Anthropogenic Sound and Fishes

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Anthropogenic Sound and Fishes

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16. Abstract

This report considers the potential effects on fishes of the underwater sounds generated by impact pile driving and, where appropriate, other underwater sound sources. Using a comprehensive review of the scientific literature since 2005, the report considers what is now known about underwater sound and the effects of anthropogenic sound on behavior, physiology, and anatomy. The report then evaluates current (2008) guidelines in the light of new data, and proposes the incorporation of newer (2014) guidelines in future regulatory decisions. The report concludes with a discussion of major data gaps that need to be filled to further improve criteria and recommends research programs that will help fill these gaps.

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FOREWORD

by Pooled Fund Team

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This following report[1] has been prepared with pooled funding provided by the California Department of Transportation (Caltrans), the Washington Department of Transportation (WSDOT), and the Oregon Department of Transportation (ODOT). Additionally, the Federal Highway Administration (FHWA) participated in the Scope of Work development and technical review. Representatives from each agency are on the Pooled Fund Team that developed the scope of work for this report and have provided review and oversight of the report. The report was prepared by Dr. Arthur N. Popper, Dr. Anthony D. Hawkins, and Dr. Michele Halvorsen under a contract between ICF and WSDOT. The discussion in this foreword provides the regulatory context that drives the need for this report.

In October 2000, fish injury and mortality occurred while large 96-inch piles were driven for the Caltrans San Francisco-Oakland Bay Bridge seismic safety project (referenced as Caltrans 2001 in report). This resulted in the awareness of potential hydroacoustic impacts on fishes and other aquatic species as a result of pile driving activities, the eventual formation of the Fisheries Hydroacoustic Working Group (FHWG), and the development of the interim peak and cumulative injury thresholds for fish.

Nearly every estuary and major stream in California, Oregon, and Washington provides habitat for one or more Federal Endangered Species Act (FESA) listed fish species administered by the National Oceanic and Atmospheric Administration Fisheries and the U.S. Fish and Wildlife Service (Services), species managed under the Essential Fish Habitat (EFH) provisions of the Magnuson-Stevens Act, or listed fish species administered by the California Department of Fish and Wildlife (CDFW) under the California Endangered Species Act (CESA). This triggers multiple FESA Section 7 consultations, CESA take authorizations and full mitigation for unavoidable impacts, and EFH consultations annually in West Coast states. Most projects that involve impact piling driving result in adverse effects and thus require formal consultation.

To support consultation, action agencies including Caltrans and FHWA must prepare a biological assessment that determines whether a proposed major construction activity under the authority of a Federal action agency (or State NEPA-assigned agency such as

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[1] The authors of the report (Drs. Popper, Hawkins, and Halvorsen, as well as the staff of ICF) did not participate in writing or reviewing this foreword. Therefore, the opinions and ideas expressed in this foreword are those of the members of the Pooled Fund Team alone.

Caltrans) is likely to adversely affect listed species, proposed species, or designated critical habitat, thereby resulting in incidental take.

When adverse effects are likely to occur, the Federal Service(s) respond with a biological opinion that documents the opinion of the Service(s) as to whether a Federal action is likely to jeopardize the continued existence of listed species or result in the destruction or adverse modification of critical habitat. The biological opinion also provides the action agency with an exemption for incidental take. For CESA, CDFW responds with either a Consistency Determination (CD 2080.1[2]) based on the biological opinion, or, if CDFW does not agree with all measures in the biological opinion, CDFW requires an Incidental Take Permit (ITP 2081[b][3]), which generally requires further measures to *fully* mitigate all impacts that cannot be completely avoided or minimized through reasonable and feasible methodologies.

Take is defined by the FESA as "harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect any threatened or endangered species." For the purposes of FESA, the Services define *harm* as "any act which actually kills or injures fish or wildlife, and emphasizes that such acts may include significant habitat modification or degradation that significantly impairs essential behavioral patterns of fish or wildlife."[4] *Take* is defined by CESA as "harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct.*"*

USFWS defines *harass* as "an intentional or negligent act or omission which creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering." NOAA Fisheries applies a definition of "intentional or negligent action that has the potential to injure an animal or disrupt its normal behaviors to a point where such behaviors are abandoned or significantly altered." What is critical to understand is that these resource agencies consider harm and harassment at the level of an individual; therefore, it only takes the potential for one protected individual to be affected in this way to reach the threshold of harm or harassment.

