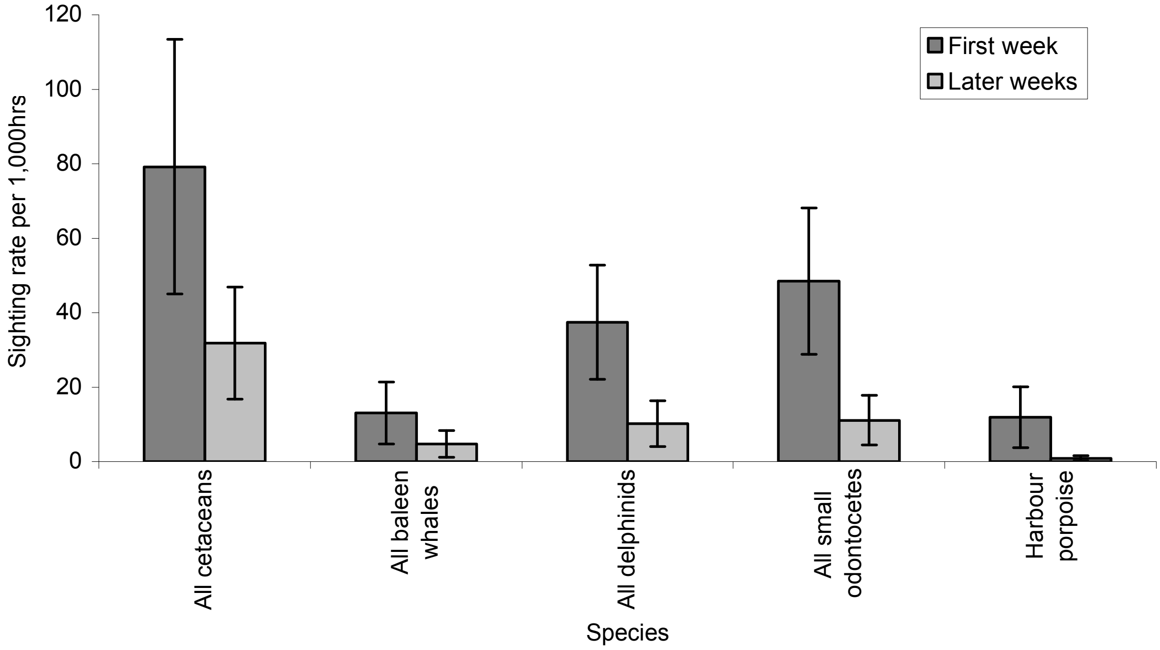

Figure 3.8. Mean sighting rate (and standard error) of marine mammals during the first week and later weeks of seismic surveys with 'small arrays'.

3.2.3 Closest distance of approach to the airguns (firing versus not firing)

On surveys with 'large arrays', marine mammals often approached closer to the airguns when they were not firing than when they were firing (Figure 3.9). This difference was statistically significant for all baleen whales combined (although not for fin whales or minke whales, the only individual baleen whale species for which sample sizes were sufficient to test), killer whales, bottlenose dolphins, white-beaked dolphins, Atlantic white-sided dolphins and the harbour porpoise (Table 3.5). For species where the results were significant, the difference in the median closest distance of approach between when the airguns were firing and when they were not firing ranged between 300m (Atlantic white-sided dolphin) and 1,500m (bottlenose dolphin).

Conversely, on surveys with 'small arrays' marine mammals often approached closer to the airguns during periods when the airguns were firing than when they were not firing (Figure 3.10), but this was only statistically significant for all baleen whales combined (Table 3.5), which on average were over 1km further away when the airguns were not firing.

Table 3.5. Statistical significance of difference in closest distance of approach of marine mammals to the airguns in relation to airgun activity, using the Wilcoxon-Mann-Whitney test (z = Wilcoxon statistic; for small samples W_x = sum of ranks of the smallest group; n = sample size; P = probability; n.s. = not significant).

Species	z	Wx	n	P
'Large arrays'				
Grey seal	0.000	-	27	n.s.
Harbour seal	-	33	10	n.s.
All baleen whales combined	9.283	-	477	< 0.001
Fin whale	1.382	-	107	n.s.
Minke whale	0.813	-	248	n.s.
Sperm whale	0.953	-	111	n.s.
Long-finned pilot whale	0.439	-	79	n.s.
Killer whale	2.099	-	81	< 0.05
Risso's dolphin	-0.281	-	23	n.s.
Bottlenose dolphin	-1.799	-	12	< 0.05
White-beaked dolphin	6.075	-	302	< 0.001
Atlantic white-sided dolphin	3.133	-	213	< 0.001
Short-beaked common dolphin	1.420	-	16	n.s.
Harbour porpoise	3.065	-	126	< 0.01
'Small arrays'				
All cetaceans combined	-0.953	-	136	n.s.
All baleen whales combined	-2.311	-	25	< 0.05
Minke whale	-0.187	-	14	n.s.
All delphinids combined	-0.428	-	66	n.s.
All small odontocetes combined	-0.530	-	72	n.s.
Atlantic white-sided dolphin	-0.147	-	18	n.s.

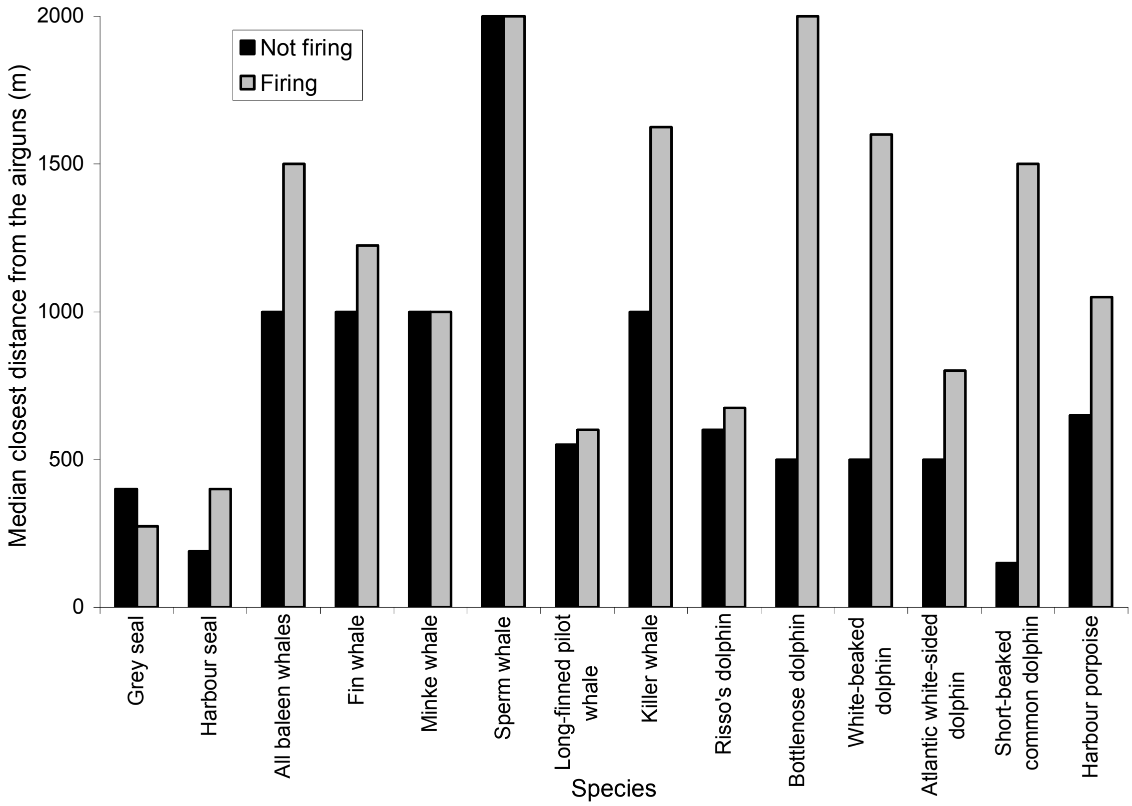

Figure 3.9. Median closest distance of marine mammals from the airguns in relation to airgun activity on surveys with 'large arrays'.

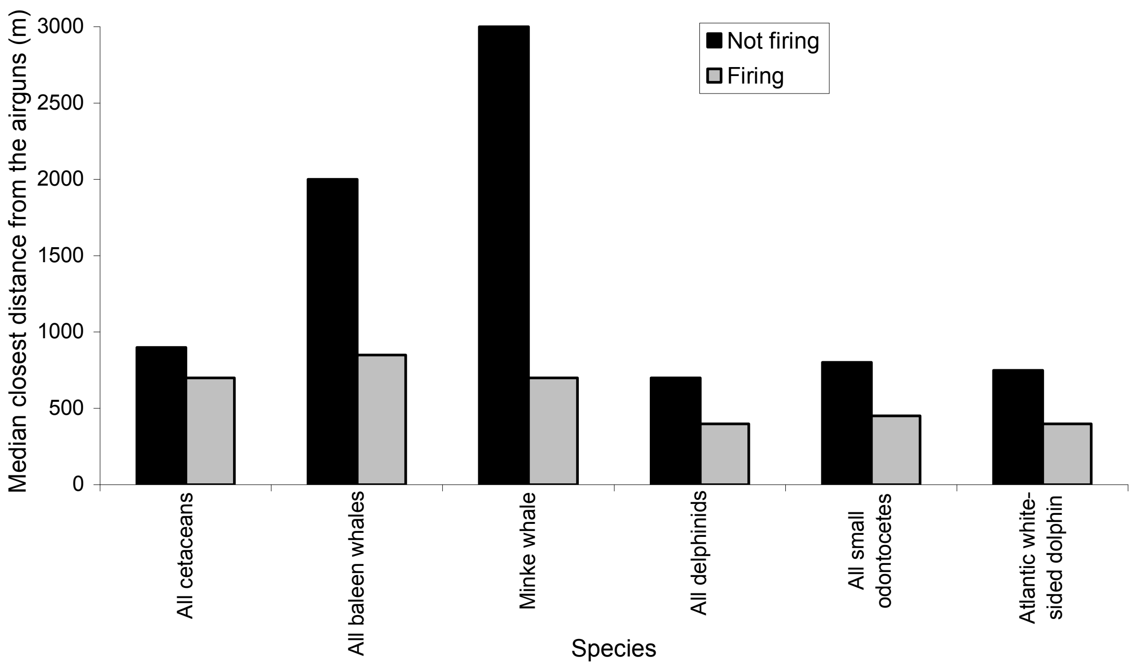

Figure 3.10. Median closest distance of marine mammals from the airguns in relation to airgun activity on surveys with 'small arrays'.

3.2.4 Behaviour

Firing of ‘large arrays’ affected the movement of cetaceans around the vessel (Table 3.6). Long-finned pilot whales, white-beaked dolphins and the combined groups of all delphinids and all small odontocetes engaged in positive interactions with the vessel or its equipment (e.g. bow-riding, approaching close to the vessel) or travelled towards the vessel more often when the airguns were silent. On surveys with ‘large arrays’, significantly more pods of fin whales, minke whales, long-finned pilot whales, white-beaked dolphins, Atlantic white-sided dolphins and harbour porpoises avoided or travelled away from the vessel during periods when the airguns were firing compared to when they were not firing. Minke whales, long-finned pilot whales and *Lagenorhynchus* spp. also altered course (in any direction, not necessarily to avoid the vessel) more often when the airguns were firing.

Minke whales, bottlenose dolphins, white-beaked dolphins and short-beaked common dolphins were more often recorded as swimming fast when ‘large arrays’ were firing and similarly *Lagenorhynchus* spp. and all baleen whales combined were more often recorded as swimming slowly when the airguns were not firing (Table 3.6). Conversely, long-finned pilot whales were more often recorded as swimming slowly when the airguns were firing. Bottlenose dolphins were more likely to breach or jump when the airguns were firing and white-beaked dolphins were more often recorded as splashing when the airguns were firing (often concurrent with fast swimming). When all cetaceans were combined fewer were porpoising during periods of firing. There was no apparent effect of firing on the tendency of cetaceans to swim in close groups or more widely spread groups, nor was there any correlation with tail-slapping or spy-hopping. The predominant behaviours exhibited by seals were bottling, diving or slow swimming but there was no significant difference in the prevalence of these behaviours in relation to airgun activity.

Other effects on surfacing/ diving behaviours were also apparent. There were indications that cetaceans may have remained close to the water surface when ‘large arrays’ were active. All cetaceans combined, all baleen whales combined and minke whales were more often recorded as surfacing frequently during periods when ‘large arrays’ were firing (Table 3.6). Although minke whales and other baleen whales contributed to the result for all cetaceans, these were not the only species recorded as surfacing frequently. Similarly, when all cetaceans were combined more were found to be logging or apparently resting at the surface when ‘large arrays’ were firing. Milling, where animals continue to surface in the same general vicinity, was more prevalent in baleen whales during periods of firing. A group including all delphinids were more often recorded both as diving and logging/ resting during periods of firing (although neither behaviour was observed more often during firing in those delphinid species that were able to be tested individually).

Cetaceans were sometimes recorded as feeding, although those animals showing behaviours indicative of feeding (e.g. holding a fish, lunging, or erratic swimming with birds diving alongside) would represent only a proportion of the animals actually feeding. Most species were recorded as feeding less often when ‘large arrays’ were firing (Table 3.6). Whilst the difference was not statistically significant for individual species, it was when all cetaceans were combined. This result was heavily influenced by delphinid species, but other species (e.g. fin whale, minke whale, harbour porpoise) were also observed feeding and contributed to the result.

Fewer effects on behaviour were evident with ‘small arrays’. When species were combined it was apparent that positive interactions with the vessel or its equipment or travel towards the vessel occurred more often when the airguns were not firing, while avoidance or travel away was more prevalent when the airguns were firing (Table 3.6). While with ‘large arrays’ there were some indications that cetaceans may sometimes remain near the surface when the airguns are firing, with ‘small arrays’ cetaceans (all species combined) were more often recorded as surfacing infrequently during periods of firing, i.e. they were remaining submerged more.

Table 3.6. Behaviour of marine mammals in relation to airgun activity (n = sample size; P = probability; n.s. = not significant).

Behaviour	Species	% of encounters while firing when behaviour was exhibited	% of encounters while not firing when behaviour was exhibited	χ^2	n	P
'Large arrays'						
Altered course	Fin whale	8.3	4.5	1.89	20	n.s.
	Minke whale	4.7	1.7	4.85	17	< 0.05
	Long-finned pilot whale	13.0	5.9	6.02	41	< 0.05
	Lagenorhynchus spp.	4.4	1.5	13.04	43	< 0.001
	Atlantic white-sided dolphin	5.6	3.0	2.32	24	n.s.
Avoidance or travel away from vessel/ equipment	Fin whale	24.3	14.6	3.95	61	< 0.05
	Minke whale	16.3	8.2	8.44	70	< 0.01
	Sperm whale	19.7	18.7	0.05	68	n.s.
	Long-finned pilot whale	13.9	5.1	9.49	41	< 0.01
	Killer whale	18.1	11.9	1.70	41	n.s.
	White-beaked dolphin	19.2	8.2	22.24	115	< 0.001
	Atlantic white-sided dolphin	12.2	6.3	5.80	51	< 0.05
	Harbour porpoise	37.5	20.0	7.78	82	< 0.01
Bottling	Grey seal	36.8	31.3	0.14	28	n.s.
Breaching, jumping, somersaulting	Minke whale	8.4	5.9	1.31	43	n.s.
	Long-finned pilot whale	4.3	5.9	0.53	23	n.s.
	Killer whale	12.1	7.8	1.24	27	n.s.
	Risso's dolphin	33.3	16.3	2.01	15	n.s.
	Bottlenose dolphin	69.2	32.1	5.43	33	< 0.05
	Lagenorhynchus spp.	41.8	37.5	1.80	706	n.s.
	White-beaked dolphin	38.8	33.2	1.84	359	n.s.
	Atlantic white-sided dolphin	46.4	44.6	0.10	282	n.s.
	Short-beaked common dolphin	40.0	25.7	1.96	68	n.s.
Close group	Long-finned pilot whale	2.4	3.0	0.12	12	n.s.
	White-beaked dolphin	2.5	2.1	0.17	23	n.s.
	Atlantic white-sided dolphin	3.1	3.5	0.08	21	n.s.
Dispersed group	Long-finned pilot whale	5.3	2.5	2.20	17	n.s.
	White-beaked dolphin	3.3	3.0	0.04	32	n.s.
	Atlantic white-sided dolphin	7.1	12.6	3.70	68	n.s.
Diving	Grey seal	47.4	25.4	2.38	26	n.s.
	Fin whale	12.5	11.2	0.11	38	n.s.
	Minke whale	7.0	10.4	1.77	59	n.s.
	Sperm whale	43.0	50.0	0.90	168	n.s.
	All delphinids combined	2.2	1.4	4.10	75	< 0.05
	Long-finned pilot whale	5.8	3.4	1.41	20	n.s.
	Lagenorhynchus spp.	1.3	1.2	0.04	23	n.s.
Fast swimming	Fin whale	9.7	5.6	1.80	322	n.s.
	Minke whale	20.5	8.2	17.30	79	< 0.001
	Long-finned pilot whale	14.4	10.6	1.34	55	n.s.
	Killer whale	20.5	11.4	3.56	42	n.s.
	Bottlenose dolphin	46.2	18.9	4.66	22	< 0.05
	White-beaked dolphin	33.3	25.7	4.31	287	< 0.05
	Atlantic white-sided dolphin	53.1	45.3	1.68	298	n.s.
	Short-beaked common dolphin	63.3	24.3	13.83	72	< 0.001
	Harbour porpoise	26.3	20.0	1.11	73	n.s.
Feeding	All cetaceans combined	8.2	10.3	7.85	706	< 0.01
	Fin whale	9.7	12.9	0.71	37	n.s.
	Minke whale	0.9	3.1	2.76	15	n.s.
	Long-finned pilot whale	6.3	8.0	0.48	32	n.s.
	Killer whale	16.9	26.5	2.34	72	n.s.
	White-beaked dolphin	12.3	11.1	0.28	118	n.s.
	Atlantic white-sided dolphin	19.9	23.4	0.73	139	n.s.
In subgroups	All cetaceans combined	0.3	0.5	0.78	31	n.s.
	All delphinids combined	0.5	0.7	0.56	29	n.s.
	Lagenorhynchus spp.	1.0	1.1	0.05	19	n.s.
Logging/ resting	All cetaceans combined	3.7	2.6	6.81	216	< 0.01
	Sperm whale	35.9	29.9	0.95	115	n.s.
	All delphinids combined	2.1	1.3	4.18	72	< 0.05
	Long-finned pilot whale	8.7	8.9	0.01	39	n.s.
	All small odontocetes combined	1.0	0.8	0.31	33	n.s.

Behaviour	Species	% of encounters while firing when behaviour was exhibited	% of encounters while not firing when behaviour was exhibited	χ^2	n	P
Milling	All baleen whales combined	3.0	0.7	10.16	22	< 0.001
	Lagenorhynchus spp.	2.5	3.1	0.43	53	n.s.
	White-beaked dolphin	2.5	3.2	0.26	31	n.s.
	Atlantic white-sided dolphin	3.1	2.8	0.03	18	n.s.
Porpoising	All cetaceans combined	8.3	10.0	5.49	694	< 0.05
	Long-finned pilot whale	4.3	5.1	0.13	21	n.s.
	White-beaked dolphin	16.7	12.9	2.04	144	n.s.
	Atlantic white-sided dolphin	28.1	33.4	1.21	198	n.s.
	Short-beaked common dolphin	23.3	25.2	0.04	62	n.s.
Positive Interactions with or travel towards vessel/ equipment	All baleen whales combined	3.9	6.1	3.22	72	n.s.
	Minke whale	6.1	6.8	0.13	42	n.s.
	All delphinids combined	9.5	18.1	47.60	725	< 0.001
	All small odontocetes combined	8.9	18.3	45.26	613	< 0.001
	Long-finned pilot whale	15.4	27.0	6.93	96	< 0.01
	White-beaked dolphin	15.2	37.1	31.02	324	< 0.001
	Atlantic white-sided dolphin	7.7	13.1	3.48	71	n.s.
	Short-beaked common dolphin	23.3	25.7	0.06	63	n.s.
Resurfaced	All cetaceans combined	0.3	0.2	0.13	16	n.s.
Slow swimming	All seals combined	8.2	11.9	0.47	20	n.s.
	All baleen whales combined	19.0	24.2	4.02	304	< 0.05
	Fin whale	16.7	16.3	0.01	53	n.s.
	Minke whale	21.9	26.6	1.28	160	n.s.
	Sperm whale	21.8	18.7	0.42	71	n.s.
	Long-finned pilot whale	51.4	33.8	8.24	187	< 0.01
	Killer whale	20.5	25.6	0.64	73	n.s.
	Risso's dolphin	50.0	34.9	0.87	27	n.s.
	Bottlenose dolphin	15.4	28.3	1.21	19	n.s.
	Lagenorhynchus spp.	6.9	12.3	10.04	196	< 0.01
	White-beaked dolphin	6.5	10.4	3.24	97	n.s.
	Atlantic white-sided dolphin	8.2	13.6	3.29	74	n.s.
	Harbour porpoise	32.5	26.2	0.89	94	n.s.
Splashing	White-beaked dolphin	8.7	4.5	6.45	58	< 0.05
	Atlantic white-sided dolphin	5.6	3.0	2.32	24	n.s.
Spy-hopping	Long-finned pilot whale	5.8	6.3	0.06	27	n.s.
Surfacing frequently	All cetaceans combined	1.6	1.0	6.30	87	< 0.05
	All baleen whales combined	4.8	2.2	7.07	44	< 0.01
	Minke whale	5.1	1.7	6.10	18	< 0.05
	All delphinids combined	0.4	0.5	0.11	22	n.s.
Surfacing infrequently	Fin whale	6.9	8.4	0.23	25	n.s.
	Minke whale	15.4	14.4	0.10	94	n.s.
	Sperm whale	4.9	2.8	1.05	13	n.s.
	Lagenorhynchus spp.	1.3	2.1	1.07	34	n.s.
	Harbour porpoise	7.5	6.5	0.08	23	n.s.
Swimming at or just below surface	All cetaceans combined	0.5	0.4	0.15	30	n.s.
Tail-slapping	Long-finned pilot whale	1.9	3.8	1.34	13	n.s.
	Lagenorhynchus spp.	1.0	1.7	1.34	27	n.s.
'Small arrays'						
Avoidance or travel away from vessel/ equipment	All cetaceans combined	18.0	8.7	12.42	103	< 0.001
	All delphinids combined	17.3	6.7	11.27	51	< 0.001
	All small odontocetes combined	22.7	9.3	11.40	61	< 0.001
Breaching, jumping, somersaulting	All small odontocetes combined	34.1	24.6	2.56	138	n.s.
	Atlantic white-sided dolphin	42.3	33.3	0.43	35	n.s.
Diving	All cetaceans combined	6.6	7.2	0.09	70	n.s.
Fast swimming	White-beaked dolphin	46.7	33.0	0.71	45	n.s.
	Atlantic white-sided dolphin	46.2	33.3	0.86	36	n.s.
Feeding	All cetaceans combined	8.2	9.2	0.16	89	n.s.
	All small odontocetes combined	12.5	10.5	0.28	57	n.s.
Porpoising	All small odontocetes combined	17.1	10.7	2.53	62	n.s.
	Atlantic white-sided dolphin	19.2	23.6	0.16	22	n.s.

Behaviour	Species	% of encounters while firing when behaviour was exhibited	% of encounters while not firing when behaviour was exhibited	χ^2	n	P
Positive Interactions with or travel towards vessel/ equipment	All cetaceans combined	11.5	26.2	13.84	232	< 0.001
	All delphinids combined	11.6	36.1	16.35	190	< 0.001
	All small odontocetes combined	13.6	39.3	13.80	185	< 0.001
	Lagenorhynchus spp.	16.3	49.5	8.94	105	< 0.01
	White-beaked dolphin	46.7	67.0	0.84	84	n.s.
Slow swimming	All baleen whales combined	25.0	16.8	0.17	30	n.s.
	All small odontocetes combined	13.6	12.5	0.07	67	n.s.
Surfacing infrequently	All cetaceans combined	8.7	4.2	6.00	50	< 0.05

3.3 Effectiveness of the soft start

Detection rates differed significantly with source activity (not firing versus soft start versus full power) for all species or species groups that were able to be tested (Table 3.7). In all cases, detection rates during the soft start were significantly lower than when the airguns were not firing (Table 3.8, Figure 3.11). Although the mean detection rates for delphinids shown in Figure 3.11 are similar when the airguns were not firing and during the soft start, the Wilcoxon statistic measures both the direction and the magnitude of the difference and in this case the vast majority (83%) of samples had a lower detection rate during the soft start. Detection rates of baleen whales (all species combined) and minke whales were also lower during the soft start than when the airguns were firing at full power.

Table 3.7. Statistical significance of difference in detection rate of marine mammals in relation to airgun activity (differentiating the soft start from full power) using Friedman two-way analysis of variance by ranks (F_r = Friedman statistic; n = sample size; P = probability; n.s. = not significant).

Species	F_r	n	P
All cetaceans combined	36.873	723	< 0.001
All baleen whales combined	11.438	723	< 0.01
Minke whale	6.077	723	< 0.05
All delphinids combined	18.919	723	< 0.001
All small odontocetes combined	23.182	723	< 0.001
White-beaked dolphin	20.000	723	< 0.001
Atlantic white-sided dolphin	10.500	723	< 0.01

Table 3.8. Multiple comparisons of treatments using the Wilcoxon signed ranks tests (z = Wilcoxon statistic; T^+ = statistic for small samples; n = sample size; P = probability; n.s. = not significant).

	Not firing versus soft start			Soft start versus full power		
Species	z	n	P	z	n	P
All cetaceans combined	3.675	57	< 0.001	-1.301	34	n.s.
All baleen whales combined	$T^+ = 55$	10	< 0.001	$T^+ = 0$	8	< 0.01
Minke whale	$T^+ = 15$	5	< 0.05	$T^+ = 0$	5	< 0.05
All delphinids combined	2.651	41	< 0.01	-0.530	24	n.s.
All small odontocetes combined	3.238	39	< 0.001	-1.169	21	n.s.
White-beaked dolphin	$T^+ = 55$	10	< 0.001	-	0	-
Atlantic white-sided dolphin	$T^+ = 45$	9	< 0.01	$T^+ = 0$	3	n.s.

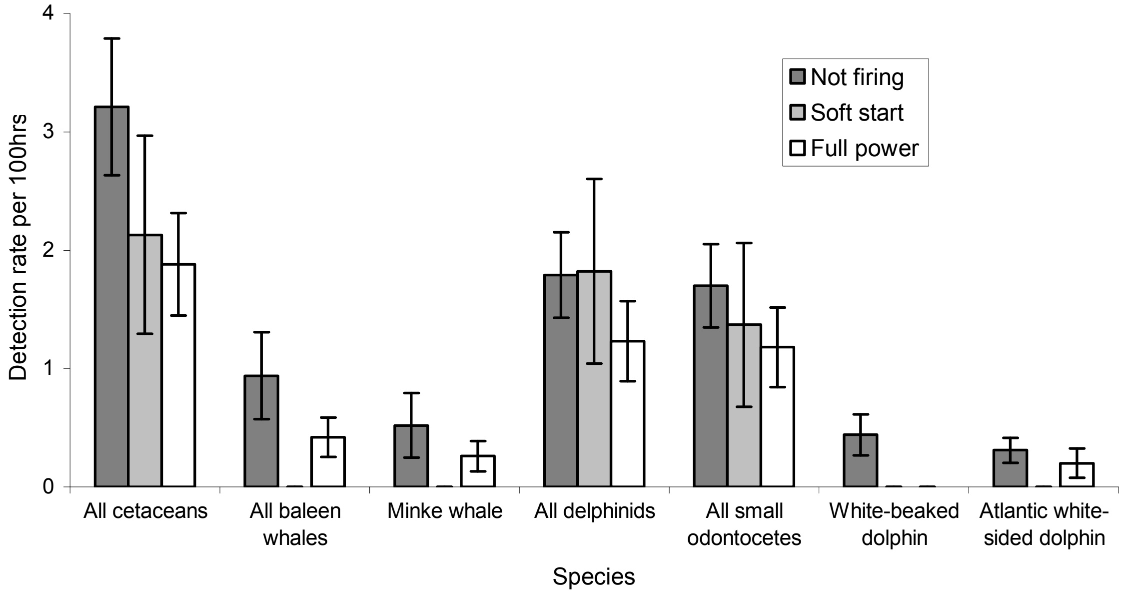

Figure 3.11. Mean detection rate (and standard error) of marine mammals in relation to airgun activity.

The closest distance that marine mammals approached the airguns differed significantly with source activity on surveys with 'large arrays' for the majority of species or species groups tested, with the exception of the minke whale (Table 3.9). Multiple comparisons of treatments to determine where the differences lay showed that for all species or species groups where the result was significant the closest distance during the soft start did not differ significantly from the closest distance at other times, but animals remained significantly further away from the airguns when they were firing at full power than when they were not firing (Figure 3.12). Sample sizes for surveys with 'small arrays' were lower, but when all cetaceans were combined there were no significant differences in the closest distance of approach with source activity (Table 3.9).

Table 3.9. Statistical significance of difference in closest distance of approach of marine mammals to the airguns in relation to airgun activity (not firing versus soft start versus full power) using the Kruskal-Wallis one-way analysis of variance by ranks (KW = Kruskal-Wallis statistic; n = sample size; P = probability; n.s. = not significant; degrees of freedom = 2 in all cases).

Species	KW	n	P
'Large arrays'			
All cetaceans combined	82.183	2,927	< 0.001
All baleen whales combined	20.898	613	< 0.001
Minke whale	5.965	342	n.s.
All delphinids combined	42.615	1,682	< 0.001
All small odontocetes combined	67.525	1,566	< 0.001
Lagenorhynchus spp.	62.672	721	< 0.001
White-beaked dolphin	44.825	391	< 0.001
Atlantic white-sided dolphin	18.045	263	< 0.001
'Small arrays'			
All cetaceans combined	4.061	296	n.s.

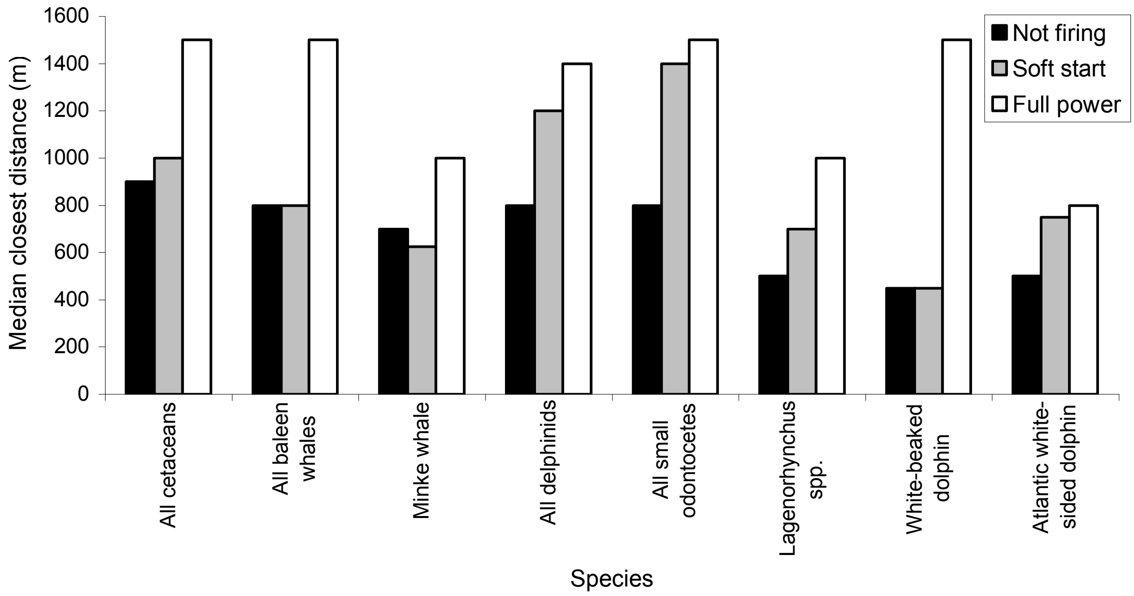

Figure 3.12. Median closest distance of approach of marine mammals to the airguns in relation to airgun activity (differentiating the soft start from full power) on surveys with 'large arrays'.