Caltrans, WSDOT, ODOT, and FHWA recognize that the analysis of the effects of sound pressure produced during pile driving on fish is not an exact science; it requires best professional judgment based on scientific research and experience. The regulatory context that State Departments of Transportation (DOTs) operate in is extremely important because the Services apply their regulatory definitions of harm and harassment when determining whether incidental take will result from impact pile driving. In the absence of solid and specific supporting science, the Services will err in favor of the species, which may result in the incorporation of excessively conservative assumptions by all parties. This allows the actions agencies to estimate their effects without the fear of project delays related to project pile numbers, strikes, or sound pressure levels exceeding estimates in the biological assessment or take authorization. Estimates prepared by State DOTs form the basis of terms and conditions (limits and constraints) that will be placed on the project by resource agencies.

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[2] https://www.wildlife.ca.gov/Conservation/CESA/Consistency-Determinations

[3] https://www.wildlife.ca.gov/Conservation/CESA/Incidental-Take-Permits

[4] https://www.biologicaldiversity.org/campaigns/esa/esatext.html

Threshold exceedances during construction translate into project delays and cost increases, and can contribute to strained relationships with agency partners and contractors.

Permit conditions related to pile driving can include a wide variety of requirements, such as daily and seasonal timing restrictions, peak and cumulative sound limitations, requirements for underwater sound attenuation systems, fish salvage or exclusion, hydroacoustic monitoring, fish monitoring, and special studies, and mitigation plans for the take of state-listed species. There are substantial costs and time delays associated with implementation of these requirements which are triggered by injury and behavioral criteria that were developed more than 10 years ago and based on a limited amount of qualified data.

The objective of this report is to obtain the best scientific data available to determine potential effects on fishes exposed to elevated levels of underwater sound produced by impulsive underwater sources in the aquatic environment, especially impact pile driving. In 2008, the FHWG used a similar literature review and research recommendation, prepared by recognized expert hydroacoustic engineering and fisheries scientists to provide science-based recommendations that helped establish interim threshold levels for underwater sound generated by impact pile driving related to physical injury and temporary threshold shift (TTS). The intent of this new effort is to evaluate research and literature that have been published since that time in order for DOTs and our partners to consider new science and data related to the current thresholds. Enough subsequent research has been conducted to justify an updated evaluation. The effectiveness of the current thresholds for protection of fish species is also considered.

This report includes a review and evaluation of literature and research on the effects of underwater sound on fishes, focusing on the effects of impulsive sound produced by impact pile driving. The primary intent of this review is to serve as a “white paper” that is a succinct source of information. This report includes the evaluation of existing literature (2005 to present) and a summary of knowledge of the injurious and sub-injurious response of fishes exposed to elevated levels of underwater sound produced during impulsive sound events such as impact pile driving and blasting. This knowledge summary will be used to evaluate the effectiveness of the 2008 Interim Criteria for fishes.

Overall the project objectives are to:

A. If possible, and based on currently available research, provide a firm scientific, technical basis on which to improve the current understanding of the sound levels that are commonly associated with the very earliest onset of physical injury to fishes (e.g., scale loss) from the exposure to impulsive underwater sound.

B. Provide a firm scientific, technical basis on which to improve the identification of the earliest onset of TTS to fish from exposure to impulsive underwater sound.

C. Evaluate effectiveness of the existing 2008 interim thresholds for fish protection based on recent research.

- What are the physical and sub-injurious effects on fish commonly associated with sound levels in excess of 206 decibel (dB) peak?

- What are the sub-injurious effects on fish commonly associated with cumulative sound exposure level values in excess of 183/187 dB?

D. Identify knowledge gaps specific to the onset of physical injury to fish and TTS such that specific research can be proposed and carried out.

E. Provide a summary of other lessons learned.

Over the past 15 years, criteria and analysis methods and tools have been developed to address hydroacoustic impacts from pile driving in the permitting process. Caltrans, WSDOT, ODOT, and FHWA realize that the interim injury and behavioral thresholds and NOAA Fisheries calculator have flaws and limitations and/or are over-simplified, but we rely on these tools to expedite project delivery, provide some level of predictability, and avoid significant delays.

We also recognize that particle motion may be the next important factor to consider for understanding our project effects on aquatic organisms. However, until particle motion is more widely understood and processes to meaningfully and expeditiously address it are standardized, which may take a significant amount of time, infrastructures agencies and our resource partners will be applying current underwater sound pressures data similar to that found in Caltrans' Compendium and other tools to predict underwater sound pressure levels to best determine when methodologies should be employed to avoid and minimize impacts on species.

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Anthropogenic Sound and Fishes

EXECUTIVE SUMMARY

Arthur N. Popper, Anthony D. Hawkins, Michele B. Halvorsen

This report considers the potential effects on fishes of the underwater sounds generated by impact pile driving. However, while the report focuses on pile driving, it also considers the effects of other underwater sound sources, where appropriate. The review updates earlier reports, including the one prepared by Hastings and Popper (2005) for the Fisheries Hydroacoustic Working Group (FHWG), which led to the Agreement in Principal for Interim Criteria for Injury to Fish from Pile Driving Activities (Fisheries Hydroacoustic Working Group 2008). These criteria are hereafter referred to as the 2008 Interim Criteria.

More recently, guidelines were developed by an international group of experts under the auspices of the American National Standards Institute (ANSI) and these were extensively peer-reviewed These guidelines are reported in *Sound Exposure Guidelines for Fishes and Sea Turtles* (Popper et al. 2014).[5] These guidelines are hereafter referred to as the 2014 Guidelines.

The 2014 Guidelines provided a critical review of the available data. The document also provided tables, based on the best available science up to 2014, that suggested interim criteria for exposure to pile driving and other anthropogenic sources. The 2014 Guidelines considered whether sufficient data existed to establish relatively simple, broad criteria, based on those sound levels that evoke particular responses. The pile driving criteria in that report were based on an extensive set of peer-reviewed studies that were funded, in part, by the FHWG and the Transportation Research Board.

This report uses new and relevant data from 2005 to mid-2018 on the effects of sounds on fishes. The documents reviewed include peer reviewed scientific papers, reports on the internet, and reports from other sources. The documents emphasize application of the best available science.

The effects on fishes that are considered include:

- Changes to hearing abilities and the ability to detect sounds, including temporary threshold shift (TTS);
- Masking of biologically important sounds;
- Behavioral responses;
- Physical injuries, including those that lead to death; and
- Physiological changes.

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[5] Disclosure: Drs. Popper, Hawkins, and Halvorsen were co-authors of the 2014 Guidelines. The development of the 2014 Guidelines was initiated by funding from the National Oceanic and Atmospheric Administration and later supported by the Bureau of Ocean Energy Management, the National Science Foundation, and other organizations.

To effectively evaluate the literature, it is important that readers first understand a number of critical issues on fish hearing, behavior, and acoustic communication, and on the potential impacts of anthropogenic sounds on fishes (Chapter 2). These include:

- The importance of particle motion rather than sound pressure to many fishes;
- The difficulties in extrapolating between species;
- The difficulties in carrying out behavioral studies on captive fishes;
- The difficulties in interpreting behavioral responses;
- Distinguishing between effects on individuals and impacts on populations; and
- The acoustic problems encountered when conducting experiments in tanks and enclosures.

Furthermore, recent advances in our understanding of the interaction of fishes with the acoustic environment make it important to consider relevant aspects of underwater acoustics. To meet this need, Chapters 3 and 4 define the terms that are used to describe sounds, the metrics employed, and methods for measuring sounds and modeling the propagation of sounds. Chapter 3 focuses on general principles of underwater sound, while Chapter 4 provides a background on particle motion. Together, the chapters describe the sounds that are generated by pile drivers within the water, and also within the substrate. The interface waves that travel along the surface of the substrate are also taken into account.

Studies of the effects of seismic surveys, conducted using airguns, have some relevance to the effects of pile driving. Airguns do not generate sounds that are as high in source level, and the pulse repetition rate is much slower, but their levels do match those at some distance from a pile driver. Studies related to airguns are an important point of reference, because the sounds are impulsive and potentially relevant to pile driving.

Effects on Hearing Abilities

Chapter 5 considers the importance of sound in the lives of fish. Fishes may use sound to communicate with one another, detect prey and predators, navigate through the aquatic environment, and select appropriate habitats. Sounds are also used by some fishes during spawning. The localization of fish that are ready to spawn by other fish may be especially important. It is evident that anthropogenic sounds have the potential to prevent fish calls being detected and may also affect the discrimination of such sounds. Fishes also obtain biologically important information about their environment by examining the acoustic scene that surrounds them.

Sound is detected by fishes using an inner ear that is sensitive to the particle motion components of the sound field. Sound pressure is only detected in fishes that have a gas-filled organ, such as the swim bladder. Being able to detect the pressure component of the sound field expands the frequency range detected and increases sensitivity to underwater sounds. Although most fishes can detect sounds from less than 50 Hz (hertz = cycles/second) to more than 500 Hz, some species, like the Atlantic herring, can detect sounds to more than 1,000 Hz and some fishes even to 3,000 Hz. Thus, there is substantial variability in hearing range and sensitivity among fishes, the most important difference being between those species that detect sound pressure, and those that only detect particle motion.