Some behaviours differed significantly with source activity on surveys with 'large arrays'. All species groups that could be tested showed an increased tendency to avoid or travel away from the vessel during the soft start than at any other time (Table 3.10), although not all animals did display such behaviours. All species and species groups tested also showed a reduced tendency to engage in positive interactions with the survey vessel or its equipment (e.g. bow-riding, etc.) or travel towards the vessel during the soft start than when the airguns were not firing and a further reduction when the airguns were firing at full power (Table 3.10). Some significant differences in swimming speed, aerial behaviours (such as breaching) and feeding were also apparent. It appeared that the soft start elicited increased avoidance and a decrease in positive interactions with the vessel or its equipment and in some cases also an increase in swimming speed (or fewer slow swimming behaviours). At full power positive interactions declined further, with some further increases in swimming speed, cetaceans (all species combined) engaged in feeding less often and small odontocetes were more likely to exhibit aerial behaviours. On surveys with 'small arrays' cetaceans (all species combined) again showed a reduced tendency to engage in positive interactions with the survey vessel or its equipment or travel towards the vessel during the soft start than when the airguns were not firing and a further reduction when the airguns were firing at full power (Table 3.10).

Table 3.10. Behaviour of marine mammals in relation to source activity (differentiating the soft start from full power) (n = sample size; P = probability; n.s. = not significant; d.f. = 2 in all cases).

Behaviour and Species	% of encounters while not firing when behaviour was exhibited	% of encounters during soft start when behaviour was exhibited	% of encounters while firing at full power when behaviour was exhibited	χ^2	n	P
'Large arrays'						
Avoidance or travel away from vessel						
All cetaceans combined	10.0	20.5	17.9	88.25	975	< 0.001
All delphinids combined	8.3	18.5	16.4	61.72	484	< 0.001
All small odontocetes combined	8.9	22.4	17.7	65.28	483	< 0.001
Lagenorhynchus spp.	7.7	24.5	16.9	35.68	186	< 0.001
Breaching, jumping or somersaulting						
All cetaceans combined	19.2	19.9	19.6	0.15	1,491	n.s.
All delphinids combined	28.9	29.6	31.3	1.71	1,321	n.s.
All small odontocetes combined	28.5	28.0	35.6	14.78	1,272	< 0.001
Lagenorhynchus spp.	37.4	32.7	41.9	2.21	679	n.s.
White-beaked dolphin	33.0	26.9	38.8	2.16	346	n.s.
Atlantic white-sided dolphin	44.8	41.2	46.8	0.17	271	n.s.
Dived						
All cetaceans combined	5.3	8.5	6.0	3.82	432	n.s.
Feeding						
All cetaceans combined	9.4	9.1	7.1	9.89	669	< 0.01
All delphinids combined	13.1	11.1	11.1	2.82	555	n.s.
All small odontocetes combined	11.3	10.3	11.5	0.15	475	n.s.
Lagenorhynchus spp.	14.4	12.2	15.1	0.28	257	n.s.
Positive interactions or travel towards the vessel						
All cetaceans combined	13.5	10.2	6.7	66.92	873	< 0.001
All delphinids combined	18.9	12.0	9.2	54.51	710	< 0.001
All small odontocetes combined	16.7	11.2	7.9	43.60	600	< 0.001
Lagenorhynchus spp.	27.3	18.4	9.2	39.71	404	< 0.001
White-beaked dolphin	36.9	26.9	13.3	32.65	314	< 0.001
Swimming fast						
All cetaceans combined	18.9	26.1	21.8	10.23	1,541	< 0.01
All delphinids combined	26.3	27.8	31.3	8.07	1,239	< 0.05
All small odontocetes combined	26.9	31.8	34.7	16.97	1,218	< 0.001
Lagenorhynchus spp.	33.6	30.6	42.4	7.50	632	< 0.05
White-beaked dolphin	25.5	26.9	33.8	4.50	278	n.s.
Atlantic white-sided dolphin	46.3	41.2	52.7	1.20	287	n.s.
Swimming slowly						
All cetaceans combined	15.4	11.9	16.2	2.21	1,201	n.s.
All baleen whales combined	24.3	10.0	19.8	4.78	299	n.s.
All delphinids combined	13.9	10.2	15.8	4.09	639	n.s.
All small odontocetes combined	11.9	10.3	9.6	3.90	473	n.s.
Lagenorhynchus spp.	12.2	4.1	6.7	11.53	186	< 0.01
'Small arrays'						
Positive interactions or travel towards the vessel						
All cetaceans combined	25.4	12.0	9.0	18.93	227	< 0.001

There were 84 encounters where marine mammals were first detected when the airguns were not firing but were still present when the soft start commenced. On 15 of these encounters (18%) responses were observed concurrent with the soft start commencing that could constitute a startle response. These responses included altering course to avoid the vessel, increasing swimming speed, diving, resurfacing after having dived, leaping, porpoising, spy-hopping, raising tail flukes and disappearing. Although there was one instance where animals that initially moved away at the onset of the soft start subsequently re-approached, there was no evidence that returning towards the vessel during the soft start was a common occurrence. Observed responses were not always consistent, demonstrating that different individuals may respond differently, e.g. one sperm whale was observed to dive when the soft start commenced while on another occasion a sperm whale that had recently dived was observed to re-surface and proceeded to swim at speed along the surface. Responses were observed both close to and further away from the airguns, up to 3km away.

In some other cases behaviours such as avoidance of the vessel, spy-hopping and tail-slapping were recorded but it was not made clear whether this was before or after the soft start began. Equally some animals were recorded as bow-riding but again it was unclear whether this was before or during the soft start. Diving was the only behaviour where sample sizes were sufficient to test the prevalence of the behaviour between encounters where the airguns were not firing throughout, those where the soft start commenced during the course of the encounter or those where the airguns were performing a soft start throughout (surveys with arrays of any size were included). More cetaceans were observed to dive if the soft start commenced during the encounter, with the difference being statistically significant (Table 3.11).

Table 3.11. Behaviour of marine mammals in relation to whether the soft start commenced during the encounter or not (n = sample size; P = probability; d.f. = 2).

Behaviour and Species	% of encounters while not firing when behaviour was exhibited	% of encounters during which the soft start commenced when behaviour was exhibited	% of encounters wholly during soft start when behaviour was exhibited	χ^2	n	P
Dived						
All cetaceans combined	5.7	10.3	9.0	6.67	358	< 0.05

3.4 General trends in compliance with the JNCC guidelines

The proportion of pre-shooting searches of adequate duration (at least 30 minutes, or at least 60 minutes in deep waters since 2009) has shown no major trends over time, ranging between 76% and 93%. The proportion of adequate soft starts has shown an increase since 2004 when alternative methods of performing a soft start were introduced for site surveys and VSP operations (Figure 3.13). Prior to this, on most site surveys no soft start was undertaken, even though this was not always agreed with the regulator and JNCC, hence the need for the guideline revisions to offer alternative soft start methods for such survey types.

There were 165 occasions when firing was required to be delayed in UK waters since the introduction of the guidelines in 1995 until the end of 2010. Delays were required most often due to the presence of white-beaked dolphins or Atlantic white-sided dolphins (the two species sighted overall most commonly from seismic survey vessels) in the mitigation zone, followed by unidentified dolphins and harbour porpoises. There were fewer delays due to other cetacean species and delays due to the presence of seals were uncommon. There was some evidence that delays were more likely at the beginning of a survey; since 2003 delays occurred for one of every 131 survey lines or airgun tests that were the first shots of the survey, but for only one of every 185 subsequent lines or tests (Stone 2015). The level of compliance with the requirement to delay firing was highly variable between years (Figure 3.14). As there were only a small number of delays, each occasion when the correct procedures were or were not implemented resulted in a substantial raising or lowering of the proportion where there was compliance for that year. Although highly variable, overall compliance with the requirement to delay firing if marine mammals are in close proximity has shown a general improvement over time (Figure 3.14), although the level of compliance with this aspect of the guidelines still lags behind that of pre-shooting searches and soft starts. Incorrect procedures in a delay situation were sometimes due to the subsequent soft start being too short (20% of all delay situations), but more often due to the delay not being long enough (27% of all delay situations).

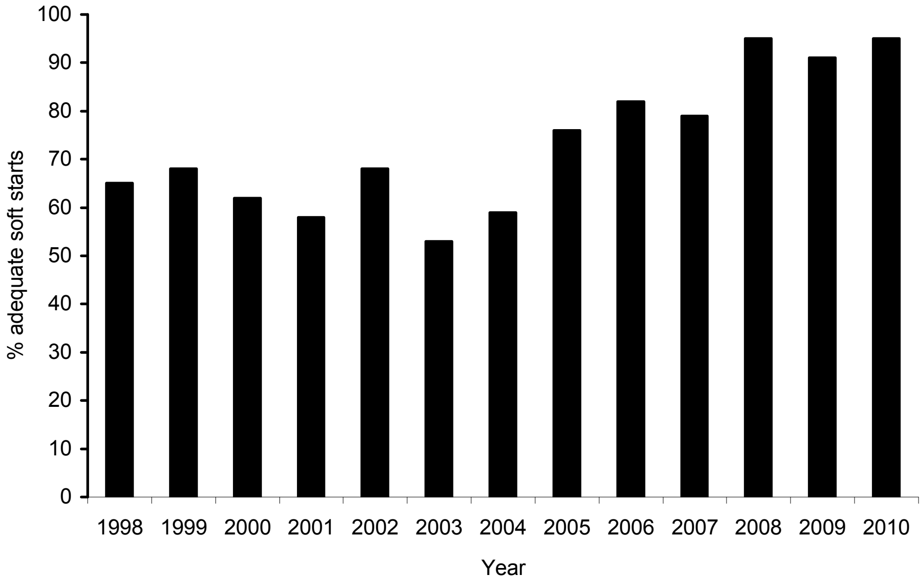

Figure 3.13. Proportion of adequate soft starts within the UKCS over time (all survey types).

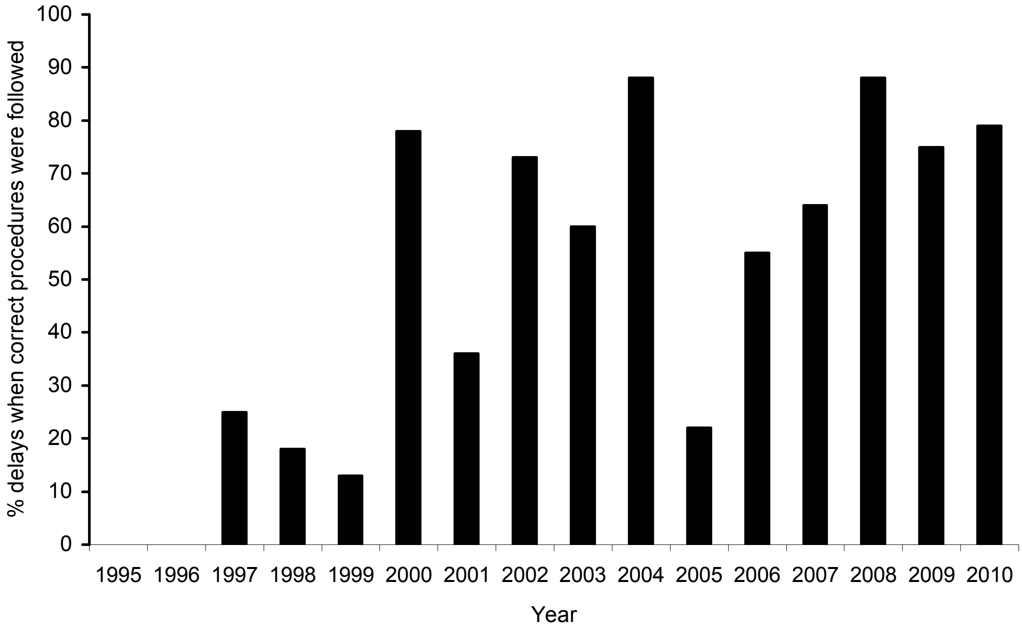

Figure 3.14. Proportion of occasions when correct procedures were followed when a delay in firing was required within the UKCS due to the presence of marine mammals (no delays were needed in 1995 or 1996).

4 Discussion

4.1 Distribution of marine mammals

The distribution of marine mammals observed from seismic survey vessels to a large extent reflects the distribution of survey effort. Any trends in distribution that may be apparent need to be treated with caution as a seismic survey vessel is not an unbiased platform and its operation may have an influence on the observed distribution and abundance. Allowing for uneven survey effort, the distribution of marine mammals observed largely agrees with previous knowledge, with high numbers of sightings in known areas of high abundance (e.g. deep waters to the west of Britain and Ireland) and low numbers in known areas of relatively low abundance (e.g. the southern North Sea).

The large whales (humpback, blue, fin, sei and sperm whales) were found mostly in shelf edge and deep waters to the north and west of Britain and Ireland, in agreement with known distribution (Clark and Charif 1998; Evans 1990; JNCC 1995; NERC 1998; Pollock *et al* 2000; Reid *et al* 2003; Skov *et al* 1995). A sighting of a humpback whale closer inshore to the east of Shetland is consistent with known records of sightings on the continental shelf, which Reid *et al* (2003) notes have come mainly from the Northern Isles. There was a sighting of a fin whale in the central North Sea, with Camphuysen and Winter (1995) having also noted occasional fin whale sightings in the North Sea. However, there was no obvious reason for the decrease in fin whale sightings to the west of Shetland in more recent years.

Sightings of beaked whales (northern bottlenose whale and Sowerby's beaked whale) occurred almost exclusively in shelf edge and deep waters. Beaked whales are known to occur mainly in deep waters (e.g. CODA 2009), although there have been some rare sightings of beaked whales, including Sowerby's beaked whale, in the Minch (Reid *et al* 2003). One northern bottlenose whale was seen off Aberdeen; occasional inshore sightings of this species have been reported previously (Weir 1999; Weir and Coles 1998).

Long-finned pilot whales were seen predominantly in shelf edge and deep waters to the north and west of Scotland, with some also to the west of Ireland and in the South-west Approaches, agreeing with their known distribution (Bloor *et al* 1996; CODA 2009; JNCC 1995; NERC 1998; Pollock *et al* 1997, 2000; Reid *et al* 2003; Skov *et al* 1995). Some were also seen along the western edge of the Rinne, with Reid *et al* (2003) also noting several sightings in this area. Killer whales were seen mainly in northern waters with a concentration to the north-east of Shetland, matching their preference for cooler waters (Evans 1992). There was one sighting of false killer whales to the west of Ireland in a similar location to a sighting reported previously (Reid *et al* 2003).

Minke whales were widely distributed, with many sightings throughout the central and northern North Sea and into deeper waters further north. Northridge *et al* (1995) and Reid *et al* (2003) found concentrations of minke whales close to land. Although some were seen relatively close inshore particularly around the coast of Scotland, most effort was further offshore so these inshore areas were less well surveyed. The SCANS and SCANS-II surveys found that a weak concentration of minke whales off southeast Scotland in 1994 had dissipated to the central North Sea in 2005 (Hammond *et al* 2013).

Atlantic white-sided dolphins, striped dolphins and Risso's dolphins were found mostly on the shelf edge and in deeper waters. For Atlantic white-sided dolphins and striped dolphins this is consistent with previous findings (Evans 1990, 1992; JNCC 1995; NERC 1998; Pollock *et al* 1997, 2000; Reid *et al* 2003; Skov *et al* 1995). However, Reid *et al* (2003) considered Risso's dolphins to be a continental shelf species and noted that although a few records were from immediately over the shelf break none were in deeper waters. More than half of the sightings of Risso's dolphins from seismic survey vessels were beyond the continental shelf with several sightings occurring in deep waters of more than 1,000m depth. Their distribution was also further north than reported by Reid *et al* (2003), although Bloch *et al* (2012) noted recent occurrences in Faroese waters and considered that there was a likely northward extension of the species' known range. Bottlenose dolphin distribution was also slightly different from that reported previously. Sightings occurred in St George's Channel and off the Aberdeenshire and Moray coasts, in areas of resident populations, although such sightings were relatively few as might be expected due to the low effort closer inshore. However, there were more sightings of bottlenose dolphins further offshore in the central and northern North Sea than were reported by Reid *et al* (2003), although offshore sightings along the shelf edge and deeper waters to the west of Britain agree with known distribution (CODA 2009; Hammond *et al* 2013; Reid *et al* 2003).

Both striped dolphins and short-beaked common dolphins in UK waters are found predominantly to the south-west (CODA 2009; NERC 1998; Reid *et al* 2003), although their range is predicted to expand progressively northwards as water temperatures increase over time (Lambert *et al* 2011; MacLeod 2009). Given the low effort from seismic survey vessels to the south-west of the UK this south-westerly skew was not evident, although a number of sightings of short-beaked common dolphins did occur in St George's Channel and the Celtic Sea. Short-beaked common dolphins were also recorded in the central and northern North Sea. A change in their range related to past changes in water temperature has been demonstrated (Lambert *et al* 2011) and there have been more northerly sightings of this species reported in recent years (Robinson *et al* 2010). The sightings of striped dolphins in the central and southern North Sea are very anomalous compared to their normal distribution to the south and west of the UK (Reid *et al* 2003).

The white-beaked dolphin was the most common identified species of marine mammal observed from seismic survey vessels operating in UK waters. Previous studies of the distribution of this species have found it to occur mostly on the continental shelf (Hammond *et al* 2013; Northridge *et al* 1995; Reid *et al* 2003). While this was also true for those seen from seismic survey vessels, more were seen beyond the shelf to the north of Scotland than have been reported previously (Reid *et al* 2003). MacLeod *et al* (2007) found that the most important variable related to white-beaked dolphin distribution was water depth, with a preference being shown for shallower waters. Although the majority of white-beaked dolphins seen from seismic survey vessels were also found in shallower waters, nevertheless notable numbers of sightings occurred in deeper waters beyond the shelf edge.

Harbour porpoises are more abundant in the North Sea and adjacent areas than white-beaked and/ or Atlantic white-sided dolphins (Hammond *et al* 2013; Reid *et al* 2003), but the latter two species were more commonly seen by MMOs during seismic surveys. The relatively low numbers of harbour porpoises seen are likely at least in part to be due to difficulties in detecting this species (particularly as sea state increases above sea state 2; Hammond *et al* 2013) and the inexperience of some observers in detecting this small marine mammal. However, it may also indicate avoidance by harbour porpoises of areas where seismic surveys are taking place (whether due to airgun noise or general vessel avoidance) and further investigation of possible reasons for the relative lack of harbour porpoise sightings is warranted. The SCANS and SCANS-II surveys showed that the total abundance of harbour porpoise in the North Sea and adjacent waters did not change significantly between 1994 and 2005 but the distribution did change, with densities lower in the north and higher in the south in 2005 than in 1994 (Hammond *et al* 2013). It was considered that a likely explanation for this change in distribution of harbour porpoises was a change in the distribution or availability of their prey (Hammond *et al* 2013). Of the harbour porpoises that were seen from seismic survey vessels, numbers of sightings increased in the southern North Sea from 2006 onwards and decreased further north (with the exception of the Outer Moray Firth), agreeing with the trend found by SCANS-II (Hammond *et al* 2013).

There were fewer sightings of harbour seals than grey seals but both species were seen at similar distances from haul-out sites such as the Moray Firth. Grey seals were thought to forage further from haul-out sites than harbour seals (Thompson *et al* 1996), but recent tagging studies have shown that harbour seals forage more extensively in offshore waters than was previously known (Sharples *et al* 2012).

4.2 Effects of seismic operations on marine mammals

The data collected by MMOs, including any monitoring to cover periods when the airguns are firing in addition to the required pre-shooting search, is valuable for investigating potential impacts of seismic operations on marine mammals. Injury to marine mammals as a result of acoustic input to the marine environment is a primary concern. Injuries to marine mammals are often not apparent unless the animal subsequently strands and even then it can be very difficult to establish the cause. Reports of possible injuries to marine mammals due to seismic surveying are very few. While there has been speculation that strandings of humpback whales in Brazil, Cuviers beaked whales in Mexico and illness/ injury of a pantropical spotted dolphin off Liberia may have been due to seismic surveying, these links were based on spatial and/ or temporal coincidence and remain inconclusive (Engel *et al* 2004; Gray and Van Waerebeek 2011; Taylor et al 2004). Observations from seismic survey vessels operating in UK waters showed no evidence of any injuries, but these would not necessarily be apparent from surface observations.

Under European and UK law, both deliberate injury and deliberate/ reckless disturbance of European protected species (EPS) are prohibited. Disturbance in this context includes disturbance that is likely to impair the animals' ability to survive, to breed or reproduce, or to rear or nurture their young, or to migrate, or disturbance that will affect significantly their local distribution or abundance. Some behavioural responses to seismic operations were evident from the observations. Southall *et al* (2007) proposed a severity scale for ranking observed behavioural responses of free-ranging marine mammals to anthropogenic sound, ranging from no observable response (response score zero) to outright panic, flight, stampede, attack of conspecifics, stranding events or avoidance behaviour related to predator detection (all response score nine). The observed responses of marine mammals to seismic operations in UK waters can be considered in the context of these response scores. JNCC draft guidance on The Protection of Marine European Protected Species from Injury and Disturbance (JNCC *in litt*) proposes that a disturbance offence is more likely to occur when there is a risk of animals incurring sustained or chronic disruption of behaviour scoring five or more on Southall *et al*'s (2007) scale, or where there is a risk of animals being displaced from the area, with redistribution significantly different from natural variation.

Displacement of animals from an area, particularly feeding and/ or breeding areas, in response to anthropogenic activities could have significant impacts on individuals and populations, particularly if the displacement is prolonged. Long term avoidance of an area, beyond the duration of operations, is ranked highly on Southall *et al*'s (2007) severity scale of behavioural responses (response score eight) and such long-term avoidance could potentially impair the animals' ability to feed (thus affecting survival), to breed or to migrate etc. Data were not collected beyond the duration of seismic surveys to see whether any displacement persisted but lateral displacement during periods of airgun activity was observed for some species, as indicated by a reduction in the number of sightings or acoustic detections and/ or animals remaining further from the source at these times (sections 3.2.1 and 3.2.3). Where detection rates were reduced this suggests lateral displacement beyond the visual range of the observer, demonstrating at least minor avoidance of the sound source, which is ranked as response score six by Southall *et al*

(2007). However, there was no evidence that such avoidance was sustained as the higher detection rates when the airguns were not firing included many periods after the end of survey lines when shooting had only recently ceased. Nor was there any evidence that the displacement resulted in redistribution significantly different from natural variation. Species exhibiting lateral displacement beyond the visual range of the observer during periods of airgun activity on surveys with ‘large arrays’ included the minke whale, killer whale, white-beaked dolphin, Atlantic white-sided dolphin, harbour porpoise and grey seal, as well as all beaked whales combined (section 3.2.1).

During surveys with ‘small arrays’ only sperm whales and harbour porpoises gave any indication of lateral displacement beyond the visual range of the observer. Lucke *et al* (2009) found aversive behavioural responses of a single captive harbour porpoise when exposed to noise from a seismic airgun and also found that the masked temporary threshold shift level was lower than for other odontocetes. Bain and Williams (2006) found that harbour porpoises appeared to be the species affected by the lowest levels of airgun noise, with apparent avoidance over 70km from airguns, although sample sizes were too small to permit statistical testing. Thompson *et al* (2013) found that seismic operations using a relatively small array (470 cu. in.) with similar sound exposure levels to those in Lucke *et al*’s (2009) study resulted in short-term avoidance of harbour porpoises, although animals were typically detected again at affected sites within a few hours and there were indications of possible habituation or tolerance as the survey progressed; those porpoises remaining in the area did however reduce their buzzing activity, indicative of prey capture or social communication, with the probability of buzzes decreasing with proximity to the source (Pirotta *et al* 2014). An increased sensitivity compared to other species may explain why, in the present study, harbour porpoises were apparently displaced during periods of firing regardless of the size of the airgun array, while some other odontocetes appeared only to be displaced by larger airgun arrays.

Although only sperm whales and harbour porpoises showed evidence of lateral displacement beyond the visual range of the observer during periods of firing on surveys with ‘small arrays’, there was nevertheless a significant decrease in overall sighting rates of groups combining all delphinids and all small odontocetes after the first week of these surveys (section 3.2.2). With repeated exposure to sound increased habituation or increased sensitisation may occur (Richardson *et al* 1995), so it is possible that an initial tolerance of smaller airgun arrays by delphinids and small odontocetes might give way to increasing sensitisation as surveys progress; alternatively there could be some other explanation for the later decrease in sighting rates, such as a delayed reaction due to prey moving out of the area. The reduction in rates of delays after the initial use of airguns on a survey might point to an adaptive response, with animals ‘warned’ by previous firing perhaps being less likely to approach close to the vessel (section 3.4).

Where animals remained significantly further from the airguns during periods of airgun activity but detection rates were not reduced this may indicate lateral displacement of a lesser degree, i.e. not beyond the visual range of the observer. Bottlenose dolphins responded in this way when ‘large arrays’ were active (section 3.2.3). More localised responses were also indicated in some species during periods of airgun activity by an increased tendency to avoid or travel away from the vessel and/ or a reduction in positive interactions with (e.g. bow-riding) or travel towards the vessel or its equipment (section 3.2.4). This more localised avoidance may indicate a level of disturbance or discomfort and was evident on surveys with ‘large arrays’ for fin whales and long-finned pilot whales, even though there was no significant lateral displacement of these species. On surveys with ‘small arrays’, localised avoidance without significant lateral displacement was indicated for groups comprising all cetaceans combined, all delphinids combined, all small odontocetes combined and *Lagenorhynchus* spp. Minor avoidance of the sound source, ranked as response score six on the severity scale of Southall *et al* (2007), could potentially have an impact on activities such as foraging, although again there is no evidence that such responses were sustained.

Where there is no avoidance of the sound source, Southall *et al* (2007) rank changes in locomotion speed, direction and/ or dive profiles as response scores three, four or five, depending on whether they are minor, moderate or extensive/ prolonged. On surveys with 'large arrays', short-beaked common dolphins showed an increase in swimming speed when the airguns were active but no behaviours that would indicate avoidance of the airgun noise (section 3.2.4). Gailey *et al* (2007) also found increased swimming speeds of cetaceans in response to airgun activity.

A reduction in foraging effort may clearly have significant consequences for individuals and populations. Although feeding may not always be apparent from surface observations, when all cetaceans were combined significantly fewer animals were recorded as feeding when 'large arrays' were active (section 3.2.4). Jochens *et al* (2008) and Miller *et al* (2009) found no horizontal avoidance of seismic operations by sperm whales but did find that there may be a decrease in foraging effort (indicated by changes in sperm whale buzz rates associated with foraging); as this species forages at depth, a reduction in foraging would not be readily apparent from the data collected by MMOs. It should be noted that although there were no observed effects of noise from 'large arrays' on species such as sperm whales, there could potentially be effects that were not observed; behavioural observations were limited to periods when animals were at the surface (representing a relatively small proportion of time for deep divers such as sperm whales). It should also be noted that some responses of marine mammals to noise could be subtle and not able to be observed by MMOs (e.g. increased stress hormones; Rolland *et al* 2012; Romano *et al* 2004).

When 'large arrays' were active there were some indications that some cetaceans may remain closer to the surface (surfacing frequently, logging, apparently resting or milling), where noise levels may be lower due to the Lloyd's mirror effect (Richardson *et al* 1995; Urick 1983), although this was not a universal response (section 3.2.4). Robertson *et al* (2013) found that bowhead whales spent less time at the surface and Gailey *et al* (2007) found that gray whales stayed underwater longer in response to seismic operations, but most other studies have indicated that cetaceans may remain near the surface in response to noise. For example, McCauley *et al* (1998, 2000) found that humpback whales spent more time at the surface during periods of seismic operations and Jochens *et al* (2008) and Miller *et al* (2009) suggested that a sperm whale responded to airgun sounds by resting near the surface until airgun exposure ceased. Also, Barkaszi *et al* (2012) found that sperm whales in the Gulf of Mexico were surfacing more when airguns were at full power than when they were silent. Robertson *et al* (2013) suggested that changes in surfacing, respiration and dive behaviours of cetaceans exposed to seismic operations may have implications for the ability to detect animals. If cetaceans remain near the surface at times of airgun activity this could make them easier to detect visually and might lead to a relative increase in sighting rates at these times. As most of the effort in the present study was visual, any behaviours which may have influenced visual detection rates could have the potential to mask any changes in numbers of animals in the vicinity. Therefore a lack of any significant difference in detection rates for some species in the present study does not necessarily rule out overall avoidance by these species. Robertson *et al* (2013) found that changes in surfacing, respiration and dive behaviours were context-dependent, depending on the circumstance and the activity of the animal; seismic operations had a greater effect when whales were travelling than when they were socialising or feeding. The response of marine mammals to airgun activity is likely to be very complex, involving many variables that may contribute to results such as those for sperm whales in the present study, which are difficult to explain (i.e. detection rates of sperm whales were reduced during periods when 'small arrays' were active, suggesting that they moved out of the area, but when 'large arrays' were active no response was observed; section 3.2.1).

All UK MMO data were examined from 1994 (just prior to the introduction of the JNCC guidelines) until the end of 2010. Some subsets of this dataset have been analysed previously (Stone 1997, 1998, 2000, 2001, 2003a, b, 2006; Stone and Tasker 2006), but these previous studies used data pooled over a maximum of four years. Pooling all data from 1994 to 2010 provided a much larger dataset, thereby resulting in larger sample sizes which permitted statistical testing of a greater range of responses over a greater range of species than was possible previously, although there were still some aspects of the analysis where sample sizes were low and species needed to be combined. For the first time beaked whales were able to be included in the analysis, although sample sizes were low and all species of beaked whale had to be combined. Nevertheless it was possible to demonstrate that detection rates of beaked whales were significantly lower when 'large arrays' were active (section 3.2.1), whereas previously there has been little evidence that beaked whales respond overtly to the noise from seismic airguns (Moulton and Holst 2010). One seismic survey has been implicated in the stranding of Cuviers beaked whales but without conclusive evidence of a link (Taylor *et al* 2004). Beaked whales are known to be sensitive to other anthropogenic noise, with cases of mass strandings related to the use of military mid-range frequency sonar (Balcomb and Claridge 2001; Cox *et al* 2006; Evans and England 2001; Fernández *et al* 2005; Tyack *et al* 2011). Southall *et al* (2007) suggested that regulatory agencies should consider adopting provisional injury criteria for beaked whales exposed to military sonar at lower levels than for other mid frequency cetaceans. Seismic airguns use predominantly low frequencies up to around 200Hz (Gausland 2001; Gulland and Walker 2001) whereas mid-range frequency sonar uses frequencies of around 3-8kHz (Evans and England 2001; Tyack *et al* 2011), so the results are not necessarily directly comparable, but nevertheless a response of beaked whales to seismic airguns has been noted here.

Responses of bottlenose dolphins also were not able to be tested previously but the larger dataset showed localised avoidance, increased swimming speed and increased incidence of breaching when 'large arrays' were active (sections 3.2.3 and 3.2.4). Furthermore, grey seals were found to have significantly lower detection rates when 'large arrays' were active (section 3.2.1), whereas previously sample sizes were too low to examine responses of seals. Harris *et al* (2001) showed some lateral displacement of seals (mostly ringed seals) during seismic surveys. However, some other species that previously could not be examined due to low sample sizes were found to show no discernible effects (humpback whale, sei whale, Risso's dolphin and harbour seal).

Baleen whales are estimated to have functional hearing within the range 7Hz to 22kHz, while most odontocetes belong to a mid-frequency hearing group with functional hearing from about 150Hz to 160kHz and porpoises belong to a high frequency hearing group with functional hearing between 200Hz and 180kHz (Southall *et al* 2007). As many anthropogenic sound sources are of low frequency it has often been assumed that baleen whales would be more vulnerable to disturbance from such sources than odontocetes. Seismic airguns, for example, produce peak energy at low frequencies up to about 200Hz (Gausland 2001; Gulland and Walker 2001). These low frequency sounds can travel long distances; for example, Nieukirk *et al* (2012) recorded airgun sounds in some cases almost 4,000km away from the source and Hildebrand (2009) noted that seismic sources contributed to low frequency ambient noise across ocean basins. Although avoidance of seismic survey vessels has been demonstrated for baleen whales such as bowhead whales, gray whales and humpback whales elsewhere (e.g. Ljungblad *et al* 1988; McCauley *et al* 1998, 2000; Moulton and Holst 2010; Richardson and Greene 1993; Richardson *et al* 1986, 1999; Yazvenko *et al* 2007), previously no effects were observed on individual baleen whale species in UK waters (Stone and Tasker 2006). However, use of the larger dataset revealed lateral displacement of minke whales (indicated by lower detection rates) when 'large arrays' were active and localised avoidance by fin whales at these times (no change in detection rates but fin whales tended to avoid or travel away from the vessel) (sections 3.2.1 and

3.2.4). It has been noted that impacts of human pressures on the minke whale are largely unknown (Thomsen *et al* 2011); localised avoidance of active airguns by minke whales has been observed in the northwest Atlantic (Moulton and Holst 2010) and the present study has confirmed that minke whales in UK waters show similar avoidance of seismic operations. Fin whales in the Mediterranean Sea modified their vocalisations and moved out of the area of a seismic survey for an extended period (Castellote *et al* 2012); although such displacement was not observed in fin whales in UK waters nevertheless some localised avoidance was found. In the present study no responses were observed in either humpback whales or sei whales, although sample sizes were low. Elsewhere responses to seismic survey vessels have been demonstrated for humpback whales (Cerchio *et al* 2014; McCauley *et al* 1998, 2000; Moulton and Holst 2010).