In considering the effects of underwater sounds, it is useful to divide fishes into different groups, depending on their anatomy, and ability to detect sounds. The 2014 Guidelines define the following groups:

- Fishes lacking swim bladders that are sensitive only to sound particle motion and show sensitivity to only a narrow band of frequencies, less than approximately 300 Hz (e.g., flatfishes, sharks, skates and rays, and some mackerel species).

- Fishes with a swim bladder where that organ does not appear to play a role in hearing. These fishes show sensitivity to only a narrow band of frequencies (e.g., salmon, and some tuna species).

- Fishes with swim bladders that are close, but not intimately connected, to the ear. These fishes show a more extended frequency range, extending up to about 500 Hz (e.g., codfishes, eels, and some drums and croakers).

- Fishes that have special structures physically linking the swim bladder to the ear. These fishes have a wider frequency range, extending in some cases to several kilohertz (kHz) (e.g., some drums and croakers, herrings, and the large group of otophysan fishes including the goldfish, *Carassius auratus*).

Fishes from within these different groups not only vary in their hearing abilities, but also in their susceptibility to hearing loss, physical injury, and physiological damage from exposure to sound. They may also vary in their behavioral responses to sound, although behavioral responses are often influenced by other factors.

Hearing loss may result from exposure to intense sounds. The loss may be permanent or temporary. Permanent threshold shift (PTS) is a loss of hearing that never recovers. In contrast, TTS is a relatively short-lived reduction in hearing sensitivity due to changes in the sensory cells of the ear, generally resulting from exposure to intense sounds for short periods of time, or somewhat longer exposures to lower sound levels. Termination of exposure eventually leads to the return of normal hearing ability. There is evidence that TTS in fishes only occurs when the potentially damaging sound is at a certain (as yet undefined) level above the auditory threshold at the frequency of exposure. However, most TTS data for fishes have been obtained with sounds that are of longer duration than those produced by pile driving.

Determining sound exposure criteria for effects on hearing is very difficult. There are many reasons for this, particularly because the levels of hearing loss will vary by numerous factors, perhaps most important of which is the hearing sensitivity of fish, as well as the characteristics of the anthropogenic sound such as duration, intensity, rise time, spectrum. Similarly, for TTS to occur, the level of the anthropogenic sound must be substantially above the hearing threshold of a fish for some extended period of time. Thus, fishes that hear well (none of which are listed under the Federal Endangered Species Act) have the potential to develop TTS at a lower sound levels than fishes that do not hear well.

The 2014 Guidelines provide conservative levels of sound that might result in TTS in several species and suggest that change in hearing sensitivity should be a minimum of 6 decibels (dB) to be considered TTS. However, even the levels for onset of TTS in the 2014 Guidelines must be taken as extremely tentative because they are based on data from only three species, while TTS could not be induced in other species even with very high sound levels.

Masking of Biologically Important Sounds

Ambient sound, also often called ambient or background noise, includes sounds from both natural and anthropogenic sources. Its presence can affect the ability of fishes to detect biologically relevant sounds (including important parts of the acoustic scene) (Chapter 5), a process called masking.

Fishes are adapted to detect biologically important signals in the presence of natural ambient sounds. However, at some frequencies, the lowest sound levels detectable by fishes are limited by the levels of ambient noise. Any increase in the level of ambient noise (generally resulting from anthropogenic sounds such as those produced by pile driving) has the potential to result in a decline in hearing sensitivity (i.e., poorer hearing) as a consequence of masking. Thus, there is considerable concern about how much the presence of anthropogenic noise may mask sounds of importance to fishes. In addition to affecting the detection of biologically relevant signals, there is evidence that increases in the levels of both natural and anthropogenic noise will have an effect on the detection distances for such sounds. Any anthropogenic sounds that interfere with the ability of fish to detect and analyze key signals have potential consequences for the fitness and survival of individuals, populations, and species.

The 2014 Guidelines define masking in fishes to be the impairment of the ability to detect sounds by greater than 6 dB and lasting more than 30 seconds. This level was also chosen because it was considered unlikely that a change of less than 6 dB, or a brief episode of masking, would result in a significant effect[6]. Species that hear well are more likely to be masked by lower anthropogenic sound levels than fishes that have poorer hearing. However, it is difficult to understand whether intermittent sounds, such as pile driving, will have any real effects on masking because the signals are very short, followed by a longer period (one second or more) of silence. Most experiments on masking have been carried out using continuous sounds. It may not be possible to develop criteria for masking at this time beyond the initial suggestion in the 2014 Guidelines.

Behavioral Responses

Although physiological and physical effects may occur relatively close to a source, behavioral responses may occur wherever the fishes can detect the sound (Chapter 6, also see the Appendix). Different behavioral responses may occur, depending on the level of the sound, the level of ambient sound, what the fish are doing at the time of the sound, and their previous exposure to the same and other sounds. Whether or not a fish responds may also depend on its condition, motivational state, and the presence of other animals including predators.