Odontocetes hear best at frequencies mostly above those at which the peak energy from seismic airguns is produced. Nevertheless, several species of mid frequency odontocete in UK waters (beaked whales, long-finned pilot whales, killer whales, bottlenose dolphin, white-beaked dolphin and Atlantic white-sided dolphin) demonstrated some degree of avoidance (sections 3.2.1 to 3.2.4), or in the absence of avoidance a change in swimming behaviour (short-beaked common dolphin; section 3.2.4). Harbour porpoises, in the high frequency group, also demonstrated avoidance (section 3.2.1). Although sound from seismic airguns is predominantly low frequency, nevertheless higher frequency sounds are also emitted that would be audible to odontocetes (De Ruiter *et al* 2006; Goold and Fish 1998; Madsen *et al* 2006; Potter *et al* 2007), although these high frequency sounds are likely to attenuate rapidly (Potter *et al* 2007). It seems that in UK waters, the tendency of cetacean groups to show a response to noise from seismic airguns does not correlate directly with what might be expected based on their hearing abilities. It could be that the responses are driven not only by the ability to hear the sound but also by how the sound is perceived; for example, animals may avoid sounds that they might interpret as indicating the presence of predators, to which smaller species may be more vulnerable. Similar responses of small odontocetes to noise from seismic airguns have been observed elsewhere; for example, Barkaszi *et al* (2012) examined MMO data from the Gulf of Mexico and demonstrated that delphinids showed spatial avoidance, displayed more surface behaviours such as breaching and porpoising and were less likely to display bow-riding behaviour during periods of airgun activity. Weir (2008a) found that Atlantic spotted dolphins showed a more marked overt response to airgun sound than either humpback whales or sperm whales.

The larger dataset also allowed greater examination of the response of marine mammals to the soft start in UK waters than has been possible previously (Stone 2006; Stone and Tasker 2006). Whether the soft start is an effective mitigation measure has been identified as a key question of interest and the recording forms in the UK were revised in 2009 to allow a distinction between effort during the soft start and at full power to aid in addressing this question (Barton *et al* 2008). The results showed that all species or species groups tested had reduced detection rates during the soft start compared to when the airguns were not firing (section 3.3). All species groups tested also showed an increased tendency to avoid or travel away from the vessel during soft starts of ‘large arrays’ (section 3.3), although not all individuals displayed such behaviours. These observed responses suggest that the soft start can be a useful mitigation tool, causing some marine mammals to move away from the immediate vicinity of airguns before full power is reached, thereby helping to avoid exposure to high levels of sound. Movement directed away from the source can only reduce exposure levels if the avoidance speed of the animal is much greater than the approach speed of the source (Von Benda-Beckmann *et al* 2014); seismic survey vessels towing airguns typically travel at relatively low speeds (around 4-5 knots), therefore movement away from the source may be effective at reducing exposure to sound from seismic operations. However, the lack of a universal response during the soft start procedure highlights the need to monitor for marine mammals prior to commencing firing airguns, with subsequent delay of firing if marine mammals are detected within the mitigation zone. For undetected animals, the soft start may also offer protection to some individuals by causing them to move out of the vicinity of the airguns before full power is reached.

Although the results here showed a high level of agreement in the response to the soft start between the species and species groups tested, the testing of individual species was limited to minke whales, white-beaked dolphins and Atlantic white-sided dolphins due to data availability. Therefore a degree of caution should be exercised as other species may respond differently. Nothing is known, for example, about the effectiveness of the soft start for sensitive species such as beaked whales (Barlow and Gisiner 2006). Moulton and Holst (2010) suggested that the effectiveness of the soft start varies with species and probably circumstances; in the north-west Atlantic they found mysticetes were observed further from the survey vessel during the soft start than when the airguns were silent but, in contrast to the present study, found no response in delphinids or toothed whales and considered that the soft start may be largely ineffective for some odontocetes. Weir (2008b) gave a detailed report on the response of a single pod of short-finned pilot whales to a soft start and although the whales showed an initial avoidance response this was limited in space and time, with the directed movement away from the source changing to milling at the surface, although it was noted that this might represent vertical avoidance. This, however, contrasts with the observation here that cetaceans were more likely to dive if the soft start commenced during the encounter (section 3.3). This observation was perhaps surprising, as sound levels may be lower at or near the surface due to interference between direct and surface-reflected sound (Richardson *et al* 1995; Urick 1983), so animals might be expected to be more likely to remain near the surface at the onset of noise.

There is clearly a need for more detailed studies on the response of marine mammals to the soft start procedure. Noise modelling can be used to investigate the effectiveness of the soft start, but needs to be set in the context of the animals' response. Von Benda-Beckmann *et al* (2014) used modelling to demonstrate that the effectiveness of soft start for sonar depended strongly on the assumed response threshold of animals and differed with soft start duration. Modelling has also indicated that the threshold levels for hearing injury for cetaceans are not reached during the initial stages of a soft start of an airgun array, although threshold criteria for pinnipeds may be approached, perhaps requiring additional mitigation for seismic surveys operating close to haul-out areas for pinnipeds (Hannay *et al* 2011). Von Benda-Beckmann *et al* (2014) noted that critical research questions that need addressing are documentation of avoidance strategies (horizontal/ vertical avoidance, swim speeds), behavioural context and estimates of sound dosage that predicts the onset of an avoidance response for sounds other than sonar.

4.3 Compliance with the JNCC guidelines

Only overall trends in compliance with the key aspects of the JNCC guidelines were considered here (for more detail on individual years see Stone 1998, 2000, 2001, 2003a, 2006, 2015), but the general picture was one of improvement over time. Pre-shooting searches were mostly of acceptable duration, but the pre-shooting search is only effective as a mitigation tool if a delay is properly implemented for any marine mammals that are detected in the mitigation zone during the pre-shooting search. Therefore, assessment of the pre-shooting search on its own should not be taken as a measure of how well the guidelines are being implemented. Compliance with the requirement to delay the commencement of firing if marine mammals are detected in the mitigation zone has shown a general improvement over time, but is highly variable and lags behind other aspects of compliance with the guidelines. Further improvement in compliance with the requirement to delay firing would increase the overall effectiveness of the pre-shooting search as a mitigation tool.

The results here have indicated that the soft start may be a useful mitigation tool, causing some marine mammals to move away from the vicinity of the airguns before full power is reached, but it will only be effective if animals have sufficient time during the soft start to move to a distance where exposure to high levels of sound would be avoided. It is therefore important that there is compliance with the specified minimum duration for the soft start. The introduction in 2004 of alternative methods of performing a soft start for site surveys and VSP operations has clearly been a successful addition to the guidelines, resulting in an improvement in the proportion of soft starts meeting the specified minimum duration.

5 Conclusions

Based on observations in UK waters over a 16 year period, some responses of marine mammals to seismic surveys were evident. These responses did not correlate directly with what might be expected based on their hearing abilities, as small and medium-sized odontocetes showed responses as well as some baleen whales. On surveys with 'large arrays' (500 cu. in. or more) all small or medium-sized odontocete species (except Risso's dolphin) showed some significant response when airguns were active, whether this was lateral displacement or more localised avoidance or a change in behaviour. This included the first indication of a response by beaked whales to airgun activity in UK waters, where lateral displacement was indicated when 'large arrays' were active. Baleen whales, based on their hearing abilities, may be expected to show greater responses to seismic operations; minke whales showed lateral displacement in response to active 'large arrays', while fin whales showed more localised avoidance.

Responses of marine mammals were less evident on surveys with 'small arrays' (less than 500 cu. in.). The only individual species where lateral displacement was evident were the sperm whale and the harbour porpoise, although testing of responses for many species was limited by lower sample sizes. Baleen whales did not show any negative behavioural responses to surveys with 'small arrays'. Although responses were generally fewer than with 'large arrays', nevertheless the variation in effects observed confirm that mitigation measures should continue to apply to all types of seismic surveys and cover the risk to all marine mammal species.

The results presented here indicate that the soft start may be an effective mitigation measure, suggesting that some marine mammals were moving away from the airguns before full power was reached. However, information was only available for a few species, so there is still a need for further studies on the effectiveness of the soft start.

There has been an increase in the standard of soft starts in UK waters over the years. In recent years most soft starts lasted at least 20 minutes from the time of commencement until full power was reached. A visual pre-shooting search generally provided adequate monitoring during daylight hours, with delays in firing due to the presence of marine mammals in the mitigation zone being relatively rare. Although compliance with the requirement to delay showed a general improvement over time, there were still a number of occasions when delays were not correctly implemented and standards were lower than for compliance with other aspects of the JNCC guidelines.

It is acknowledged that MMO observations of cetacean behaviour have the potential to be biased given the difficulty in observing cetaceans, the subjective nature of interpreting behaviour and the possibility that the MMO may have an expectation, even if subconsciously, that animals will respond differentially between when seismic airguns are firing and when they are not firing. However, blind field trials cannot be achieved on board seismic surveys and experimental set ups (e.g. Cato *et al* 2013) would not be feasible over the same spatial and temporal scales and range of species as can be provided by MMO data. MMO data thus provide a valuable resource for investigating the potential impacts of industrial activities on marine mammals across a range of species and geographical areas. This report represents one of the longest term analyses of MMO data to date; every attempt has been made to limit bias potential where possible, for example by using matched pairs in the statistical analyses. There is a need to continue to collect MMO data to test the effectiveness of the guidelines and compliance therewith. Such studies should aim to improve mutual understanding between regulators/ advisors and industry in order that mitigation is applied correctly, is logistically feasible, is well justified and is proportional to the risk to species.

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7 References

BAIN, D.E. & WILLIAMS, R. 2006. Long-range effects of airgun noise on marine mammals: responses as a function of received sound level and distance. *Paper SC/58/E35 presented to the IWC Scientific Committee, June 2006, St Kitts and Nevis, WI*. Available from: http://iwc.int/document_1822 [Accessed 29th August 2013].

BALCOMB, K.C. & CLARIDGE, D.E. 2001. A mass stranding of cetaceans caused by naval sonar in the Bahamas. *Bahamas Journal of Science*, **5**, 1-12.

BARKASZI, M.J., BUTLER, M., COMPTON, R., UNIETIS, A. & BENNET, B. 2012. Seismic survey mitigation measures and marine mammal observer reports. U.S. Dept. of the Interior, Bureau of Ocean Energy Management, Gulf of Mexico OCS Region, New Orleans, LA. *OCS Study BOEM* 2012-015. Available from: http://www.data.boem.gov/homepg/data_center/other/espis/espismaster.asp?appid=1 [Accessed 15th November 2014].

BARLOW, J. & GISINER, R. 2006. Mitigating, monitoring and assessing the effects of anthropogenic sound on beaked whales. *Journal of Cetacean Research and Management*, **7(3)**, 239-249.

BARTON, C. 2012. JNCC MMO database manual. Available from JNCC on request.

BARTON, C.J.S., JAQUES, R. & MASON, M. 2008. Identification of potential utility and collation of existing marine mammal observer data. *Report to the Joint Industry Programme Sound and Marine Life Programme,* 109pp. Available from: http://www.soundandmarinelife.org/library/project-reports.aspx [Accessed 6th February 2015].

BLOCH, D., DESPORTES, G., HARVEY, P., LOCKYER, C. & MIKKELSEN, B. 2012. Life history of Risso's dolphin (*Grampus griseus*) (G. Cuvier, 1812) in the Faroe Islands. *Aquatic Mammals*, **38(2)**, 250-266.

BLOOR, P.D., REID, J.B., WEBB, A., BEGG, G. & TASKER, M.L. 1996. The distribution of seabirds and cetaceans between the Shetland and Faroe Islands. *JNCC Report* No. 226.

CAMPHUYSEN, C.J. & WINTER, C.J.N. 1995. Feeding fin whales *Balaenoptera physalus* in the North Sea. *Lutra*, **38**, 81-84.

CASTELLOTE, M., CLARK, C.W. & LAMMERS, M.O. 2012. Acoustic and behavioural changes by fin whales (*Balaenoptera physalus*) in response to shipping and airgun noise. *Biological Conservation,* **147**, 115-122.

CATO, D.H., NOAD, M.J., DUNLOP, R.A., MCCAULEY, R.D., GALES, N.J., SALGADO KENT, C.P., KNIEST, H., PATON, D., JENNER, K.C.S., NOAD, J., MAGGI., A.L., PARNUM, I.M. & DUNCAN, A.J. 2013. A study of the behavioural response of whales to the noise of seismic air guns: design, methods and progress. *Acoustics Australia*, **41**, 91-100.

CERCHIO, S., STRINDBERG, S., COLLINS, T., BENNETT, C. & ROSENBAUM, H. 2014. Seismic surveys negatively affect humpback whale singing activity off northern Angola. *PLoS ONE,* **9(3)**, e86464. doi:10.1371/journal.pone.0086464.

CLARK, C.W. & CHARIF, R.A. 1998. Acoustic monitoring of large whales to the west of Britain and Ireland using bottom-mounted hydrophone arrays, October 1996 - September 1997. *JNCC Report* No. 281.

CODA 2009. *Cetacean offshore distribution and abundance in the European Atlantic*. Available from: http://biology.st-andrews.ac.uk/coda/ [Accessed 28th August 2012].

COX, T.M., RAGEN, T.J., READ, A.J., VOS, E., BAIRD, R.W., BALCOMB, K., BARLOW, J., CALDWELL, J., CRANFORD, T., CRUM, L., D'AMICO , A., D'SPAIN, G., FERNÁNDEZ, A., FINNERAN , J., GENTRY, R., GERTH , W., GULLAND, F., HILDEBRAND, J., HOUSER, D., HULLAR, T., JEPSON, P.D., KETTEN, D., MACLEOD, C.D., MILLER, P., MOORE , S., MOUNTAIN, D.C., PALKA, D., PONGANIS, P., ROMMEL, S., ROWLES, T., TAYLOR, B., TYACK, P., WARTZOK, D., GISINER, R., MEADS, J. & BENNER, L. 2006. Understanding the impacts of anthropogenic sound on beaked whales. *Journal of Cetacean Research and Management,* **7(3)**, 177-187.

DE RUITER, S.L., TYACK, P.L, LIN, Y-T., NEWHALL, A.E., LYNCH, J.F. & MILLER, P.J.O. 2006. Modeling acoustic propagation of airgun array pulses recorded on tagged sperm whales (*Physeter macrocephalus*). *Journal of the Acoustical Society of America*, **120(6)**, 4,100-4,114.

ENGEL, M.H., MARCONDES, M.C.C., MARTINS, C.C.A., LUNA, F.O., LIMA, R.P. & CAMPOS, A. 2004. Are seismic surveys responsible for cetacean strandings? An unusual mortality of adult humpback whales in Abrolhos Bank, northeastern coast of Brazil. *Paper SC/56/E28 presented to the IWC Scientific Committee, IWC Annual Meeting, 19-22 July, Sorrento, Italy.*

EVANS, D.L. & ENGLAND, G.R. 2001. *Joint interim report Bahamas marine mammal stranding event of 14-16 March 2000.* Washington, DC: National Oceanic and Atmospheric Administration, US Department of Commerce and US Navy.

EVANS, P.G.H. 1990. European cetaceans and seabirds in an oceanographic context. *Lutra*, **33**, 95-125.

EVANS, P.G.H. 1992. *Status review of cetaceans in British and Irish waters.* Report of the UK Mammal Society Cetacean Group, University of Oxford.

FERNÁNDEZ, A., EDWARDS, J.F., RODRÍGUEZ, F., ESPINOSA DE LOS MONTEROS, A., HERRÁEZ, P., CASTRO, P., JABER, J.R., MARTÍN, V. & ARBELO, M. 2005. "Gas and fat embolic syndrome'' involving a mass stranding of beaked whales (Family *Ziphiidae*) exposed to anthropogenic sonar signals. *Veterinary Pathology*, **42**, 446-457.

GAILEY, G., WÜRSIG, B. & MCDONALD, T.L. 2007. Abundance, behavior, and movement patterns of western gray whales in relation to a 3-D seismic survey, Northeast Sakhalin Island, Russia. *Environmental Monitoring and Assessment*, **134**, 75-91.

GAUSLAND, I. 2001. Physics of sound in water. *In:* M.L. Tasker & C.R. Weir, eds. 2001 Proceedings of the seismic and marine mammals workshop, London, 23-25 June 1998, Chapter 3. Available from: http://www.anp.gov.br/brnd/round9/round9/guias_R9/sismica_R9/Bibliografia/Tasker%20and%20Weir%201998%20-%20workshop%20seismic%20LONDRES.PDF [Accessed 31st August 2012].

GOOLD, J.C. & FISH, P.J. 1998. Broadband spectra of seismic survey air-gun emissions, with reference to dolphin auditory thresholds. *Journal of the Acoustical Society of America*, **103**, 2,177-2,184.

GRAY, H. & VAN WAEREBEEK. K. 2011. Postural instability and akinesia in a pantropical spotted dolphin, *Stenella attenuata*, in proximity to operating airguns of a geophysical seismic vessel. *Journal for Nature Conservation*,**19**, 363-367.

GULLAND, J.A. & WALKER, C.D.T. 2001. Marine seismic overview. *In:* M.L. Tasker & C.R. Weir, eds. 2001 Proceedings of the seismic and marine mammals workshop, London, 23-25 June 1998, Chapter 2. Available from: http://www.anp.gov.br/brnd/round9/round9/guias_R9/sismica_R9/Bibliografia/Tasker%20and%20Weir%201998%20-%20workshop%20seismic%20LONDRES.PDF [Accessed 15th March 2013].

HAMMOND, P.S., MACLEOD, K., BERGGREN, P., BORCHERS, D.L., BURT, L., CAÑADAS, A., DESPORTES, G., DONOVAN, G.P., GILLES., A., GILLESPIE, D., GORDON, J., HIBY, L., KUKLIK, I., LEAPER, R., LEHNERT, K., LEOPOLD, M., LOVELL, P., ØIEN, N., PAXTON, C.G.M., RIDOUX, V., ROGAN, E., SAMARRA, F., SCHEIDAT, M., SEQUEIRA, M., SIEBERT, U., SKOV, H., SWIFT, R., TASKER, M.L., TEILMANN, J., VAN CANNEYT, O. & VÁZQUEZ, J.A. 2013. Cetacean abundance and distribution in European Atlantic shelf waters to inform conservation and management. *Biological Conservation*, **164**, 107-122. Available from: http://research-repository.st-andrews.ac.uk/handle/10023/3859 [Accessed 27th January 2014].

HANNAY, D., RACCA, R. & MACGILLIVRAY, A. 2011. Model based assessment of underwater noise from an airgun array soft-start operation. *OGP Report* No. 451. Available from: http://www.ogp.org.uk/pubs/451.pdf [Accessed 12th November 2013].

HARRIS, R.E., MILLER, G.W. & RICHARDSON, W.J. 2001. Seal responses to airgun sounds during summer seismic surveys in the Alaskan Beaufort Sea. *Marine Mammal Science*, **17(4)**, 795-812.

HEINEMANN, D. 1981. A range finder for pelagic bird censusing. *Journal of Wildlife Management*, **45**, 489-493.

HILDEBRAND, J.A. 2009. Anthropogenic and natural sources of ambient noise in the ocean. *Marine Ecology Progress Series*, **395**, 5-20.

JNCC. 1995. *European seabirds at sea database: seabird and cetacean UKDMAP datasets version 2.1.* Peterborough: JNCC.

JNCC. 2010. JNCC guidelines for minimising the risk of injury and disturbance to marine mammals from seismic surveys. Peterborough: JNCC. Available from: http://jncc.defra.gov.uk/page-1534 [Accessed 6th July 2012].

JNCC. 2012a. Marine mammal recording forms. Peterborough: JNCC. Available from: http://jncc.defra.gov.uk/page-1534 [Accessed 6th July 2012].

JNCC. 2012b. Guide to using marine mammal recording forms. Peterborough: JNCC. Available from: http://jncc.defra.gov.uk/page-1534 [Accessed 18th March 2013].

JOCHENS, A., BIGGS, D., BENOIT-BIRD, K., ENGELHAUPT, D., GORDON, J., HU, C., JAQUET, N., JOHNSON, M., LEBEN, R., MATE, B., MILLER, P., ORTEGA-ORTIZ, J., THODE, A., TYACK, P. & WÜRSIG, B. 2008. Sperm whale seismic study in the Gulf of

Mexico: Synthesis report. U.S. Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA. *OCS Study MMS* 2008-006. Available from: http://www.data.boem.gov/homepg/data_center/other/espis/espismaster.asp?appid=1 [Accessed 15th November 2014].

LAMBERT, E., MACLEOD, C.D., HALL, K., BRERETON, T., DUNN, T.E., WALL, D., JEPSON, P.D., DEAVILLE, R. & PIERCE, G.J. 2011. Quantifying likely cetacean range shifts in response to global climatic change: implications for conservation strategies in a changing world. *Endangered Species Research*, **15**, 205-222.

LJUNGBLAD, D.K., WÜRSIG, B., SWARTZ, S.L. & KEENE, J.M. 1988. Observations on the behavioral responses of bowhead whales (*Balaena mysticetus*) to active geophysical vessels in the Alaskan Beaufort Sea. *Arctic*, **41**, 183-194.

LUCKE, K., SIEBERT, U., LEPPER, P.A. & BLANCHET, M-A. 2009. Temporary shift in masked hearing thresholds in a harbour porpoise (*Phocoena phocoena*) after exposure to seismic airgun stimuli. *Journal of the Acoustical Society of America*, **125(6)**, 4,060-4,070.

MACLEOD, C.D. 2009. Global climate change, range changes and potential implications for the conservation of marine cetaceans: a review and synthesis. *Endangered Species Research*, **7**, 125-136.

MACLEOD, C.D., WEIR, C.R., PIERPOINT, C. & HARLAND, E. 2007. The habitat preferences of marine mammals west of Scotland (UK). *Journal of the Marine Biological Association of the United Kingdom*, **87**, 157-164.

MADSEN, P.T., JOHNSON, M., MILLER, P.J.O., AGUILAR SOTO, N., LYNCH, J. & TYACK, P. 2006. Quantitative measures of air-gun pulses recorded on sperm whales (*Physeter macrocephalus*) using acoustic tags during controlled exposure experiments. *Journal of the Acoustical Society of America*, **120(4)**, 2,366-2,379.

MCCAULEY, R.D., FEWTRELL, J., DUNCAN, A.J., JENNER, C., JENNER, M-N., PENROSE, J.D., PRINCE, R.I.T., ADHITYA, A., MURDOCH, J. & MCCABE, K. 2000. Marine seismic surveys - a study of environmental implications. *Appea Journal,* **2000**, 692-708.

MCCAULEY, R.D., JENNER, M-N., JENNER, C., MCCABE, K.A. & MURDOCH, J. 1998. The response of humpback whales (*Megaptera novaeangliae*) to offshore seismic survey noise: preliminary results of observations about a working seismic vessel and experimental exposures. *Appea Journal*, **1998**, 692-707.

MILLER, P.J.O., JOHNSON, M.P., MADSEN, P.T., BIASSONI, N., QUERO, M. & TYACK P.L. 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 I*, **56**, 1,168-1,181.

MOULTON, V.D. & HOLST, M. 2010. Effects of seismic survey sound on cetaceans in the northwest Atlantic. *Environmental Studies Research Funds Report* No. 182. St. John's. 28pp.

NERC. 1998. *United Kingdom Digital Marine Atlas (UKDMAP) - Version 3, July 1998*. Bidston Observatory, Birkenhead, Merseyside: National Environment Research Council/ British Oceanographic Data Centre.

NIEUKIRK, S.L., MELLINGER, D.K., MOORE, S.E., KLINCK, K., DZIAK, R.P. & GOSLIN, J. 2012. Sounds from airguns and fin whales recorded in the mid-Atlantic Ocean, 1999-2009. *Journal of the Acoustical Society of America*, **131(2)**,1,102-1,112.

NORTHRIDGE, S.P., TASKER, M.L., WEBB, A. & WILLIAMS, J.M. 1995. Distribution and relative abundance of harbour porpoises (*Phocoena phocoena* L.), white-beaked dolphins (*Lagenorhynchus albirostris* Gray), and minke whales (*Balaenoptera acutorostrata* Lacepède) around the British Isles. *ICES Journal of Marine Science*, **52**, 55-66.

PIROTTA, E., BROOKES, K.L., GRAHAM, I.M. & THOMPSON, P.M. 2014. Variation in harbour porpoise activity in response to seismic survey noise. *Biology Letters*, **10(5)**, 20131090.

POLLOCK, C.M., MAVOR, R., WEIR, C.R., REID, A., WHITE, R.W., TASKER, M.L., WEBB, A. & REID, J.B. 2000. *The distribution of seabirds and marine mammals in the Atlantic Frontier, north and west of Scotland*. Peterborough: JNCC.

POLLOCK, C.M., REID, J.B., WEBB, A. & TASKER, M.L. 1997. The distribution of seabirds and cetaceans in the waters around Ireland. *JNCC Report* No. 267.

POTTER, J.R., THILLET, M., DOUGLAS, C., CHITRE, M.A., DOBORZYNSKI, Z. & SEEKINGS, P.J. 2007. Visual and passive acoustic marine mammal observations and high-frequency seismic source characteristics recorded during a seismic survey. *IEEE Journal of Oceanic Engineering*, **32(2)**, 469-483.

REID, J.B., EVANS, P.G.H. & NORTHRIDGE, S.P. 2003. *Atlas of cetacean distribution in north-west European waters.* Peterborough: JNCC.

RICHARDSON, W.J. & GREENE, C.R. JR. 1993. Variability in behavioural reaction thresholds of bowhead whales to man-made underwater sounds. *Journal of the Acoustical Society of America*, **94**, 1,848.

RICHARDSON, W.J., GREENE, C.R. JR, MALME, C.I. & THOMSON, D.H. 1995. *Marine mammals and noise.* San Diego: Academic Press.

RICHARDSON, W.J., MILLER, G.W. & GREENE, C.R. JR. 1999. Displacement of migrating bowhead whales by sounds from seismic surveys in shallow waters of the Beaufort Sea. *Journal of the Acoustical Society of America*, **106**, 2,281.

RICHARDSON, W.J., WÜRSIG, B. & GREENE, C.R. JR. 1986. Reactions of bowhead whales, *Balaena mysticetus*, to seismic exploration in the Canadian Beaufort Sea. *Journal of the Acoustical Society of America*, **79**, 1,117-1,128.

ROBERTSON, F.C., KOSKI, W.R., THOMAS, T.A., RICHARDSON, W.J., WŰRSIG, B. & TRITES, A.W. 2013. Seismic operations have variable effects on dive-cycle behavior of bowhead whales in the Beaufort Sea. *Endangered Species Research*, **21**, 143-160.

ROBINSON, K.P., EISFELD, S.M., COSTA, M. & SIMMONDS, M.P. 2010. Short-beaked common dolphin (*Delphinus delphis*) occurrence in the Moray Firth, northeast Scotland. *Marine Biodiversity Records*, **3,** e55.

ROLLAND, R.M., PARKS, S.E., HUNT, K.E., CASTELLOTE, M., CORKERON, P.J., NOWACEK, D.P., WASSER, S.K. & KRAUS, S.D. 2012. Evidence that ship noise increases stress in right whales. *Proceedings of the Royal Society B*, **279**, 2363-2368.

ROMANO, T.A., KEOGH, M.J., KELLY, C., FENG, P., BERK, L., SCHLUNDT, C.E., CARDER, D.A. & FINNERAN, J.J. 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**, 1124-1134.

SHARPLES, R.J., MOSS, S.E., PATTERSON, T.A. & HAMMOND, P.S. 2012. Spatial variation in foraging behaviour of a marine top predator (*Phoca vitulina*) determined by a large-scale satellite tagging program. *PLoS ONE*, **7(5)**, e37216.

SIEGEL, S. & CASTELLAN, N.J. JR. 1988. *Nonparametric statistics for the behavioral sciences.* Singapore: McGraw-Hill Book Co.

SKOV, H., DURINCK, J., DANIELSEN, F. & BLOCH, D. 1995. Co-occurrence of cetaceans and seabirds in the northeast Atlantic. *Journal of Biogeography*, **22**, 71-88.

SOUTHALL, B.L., BOWLES, A.E., ELLISON, W.T., FINNERAN, J.J., GENTRY, R.L., GREENE, C.R. JR., KASTAK, D., KETTEN, D.R., MILLER, J.H., NACHTIGALL, P.E., RICHARDSON, W.J., THOMAS, J.A. & TYACK, P.L. 2007. Marine mammal noise exposure criteria: initial scientific recommendations. *Aquatic Mammals*, **33(4)**, 411-414.

STONE, C.J. 1997. Cetacean observations during seismic surveys in 1996. *JNCC Report* No. 228*.* Available from: http://jncc.defra.gov.uk/page-1534 [Accessed 12th July 2014].

STONE, C.J. 1998. Cetacean observations during seismic surveys in 1997. *JNCC Report* No. 278*.* Available from: http://jncc.defra.gov.uk/page-1534 [Accessed 12th July 2014].

STONE, C.J. 2000. Cetacean observations during seismic surveys in 1998. *JNCC Report* No. 301. Available from: http://jncc.defra.gov.uk/page-1534 [Accessed 12th July 2014].

STONE, C.J. 2001. Marine mammal observations during seismic surveys in 1999. *JNCC Report* No. 316. Available from: http://jncc.defra.gov.uk/page-1534 [Accessed 12th July 2014].

STONE, C.J. 2003a. Marine mammal observations during seismic surveys in 2000. *JNCC Report* No. 322. Available from: http://jncc.defra.gov.uk/page-1534 [Accessed 12th July 2014].

STONE, C.J. 2003b. The effects of seismic activity on marine mammals in UK waters, 1998-2000. *JNCC Report* No. 323. Available from: http://jncc.defra.gov.uk/page-1534 [Accessed 12th July 2014].

STONE, C.J. 2006. Marine mammal observations during seismic surveys in 2001 and 2002. *JNCC Report* No. 359. Available from: http://jncc.defra.gov.uk/page-1534 [Accessed 12th July 2014].

STONE, C.J. & TASKER, M.L. 2006. The effects of seismic airguns on cetaceans in UK waters. *Journal of Cetacean Research and Management,* **8(3)**, 255-263.

STONE, C.J. 2015. Implementation of and considerations for revisions to the JNCC guidelines for seismic surveys. *JNCC Report* No. 463b in prep.

TAYLOR, B., BARLOW, J., PITMAN, R., BALLANCE, L., KLINGER, T., DEMASTER, D., HILDEBRAND, J., URBAN, J., PALACIOS, D. & MEAD, J. 2004. A call for research to assess risk of acoustic impact on beaked whale populations. *Paper SC/56/E36 presented to the IWC Scientific Committee, July 2004, Sorrento, Italy.* Available from: https://www.awionline.org/sites/default/files/uploads/legacy-uploads/documents/Taylor_et_al_2004-1238105868-10185.pdf [Accessed 7th May 2014].

THOMPSON, P.M., BROOKES, K.L., GRAHAM, I.M., BARTON, T.R., NEEDHAM, K., BRADBURY, G. & MERCHANT, N.D. 2013. 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 B,* **280**, 20132001. Available from: http://dx.doi.org/10.1098/rspb.2013.2001 [Accessed 4th October 2013].

THOMPSON, P.M., MCCONNELL, B.J., TOLLIT, D.J., MACKAY, A., HUNTER, C. & RACEY, P.A. 1996. Comparative distribution, movements and diet of harbour and grey seals from the Moray Firth, NE Scotland. *Journal of Applied Ecology*, **33**, 1,572-1,584.

THOMSEN, F., MCCULLY, S.R., WEISS, L.R., WOOD, D.T., WARR, K.J., BARRY, J. & LAW, R.J. 2011. Cetacean stock assessments in relation to exploration and production industry activity and other human pressures: review and data needs. *Aquatic Mammals*, **37(1)**, 1-93.

TYACK, P.L., ZIMMER, W.M.X., MORETTI, D., SOUTHALL, B.L., CLARIDGE, D.E., DURBAN, J.W., CLARK, C.W., D'AMICO, A., DIMARZIO, N., JARVIS, S., MCCARTHY, E., MORRISSEY, R., WARD, J. & BOYD, I.L. 2011. Beaked whales respond to simulated and actual navy sonar. *PLoS ONE*, **6(3)**, e17009.

URICK, R.J. 1983. *Principles of underwater sound, 3rd edition*. Los Altos: Peninsula Publishing.

VON BENDA-BECKMANN, A.M., WENSVEEN, P.J., KVADSHEIM, P.H., LAM, F.A., MILLER, P.J.O., TYACK, P.L. & AINSLIE, M.A. 2014. Modelling effectiveness of gradual increases in source level to mitigate effects of sonar on marine mammals. *Conservation Biology*, **28(1)**, 119-128.

WEIR, C.R. 1999. Northern bottlenose whales, Isle of Skye. *Birding Scotland*, **2**, 12-13.

WEIR, C.R. 2008a. Overt responses of humpback whales (*Megaptera novaeangliae*), sperm whales (*Physeter macrocephalus*), and Atlantic spotted dolphins (*Stenella frontalis*) to seismic exploration off Angola. *Aquatic Mammals*, **34(1)**, 71-83.

WEIR, C.R. 2008b. Short-finned pilot whales (*Globicephala macrorhynchus*) respond to an airgun ramp-up procedure off Gabon. *Aquatic Mammals*, **34(3)**, 349-354.

WEIR, C.R. & COLES, P. 1998. Northern bottlenose whales at Broadford Bay, Isle of Skye. *Soundings*, **4**, 4-5.

YAZVENKO, S.B., MCDONALD, T.L., BLOKHIN, S.A., JOHNSON, S.R., MEIER, S.K., MELTON, H.R., NEWCOMER, M.W., NIELSON, R.M., VLADIMIROV, V.L. & WAINWRIGHT, P.W. 2007. Distribution and abundance of western gray whales during a seismic survey near Sakhalin Island, Russia. *Environmental Monitoring and Assessment*, **134**, 45-73.

Appendix 1

On all maps the short dashed line = 200m isobath; the long dashed line = 1,000m isobath.

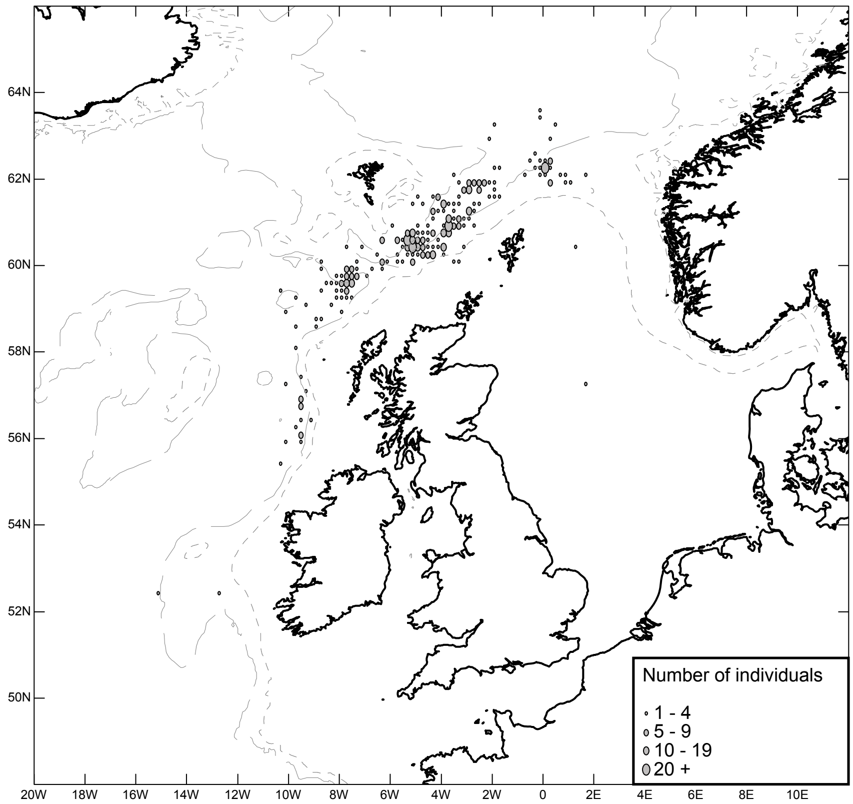

Figure 8.1. Fin whales encountered during seismic surveys, 1994-2010.

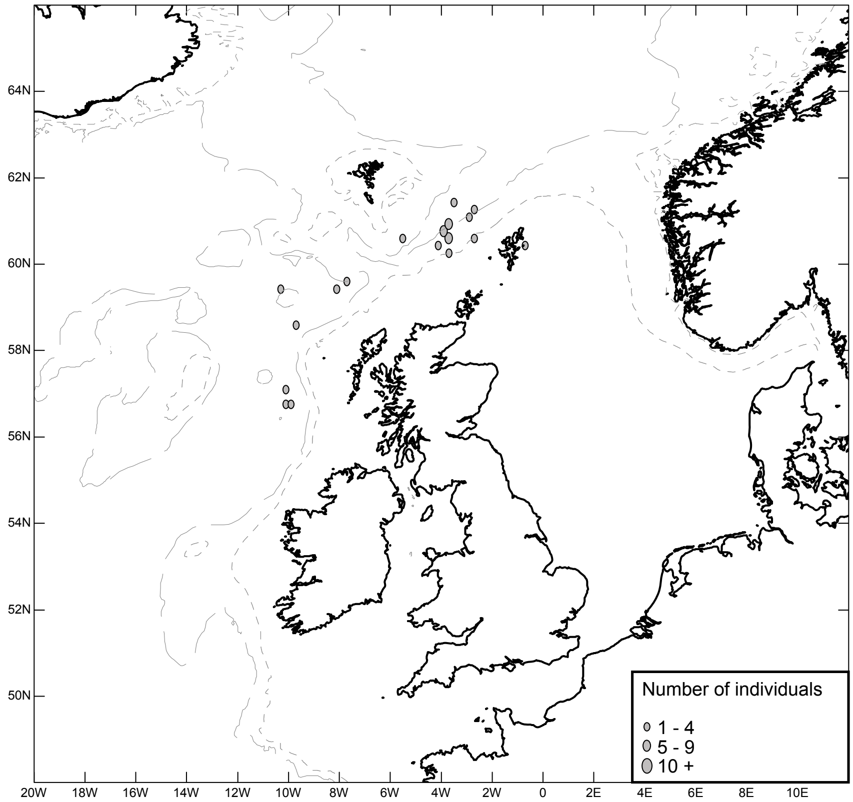

Figure 8.2. Humpback whales encountered during seismic surveys, 1994-2010.

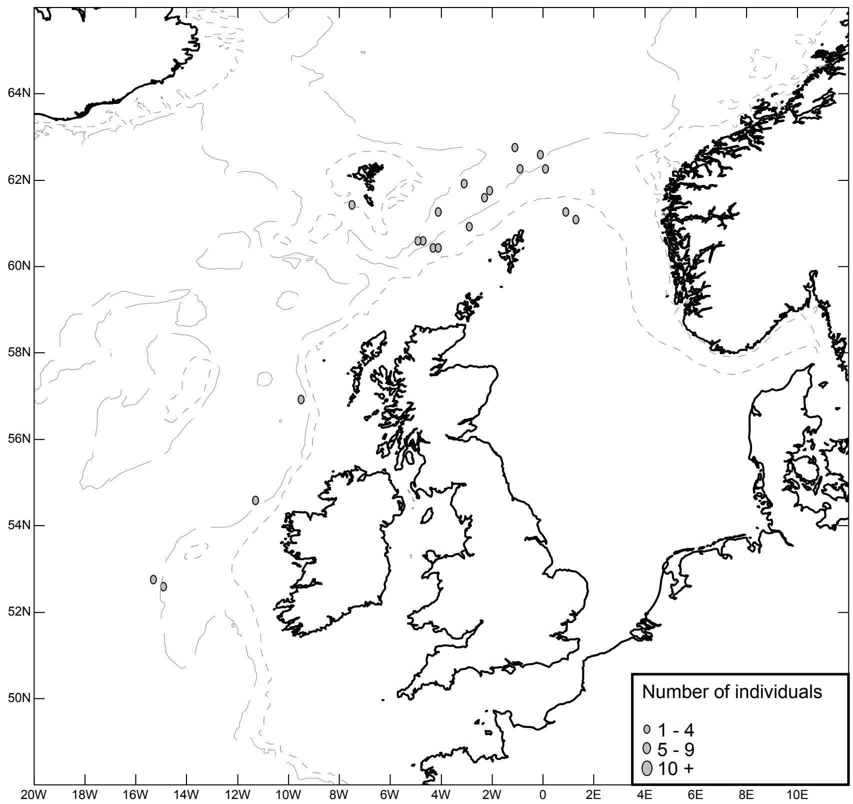

Figure 8.3. Sei whales encountered during seismic surveys, 1994-2010.

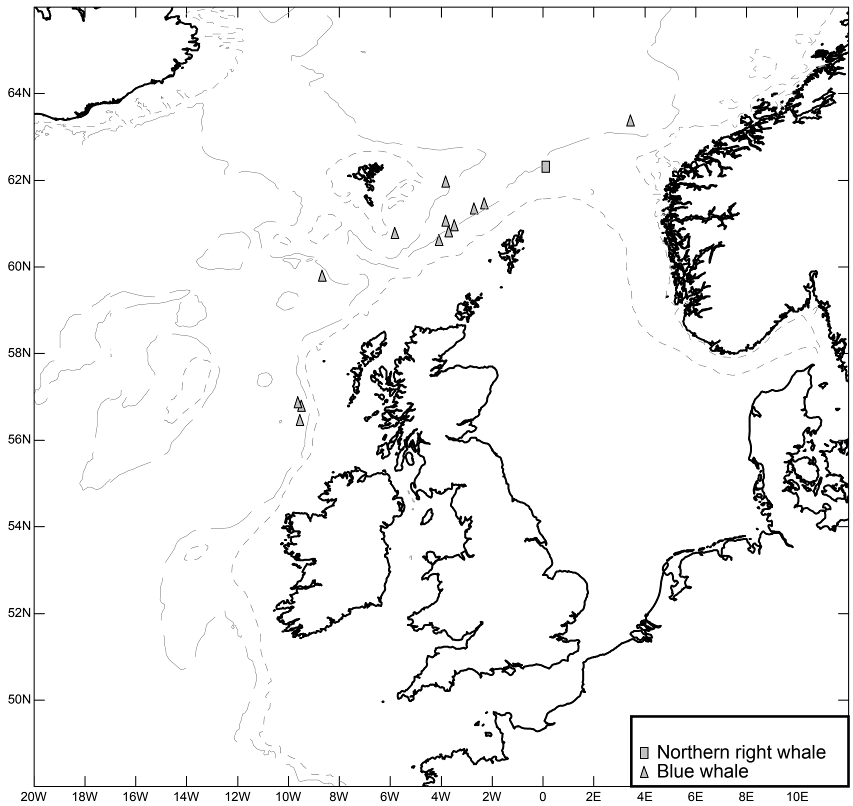

Figure 8.4. Northern right whale (probable) and blue whales encountered during seismic surveys, 1994-2010.

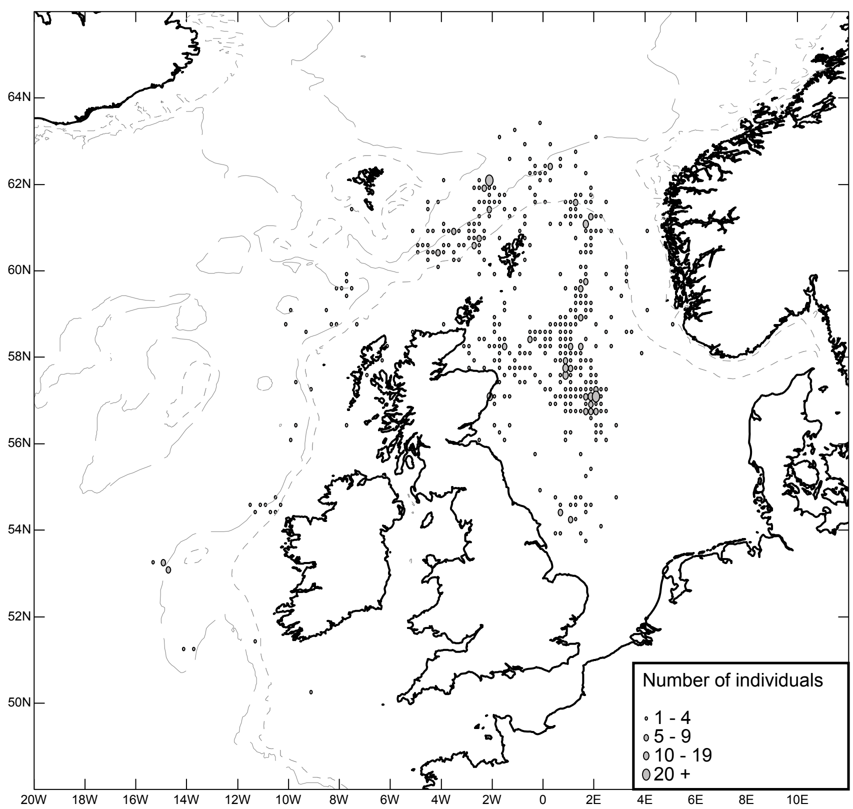

Figure 8.5. Minke whales encountered during seismic surveys, 1994-2010.

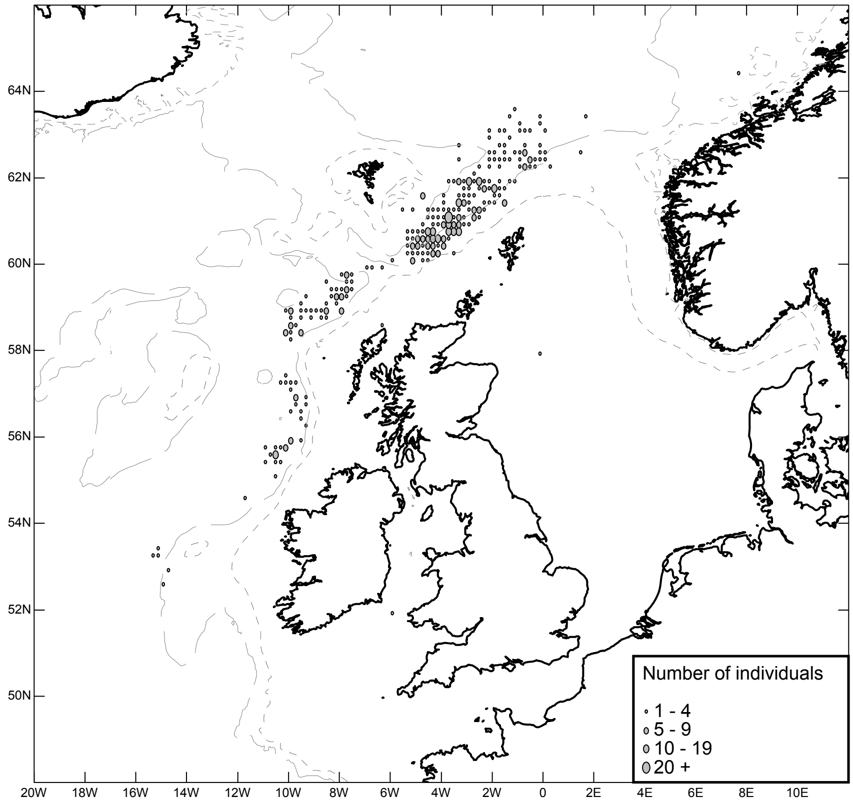

Figure 8.6. Sperm whales encountered during seismic surveys, 1994-2010.

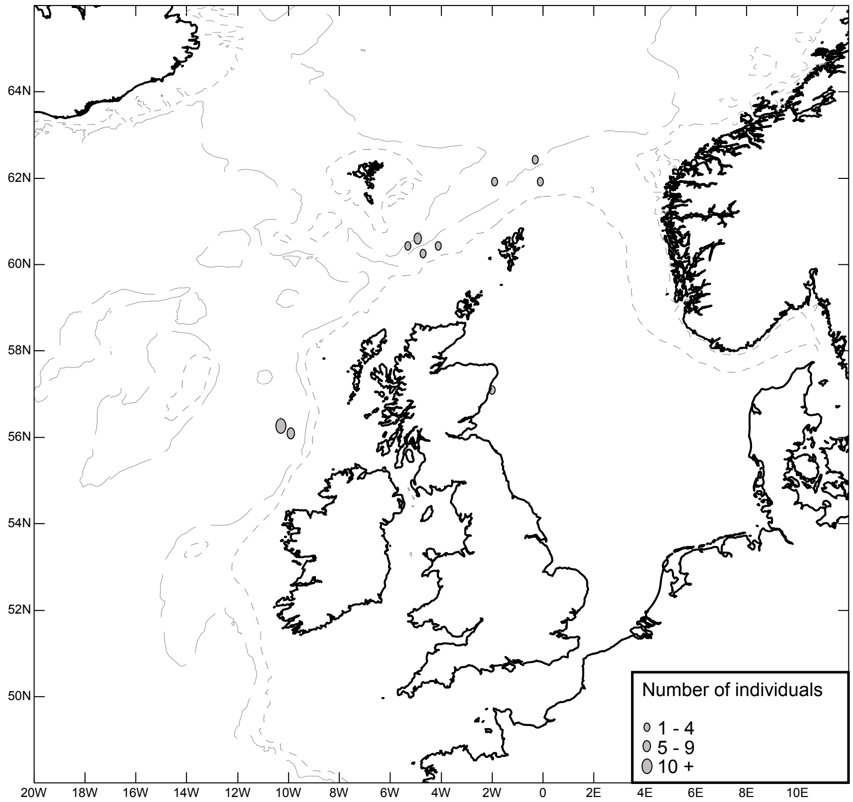

Figure 8.7. Northern bottlenose whales encountered during seismic surveys, 1994-2010.

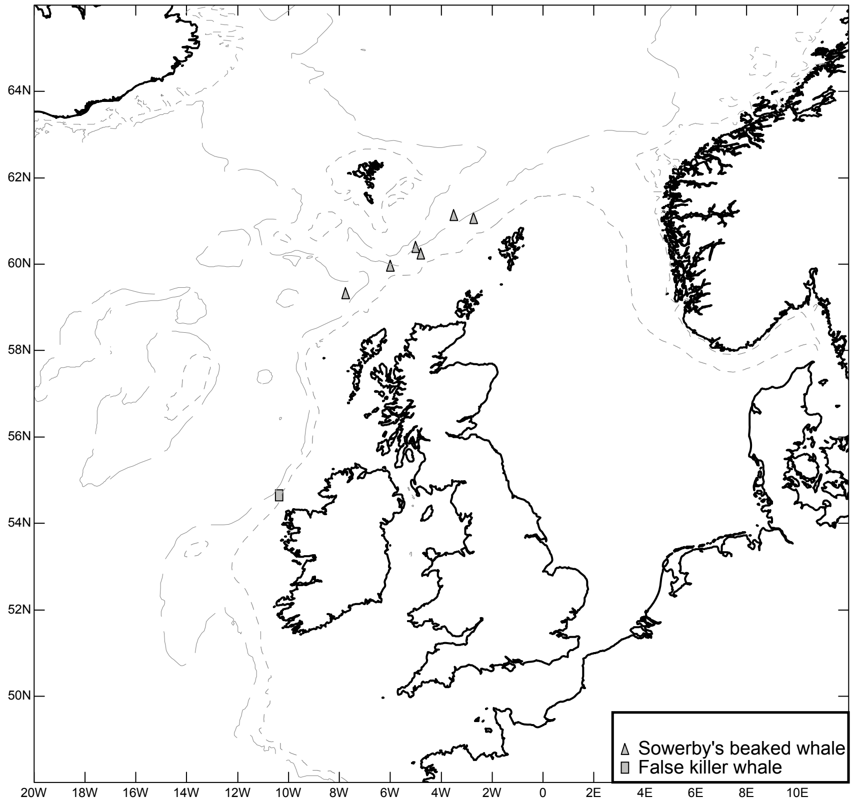

Figure 8.8. Sowerby's beaked whales and false killer whales encountered during seismic surveys, 1994-2010.

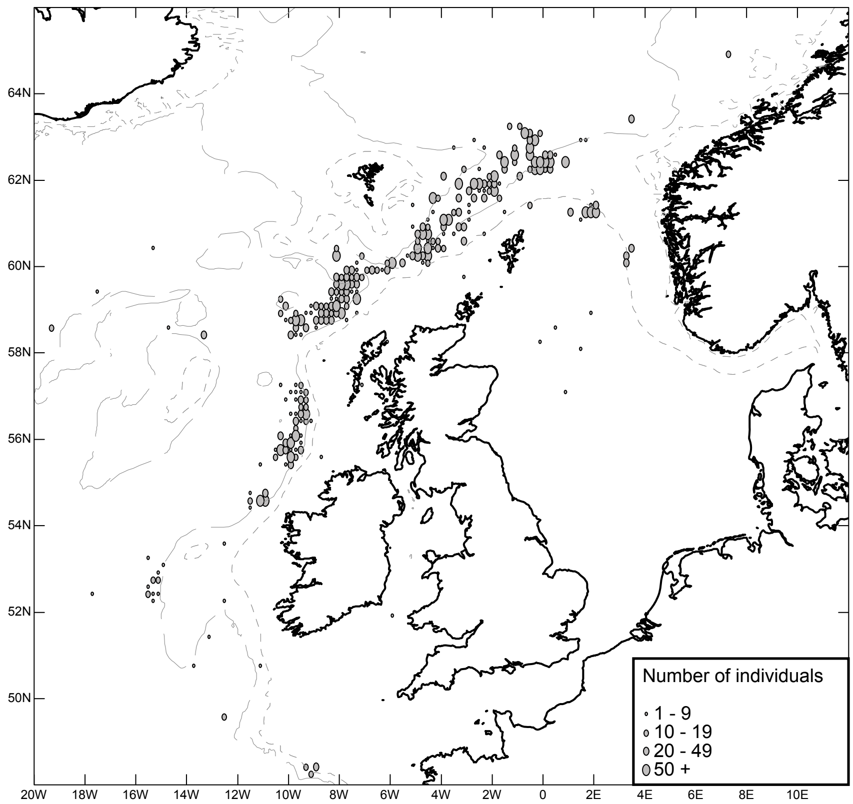

Figure 8.9. Long-finned pilot whales encountered during seismic surveys, 1994-2010.

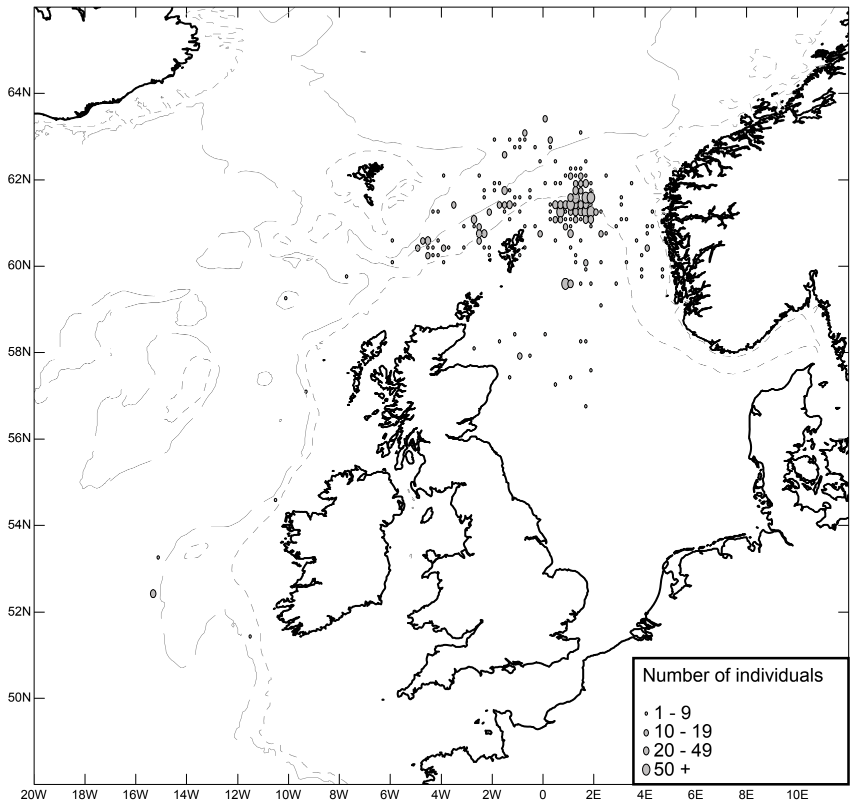

Figure 8.10. Killer whales encountered during seismic surveys, 1994-2010.

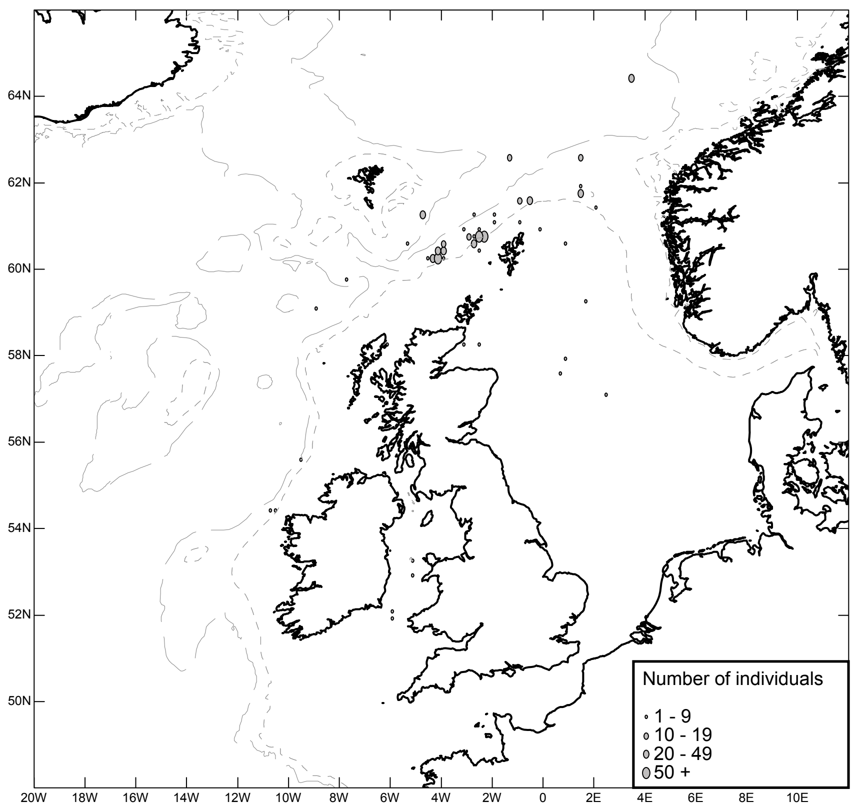

Figure 8.11. Risso's dolphins encountered during seismic surveys, 1994-2010.

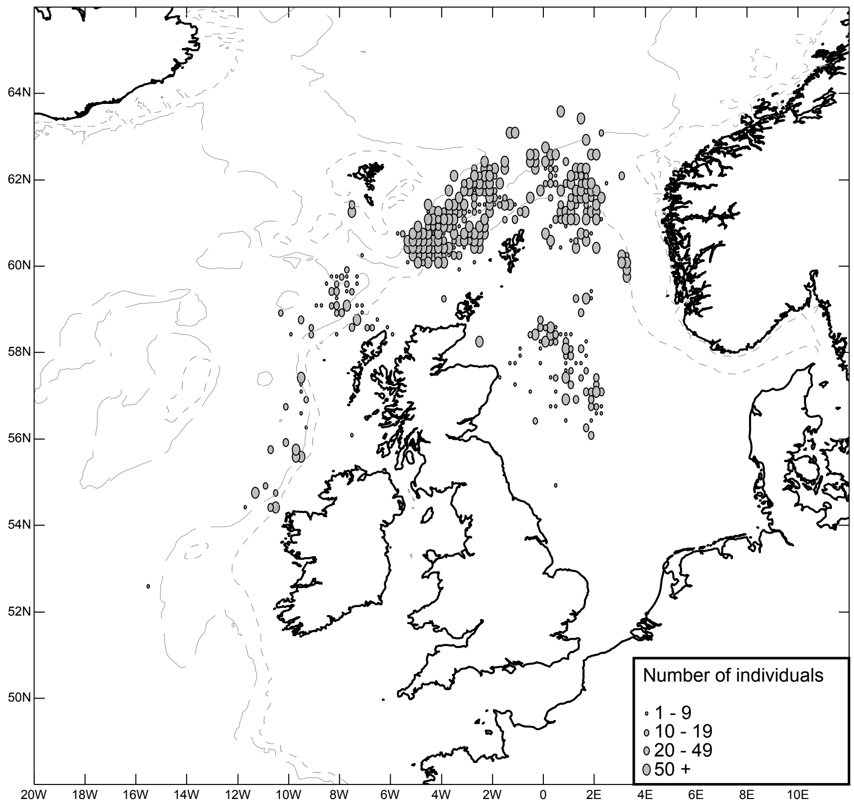

Figure 8.12. Atlantic white-sided dolphins encountered during seismic surveys, 1994-2010.

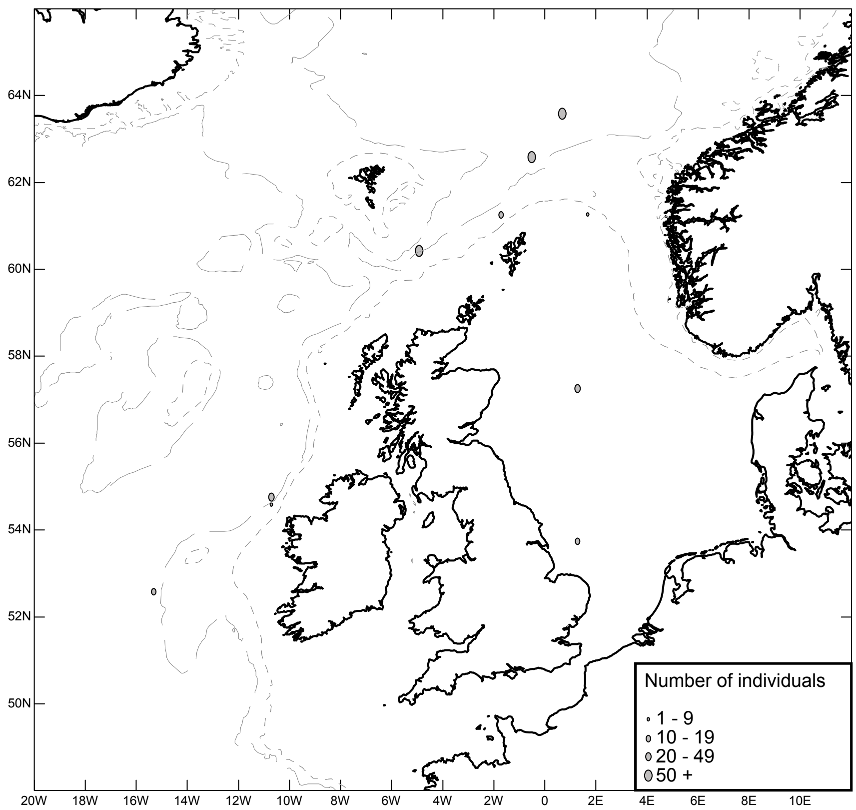

Figure 8.13. Striped dolphins encountered during seismic surveys, 1994-2010.

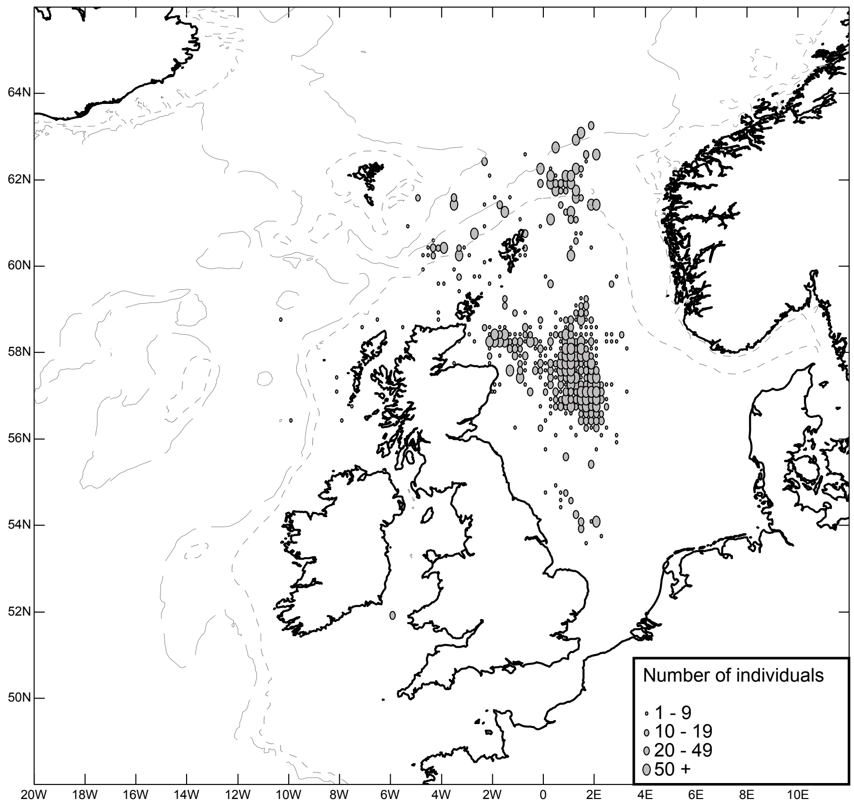

Figure 8.14. White-beaked dolphins encountered during seismic surveys, 1994-2010.