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[6] Six dB was also used in the NMFS (2018) report on TTS for marine mammals, because it is the minimum that can easily be determined using behavioral hearing methodology.

A better understanding of the sensitivity of fishes to sound requires studies that examine variation in levels of behavioral response in parallel with detailed characterization of the sound fields, ideally using a variety of different sound measurement metrics to ascertain which aspects of the sounds are most important. There have been recent studies of the effects of pile driving and other intermittent sound stimuli upon the behavior of fishes. Many of these studies have been carried out using captive fish, often under imperfect acoustic conditions. However, it is possible to draw some general conclusions, especially from those studies carried out on wild fishes in their natural environment.

It is generally assumed that fishes with better hearing abilities are more likely to respond to sounds than less sensitive species. However, a number of studies have suggested that this may not always be the case. For example, a study on sprat and mackerel showed that the sound pressure levels to which fish schools responded, estimated from dose response curves, were remarkably similar for both species, despite major differences in their hearing abilities.

Fish exposed to pile driving sounds may show alarm responses. They may increase their swimming speeds (often showing a directional response), change their ventilation and heart rates, and show startle responses. Such transient escape reflexes (startle responses) are unlikely to result in adverse impacts because the fish may rapidly return to their normal behavior. However, stronger more sustained behavioral responses may generate oxygen debt and place an energetic load on the fish. Playback of pile driving sounds has been shown to cause both the break-up of fish schools (often termed *shoals*), and the consolidation of schools (which may have adverse effects through a reduction in oxygen levels or the accumulation of waste material). Disrupting the collective dynamics of fish schools could have significant implications for some species. Anthropogenic sounds may also interfere with foraging behavior, either by masking the relevant sounds or by resembling the sounds that the prey may generate. Moreover, elevated noise levels have the potential to affect anti-predator behavior of different species in different ways. The presence of noise may interfere with the detection, location, and identification of predator sounds by fishes, perhaps rendering them more likely to be captured.

The National Oceanic and Atmospheric Administration Fisheries (NOAA Fisheries) currently identifies an SPL_{rms} of 150 dB re 1 µPa[7] as the sound pressure level that may result in onset of behavioral effects (Caltrans 2015). There are substantial issues with this criterion. Its origin is unknown (Hastings 2008), and the scientific basis for it has not been documented. Perhaps most importantly, a single criterion value for behavior does not take into consideration the very substantial species differences in hearing sensitivity and behavior, nor does it take into consideration response changes with animal age, season, or even motivational state.

There have been a few studies of the effects of pile driving and other impulsive sound stimuli upon the behavior of wild fishes in their natural environments that may be relevant to the setting of future criteria. Indeed, it is remarkable how similar the sound pressure levels that evoked behavioral responses were in three of these key experiments dealt with in Chapter 6. Although the sound levels were specified in terms of the peak-to-peak and single strike sound exposure levels, rather than RMS levels, it is notable that the levels were only slightly higher than the SPL_{rms} of

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[7] See Chapter 3 for a discussion of acoustic metrics.

150 dB re 1 µPa level specified by NOAA Fisheries for behavioral effects. The actual sound pressure levels to which the fish schools responded on 50% of presentations, derived from dose response curves, were 163.2 and 163.3 dB re 1 µPa peak-to-peak, and the single strike sound exposure levels were 135.0 and 142.0 dB re 1 µPa^2·s for sprat and mackerel, respectively. Importantly, however, these studies did not evaluate some critical aspects of behavioral responses, including how long animals responded to the sound sources, whether their responses habituated over time, and whether animals would have moved away from the source had the sounds continued. They indicated, however, that further studies on wild fishes in their natural environments may be productive.

Physical Injuries and Death

Exposure to high-intensity pile driving sounds has the potential to damage the organs and tissues of fishes, reducing their fitness, and in some cases perhaps leading eventually to death (Chapter 7). A series of recent studies, all of which have been carefully designed with appropriate controls and statistical power, has investigated the physical effects of pile driving on fishes. The studies have provided greater insight into potential effects from exposure to pile driving. The funding for the work described that was done in the United States was a direct outcome of the 2008 Interim Criteria set forth by the FHWG in 2008.[8]

Data on direct mortality due to exposure to pile driving exist for only a few instances (e.g., Caltrans 2001), with 96-inch shell piles and only when fishes were within 10–12 meters (m) (33–39 feet) of the piles being driven. The majority of earlier studies on potential effects of pile driving, upon which the 2008 Interim Criteria are largely based, appeared in grey literature, and concerns have been expressed regarding the experimental design, including the need for controls for such studies. Despite lack of peer-reviewed data, there has been concern that damage to the tissues and organs of fishes could result in death at some time after the animals have left the pile driving region, as pointed out by Hastings and Popper (2005).