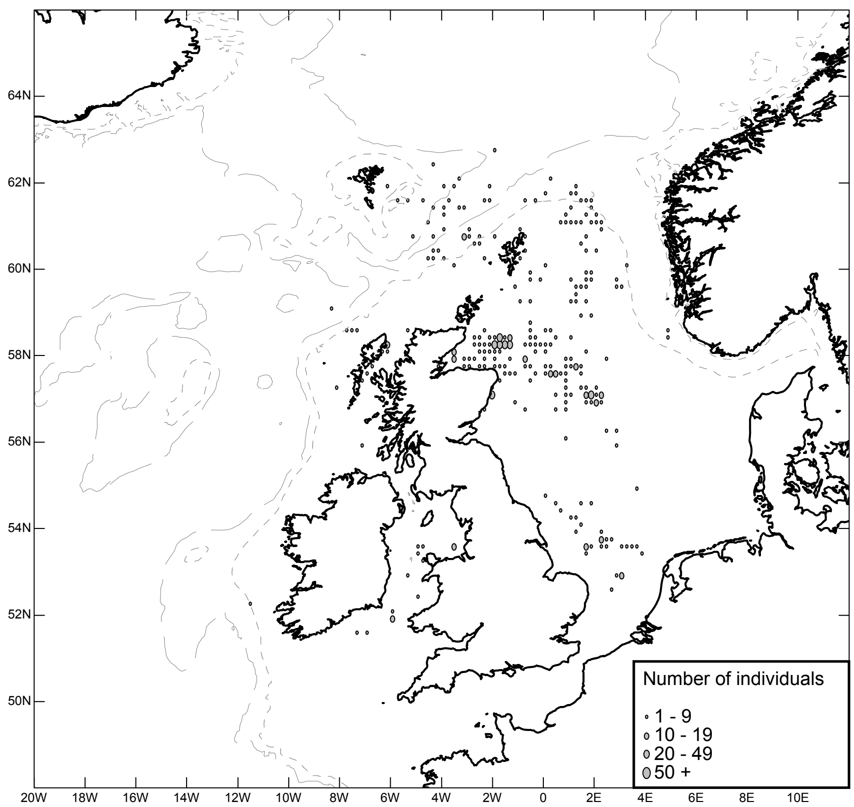

Figure 8.15. Harbour porpoises encountered during seismic surveys, 1994-2010.

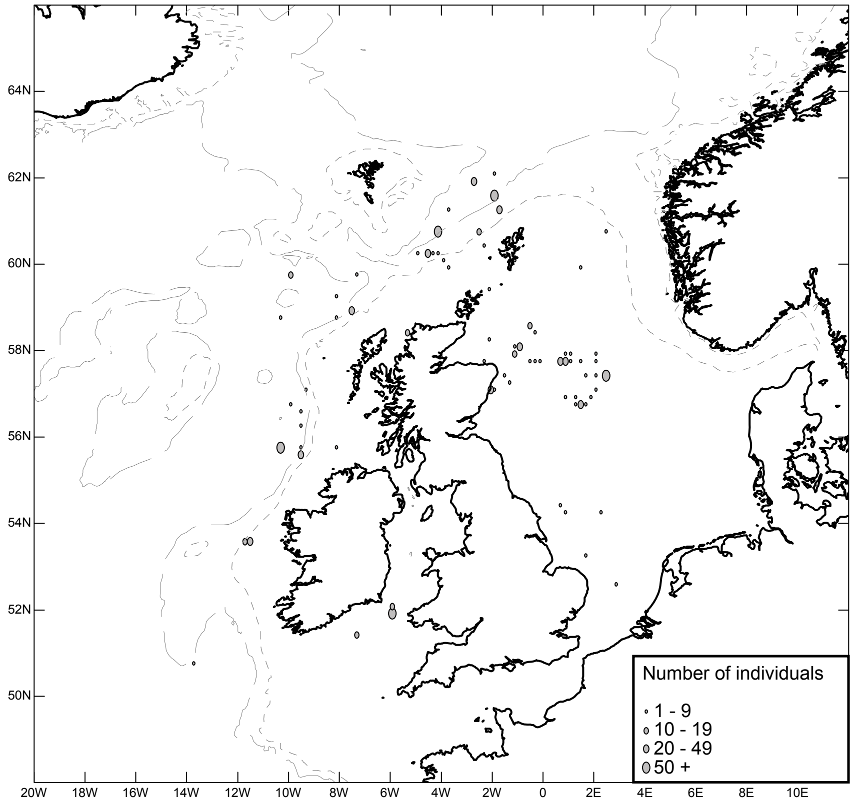

Figure 8.16. Bottlenose dolphins encountered during seismic surveys, 1994-2010.

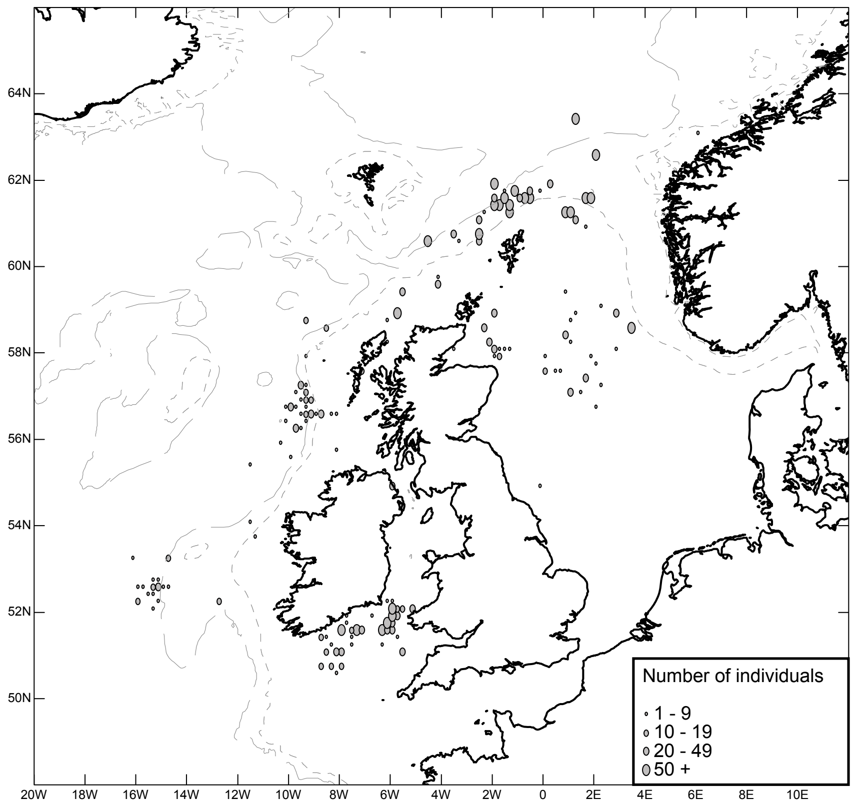

Figure 8.17. Short-beaked common dolphins encountered during seismic surveys, 1994-2010.

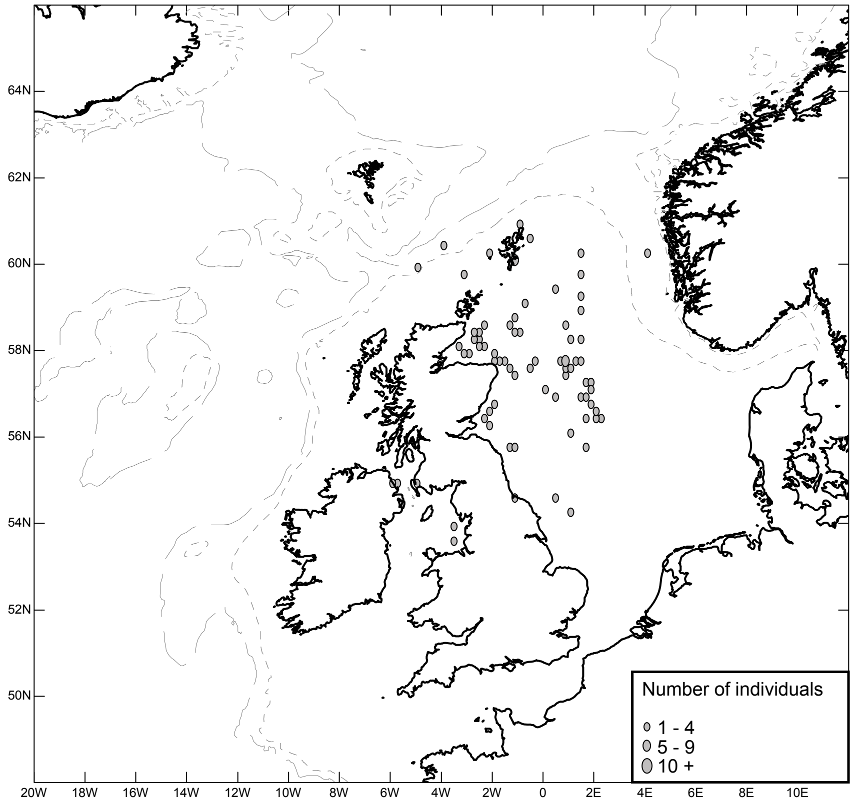

Figure 8.18. Grey seals encountered during seismic surveys, 1994-2010.

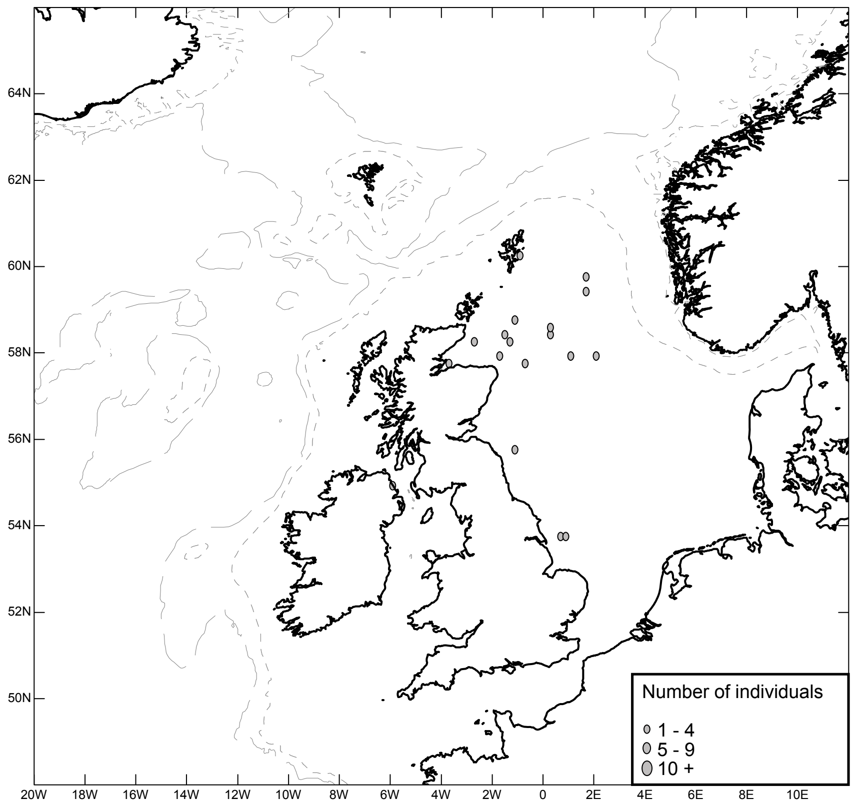

Figure 8.19. Harbour seals encountered during seismic surveys, 1994-2010.

Appendix 2

Scientific names of species mentioned in the text:

Harbour seal	*Phoca vitulina*
Grey seal	*Halichoerus grypus*
Ringed seal	*Pusa hispida*
Bowhead whale	*Balaena mysticetus*
Northern right whale	*Eubalaena glacialis*
Gray whale	*Eschrichtius robustus*
Humpback whale	*Megaptera novaeangliae*
Blue whale	*Balaenoptera musculus*
Fin whale	*Balaenoptera physalus*
Sei whale	*Balaenoptera borealis*
Minke whale	*Balaenoptera acutorostrata*
Sperm whale	*Physeter macrocephalus*
Northern bottlenose whale	*Hyperoodon ampullatus*
Sowerby's beaked whale	*Mesoplodon bidens*
Cuvier's beaked whale	*Ziphius cavirostris*
Short-finned pilot whale	*Globicephala macrorhynchus*
Long-finned pilot whale	*Globicephala melas*
Killer whale	*Orcinus orca*
False killer whale	*Pseudorca crassidens*
Risso's dolphin	*Grampus griseus*
Bottlenose dolphin	*Tursiops truncatus*
White-beaked dolphin	*Lagenorhynchus albirostris*
Atlantic white-sided dolphin	*Lagenorhynchus acutus*
Short-beaked common dolphin	*Delphinus delphis*
Striped dolphin	*Stenella coeruleoalba*
Pantropical spotted dolphin	*Stenella attenuata*
Atlantic spotted dolphin	*Stenella frontalis*
Harbour porpoise	*Phocoena phocoena*

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

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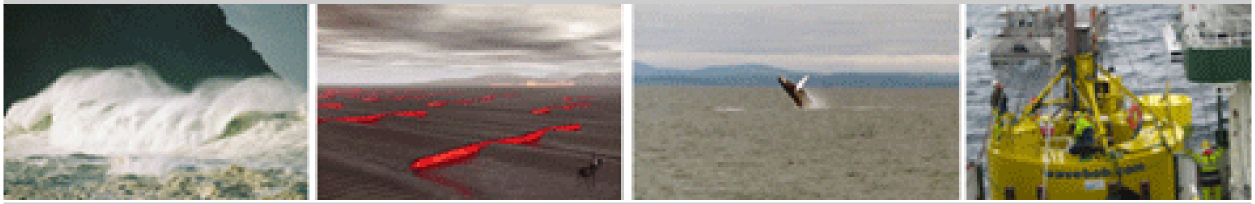
The Bureau of Ocean Energy Management (BOEM) promotes energy independence, environmental protection and economic development through responsible, science-based management of offshore conventional and renewable energy resources.



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.

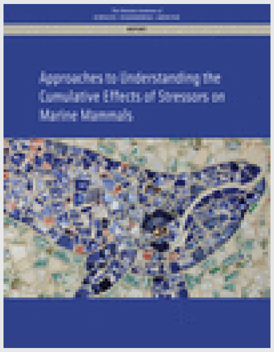
Click here for the fact sheet on Atlantic G&G Surveys Record of Decision.

- 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.

ATTACHMENT E

to July 21, 2017
Letter of IAGC/API/NOIA



Approaches to Understanding the Cumulative Effects of Stressors on Marine Mammals

DETAILS

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CONTRIBUTORS

Committee on the Assessment of the Cumulative Effects of Anthropogenic Stressors on Marine Mammals; Ocean Studies Board; Division on Earth and Life Studies; National Academies of Sciences, Engineering, and Medicine

Approaches to Understanding the Cumulative Effects of Stressors on Marine Mammals

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Ocean Studies Board

Division on Earth and Life Studies

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COMMITTEE ON THE ASSESSMENT OF THE CUMULATIVE EFFECTS OF ANTHROPOGENIC STRESSORS ON MARINE MAMMALS

PETER L. TYACK, *Chair*, University of St. Andrews, Scotland
HELEN BAILEY, University of Maryland Center for Environmental Science, Solomons
DANIEL E. CROCKER, Sonoma State University, Rohnert Park, California
JAMES E. ESTES, University of California, Santa Cruz
CLINTON D. FRANCIS, California Polytechnic State University, San Luis Obispo
JOHN HARWOOD, University of St. Andrews, Scotland
LORI H. SCHWACKE, Hollings Marine Laboratory (National Oceanic and Atmospheric Administration), Charleston, South Carolina
LEN THOMAS, University of St. Andrews, Scotland
DOUGLAS WARTZOK, Florida International University, Miami

Staff

KIM WADDELL, Study Director, Gulf Research Program
STACEE KARRAS, Associate Program Officer, Ocean Studies Board
PAYTON KULINA, Senior Program Assistant, Ocean Studies Board
NICHOLAS MACFARLANE, Mirzayan Fellow, Ocean Studies Board

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
ANNA FAIRBROTHER, Exponent EcoSciences
FRANCES GULLAND, Marine Mammal Center
ALAN HASTINGS, University of California, Davis
GARY ISAKSEN, ExxonMobil Exploration Company
CARRIE KAPPEL, National Center for Ecological Analysis and Synthesis
JOHN ORCUTT, Scripps Institution of Oceanography
BRANDON SOUTHALL, Southall Environmental Associates, Inc.
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

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 cumula-

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.

tive 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.

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

[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

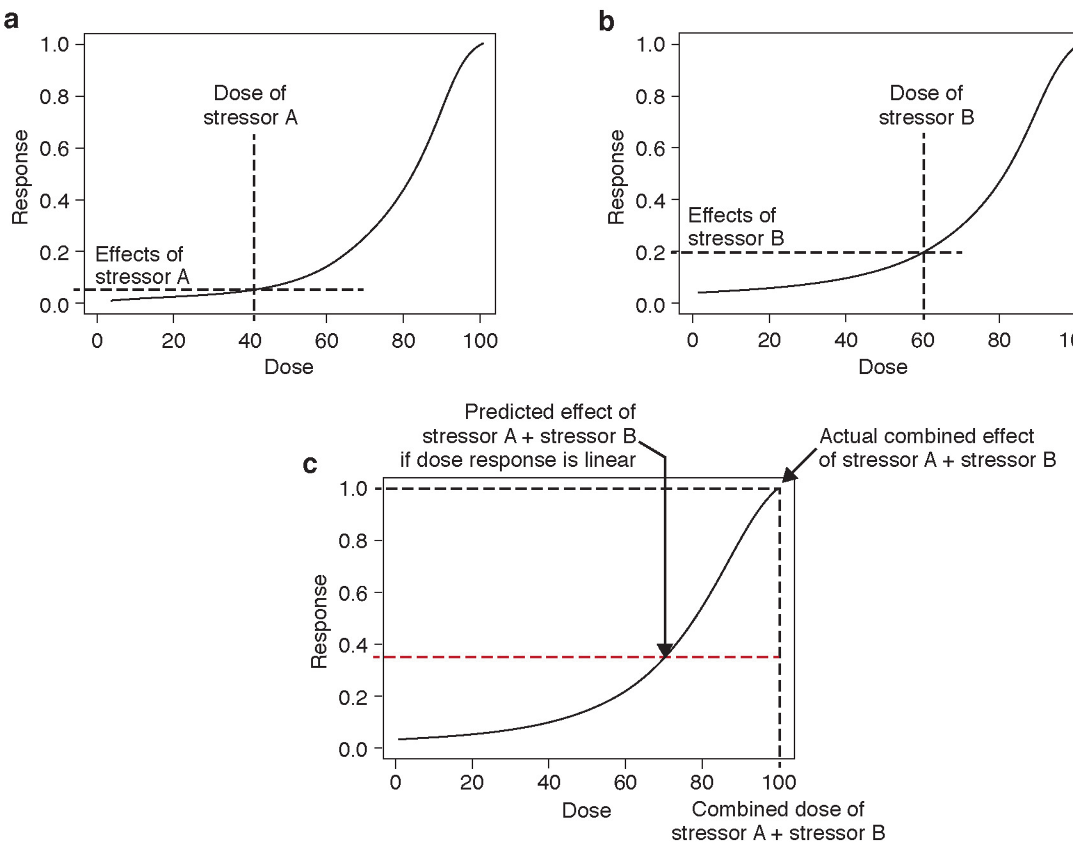

FIGURE S.1 This figure illustrates how the potential for interaction between two stressors (A and B) that share a common mechanism of action depends on the form of the dose–response relationship. (a) Effect of stressor A alone. (b) Effect of stressor B alone. (c) Effect of a combined dose of stressor 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

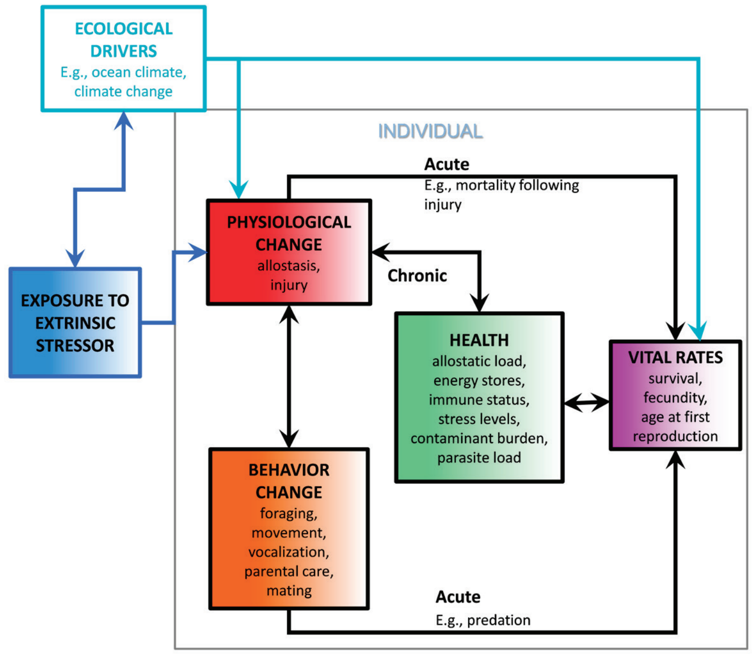

FIGURE S.2 The Population Consequences of Multiple Stressors (PCoMS) framework for a single individual exposed to one stressor. Each compartment in the framework represents one or more quantities (variables) that evolve over time. Compartments are connected by arrows that represent causal flows ("transfer 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

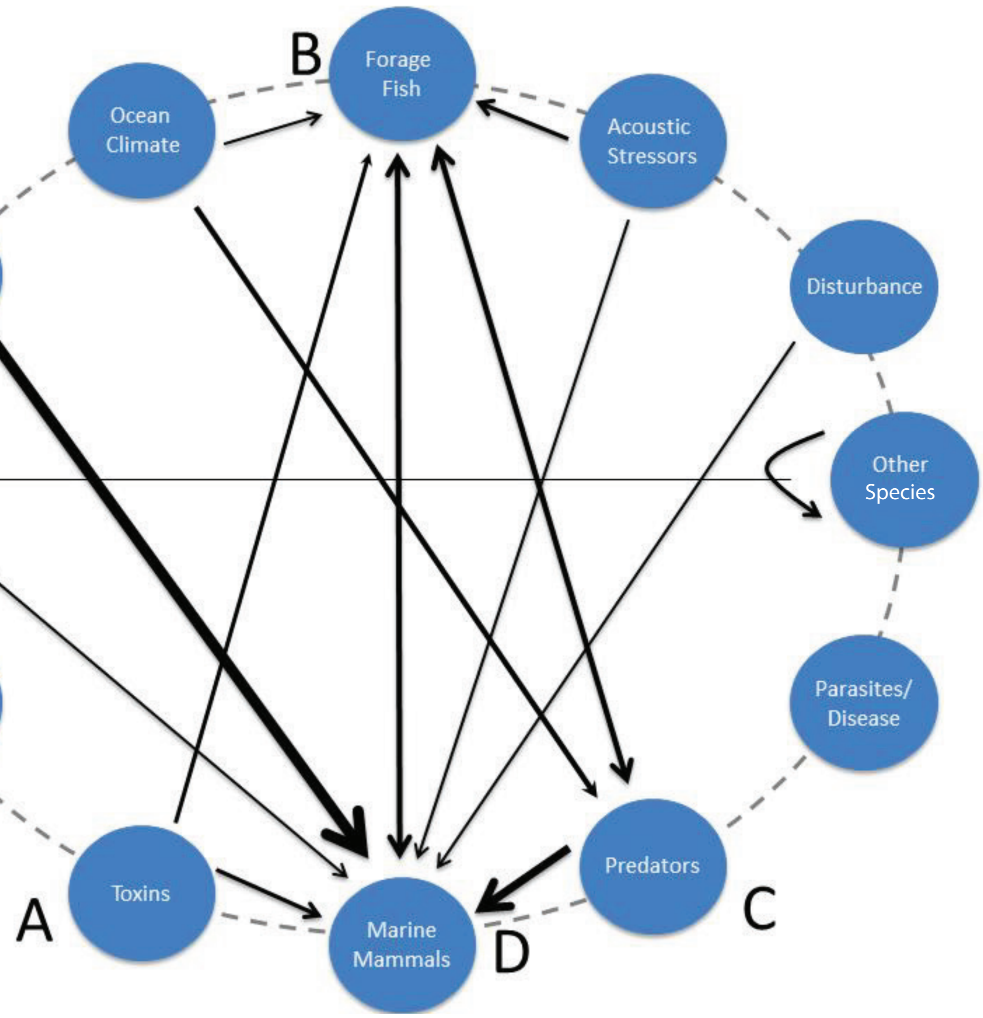

FIGURE S.3 Schematic illustration of an interaction web. Circles around the perimeter of the dashed oval represent species or elements of the abiotic environment (collectively referred to as nodes), and arrows between circles represent species interactions or interactions between species and the abiotic 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.

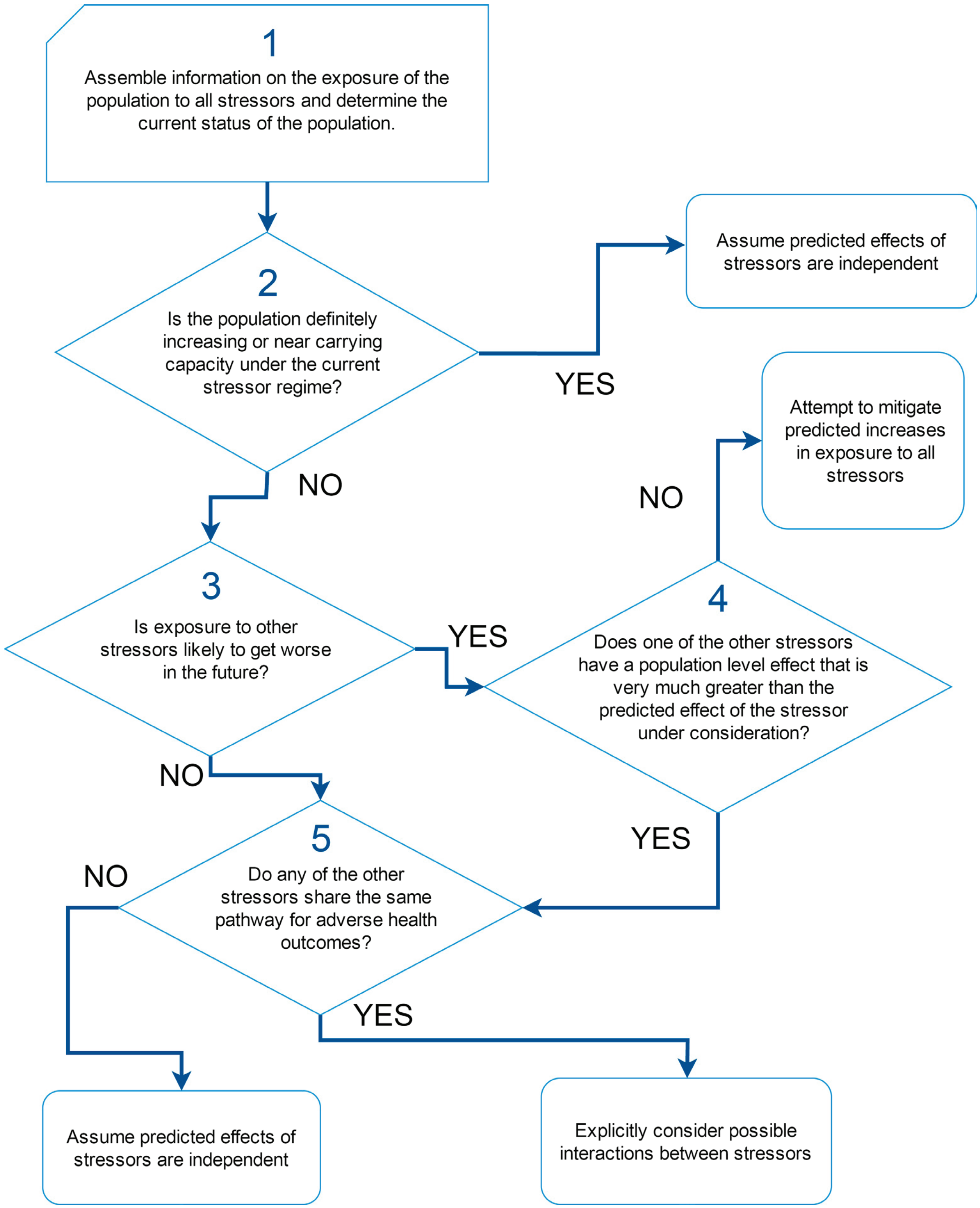

FIGURE S.4 A decision tree for identifying situations where studies of the possible interactions between stressors should be given a high priority when considering the effect of a focal stressor on a population.

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

[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

[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

[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.

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.

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.

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).

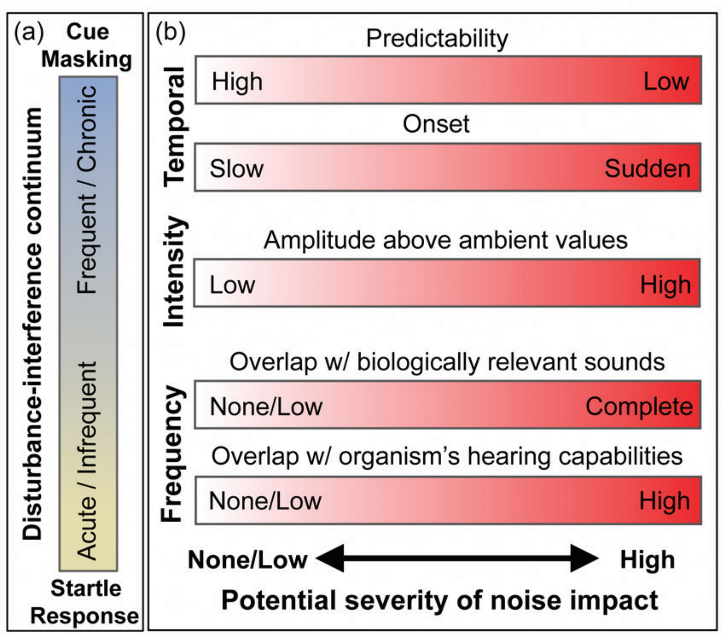

FIGURE 2.1 (a) The disturbance–interference continuum can range from acute or infrequent noise stimuli that will likely trigger startle, flight, or hide responses to frequent or chronic noises that interfere with cue detection. (b) The severity of an impact from a noise stimulus will depend on the temporal,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

[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

[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.

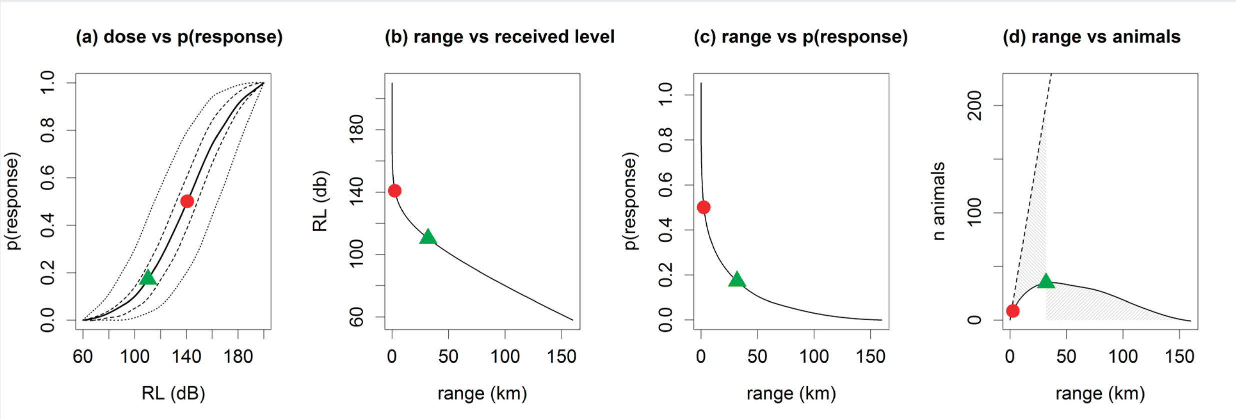
[6] See http://www.westcoast.fisheries.noaa.gov/protected_species/marine_mammals/threshold_guidance.html.

[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).

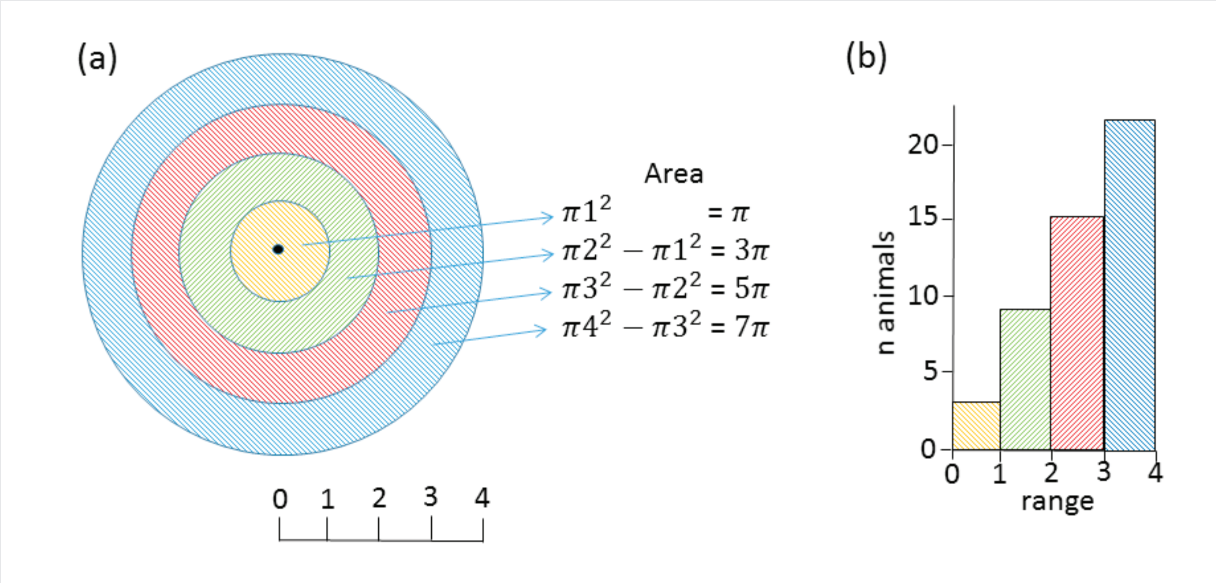
Regarding the second issue, uncertainty on inputs can be translated into uncertainty on take estimates readily through stochastic simulation. Regulators may then choose the level of risk they wish to use in deciding whether to permit an activity (e.g., Taylor et al., 2000). For simple cases, simulation 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_{SEL} for high-frequency (HF) cetaceans to 193 dB_{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_{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),

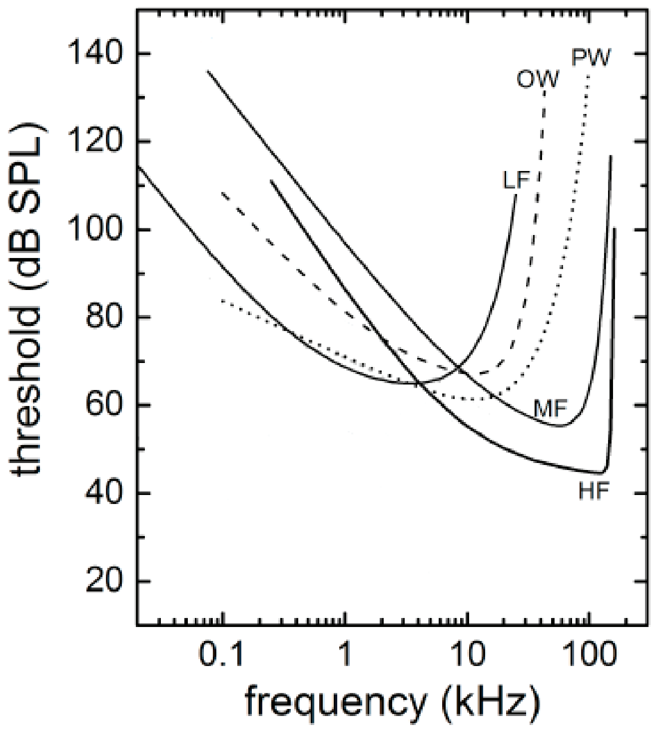


FIGURE 2.2 Composite audiograms obtained through behavioral testing except for LF that was calculated. NOTE: HF = high-frequency cetaceans; LF = low-frequency cetaceans; MF = mid-frequency cetaceans; OW = otariids, walrus, and sea otter in water; PW = phocids in water. Thresholds are expressed in dB_{RMS} 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 harrasment 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.

[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 $dB_{P\text{-}P}$ 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 $dB_{P\text{-}P}$ 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 $dB_{P\text{-}P}$ 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_{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_{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

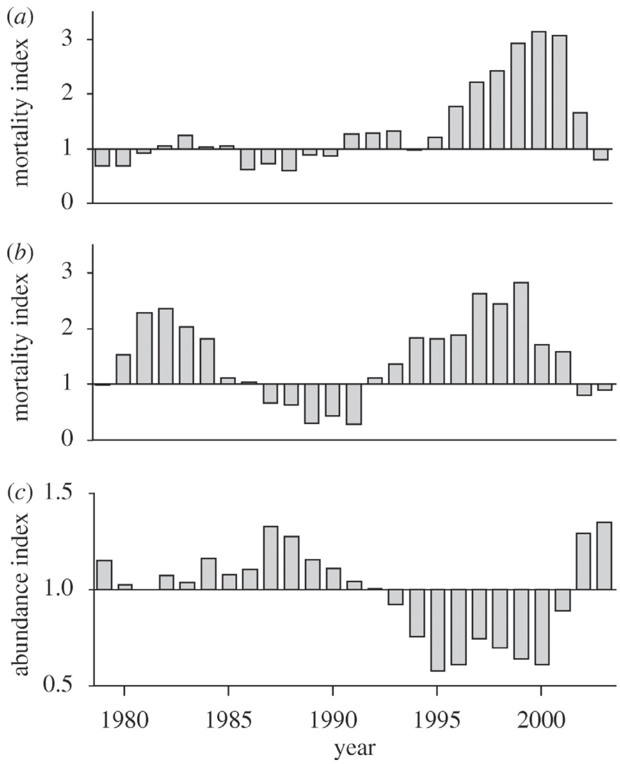


FIGURE 3.1 Mortality of (a) northern and (b) southern resident killer whales negatively covary with (c) abundance of Chinook salmon. (a, b) Values above or below 1 reflect higher or lower mortality rates than expected or (c) higher or lower abundance of Chinook salmon than the average for the time series.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

[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."

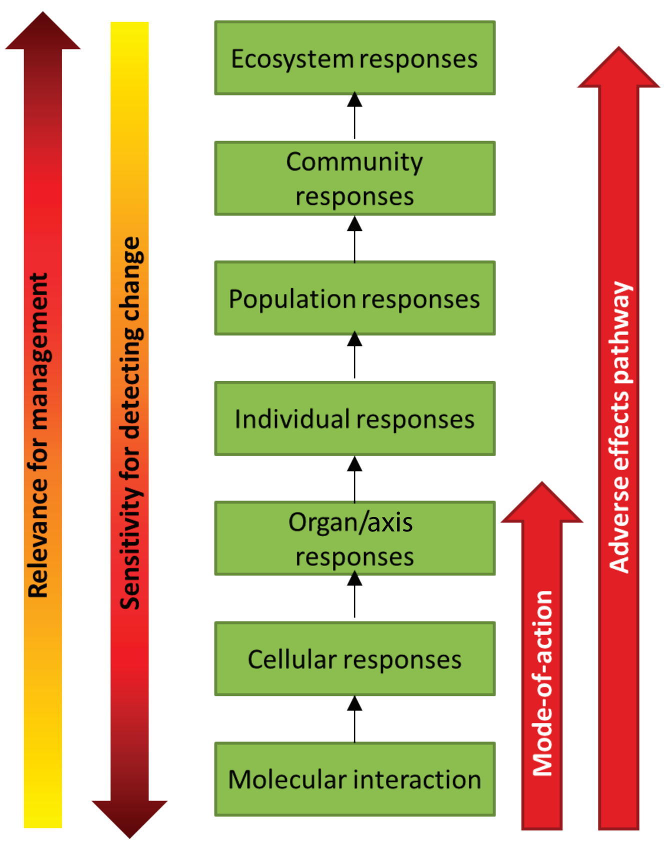


FIGURE 4.1 The hierarchy of responses to a stressor across multiple levels of biological organization.

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:

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

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 nonchemical 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

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

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

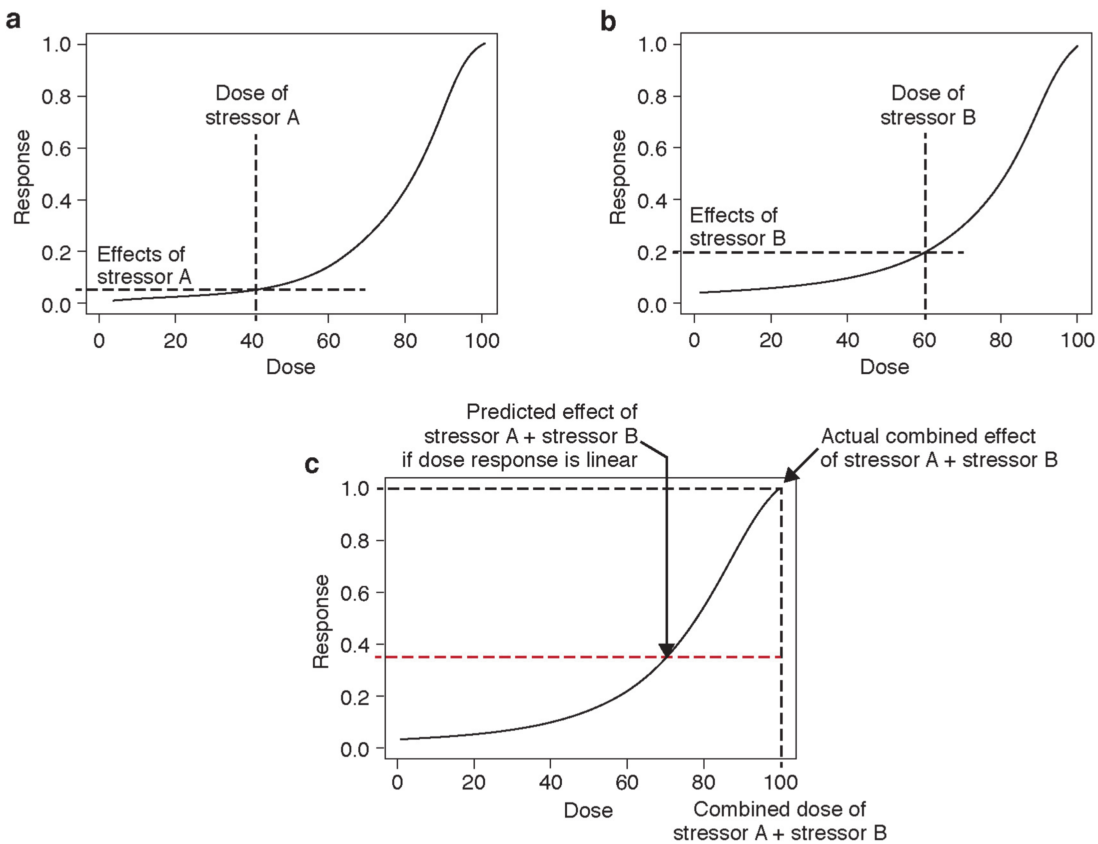


FIGURE 4.2 This figure illustrates how the potential for interaction between two stressors (A and B) that share a common mechanism of action depends on the form of the dose–response relationship. (a) Effect of stressor A alone. (b) Effect of stressor B alone. (c) Effect of a combined dose of stressor 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

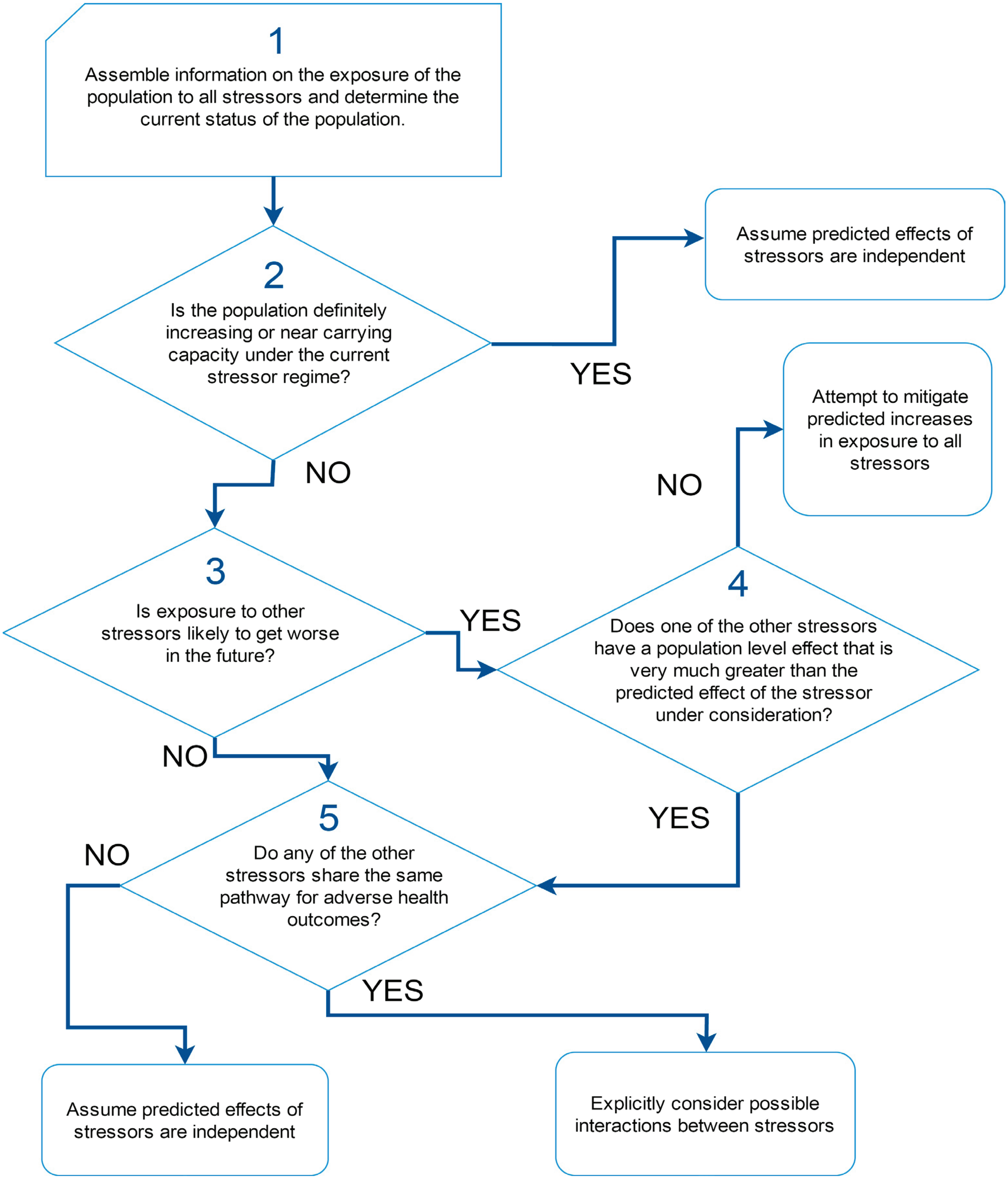


FIGURE 4.3 A decision tree for identifying situations where studies of the possible interactions between stressors should be given a high priority when considering the effect of a focal stressor on a population. See text for a detailed description of the decision-making process.

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

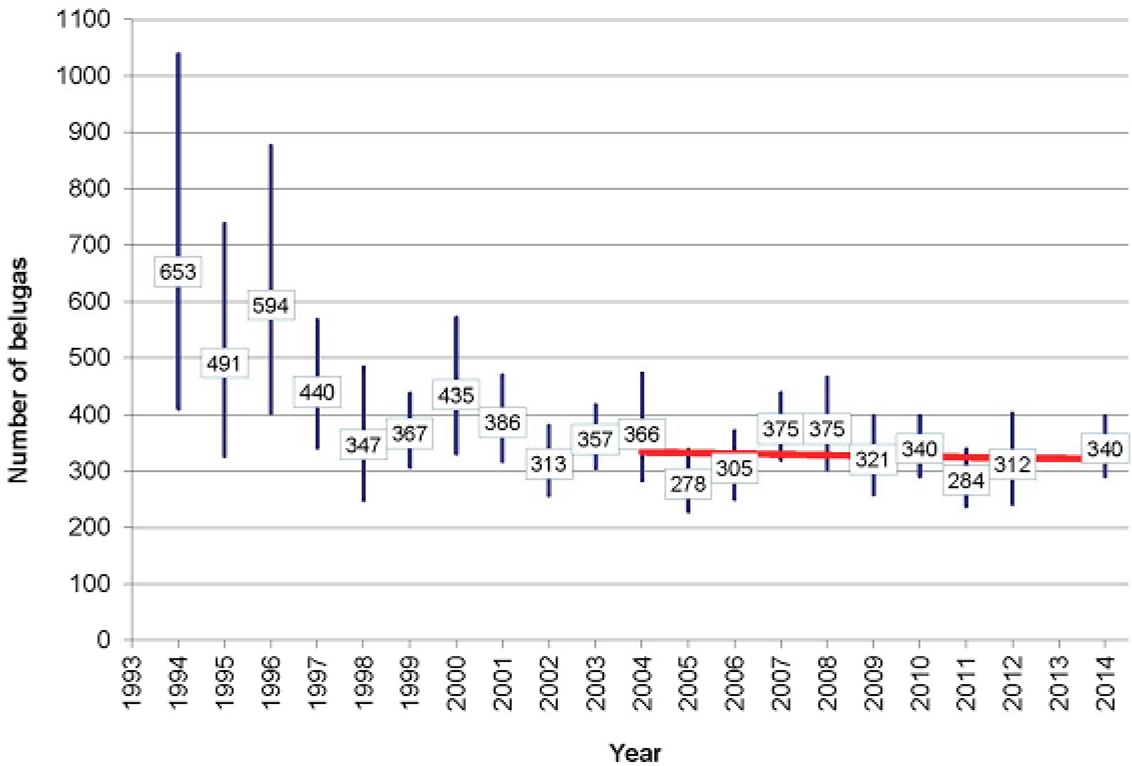


FIGURE 4.4 Figure 13 from NMFS (2015) showing abundance estimates for Cook Inlet beluga whales between 1994 and 2014. Vertical bars indicate the 95% confidence interval for each estimate. The trend from 1999 (when the hunt was managed) to 2014 was −1.3% per year (standard error [SE] = 0.7%).

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

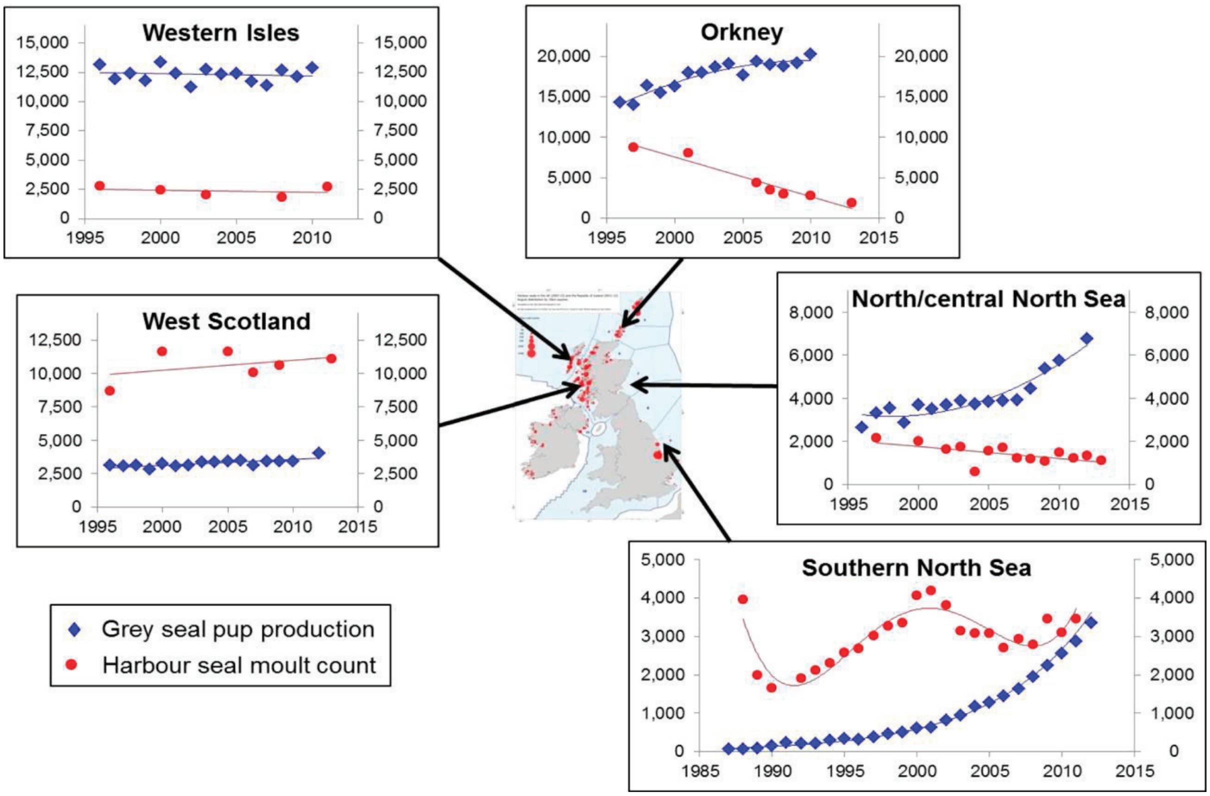


FIGURE 4.5 Changes in harbor seal molt counts and grey seal pup counts for the United Kingdom over the period 1996-2013. SOURCE: Taken from Figure 1 of Hall et al. (2015).

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

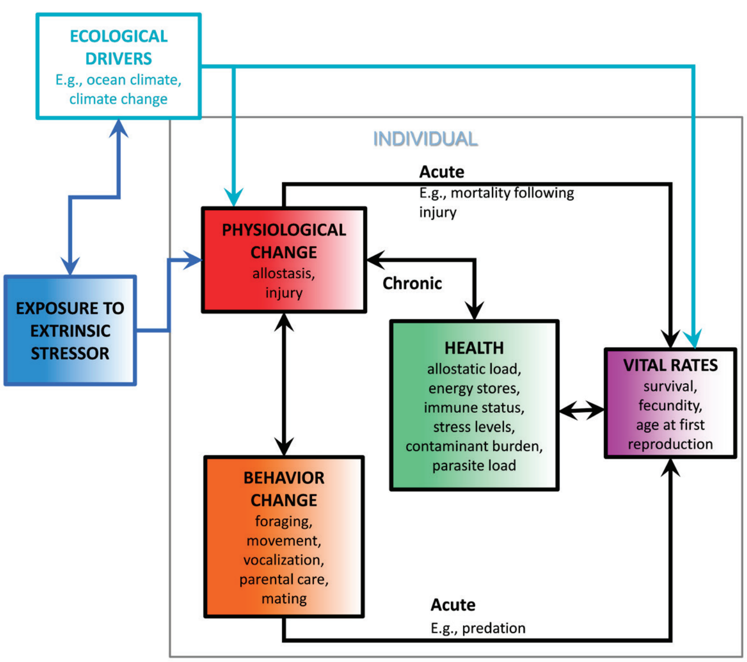


FIGURE 5.1 The Population Consequences of Multiple Stressors (PCoMS) framework for a single individual exposed to one stressor. Each compartment in the framework represents one or more quantities (variables) that evolve over time. Compartments are connected by arrows that represent causal flows ("transfer 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

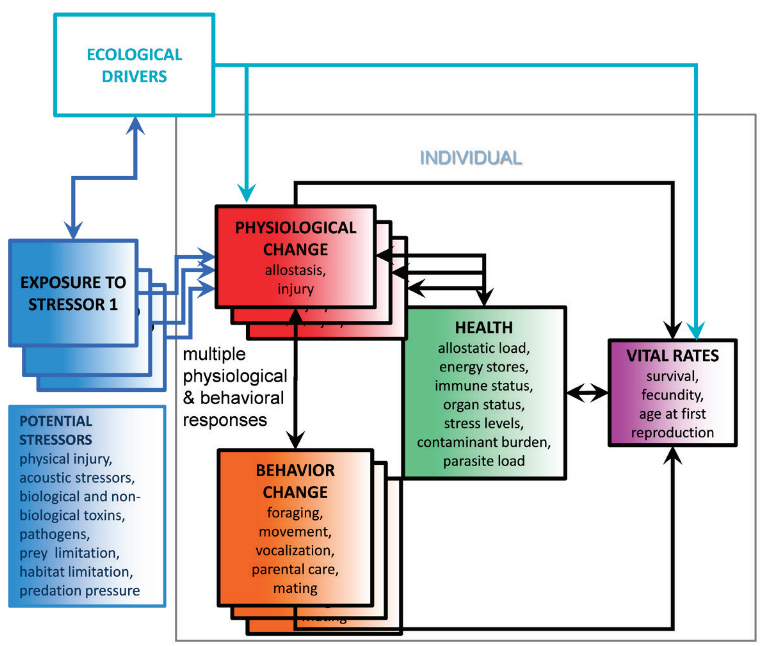


FIGURE 5.2 An expanded version of the framework shown in Figure 5.1 that includes the effect of multiple stressors on a single individual.

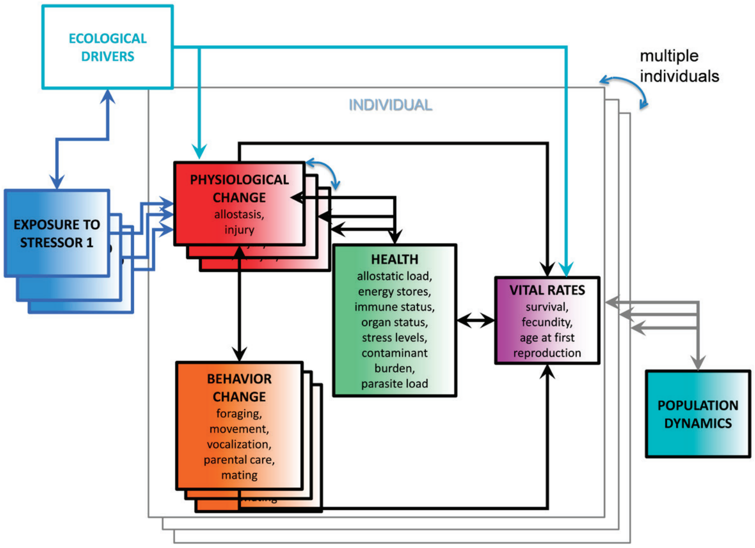


FIGURE 5.3 An expanded version of the framework shown in Figure 5.2 that includes multiple individuals and population-level consequences.

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 non-chemical 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,

[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

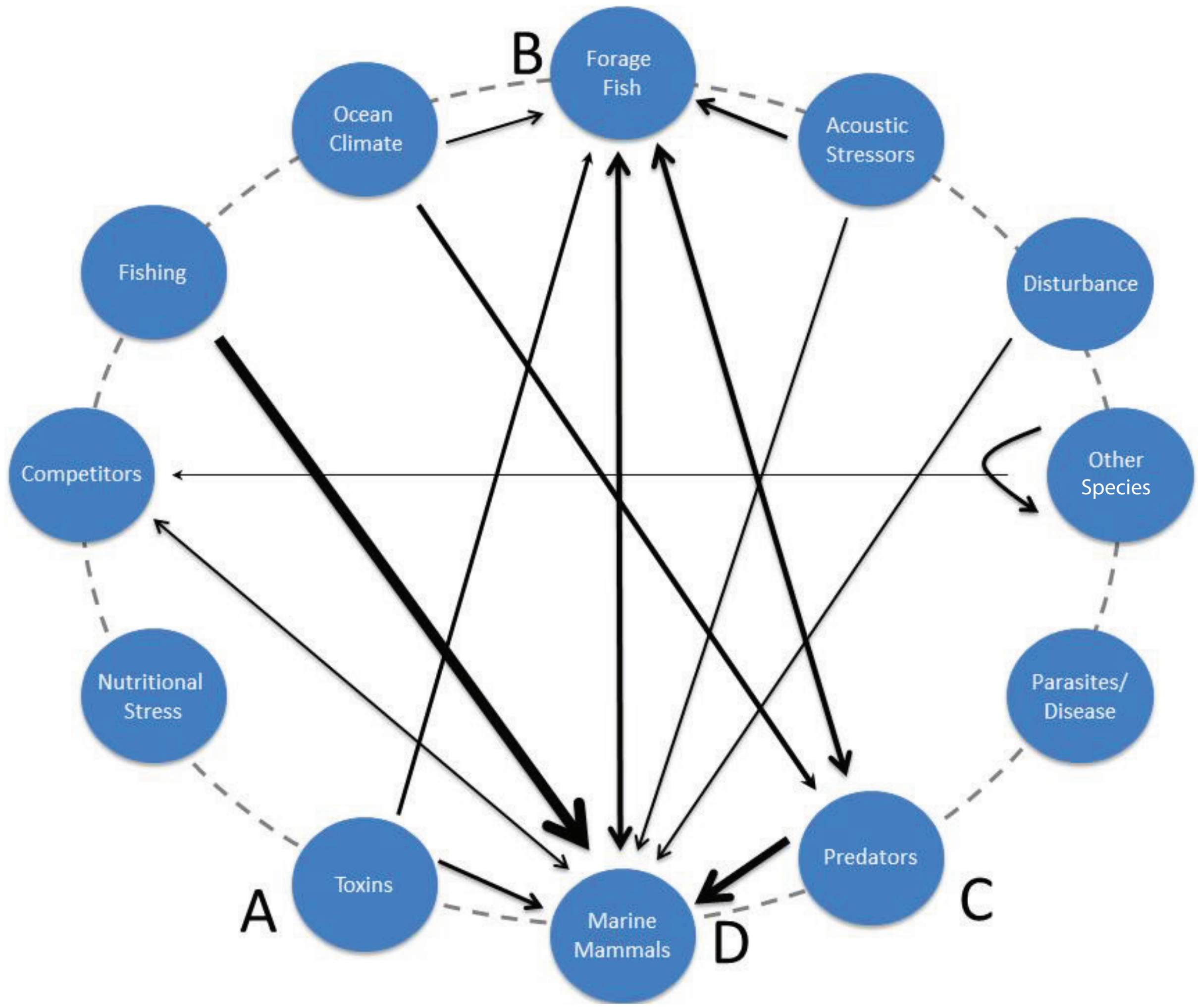


FIGURE 6.1 Schematic illustration of an interaction web. Circles around the perimeter of the dashed oval represent species or elements of the abiotic environment (collectively referred to as nodes), and arrows between circles represent species interactions or interactions between species and the abiotic 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

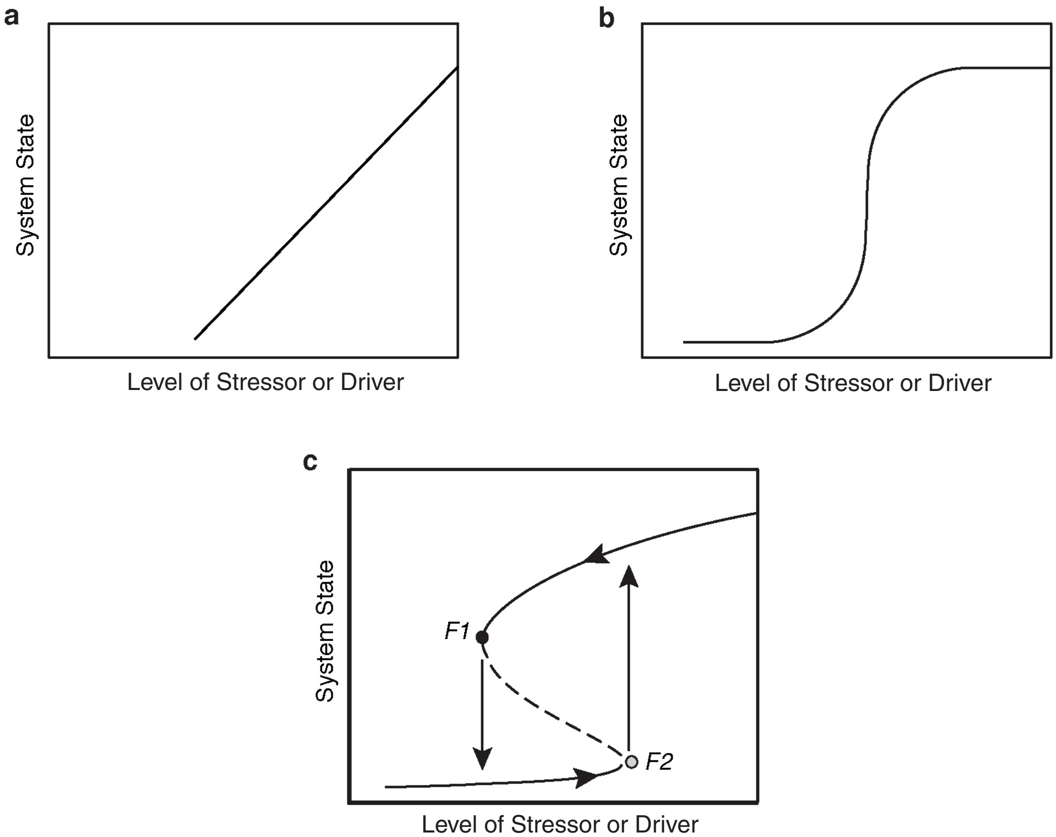


FIGURE 6.2 State-space graphs capture the functional relationships (all direct and indirect interactions) between a stressor and its effect on the state of a system. This relationship may be (a) linear or (b, c) nonlinear. The abrupt transitions depicted in (b) and (c) are often referred to as *phase 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.