There are two potential physical mechanisms by which fishes could be affected by pile driving sounds. One, about which very little is known, is from direct impact of the particle motion component of the signal on the fish that may result in intense "shaking" of the animal. The second physical mechanism that can cause damage is barotrauma, which is injury to tissues caused by a rapid change in pressure (or pressure difference) across an anatomical structure. As a result of the rapid pressure changes associated with the impulsive pile driving sound, contained gas, such as the swim bladder, may quickly and repeatedly expand and contract. Thus, the walls of the swim bladder (or other gas bodies) may then move with sufficient magnitude and rapidity to cause damage to surrounding organs and tissues as well as to the swim bladder itself. Impulsive signals created by pile driving, and explosions, carry the highest potential for causing barotrauma because of their short rise time and high amplitude.

The series of recent peer-reviewed papers summarized in Chapter 7 provides a quantitative approach to assessing onset of physical damage to fishes. The studies have provided the first dose response curves for injury to fishes, and the first quantified data on potential physical effects and

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[8] Disclosure: This work was done in Dr. Popper's laboratory at the University of Maryland. Dr. Halvorsen was co-PI on the work and Dr. Hawkins was one of the project advisors.

recovery from pile driving sounds for several fish species. They have provided data on the exposure levels that resulted in the onset of barotrauma as well as damage to the sensory cells of the inner ear. Furthermore, it has been shown that the number and severity of injuries increase with higher single strike sound exposure levels (SEL_{ss}). The results also demonstrated the complexity of the effects with regard to the acoustic metrics, including the SEL_{ss}, how sound energy is cumulated, and the number of pulses, and that physical effects will likely never be described by a single metric like the cumulative sound exposure level (SEL_{cum}), or provide a simple formula describing the relationship between the number of injuries and sound parameters. However, the studies provided a good estimate of the sound pressure levels that result in onset of even minor injuries and further demonstrated that onset levels vary by species.

While data are not yet conclusive, it is likely that a major variable for injury onset is the condition of the swim bladder, including such issues as how it is filled and the size of the individual animal. Other morphological swim bladder characteristics that might be relevant include placement of the swim bladder and the rigidity of tissues surrounding the swim bladder. The likelihood of injury seems to be lower in fishes without a swim bladder, which may extend to sharks and rays, even though only teleosts have been tested so far.

These recent studies clearly demonstrate that the onset of physical injury from pile driving signals begins at sound exposure levels that are substantially higher than those listed in the applied 2008 Interim Criteria. Indeed, the studies described in Chapter 7 demonstrate that onset of physical responses occurs at least 16 dB above the levels in the SEL_{cum} 2008 Interim Criteria and is probably over 23 dB higher. Other factors are likely to play a major role in potential physical effects of impulsive sounds. For example, pulse rise time is likely to have an impact on the movement of the swim bladder and, thus, the way that its wall strikes the surrounding tissues. Interpulse interval is potentially of importance because a longer time between pulses may allow for some recovery from tissue damage. Other factors may include the overall duration of exposure, and the frequency spectrum of the source. It is therefore recommended that the levels proposed by the 2014 Guidelines be adopted until more data are available.

It has been suggested that, regardless of how acoustic energy is accumulated (e.g., combination of SEL_{ss} and number of impulses), the impacts on the organism will be the same – "the equal energy hypothesis" (see also Woodbury and Stadler 2008; Stadler and Woodbury 2009). However, recent studies of physical effects discussed in Chapter 7 demonstrate that this equal energy hypothesis does not apply to impulsive pile driving signals, indicating that the SEL_{cum} metric is insufficient as a predictor. It was demonstrated that fewer high-level impulses resulted in more damage than a larger number of lower level impulses, despite the SEL_{cum} being at the same level (Halvorsen et al. 2012a; Casper et al. 2017).

Physiological Changes

Anthropogenic sounds may also have effects on some aspects of the physiology of fishes (Chapter 8) that may have long-term consequences for their fitness. These effects are generally referred to as *stress*, which can be defined as a state of biological strain or tension resulting from adverse circumstances. In such circumstances, animals, including fishes, may show hormonal, autonomic, immune, and heart rate and respiratory responses that may initially allow them to adapt to adverse conditions. However, some stressors may change the state of physiological processes, and affect homeostasis, thus having an adverse effect on the animals' health and well-being.