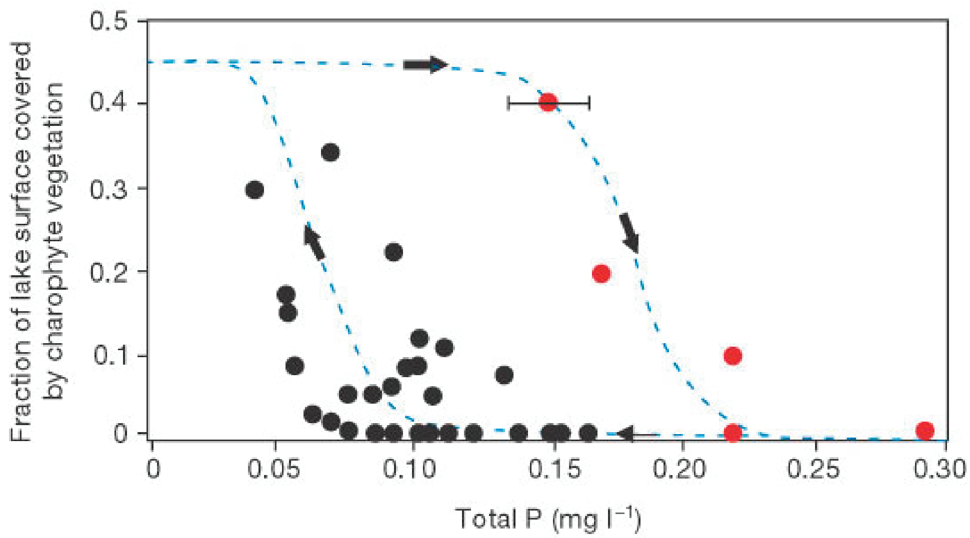
Ecosystems can shift among different basins of attraction (Scheffer et al., 2001)—different configurations to the distribution and abundance of species, in which movement from one basin to another requires a strong perturbation. This situation can be likened to the behavior of a ball over a three-dimensional 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

Panel A



Panel B

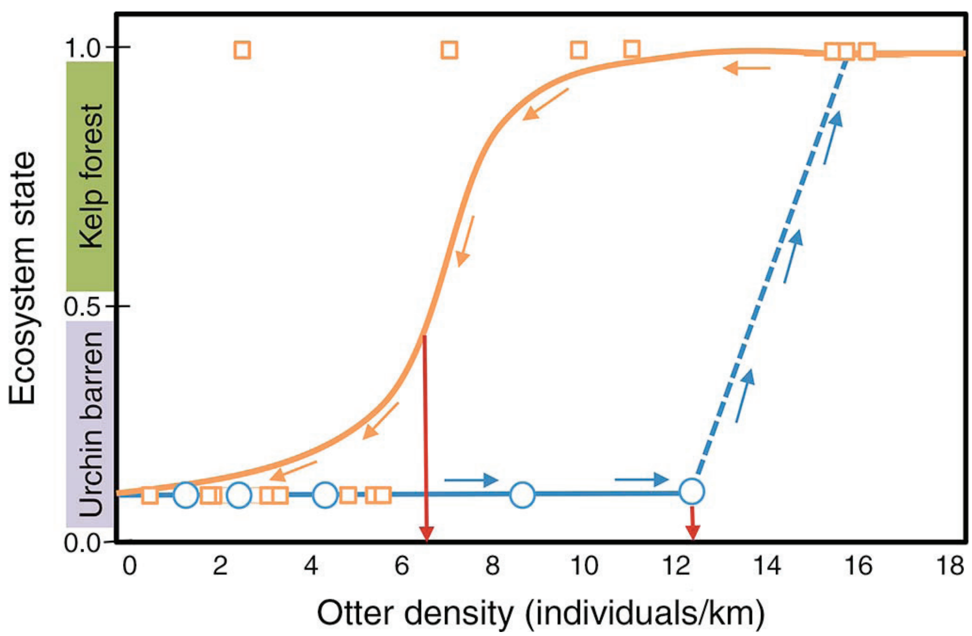


FIGURE 6.3 Two empirical examples of hysteresis: Panel A shows the differing response of charophyte vegetation in a shallow European lake to an increase (red dots) followed by a decline (black dots) in phosphorous concentration. SOURCE: Scheffer et al. (2001). Panel B demonstrates the differing sea otter 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

[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.

[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

[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 up-sweeps. *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

[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.

[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

[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

[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 non-impulsive 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.

[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).

ATTACHMENT F

to July 21, 2017

Letter of IAGC/API/NOIA

efficiency in recent years. Using standard hardware, we now acquire more and better quality data due to advancements in vessels, configurations, acquisition planning and execution, and data processing. 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 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 and 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 the most modern technology to make improved evaluations. Moreover, because G&G activities are temporary and transitory, 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.

Finally, 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 to make the technology commercially viable. Moreover, the hypothetical environmental benefits of alternative technologies have not been demonstrated.

B. Chapters 6 and 7 of the Application Are Substantially Flawed

The MMPA implementing regulations require an application for ITRs to describe, among other things:

- "The type of incidental taking authorization that is being requested . . . and the method of incidental taking" (50 C.F.R. § 216.104(5));
- "[T]he number of marine mammals (by species) that may be taken by each type of taking . . . and the number of times such takes by each type of taking are likely to occur" (*id*. § 216.104(6)); and
- "The anticipated impact of the activity upon the species or stock of marine mammal" (*id*. § 216.104(7)).

The purpose of this information is to allow NMFS to assess the impacts that are "reasonably likely" or "reasonably expected" to occur based on the best scientific information available. 50 C.F.R. §§ 216.102(a), 216.103.

Unfortunately, the Application presents an unrealistic and inaccurate assessment of the number of marine mammals that may be incidentally taken and the associated impacts. Specifically, the Application (1) is intentionally designed to overestimate take, (2) is based upon biased modeling derived from flawed assumptions, (3) does not utilize all of the best available scientific information, and (4) improperly fails to incorporate the known beneficial effects of mitigation measures. As a result, the Application does not present the number of incidental takes that are "likely to occur," does not describe the "anticipated" impact of the geophysical activities, and ultimately prevents NMFS's from determining the "reasonably expected" or "reasonably likely" impacts of the contemplated ITRs. These flaws are addressed in the following subsections.

1. Chapter 6 is designed to substantially overestimate the amount of potential incidental takes

By BOEM's admission, the modeling used to estimate the anticipated number of incidental takes is intentionally designed to overestimate takes and impacts. *See* Application at 93 (the "modeling results are meant to be precautionary and likely overestimate 'exposures' and therefore 'takes'"; "modeling inputs and results are purposely precautionary in order to avoid underestimating potential impacts to marine mammals"). BOEM candidly describes the modeling effort in the DPEIS as follows:[11]

> This estimate 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. Biological significance to marine mammals is left to interpretation by the subject-matter experts.

DPEIS at 1-16.

The estimates of "exposures" that are used in the Application as surrogates for estimated takes "are based on acoustic and impact models that are, by their nature, conservative and complex." DPEIS at 1-19. Indeed, "[e]ach of the inputs into the models is purposely developed to be conservative, and this conservativeness accumulates throughout the analysis." *Id.* (emphasis added). As a result, the exposure estimates are "higher than BOEM expects would

[11] The same modeling results were used for both the DPEIS and the Application. These results are described in Appendix D to the DPEIS and in the Appendix to the Application, which are identical.

actually occur in a real world environment." *Id*.; *id*. at 1-20 ("This estimate does not reflect an actual expectation that marine mammals will be injured or disturbed. It is an overly conservative estimate."). BOEM has further admitted that using this methodology "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*. (emphasis added).[12]

The Associations appreciate BOEM's candor in describing the substantial shortcomings of the exposure modeling. However, such candor does not excuse BOEM from accurately estimating the number of likely takes and the associated anticipated impacts, as is required by the MMPA's implementing regulations. An estimate that "does not reflect BOEM's determination of the actual expected physical or behavioral impacts to marine mammals" is plainly not a description of the "anticipated" impact or the number of incidental takes that are "likely to occur." 50 C.F.R. § 216.104(6), (7). Chapters 6 and 7 of the Application (and the Appendix) are intentionally designed to be inaccurate by evaluating the worst possible consequences that could hypothetically result from unmitigated seismic surveying, based on overly conservative modeling. By taking this approach, BOEM has skirted the regulatory requirements for MMPA incidental take authorization applications.

2. The modeling relied upon by BOEM is biased and premised upon unrealistic scenarios that are unsupported by actual data

The exposure modeling set forth in the Appendix makes many biased assumptions that substantially contribute to the inaccuracy of the Application's take and impact analyses. Specifically, the modeling analyses in the Appendix contain multiple layers of precaution that aggregate in the annual and 10-year estimates. Attachment A to this letter provides a more detailed assessment of the overly conservative (*i.e.*, unrealistic) assumptions used in the modeling. These assumptions result in an exposures outcome that is anywhere from 10% to multiple orders of magnitude above the mean or most likely exposures outcome (*i.e.*, 100 to 1,000 times the "most likely" number of exposures) for any given single variable. In the

[12] This "worst-case scenario" includes repeated exposures, but does not identify the number of repeated exposures. Instead, the Application simply presents a total number of estimated exposures by species. Application at 97 ("the numbers of exposures in the following tables does not equate to the number of individual animals exposed"). This generalized presentation of exposures is insufficient because the MMPA's "small numbers" standard is based upon the number of marine mammals that are anticipated to be incidentally taken, regardless of how many times each of those marine mammals may be taken. The Application must separately present (1) the total number of anticipated incidental takes, including repeats (for the "negligible impact" assessment) and (2) the number of marine mammals, by species, anticipated to be incidentally taken, regardless of repeats (for the "small numbers" assessment). *See* 16 U.S.C. § 1371(a)(5)(A); 50 C.F.R. § 216.104(6).

aggregate, these compounding conservative assumptions produce a predicted number of exposures across all variables together that is thousands to millions of times greater than the average or most likely outcome.

For example, the Phase II model assumes a seismic source array of 8,000 cubic inches. This is at, or very near, the upper limit of the largest source arrays used in the GOM. *See* Appendix at D-25. The actual distribution of array sizes in the GOM ranges from 8,400 cubic inches to less than 2,000 cubic inches, with a mean value of 5,600 cubic inches. The scaling differences in the range to threshold criteria produced by an overestimated array size of 8,000 cubic inches cascade down through the calculations, so that when a threshold range four times larger than produced by a typical survey source is established using hearing injury thresholds 10 or 100 times lower than actual measured thresholds, and applied to numbers of animals that can be up to 10 times higher than any previous federal estimates (*see infra* § III.C), the outcome is a prediction that 10,000 to 100,000 times more exposures might occur than use of the "best available data" values might otherwise have calculated. *See* Attachment A. Instead of this overly precautionary and unrealistic approach, BOEM could have used the data for all array sizes used in the GOM in the past 10 or 20 years, plotted them on a typical bell-shaped curve, and calculated the mean or median and variance.

Further overestimation is caused by the accumulation of sound without hearing recovery during calculation of both SPL_{rms} and SEL exposure thresholds, for which sound is summed over 24 hours. *See* Appendix, Section 6.5.1.2.2, page D-64. For an intermittent source, such as a seismic survey, there is a considerable interval of 10-20 seconds or longer between individual pulses that are only a fraction of a second in duration. However, the model inappropriately sums multiple exposures that may be many hours apart as if the separate exposures are one continuous block of sound. This is not a biologically realistic assumption—hearing recovery takes place during intervals as short as a few seconds and exposures separated by hours are almost certain to involve full recovery from prior sub-threshold encounters. *See* Finneran (2015).[13] The result of this biologically unrealistic assumption that SEL accumulates without recovery over a 24-hour window is an overestimation of SEL threshold exceedance that may be at least twice the actual value and possibly many times greater. The fact that the exact hearing recovery function has not yet been empirically derived for marine mammals should not be used to ignore this well-known aspect of mammalian hearing that has been repeatedly observed during the temporary threshold shift ("TTS") data collections that form the basis for NOAA's *Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing* (Aug. 2016) (the "Guidance"). *See infra* note 21 (including comments in referenced attachment).

[13] Finneran J. J. 2015. Noise-induced hearing loss in marine mammals: A review of temporary threshold shift studies from 1996 to 2015. J. Acoust. Soc. Am., 138 (3): 1702-26. http://dx.doi.org/10.1121/1.4927418.

Additionally, as Section 6.5.1.3.2 of the Appendix acknowledges, the single-day overestimates are then used in a way that creates additional overestimation during the calculation of takes for a survey period of 30 days or more. Paradoxically, BOEM states on page D-65 of the Appendix that this simple multiplication of 24-hour values should not be done: "It is, therefore, inappropriate to scale the 24 h exceedance times to estimate the exceedance times for longer durations." Nonetheless, this method is used in the Phase II modeling (Appendix at D-180) to produce the final exposure estimates (Appendix, Section 7.3.4).

Next, Section 6.5.2 of the Appendix analyzes potential contributions to uncertainty from the sound source characterization modeling, and from sound speed profiles, geoacoustic parameters, bathymetric data, and sea state inputs to the acoustic propagation modeling. This analysis concludes that the various uncertainties in the acoustic field represent a "multi-dimensional envelope" and that these different dimensions "cannot be summed to yield a 'total' uncertainty as this would be a meaningless quantity." However, this conclusion is incorrect. There are ways to quantify the uncertainty in a meaningful way despite challenges to directly calculating the total uncertainty (or statistical variance). For example, the combined uncertainty contributed by environmental and model parameters could be further evaluated by comparing the outputs from multiple runs of the entire modeling process (both acoustic propagation modeling and exposure modeling) in which one or more of the parameters are adjusted across reasonable levels in each competing model run. The parameter-specific uncertainty analyses presented in Phase I of the Appendix are useful for identifying which parameters to adjust within the competing full modeling runs, but alone they only reinforce the fact that significant uncertainty is present at many steps within the modeling process. Multiple runs of the full modeling process using alternative parameter estimates should be conducted to improve the understanding of the total uncertainty surrounding the final results.

Furthermore, the analyses set forth in Section 6.5.2 of the Appendix use various methods to assess uncertainty around the parameters used in acoustic propagation modeling. However, in all examples, only the "typical" (average or median) and "worst case" values are evaluated. As a result, uncertainties are only characterized in <u>one direction</u> from the typical or expected result, and that direction results in longer-range propagation of sounds. When characterizing uncertainty around estimates, it is common practice to not only report the upper confidence limits ("worst case" results in this example), but to also report the lower confidence limits. Without an understanding of the lower confidence limit values, it is not possible to properly bound and assess the range of outcomes from the modeling and interpret the likelihood of potential impacts. The failure to characterize the lower confidence limits results in a flawed and significantly biased analysis.

In sum, BOEM summarizes the significant biases of the modeling as follows:

> The existing modeling largely does not account for uncertainty in the data inputs and also selects highly conservative data inputs. <u>This bias often produces unrealistically high exposure numbers and "takes" that *exponentially* increase uncertainty throughout each</u>

> step of the modeling. The modeling does not incorporate mitigation or risk reduction measures designed to limit exposure. The modeling is an overestimate and should be viewed with that understanding.

DPEIS at 4-47 (emphases added). As demonstrated above, these biases result in modeled overestimates of exposures that are thousands to millions of times greater than the average or most likely outcome. Again, this approach is contrary to the MMPA regulations, which require BOEM to estimate the number of takes that are "likely to occur" and the "anticipated" impact. 50 C.F.R. § 216.104(6), (7).[14]

3. The Application's take estimates and impact analyses are not based upon all of the best available scientific information

As addressed above, and in Attachment A, Chapters 6 and 7 of the Application are based on overly conservative, unrealistic, and biased modeling of "exposures." Aside from the legal and methodological flaws with this approach, there is a wealth of available information, including new acoustic criteria, as forth in the Guidance, that actually informs the analysis of the reasonably anticipated impacts of geophysical activities. This information, as addressed below, is either minimized or not addressed at all in the Application.

a. The history of formal assessments of offshore seismic activities

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] Indeed, more than four decades of worldwide seismic surveying and scientific

[14] The Application also erroneously requests take authorization for all estimated exposures even though, as BOEM acknowledges, not all exposures result in incidental take. Application at 93; *see* 75 Fed. Reg. 49,709, 49,716 (Aug. 13, 2010) ("Although it is possible that marine mammals could react to any sound levels detectable above the ambient noise level within the animals' respective frequency response range, this does not mean that such animals would react in a biologically significant way. According to experts on marine mammal behavior, the degree of reaction which constitutes a take, *i.e.*, a reaction deemed to be biologically significant that could potentially disrupt the migration, breathing, nursing, breeding, feeding, or sheltering, etc., of a marine mammal is complex and context specific, and it depends on several variables in addition to the received level of the sound by the animals."). Again, the numbers of incidental takes that are "likely to occur" are not reported in the Application. Table 7-4 of the Appendix appears to vaguely address the topic of translating exposures into incidental takes, but it is not apparent whether or how this table is considered in the Application.

[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/

(continued . . .)

research indicate that the risk of physical injury to marine life from seismic survey activities is extremely low. *See supra* § II. As BOEM concludes in the DPEIS, "within the GOM, there is a long-standing and well-developed OCS [oil and gas] Program (more than 50 years) and there are

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(. . . continued)
("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, 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 ").

no data to suggest that activities from the previous OCS Program are significantly impacting marine mammal populations." DPEIS at 4-77 (emphasis added).

In addition, the 2016 report from the National Academy of Sciences, Ocean Studies Board (the "NAS Report"),[16] 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.' That statement is still true…." (NAS Report at 16);

- "Evidence of the effects of noise on marine mammal populations is largely circumstantial or conjectural" (NAS Report 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" (NAS Report at 35);

- "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. Madsen et al. (2002) studied responses of sperm whales in Norwegian waters to seismic surveys at ranges > 20 km, and reported no responses at exposure ranging up to 123-130 dBRMS." (NAS Report at 56).

Consistent with the NAS Report's findings, there are well-documented examples of long-term exposures of acoustically sensitive species where 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, and despite annual lethal takes by Alaska Natives engaged in subsistence activities, bowhead whales have consistently increased in abundance to the point that they are believed to have

[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#.

reached carrying capacity. Similarly, no effects of G&G activities have been observed in Arctic ice seal populations.[17]

Finally, BOEM's Environmental Studies Program has spent more than $50 million on protected species and sound-related research over more than four decades without finding evidence of adverse effects. *See* http://www.boem.gov/BOEM-Science-Note-August-2014/ (*Science Notes*, Aug. 22, 2014) ("Since 1998, BOEM has partnered with academia and other experts to invest more than $50 million on protected species and noise-related research."). The geophysical and oil and gas industries, the National Science Foundation, the U.S. Navy, and others have spent a comparable amount of money on researching potential impacts of seismic surveys on marine life and have found no evidence of significant effects. *See* http://www.scandoil.com/moxie_issue-bm2/bm.doc/sogm_1-2-16_sml-jip.pdf; www.soundandmarinelife.org.

None of the information above is meaningfully discussed in the Application. Yet, this information is plainly relevant to the development of an accurate assessment of the "anticipated" impacts of geophysical activities on marine mammals in the GOM. 50 C.F.R. § 216.104(7). This information is also indisputably part of the best available scientific information relevant to the Application.

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[17] *See, e.g.*, 84 Fed. Reg. 25,829, 25,834 (May 1, 2012) ("Bowhead whales have continued to travel to the 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."); *see also* 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.").

b. PSO monitoring data

The Application also fails to present and consider the accumulated observational data collected by Protected Species Observers ("PSOs") on survey vessels in the GOM. This information is clearly relevant to the assessment of the potential effects of seismic vessels operating in the GOM. Not surprisingly, the PSO data indicate a negligible level of effects that undermines the results of the exposure modeling presented in the Appendix. For example, the Application implausibly concludes that many thousands of marine mammals will experience incidental take as a result of seismic activities. These estimates would result in tens of thousands of shutdown events per year. However, based on actual monitoring data, as reported in relatively recent environmental assessments, an average of only 55 shutdowns occur per year in the GOM with operations conducted under the Standard Mitigation Measures. *See also* Attachment B; Barkaszi et al. (2012) (reporting a total of 144 shutdowns from 2002 to 2008, or 24 per year).[18] The PSO data must be fully disclosed and evaluated in the Application because they are relevant to an accurate estimate of the incidental takes that are "likely to occur" and the "anticipated" impact. 50 C.F.R. § 216.104(7).[19] These data are also part of the best available science.

c. The take estimates and impact analyses are not based on the best available acoustic criteria

The Guidance establishes acoustic criteria for evaluating Level A harassment and TTS. Despite the availability of drafts of the Guidance and the scientific basis for the Guidance for many months prior to August 2016, the Application's exposure modeling analysis does not use the Guidance:

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[18] A study of more than a decade's worth of marine mammal observation data performed by the Joint Nature Conservation Committee ("JNCC") demonstrates that mitigation measures significantly reduce the effects of seismic activities on marine mammals. The JNCC study's results should be addressed in the Application. *See* http://jncc.defra.gov.uk/page-6985.

[19] Under the MMPA, Level A harassment is defined as "any act of pursuit, torment, or annoyance which . . . has the potential to injure a marine mammal or marine mammal stock in the wild." 16 U.S.C. § 1362(18)(A)(i) (emphasis added); *see also* 50 C.F.R. § 216.3. As described above, there is no scientific evidence demonstrating that G&G activities have resulted in the injury of marine mammals. Rather, the record shows that commonly employed avoidance and mitigation measures are effective in avoiding Level A harassment and minimizing the amount of Level B harassment. For this additional reason, the Associations are opposed to the modeled Level A exposures presented in the Application. At the very most, a *de minimus* amount of Level A incidental takes could be requested based on an approach that calculates a rate of reported shutdowns during seismic surveys in the GOM over the past several years and applies that rate to the levels of activity projected in the Application, using a multiplier to address the potential unmitigated exposures that may occur.

> The NMFS has advised BOEM that the use of the previous acoustic criteria to model exposure estimates is acceptable given the timing of the petition being complete and the issuance of the revised acoustic guidelines. BOEM does anticipate, however, that the July 2016 changes to NMFS' acoustic criteria likely mean the Level A exposures predicted in the modeling used for the [DPEIS and the Application] are, in most cases, overestimates.

Application at 94-95. The Application does present estimates using metrics similar to those set forth in the Guidance, but the amount of Level A incidental take for which the Application requests authorization is inexplicably based upon the outdated 1995 criteria. *See* Application, Table 6-14. Similarly, the Application presents Level B incidental take estimates generated from both the outdated 1995 criteria and newer criteria based upon Wood et al. (2012). However, again, the amount of Level B incidental take for which authorization is requested is inexplicably based upon the 1995 criteria. *Id*.

Additionally, the analytical methods and criteria that are used in the acoustic analyses supporting the Appendix modeling are less than straightforward. For example, the Appendix uses the outdated 1995 criteria, but applies Southall et al. (2007) M-1 weighting to those values, which were originally unweighted values. The Appendix modeling also uses Southall et al. (2007) SPL peak Permanent Threshold Shift ("PTS") onset values, but for low-frequency cetaceans creates its own PTS onset threshold of 192 dB re 1 μPa^2 s SEL by subtracting 6 dB from the mid-frequency cetacean onset value of 198 dB re 1 μPa^2 s (another precaution layered on top of already precautionary numbers). Appendix at D-55. Another example of unclear development of a threshold value appears in the very next paragraph where the analysis cites a value of 187 dB SEL as the mid-frequency cetacean threshold, derived by using a beluga TTS onset of 186 dB, applying Finneran and Jenkins (2012) Type II M-weighting to derive a weighted value of 172 dB and then adding 15 dB to produce a PTS threshold for mid-frequency cetaceans of 187 dB. In short, the methods for deriving the criteria used in the analysis are hardly clear.

BOEM is required to use the best available scientific information when preparing the application. *See* 50 C.F.R. §§ 216.102(a), 216.104(c), 216.105(c). It is undisputed that NMFS's 1995 acoustic criteria for Level A and Level B incidental take by harassment are no longer the best available science. For Level A incidental take (and TTS), the best available science is, by NMFS's own assertion, currently the Guidance. For Level B incidental take, the criteria set forth in Wood et al. (2012) is more current than NMFS's 1995 criteria and more consistent with a large number of similar behavioral effects models (*e.g.*, as cited in Southall et al. (2016)[20]).

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[20] Southall, B., Nowacek, D., Miller P., and Tyack, P. 2016. Experimental field studies to measure behavioral responses of cetaceans to sonar. Endangered Species Res. 31:293-315. doi: 10.3354/esr00764.

Accordingly, the Application, and the subsequent rulemaking, must use the more current sources of information that are the "best available."[21]

4. The Application's incidental take estimates and impact analyses improperly ignore mitigation measures

By BOEM's admission, the Application's incidental take estimates and impact analysis do not take into account the beneficial effects of the mitigation measures that will be required of operators who receive authorizations under the contemplated ITRs. *See* Application at 93 ("the model is not able to consider the effect of reduction of exposures from any of the 19 mitigation measures analyzed in the associated [DPEIS]"); *id*. at 129 (the mitigation measures are "meant to decrease and reduce the potential for Level A and Level B exposures[, but] [t]he modeled exposures largely do not take into account the effect these mitigations have in reducing exposures (and therefore potential for take)."[22]

BOEM's decision to ignore the beneficial effects of mitigation measures is particularly arbitrary because BOEM knows—unconditionally—that the mitigation measures will substantially decrease any adverse effects postulated by the overly conservative exposure modeling. *See, e.g.*, Application at 83, 129. In addition, the Appendix demonstrates the likely effectiveness of currently employed mitigation measures. Specifically, in Phase I of the exposure modeling described in the Appendix where various modeling methods, inputs, and assumptions are assessed, Sections 6.5.3 and 6.5.4 consider the effects of incorporating mitigation measures and aversive responses into the exposure modeling. Tables 40 and 44 show that the implementation of shutdowns may reduce the number of estimated Level A exposures by 10% to 80%.[23] Similarly, the effect of modeling aversive responses by marine mammals also

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[21] As the Associations addressed in three comment letters submitted during the process for developing the Guidance, there are technical flaws in the Guidance. We have attached those three comment letters to this letter, and request that they be included in the administrative record for the contemplated ITRs. *See* Attachment C. There are also flaws with Wood et al. (2012), but that paper is more current than the 1995 criteria.

[22] *See also* DPEIS at 1-16 ("The modeling is conservative because it did not apply any of the 19 different mitigations analyzed in [the DPEIS]."); *id.* at 1-19 ("The modeling effort in Appendix D does not, for example, take into account any mitigation measures incorporated into the alternatives because the effect of those measures cannot be quantified with statistical confidence at this time."); *id.* at 4-14 (mitigation measures not considered as part of effects analysis).

[23] The effectiveness of mitigation varies by species as it is related to the probability of detecting each species; however, those species that form large groups and/or are most abundant are the ones for which mitigation is most effective. Thus, the percent reduction in estimated exposures is likely greatest for the species with the highest absolute estimated exposures.

shows potentially large reductions in the percentages of animals exposed above Level A criteria (40% to 85% for the peak SPL criteria and 14% to 20% for the rms SPL).

Despite these demonstrations of significant and meaningful reductions in the number of estimated exposures as a result of mitigation measures and aversive responses, and the fact that both are very likely to occur, they are inexplicably not included in the final (Phase II) modeling used to estimate exposures for the impact assessments and ultimately not considered as part of the effects analysis. Although there are uncertainties associated with including these measures in the modeling process, those uncertainties are not substantially different than uncertainties associated with other inputs to the modeling process, and they should not be disqualified from use for that reason.

BOEM's failure to incorporate the known benefits of mitigation measures, many of which are standard best practices that the geophysical industry already implements, results in take estimates that, by BOEM's admission, are not "likely to occur" and an assessment of impacts that are not "anticipated." *See, e.g.*, DPEIS at 1-16 ("This estimate 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."). BOEM's approach is arbitrary, unsupported, and contrary to the MMPA. *See* 50 C.F.R. § 216.104(6), (7).

5. Conclusions—Chapters 6 and 7

As set forth above, the estimates, analyses, and conclusions presented in Chapters 6 and 7 are unrealistic, flawed, incomplete, and unlawful. The conclusions are exclusively based upon a modeling exercise that uses a multiplicative series of conservatively biased assumptions for all uncertain parameter inputs. These assumptions lead to accumulating bias as the cumulative conservative assumptions add up to increasingly unlikely statistical probabilities that are not remotely representative of real-world conditions. Consequently, the results quickly become little more than improbable worst case scenarios—not fair simulations or representations of likely effects.

Aside from being scientifically and legally indefensible, BOEM's conclusions are not supported by the best available information, which demonstrates that no significant impacts to marine mammal populations from seismic activities have occurred in the GOM. Furthermore, the scenario presented in the Application is unrealistic and not representative of real-world activities as there is no meaningful consideration of mitigation measures and their effectiveness. Insofar as we are aware, no seismic activities in the United States OCS have caused impacts amounting to anything more than temporary changes in behavior, without any known injury, mortality, or other biologically significant consequence to any marine mammal species or stocks.[24] For the reasons detailed above, Chapters 6 and 7 of the Application must be substantially revised and resubmitted, on the schedule set forth in the Settlement Agreement and the Stipulation to Amend, to comply with applicable MMPA regulations.[25]

[24] 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).

[25] Additional technical comments are provided in Attachment D to this letter.

ATTACHMENT A

BOEM

BAS

BOEM'S PRECAUTIONARY MODEL VS BEST AVAILABLE SCIENCE (BAS) MODEL

1

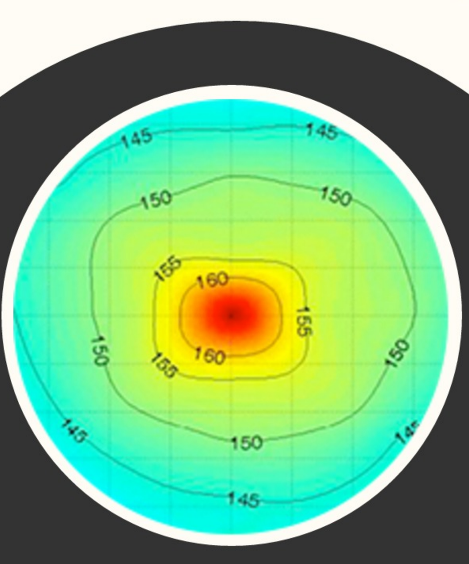
Sound Source

According to the Precautionary Model, the sound source is 1.5 - 2 times bigger than in reality.

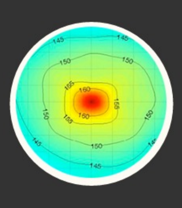
Actual size of a seismic sound source.

2

SOUND FIELD



Precautionary Method Sound Field 10-60 Times Bigger

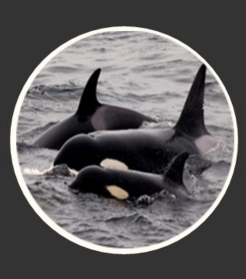


Best Available Science Actual Sound Field

3

AFFECTED ANIMAL POPULATION

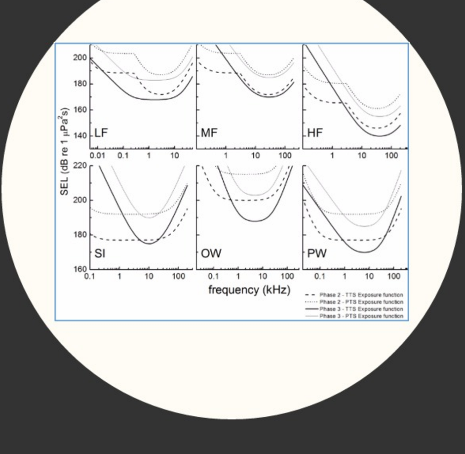
The Precautionary Model Shows 2-80 Times as Many Affected Animals



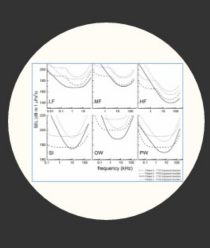
BAS Population Estimate for Affected Animals

4

ACOUSTIC THRESHOLDS



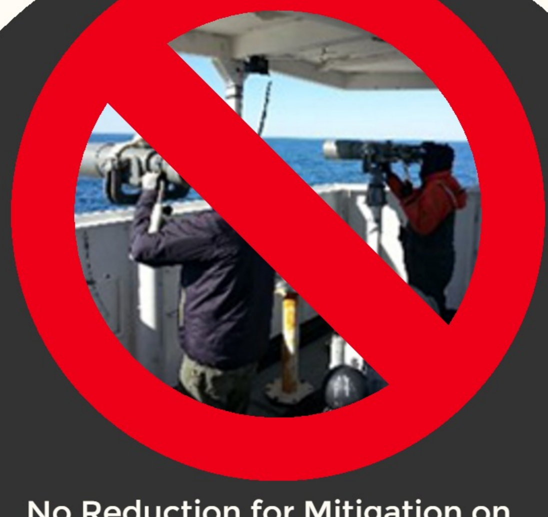
Precautionary Acoustic Threshold are 10-30 dB Aboe Acoustic Threshold Established by BAS



BAS Acoustic Threshold

5

MITIGATION



No Reduction for Mitigation on Precautionary Method

Some Reduction for Mitigation on BAS

Results

FINAL RESULTS OF PRECAUTIONARY METHOD VS. BAS

The results of over-statements in the precautionary method equation results in millions of estimated takes.