It is often difficult to distinguish between physiological and physical effects because they may be intertwined. For example, a physical effect on the kidney may result in physiological changes as well, whereas a physiological effect on the kidney may result in physical effects elsewhere in the body. In the literature on effects of noise on aquatic animals, the terms *physical* and *physiological* are often used interchangeably.

There has been only a limited number of studies of the effects of sound on the physiology of fishes, and most have involved exposure to continuous sounds. Studies on captive fishes using relatively short-term continuous white noise or simulated boat sounds have shown an increase in the secretion of cortisol, a stress hormone, as well as other physiological effects. In all cases reported, stress hormone levels returned to normal after cessation of the sound. Moreover, one study showed cortisol changes after short exposure but none after somewhat longer exposure.

There are still so few data that it is not possible to come to any general conclusions regarding real or potential effects of anthropogenic sound of any kind on fish stress levels. Moreover, there are significant issues for all of those studies showing effects, starting with the fact that they were conducted in enclosed areas where the acoustics were not properly calibrated or measured, and where the acoustics had no relationship to the sounds that a fish would normally encounter in the wild.

The effects of sound exposure on physiology, as measured by various stress parameters, are highly variable and are not yet sufficient in scope to provide real information on how exposure to pile driving might impact fishes. In particular, all of the studies, including both long- and short-term exposures, were done on captive animals in enclosed areas where the fish could not avoid the sounds.

One of the most important questions is whether there is a correlation between physical and physiological effects and whether it will be possible to define thresholds for onset of such effects. There are significant issues that have yet to be defined, and for which there are virtually no data. For example, there are few data for fishes that can yet provide guidance as to levels of sound that may result in the onset of physiological effects like stress responses, changes in hormone levels, effects on reproductive physiology, changes in heart rate, and all of the other things that are likely to occur in fishes under the presence of high-intensity sounds. Determination of levels for onset of physiological effects will likely be far more difficult to measure than for physical effects since analysis of physiological effects often require sophisticated tools and methodologies, compared with analysis of physical effects. It may be difficult at this time to decide upon onset levels for sound exposure that would result in physiological effects. However, determining onset for physiological effects may not be needed if more easily observed physical and behavioral effects can be defined more precisely.

Current Status of Interim Criteria

The 2008 Interim Criteria (Chapter 9) that are currently applied on the West Coast come from an agreement that was reached among FHWA, California, Oregon, and Washington Departments of Transportation, NOAA Fisheries, the U.S. Fish and Wildlife Service (USFWS), and the California Department of Fish and Wildlife (CDFW) as part of the FHWG in 2008 (as described in the Foreword to this report). This agreement is documented in the Agreement in Principal for Interim Criteria for Injury to Fish from Pile Driving Activities (Fisheries Hydroacoustic Working Group 2008). At that time, there was very limited scientific research on the topic of pile driving effects on fish. The criteria were intentionally called "interim" because it was understood by all parties that the criteria were based on limited scientific information and would need to be updated as new research emerged.

New peer-reviewed findings that are discussed in this report and new interim guidelines that were presented in an ANSI-approved report (Popper et al. 2014) support updating the 2008 Interim Criteria. Although the SEL_{cum} metric may be correlated with effects, it is important to also specify the SEL_{ss} and the number of impulses, as these can influence the effects. Specifying the SEL_{cum} alone does not take account these other key factors. No definition of how the energy was accumulated was provided in 2008. The scientific basis for the current behavioral criteria is unclear, and these criteria do not take into account the differences in fish behavioral responses. Finally, many of the sound propagation models that are currently utilized do not consider variables such as the sound propagation characteristics in shallow water, substrate variation, and hammer size.

The 2014 Guidelines better reflect the recent data, including much of the material reviewed in this report, and raise the sound level of the criteria for onset of effects by a significant magnitude. In particular, criteria for onset of physical effects from pile driving increase by at least 16 dB relative to the current interim criteria. Additional suggestions for interim guidelines come from reviews in Europe, and, although these reports continue to suggest the need for new data, they tend to adopt the 2014 Guidelines as the basis for regulatory activities until more data are available.

One issue is that agencies currently use onset to be the start of a single injury, while the 2014 Guidelines are not that specific. The data from which the 2014 Guidelines derive show that at levels a few decibels below the criteria for onset of injuries there is no injury. However, these levels vary by species, fish size, and whether there is a swim bladder present. Because there is so much potential variability for the onset of a single injury, it is recommended that each consultation should modify the onset criteria to be utilized, based on the different species groups presented in the 2014 Guidelines. At the same time, it is very clear that any recommendations for onset criteria in individual consultations must be far closer to those recommended in the 2014 Guidelines than to the 2008 Interim Criteria.