The results of a realistic equation based on Best Available Science shows a much more realistic number of takes.

BOEM has analyzed a strawman through the DPEIS by relying on unrealistic assumptions as opposed to the best available science as U.S. law requires. The above graphic shows how precautionary assumptions stacked upon each other lead to a multiplicative over estimation of impacts that will never occur.

Total Multiplicative Precautions (short list)

[Source+Propagation (90-120x)] x [abundance (2x)] x [conservative threshold criteria (100-10,000x)]x [no recovery factor (10-100x)] x [no allowance for aversion (6.7 x Level A)] x [no mitigation (1.1 – 2x)] = 1.3 million to 3.2 billion more takes than the number that would be produced by using average or most likely values for all variables.

SYNOPSIS OF PRECAUTIONARY ASSUMPTIONS

GULF OF MEXICO DPEIS

Bob Gisiner, IAGC

BACKGROUND

The BOEM Gulf of Mexico DPEIS is structurally very similar to most recent NEPA analyses for environmental risk from manmade sound in the marine environment. The interaction of the source, the propagation of the sound from source to animals, and the resulting sound exposures interact to produce a calculated estimate of effect, usually stated as MMPA Level A and Level B "takes", since the MMPA requires that the impact of an activity be quantified in those terms (NEPA and ESA do not have such strictly numerical requirements for estimating impact).

Historically and in this EIS, each element of the model is assessed relative to the available information and a value is selected that is considered sufficiently conservative or precautionary, given uncertainties about the scientific data or about natural variability in factors such as animal distribution, location and movement of the sound source or the sound propagating properties of the water column. Selection of conservative values in multiple steps of the model leads to an outcome that is not an average of the precautionary assumptions, or even an addition of uncertainty, but multiplication of each uncertainty by the uncertainty in the other steps. Simply put, doubling the expected value for four different parts of the model does not double the outcome, nor does it result in a 2+2+2+2 = 8-fold increase in the predicted outcome. Instead the effect of multiple precautions is multiplicative, and the outcome is 2x2x2x2 = 16-fold more than if the model was run with 'most likely' values like averages. Doubling all values out of precaution therefore does not predict an outcome of 200 takes when 100 was the most likely expected outcome, but instead produces an outcome of 1,600 takes.

As we will see from the following quick-look at the GOM DPEIS, there are many more variables in the model than the simple four variable example described above. And the levels of precaution are not simple doubling of expected values, but multiples that may range from addition of some percentage (less than doubling) to increases that are orders of magnitude greater than the "most reasonable" value (orders of magnitude are multiples of ten, such as 10, 100, 1000, etc.). The downstream consequences are also more complicated than the simple two times two example above, with some variables interacting in other than simple multiplicative ways.

For example, use of an 8000 cubic inch sound source rather than the mean or median of sizes actually used (5,600-5,100 cubic inches) would appear to only create a difference of about 30-37%, but that difference in size produces a difference in source sound level of 3-6 decibels, depending also on the number of elements in the source array. The difference in source level needs to get translated into a difference in the area covered by the sound from the two different sources, because that will change how many animals are within the two respective areas, all other factors being equal. The 33-37% difference in the size of the two arrays translates into an increase of some 45-50% (roughly) in the area exposed and therefore the number of animals taken. That is, if one uses an 8000 cubic inch array as the precautionary standard and that results in a take estimate of 150 individuals, then use of the more likely mean value of 5,600 cubic inches will result in a take of 100 individuals. Needless to say, this is a pretty large downstream consequence from alteration of a single value by what might superficially look like a pretty small amount. As we will see, factoring in the other parts of the model where similar conservative assumptions are exercised results in a prediction of takes that is millions, possibly billions, of times greater than the outcome predicted by using most likely outcomes only.

[for ease of locating information, references to the DPEIS are to the .pdf file page number, not the page numbers on the document itself]

SUMMARY OF PRECAUTIONARY ASSUMPTIONS IN THE BOEM DPEIS

This list includes only the most obvious and clearly unsupported precautionary assumptions of the model:

- Source
  - Extreme array size and number of elements increases exposures by 1.5 to 2 times.
  - Six additional precautionary assumptions were not analyzed.
- Propagation
  - Conservative or simplifying assumptions about the propagating environment add 10-16 dB minimum to the propagated sound.
  - Combined with the precautionary source assumptions, this results in a 90-120 time increase in estimated takes, all other variables being equal.
  - Six additional precautionary assumptions were not analyzed.
- Animal Abundance, Density and Movements
  - NMFS's Stock Assessment Reports ("SARs") and Duke Model differ on average by a factor of 2. A minimum compromise for uncertainty would be to reduce abundance and density estimates by 25% to 1.5 times SAR.
  - Three specific groups showed even more extreme differences, but were not separated in this simple analysis: expansion of Bryde's whale habitat leading to more takes; large increases in numbers of deep divers (beaked whales, sperm whales, Kogia); extremely large increases in pelagic dolphin numbers (over 80 times for two species)
  - Five additional precautionary assumptions were not analyzed.
- Threshold Criteria
  - Level A calculations from SPLrms and SEL used precautionary assumptions that overestimated take by 10-1,000 times. SPLpeak takes were overestimated at least twofold by using 6 dB instead of 15 dB to derive PTS from TTS.

  - Level B calculations make generous assumptions about the likelihood of response and assume all exposures that exceed threshold are biologically significant, over-estimated biological consequence by at least 1,000 to more than 100,000 times.
  - No allowance for reduced Level A due to behavioral avoidance of the source (reductions of Level A up to 85%).
  - No allowance for hearing recovery between pulses (likely reduction of cumulative SEL from a continuous pulse train of 50% or more); no allowance for hearing recovery between passes separated by hours or days (fewer than 1% of successive passes, those within 8 hours or less, will accumulate and trigger Level A criteria).
  - Four additional contributors to precautionary over-estimation were not analyzed, including application of weighting functions to impulse SPL metrics.
- Mitigation
  - No reduction in take was allocated for mitigation. While setting a specific value for mitigation may be difficult, it clearly is not zero and therefore some reduction of takes due to mitigation should be factored into the model.
  - Reductions from multiple proposed mitigations were not estimated.
    - Vessel separation and dolphin shutdowns modeled, with questionable effectiveness
    - Increased time/area closures and 10-25% effort reductions were not estimated.
- Total Multiplicative Precautions (short list)
  - [Source+Propagation (90-120x)] x [abundance (2x)] x [conservative threshold criteria (100-10,000x)]x [no recovery factor (10-100x)] x [no allowance for aversion (6.7 x Level A)] x [no mitigation (1.1 – 2x)] =
  - **1.3 million to 3.2 billion** more takes than the number that would be produced by using average or most likely values for all variables.

RECOMMENDATION

Re-calculate takes using average or most-likely values, quantify and report the overall level of uncertainty in the modeling results, and add an agreeable level of precaution to the final results, not the individual elements.

- Maybe double is reasonable?
- A statistical measure of extreme confidence like 3 sigma still covers 99.7% of all possible outcomes (370 times the central value) and is not nearly so unreasonable as the present model
- It seems unlikely that 1 million to 3 billion times the most likely outcome, which covers 99.9999% or more of all possible outcomes, is a reasonable level of 'precaution'.

PRECAUTIONARY ASSUMPTIONS

The Sound Source.

As discussed above, BOEM treats all geophysical surveys as if they were all conducted with the largest arrays in use. The nominal value of 8000 cubic inches is an approximation of the maximum array size currently used in the Gulf, typically 7900 to 8500 cubic inches. Based on a quick survey of IAGC members over the past decade, a little less than one third of all surveys use arrays of that size. The other two-thirds of surveys in the GOM use arrays that range in size from 6000-2000 cubic inches, for a mean array size of 5600 cubic inches. Since the different sizes are not distributed normally around that mean value (i.e. not a smooth bell shaped distribution), some other value of central tendency, like the median (5100 cubic inches) might be deemed a more appropriate central value. But in any case, using 8000 cubic inch sources for all modeled surveys greatly overestimates actual use.

The source level of a compressed air array increases as the cube root of its volume, all else being equal, so a difference of 8000 and 5600 cubic inches might seem trivial. But we have seen that it is not trivial in terms of the outcome of concern; the number of animals exposed, because of the resulting expansion of the acoustic 'footprint' of the array and the number of animals likely to be found within that footprint.

Furthermore, the modeled array is not only extreme in the total volume modeled, but also in the number of elements within the array. A typical large array of 8000 cubic inches might include 48 elements and sometimes as many as 60, but the BOEM DPEIS used 72 elements. Why is this important? Because array source level may only increase trivially with total volume, but it is directly proportional to the number of elements. An array with 72 elements has double the amplitude of an array of 36 elements; volume and air pressure being equal.

Therefore the combination of using an array at the extreme upper end of normally used array sizes, coupled with a number of elements in that array which also greatly exceeds the average, can by itself produce estimates of takes that are 1.5 to over 2 times as large as would be predicted by using the normal range of array sizes and numbers of elements actually in use. Based on this variable alone one would be justified in taking the final model predictions and halving them. But there are many more conservative assumptions in the model.

Also potentially capable of altering the model outcome, but not addressed in this quick analysis, are:

- The number of source vessels. When multiple source vessels are used they are used at intervals that are similar to a single source. The total acoustic energy is therefore not increased over using a single source operated at the same inter-pulse intervals, but the total area ensonified is slightly increased, depending on the spatial separation of the vessels. This may be compensated by the fact that each vessel is only producing sound every 60 seconds instead of every 15 seconds for a single source vessel). In the BOEM DPEIS, the maximum number of source vessels, four, is used for all surveys that might use multiple sources, even though many of those surveys, such as NAZ, WAZ and coil surveys, might more often use only one or two sources, and rarely use as many as four source vessels.
- Longitudinal tracks were only used during modeling on the slope region of the Gulf, which has the potential to alter sound fields and estimated takes relative to using both lateral and longitudinal tracks typical of most surveys.
- The choice of depth at which the array was towed was set at 8 meters, but other tow depths are common (6 meters is considered the default 'standard') and the choice of tow depth affects the frequency structure and propagation of the resulting sound field.
- The choice of pulse intervals typically varies from 10 to 20 seconds, with the DPEIS selection of 15 seconds being fairly typical. A four source survey would result in each source operating at 60 second intervals.

- Durations of surveys were not clear. On page 3-23 a nominal survey duration of 10.5 months was applied to all surveys, but elsewhere in the document, e.g. D-177, the survey durations varied.
- Survey areas, line separations, and other parameters on page D-177 appear to be in the same conservative direction as the array size and element count; suggesting that line spacing and area covered by a modeled 2D, 3D, WAZ or other survey may be greater than average and thus produce elevated sound exposures and take estimates.

Sound Propagation.

BOEM is to be commended for having run some preliminary models (Phase I modeling in Appendix D) to quantify some of the consequences of using simplifying or conservative assumptions (e.g. see pages D-100; D-106; D-113; D-122). Therefore we can assign some quantities to what is otherwise a very complicated variable, the day-to-day fluctuations in wind, temperature, currents, and other factors that affect sound propagation through the water between the sound source and the animals of concern.

The modeling of sources of variance yielded a 10 decibel difference in sound transmission between an average sound speed profile in the water and the extreme case used in the model (10 decibels is an order of magnitude or ten times the average). Use of hard or median properties for the seafloor added another 4 dB over the most likely outcome, with most of the Gulf being covered with soft sediment that is a poor reflector of sound). Use of a flat sea surface instead of a rough sea surface adds another 2 dB minimum, resulting in a conservative value of over-estimated propagation of 16 decibels or 60 times (!) the amount of energy propagated than would be expected on average. Add this to the conservatism we saw for the source itself, and we already have an ensonified area and number of animals ensonified that would be 90 to 120 times the reasonably expected exposures. A "best reasonable estimate" of 100 would become an estimate of 9,000 to 12,000 from these two precautionary measures alone.

Also potentially capable of altering the model outcome, but not addressed in this quick analysis, are:

- A single uniform propagation regime is used for the entire deepwater zone (Zone 7). Assumptions of flat bottom and maximum depth are not met in all cases and propagation is therefore subject to additional over-estimation factors in the deep water region.
- Survey days and survey effort appear to have been evenly distributed across the area and seasons, although this is likely not the case for actual survey effort. Theoretically this might average out, but it is also possible that fewer actual survey days in winter, when propagation conditions are best, will lead to actual surveys producing fewer takes than the model estimated by using equal division across winter and summer.
- SPLrms for longer range propagation is derived from the SEL values produced by the model. As JASCO acknowledges (D-49), modeled SEL at range tends to over-predict SPLrms as the signal is spread over time. Time resolution of the model also hinders accurate modeling of SPLrms based on proper analytic units such as rms.90 (average sound pressure over the time than encompasses 90% of the total pulse energy).
- Single frequency long range propagation modeling leads to increased errors in pulse properties with range. For modeling purposes a single frequency at the center of each 1/3 octave band is treated as 'representative' of all the sound energy within that frequency band. In practice, selection of a non-representative frequency (e.g. located at a ghost notch or filtered by

propagating environment) can lead to errors in weighted SEL values needed for determining effects thresholds.

- Use of "maximum over depth" in some model estimates of take creates a worst-case scenario where all individuals are assumed to be at the depth of highest sound exposure all the time. It is not clear in what context JASCO used maximum over depth as a simplifying step in modeling, but it will always greatly over-estimate takes when used.(D-296)
- Ranges to effect for mitigation monitoring and shutdown (but not for take estimation?) were calculated from unweighted values, whereas hearing frequency weighting needs to be applied to SEL threshold values (JASCO also seems to have applied weighting to SPLrms data, which may also be inappropriate – see section on Threshold Criteria, below).

Animal Abundance, Density and Movements.

This is a complex set of variables, with precautionary assumptions literally varying for each of the species modeled. But overall, the use of the Duke model creates an increase in predicted abundance that is about double the official NMFS abundance numbers in the SARs. Some additional modifications in the use of those data by JASCO add to the conservatism (over-prediction) by a fractional amount, in most cases.

The Duke model is a novel approach to forecasting animal distribution and density from historical correlations with readily available environmental data, typically not the true environmental predictors like prey patches or features like fronts, currents and eddies that are less easy to predict or track. As such, there are some things that the Duke model likely does better than the SARs, such as predicting average abundance of pelagic dolphins that move in and out of the US EEZ from one survey to the next, leading to large sampling variability. However, other similar models for the US west coast, for the UK, and for global oceans, have shown some extreme misses in their predictions, an expected outcome for models in the early stages of development for species that are infrequently counted and whose habits are still poorly understood relative to land animals for example. Too great dependence on a single very new model like the Duke model can therefore be expected to result in some improvements on the SAR or US Navy NODES data resources, but is also likely to produce some extreme "misses". Species with wide disparities between historical data and Duke model predictions include Atlantic spotted dolphins (from no historic estimates in SAR, to over 45,000 animals predicted by the Duke model, making them the third most abundant species in the Gulf, virtually overnight. Duke predictions of Clymene dolphin abundance are about 85 times higher than the SAR figures, Kogia numbers are increased by a factor of 12, rough-toothed dolphins by a factor of 8 and killer whales by a factor of more than 7. These are radical changes to our understanding of marine mammal abundance in the Gulf that require more than blind acceptance of a new model simply because it is generally "better" than the SARs (D-65).

Some of the animal abundance and distribution modeling may be unfamiliar and counter-intuitive to the average reader. The model in the BOEM DPEIS uses electronic representations of individual animals, or 'animats', to construct time series of exposure for a realistic number of animals, 'behaving' in realistic ways, so that the animats move about and dive at realistic speeds and distances relative to the sound source, which is also moving. As might be expected, capturing the complexities of animal behavior and all of the other variability of the sound source and the propagating ocean is impossible, so certain statistical techniques are used to smooth out some of the variability in outcome that can occur just from sampling errors alone. These techniques, such as over-populating the sound field with hundreds or thousands of times more animats than animals (and then reducing the result proportionally to the actual population) do not affect the outcome but do reduce the likelihood of random extreme variation in outcomes. Monte Carlo methods, or running the same simulation over and over hundreds or thousands of times also helps smooth out the distribution of outcomes. Because the animats are seeded randomly for each model run and because they run independently according to user-specified rules, no single model run will produce the same result (as in real life) and so the model must be run many, many times in order to arrive at a statistical average. This process, which is widely accepted as statistically legitimate and even necessary to producing realistic model outcomes, should not be confused with the selection of variables to put into the animat models and Monte Carlo simulations: those variables, like the source and propagating environment variables, can and do produce biases in the outcome, as will be discussed in detail below.

Animal survey data for the Gulf of Mexico is sparse overall, and therefore statistically weak. Various techniques have been applied to the data to generate estimates of population abundance, density and distribution. The official NMFS Stock Assessment Reports (SAR) are an official estimate by NMFS of the best estimate of population abundance in a region, but they do not offer information about animal distribution, forcing the user to either evenly distribute the animals even across the habitat, even though it is known the animals do not use all of the habitat equally. Alternatively, the modeler can generate 'expert' assumptions about how the animals use the habitat, but those assumptions can create unrealistic estimates of take if the assumptions are not good. For example, JASCO placed all sperm whale animats in water depths greater than 1000 meters because sperm whales are deep divers that tend to occupy deep water. However, a look at the data show that many, if not most, sightings of sperm whales occur in water depths of 400-800 meters, and this is largely confirmed by tagged whale data from the BOEM SWSS research project.

Alternative to applying a population estimate for the entire Gulf evenly or selectively across the Gulf is to use habitat features correlated with animal sightings to predict where animals are most likely to be seen based on 'suitability' of habitat. The statistical aspect of this process is quite well worked out as in the Duke University model applied in the BOEM DPEIS, but there are still 'human-in-the-loop' decisions that can affect model outcome. Something like the Duke model is therefore a "work in progress" in which model predictions may be more or less accurate, depending on the habitat variables available to the modeler and whether they are in fact strongly predictive of where animals will in fact be. A few "warning flags" about the novel predictions by the Duke model are:

- The distribution of Bryde's whales across the entire GOM shelf edge by the inclusion of "unidentified baleen whale" data as Bryde's whale data. Actual observations suggest that the Bryde's whales are confined to a relatively small area of habitat around DeSoto Canyon in the Eastern Planning Area (EPA), and in fact this site has been selected as a special mitigation zone. But the Duke model "places" Bryde's whales across large swaths of area where they have never been seen, greatly elevating the predicted takes in the WPA and CPA by what are probably orders of magnitude (hundreds or even thousands of modeled takes not supported by the real data).
- Several species for which there are low sighting data produced low likelihoods of occurrence across vast areas of the Gulf in the Duke model, which were further simplified to even probabilities across entire modeling zones: false killer whales, killer whales and several other species are therefore equally likely of being taken wherever surveys occur, when in reality there

are probably higher and lower areas of likelihood. It is hard to predict how the "fuzzy" predictions of the Duke model, and the modifications of the JASCO model affect take outcomes but generally speaking, these species tend to have predicted abundances derived from Duke density models that are among the highest deviations of the Duke model from SARs (e.g. 6 times SAR for killer whale, 14 times SAR for pygmy killer whale).

- Deep divers that are seldom seen during visual surveys were subjected to some assumptions about sightability that greatly elevated predicted abundance and greatly expanded habitat occurrence over the SARs; 12 times the SAR for Kogia and about 8 times the abundance for beaked whales (based on Cuvier's beaked whale modeling). This radical departure from historical estimates of abundance is somewhat consistent with comparisons elsewhere (Atlantic, California, Bahamas, eastern north Atlantic sites), but on the high side. It is also higher than predictions by passive acoustic surveys and modeling by Hildebrand, Moretti, and others. Just how "precautionary" the Duke model is for these species is hard to estimate at this time, but it is fairly clear that the Duke model is over-predicting deep diver abundance and distribution leading to excessive estimates of takes.

Additional aspects of animal distribution and movements information that may lead to over-prediction of takes include:

- Assumptions used to deal with the large number of modeling cells that yield zero abundance and zero takes can lead to over-prediction of takes. JASCO notes that the outcomes that yielded a probability of Level A take greater than one (1) was less than 0.2% (i.e., only 2 out of a thousand model results yielded a take of 1 or more animals)(D-123, D-129). The average number of Level A takes was 0.0195 or about 2 per 100, the result of a very small number of model outcomes that yielded more than one Level A take.
- The 3MB model used to set swimming and dive parameters for the animals rely on limited data, quite often from related species studied at different locations than the Gulf. It is therefore hard to predict whether the overall effect of the values entered into the 3MB model resulted in over-prediction of takes or under-prediction, but the most likely outcome is that the values used were conservative, precautionary values that added to the over-prediction of takes.
- The modelers assumed that the animals did not undergo long-term, large-scale movements. Certainly it is widely assumed that animals do not migrate in and out of the Gulf in great numbers, although sperm whales, a variety of baleen whales, and probably many other species do move between the Gulf and Atlantic or Caribbean. But the currently available data do not offer enough information, especially for winter months, to determine whether other species exhibit moderate north-south or east-west movements with the seasons similar to the inshore-offshore movements of estuarine bottlenose dolphins in the late winter and spring, or during other seasons. It is well known that large numbers of animals may travel from east to west, tracking the warm core rings spun off by the Loop Current, but this phenomenon is not sufficiently documented to inform the model.
- JASCO modeled the effect of group size on outcome. They did not see a significant difference in average outcome from using single, ungrouped animats, although they did note that obtaining the same outcome regardless of group size means that there will be more zero-take model runs as group size increases (D-135; D-174).

- As animats move over time, and if animats are removed once they exceed a take threshold, then the probability of take will decline over time as there are fewer and fewer animats in the field. JASCO used a common technique for keeping the number of animats constant and thus keeping probability of take constant over time by introducing new animats on the opposite side from which an animat had just left (D-49; D-82; D201). It is also not clear if and how animals were removed or replaced once taken. This is especially important where animats were left in the field to accumulate SEL for days or weeks. There are other nuance to re-seeding the sound fields that can result in skewed results, but a full treatment is beyond the scope of this short review.

<u>Take (Acoustic Risk) Thresholds</u>.

Both Level A and Level B thresholds range from more than 100 times higher than best scientific evidence to over 100,000 times higher. There are multiple conservative assumptions that produce this extraordinary outcome: the assumption that exposure equals take, the conservative linkage of permanent hearing decrements to temporary hearing decrements, assumptions about the accumulation of hearing effects over time without recovery between exposures, and assumptions about how many of these exposures actually have any meaningful biological consequences.

The MMPA defines “harassment” with reference to two categories: Level A harassment (potential to “injure”) and Level B harassment (potential to “disturb”). NMFS applies acoustic thresholds to estimate the amount of harassment for each category that may result from an activity. The acoustic thresholds are often mistakenly assumed to mean that an injury or mortality will occur, with 100% of the exposed animals being injured or killed, or that 100% of exposures at behavioral thresholds will cause behavioral change and that the consequences of the change are a significant and meaningful loss of food, energy, or some other key biological function. In fact, both thresholds imply a probability of there being an effect upon exposure. BOEM was quite emphatic in stating that exposure does not equal take, but the model still treats any exposure that exceeds threshold as a take. This is the first of many features within the Acoustic Risk Threshold part of the model that lead to large over-estimates of take.

Additionally, the DPEIS is not always clear when and how animals are removed from the model to prevent multiple takes of the same individual (e.g., being counted as a Level B take and then exceeding Level A criteria and also being counted as a Level A take). Removals need to be handled carefully to prevent gradual reductions of model ‘animats’ in the sound field as “taken” animats are removed.

The most recent threshold criteria for Level A takes are based on empirical data for the threshold at which a temporary decrease in hearing sensitivity (TTS) occurs across a narrow frequency range of hearing (NMFS, 2016; Finneran, 2015). BOEM also variously cites NMFS 1995; Southall et al 2007; Finneran and Jenkins, 2012: it is not yet clear which criteria they plan to use in the Final EIS, making analysis of the DPEIS difficult. JASCO in Appendix D modeled the 1995 threshold

The simplest Level A threshold, long since superseded by scientific data but still in use by NMFS, is 180 dB SPLrms (root mean squared – an average over some specified time period, and since it is an average of a logarithmic scale, dB, a square root of the mean of summed square values is required rather than a simple average). Despite being outdated by more than 20 years, BOEM still modeled takes using this hyper-precautionary threshold. This provides a threshold that is some 10 to 1,000 times more precautionary than the current best data derived from TTS thresholds for both impulse and tonal sources; the peak SPL or the summed sound energy over time (SEL), although we shall see later in this section that the SEL has also been subjected to additional conservative assumptions that render it some 10-1,000 times more conservative than SPLpeak. The values of 10 to 1000 times are based on SPLpeak thresholds of 230-200 dB SPLpeak, and an estimate of 180 dB SPL rms being comparable to 190 dB SPL peak (200 dB is ten times 190 dB and 2230 dB is one thousand times 190 dB on the same scale, in this case SPLpeak).

Permanent Threshold Shift (PTS) is not tested directly, and is assumed to occur at a level above TTS consistent with marine mammal TTS data and human/lab animal data. PTS, as for TTS, is not a threshold for deafness or major loss of hearing, but for a small decrement of hearing sensitivity within a narrow frequency range, a 'hearing notch'. This is a liberal interpretation of "injury", since the original sense of the term in MMPA was intended for animals that lost eyes, limbs, or suffered broken bones and spinal injuries during interactions with fisheries or due to being struck by ships, shot at, or otherwise seriously injured.

The criterion is rendered even more conservative by the use of a 15 decibel difference between TTS and PTS when the data from other species, including humans, indicates PTS onset at 20-40 dB above TTS threshold. Since even this conservative addition of only 15 dB to TTS produces thresholds of PTS above the source level of the sound source, Southall et al (2007) and subsequent criteria (NMFS 2016) have arbitrarily set the SPL peak metric for PTS at a mere 6 dB above TTS threshold, or almost ten times lower (and therefore productive of ten times as many exposures and takes).

The best predictor of TTS and therefore PTS, at least for tonal sounds, is SEL, a product of both signal intensity (not amplitude) and duration. It is not clear how well this relationship holds up for an impulse signal like compressed air (CA) sources, so relationships for tonal signals are applied to impulse thresholds. SEL is referenced to a time duration, typically one second, but for sounds less than 1 second long, like impulse sounds, SEL does not always hold up.

Furthermore, models like the BOEM DPEIS treat multiple exposures separated by many seconds or even hours or days, as if the sound exposure had been continuous. Near the source a geophysical survey produced 0.1 s of sound every 10-20 seconds, expressed as a "duty cycle" of approximately 1-2%. Further from the source the energy in the impulse may spread in time, increasing the duty cycle, but at ranges meaningful for Level A determination, the duty cycle remains below 10%, meaning that 90% of the time the ear is capable of recovering from some of the induced fatigue or threshold shift. Early TTS studies noted that the animals recovered from low levels of TTS within seconds or minutes, and subsequent ongoing studies are consistent, suggesting that it make take considerably more intermittent exposures to produce TTS or PTS than would be predicted by simply adding up multiple pulses as if they all occurred in succession without any time for recovery (In other words 12 pulses of 0.1 second duration each are treated as a continuous 1.2 second pulse and not what they are, which 1.2 seconds of sound within ten 15 second intervals or 150 seconds of ambient sound only).

The case for some sort of recovery function is even stronger for intermittent passes of an array that may be separated by 4, 8, 16 or more hours, in which case hearing is likely fully recovered and no accumulation of SEL should be carried forward. NMFS has traditionally carried SEL forward for 24 hours, a scientifically unwarranted precaution that leads to over-estimations of take by another 10-100 times, if not more. The current modeling exercise suggests in places that SEL accumulation was carried forward even further for weeks or even months. Appendix K offers annual summations of SEL and a similar cumulative sound metric, Leq, for an entire year. This is not scientifically justified and leads to overestimates of takes by tens or even hundreds of thousands of takes, both Level A and Level B.

Because we do not have a specific recovery function to offer yet, BOEM has not included ANY recovery in their model, whereas a model consistent with best available science should include at the very least a recovery function consistent with human and other mammalian hearing. Absence of a recovery function is likely adding another 10 to 100 fold over-estimation to Level A take.

Thresholds for Level B take have been difficult to derive, although more and more publications have offered data and a proposed threshold function: most of these papers are not cited or reviewed in the EIS, or in the reference used by the Phase II model (Appendix D), which is an unpublished contract report to a California utility company (Wood et al 2012). Wood et al (2012) also presents a potential conflict of interest, since the author of Appendix H (Brandon Southall) is also a co-author of the Wood et al (2012) report. The industry is sponsoring a review of the behavioral effects literature, but that review will not be published in time to inform the current PEIS.

In any case, the Wood et al recommendation was a step function of increasing behavioral response at increasing exposure levels, and in this respect Wood et al (2012) is similar to other Level B risk assessments like the US Navy Programmatic EISs (2009; 2014, draft 2017). All recognize that out of a given group of animals, a few will respond at low levels, with increasing recruitment up to an exposure level that approaches thresholds for TTS and PTS. BOEM also applied the outdated NMFS 1995 Level B threshold of 160 dB SPLrms.

The outcome of applying any of these thresholds is the generation of tens of thousands to millions of Level B takes in which the vast majority of “takes” are transitory disturbances that last hours or a day or two and have no impact at all on foraging success, breeding success, growth, health or any other biologically meaningful metric. The hypothetical possibility that cessation of feeding for a day or movement a few miles from the source, or a change in vocal behavior “might” lead to biologically meaningful consequences means that the model calculations are treated as “takes” under MMPA even though all acknowledge that exposures don’t equal takes and takes do not equal meaningful effects. The development of the PCOD model, and population of that model with data, confirm that behavioral disturbance from sound needs to be reduced to a “biologically significant” number that is a fraction of the counted exposures; anywhere from a conservative 1% to a more realistic 0.001% or less. In other words, estimates of thousand to millions of takes in the model are like to result in fewer than 1 to 1000 takes with actual biological consequences. These numbers, spread across large areas like the Gulf and multiple species are mathematically too low to result in a population level consequence from Level B takes (e.g. elevation of baseline mortality, decrease in baseline fecundity). This is consistent with history, where more than five decades of regular geophysical survey effort all over the globe has not generated any evidence that observed behavioral responses to the sound has any biological consequence.

Calculation of grossly inflated Level B take numbers in the GOM DPEIS is not consistent with current best information, and greatly over-estimates the consequences for the stocks of marine mammals being managed.

Finally, behavioral aversion was not applied to this model, even though a preliminary Phase I model showed that even small amounts of aversive greatly affected both Level A and Level B takes. If behavioral aversion is a trigger for Level B take then it cannot subsequently be omitted from modeling of Level A takes, since the low level exposures that trigger aversion will reduce the likelihood of higher levels of exposure.

Additional aspects of threshold assessment that may lead to over-prediction of takes include:

- Conservative thresholds for low frequency whales. Current conservative thresholds for whales increase the estimated Level A and Level B takes for these species by some 4 to 10 times over best available science predictions. Arguments for unreasonable precaution in the face of uncertainty are not consistent with mammalian auditory biology in general.
- JASCO applied novel uses of weighting functions, using outdated M1 weighting functions from Southall et al (2007) on SPL thresholds, where weighting functions should not be applied.
- Kogia are considered to have the same hearing thresholds as porpoises, even though they are unrelated and the evidence for high sensitive is based largely on data about Kogia vocal behavior and some inconsistent evoked potential audiometry.
- Modifications to beaked whale Level B thresholds unique to this EIS are applied without justification other than precaution.

<u>Mitigation.</u>

BOEM allowed no reduction in the estimated take for mitigation. This is a highly over-conservative assumption, justified by the relatively little data available on mitigation effectiveness, together with the likely variability in mitigation effectiveness between mitigation service providers, types of marine species present, monitoring conditions and other variables. Some analysis on page D-151 suggests ranges of observer mitigation effectiveness from near zero to over 70%. One cannot require mitigation and at the same time treat it as if it provides no reduction in takes. BOEM needs to come up with some metric for the benefits from required mitigation.A variety of other possible mitigations have been proposed in the GOM DPEIS, ranging from alternative source technologies and active acoustic mitigation to time/area closures, vessel separation schemes, and reduced quantities of geophysical survey effort of 10-25%. At least two of the suggested mitigation measures, vessel separation (Table ES-1; page 1-10; page 2-10; B-32; page 2-38; and D-162-163) and shutdowns for dolphins approaching vessels or bowriding (p. 2-24) offer the possibility of actually increasing takes through expansion of ensonified areas (vessel separation), or extremely high increases in shutdowns with associated prolongation of survey effort (and sound exposure) to achieve survey completion (an estimated 35-40% increase).