Finally, in order to develop future criteria, considerably more data will be needed on a variety of topics, including: onset of injury and the metrics that should be used for onset; onset of physiological effects, as well as determination of what level of such effects has the potential to do lasting harm to fishes: hearing (including masking and TTS); and behavioral effects. The issue, in each case, is the complexity of obtaining appropriate data, and then applying the data to improve acoustic criteria.

Research Needed for the Development of Future Guidelines

A full discussion of the broad range of research gaps on effects of anthropogenic noise on fishes was identified in a conference organized by the Bureau of Ocean Energy Management (BOEM) in March 2012 and published in two publications (see Normandeau 2012; Hawkins et al. 2015). These papers provide a broad overview of the most critical issues that apply to all fishes, including those exposed to pile driving. An important point is that, in most cases, it is going to be imperative to obtain data from multiple species, and a range of sizes and ages of fish within each species. This is because there is likely to be substantial variation in potential effects depending on differences in species anatomy, physiology, and behavioral responses to various stimuli. Effort needs to be made to include samples of species that cover the full diversity of fishes, including key species that are likely to be exposed to pile driving sounds. The 2014 Guidelines have suggested dividing fishes into those morphological groups that relate to the presence/absence and configuration of the swim bladder.

Of the research gaps identified earlier, and from the data reviewed in this report, a number of the most critical research gaps for understanding effects of pile driving on fishes are identified in Chapters 8 and 9. In considering research gaps and recommendations, the most immediate and important areas of research include:

- *Selection of species*, so that those examined represent the broad range of species potentially affected by anthropogenic sounds. This is needed since it will never be possible to get sufficient data on even a small number of the species that are likely affected by such sounds.

- *Behavioral responses* to pile driving sounds. There are numerous behavioral issues that need to be examined, from the sound levels that are likely to elicit behavioral responses (e.g., based on hearing studies, and studies of hearing in the presence of maskers), to actual responses to sound pressure vs. responses to particle motion. Data are needed on general behavioral responses to sounds at different sound levels and how these responses change during the course of a pile driving operation, perhaps as fishes habituate to the sounds and/or temporarily show hearing losses due to the presence of persistent sounds. Moreover, it is important to determine whether the responses of fishes differ when they are at different distances from the source and in the acoustic near field vs. the acoustic far field.

  In addition, long-term, realistic field studies are needed on the effects of pile driving on the behavior of fishes, taking account of cumulative and synergistic effects, along with stress indicators. If we are to better understand the sensitivity of marine fishes to sound, there is a particular need for studies that examine variation in levels of behavioral response in parallel with detailed characterization of the sound fields, ideally using a variety of different sound measurement metrics to ascertain which aspects of the sounds are most important.

- *Development of dose response data* for different sound parameters, such as signal intensity, cumulative effects, and number of strikes. Such data will provide insight not only for understanding the onset of physical effects or behavioral effects, but also for determining those levels above the onset level at which potentially harmful effects start to occur. Such information will enable regulators and others to be able to make better decisions on criteria, particularly if they are willing to accept the idea that a small effect may not have any impact on the fitness of the animal.

- *Hearing*. There is a clear need for more data on the hearing sensitivity of fishes of interest, and for standardization of hearing measurements, in order to correlate anthropogenic sound levels with behavioral responses, masking and TTS. Such studies need to determine hearing thresholds not only under quiet conditions but also in the presence of masking signals, to determine the ability of the fish to discriminate signals of particular interest to them in the presence of anthropogenic noise, including the impulsive sounds generated by pile driving.

- *Modeling of Sound Fields*. Once the effects of sounds on fishes have been defined, it is necessary to estimate the extent of those geographic areas over which those effects might take place. However, making such measurements is often difficult and time-consuming, and, in many cases, modeling of a sound field may be more efficient and effective in providing the information needed for regulatory purposes. Thus, it is necessary to continue to develop models that can be used to predict sound levels around pile driving operations. These models need to take into account the shallow-water environments, including variance in substrates, because they have a significant effect on sound propagation.

- *Particle Motion*. Although it is clear that the use of particle motion for establishing criteria is something that should be done in the future, the lack of data on how particle motion impacts fishes, as well as the lack of easily used methods to measure particle motion, precludes the use of particle motion at this time. To ultimately incorporate particle motion into regulatory activities, two important approaches need to be taken in the coming years:

    1. *Determine the Potential Effects of Particle Motion on Fishes*. Studies are needed to address behavioral responses, physical and physiological responses to particle motion, and hearing of the particle motion component of the sound.

    2. *Develop a Methodology for the Measurement and use of Particle Motion in Regulatory Activities that is approved by ANSI and/or the International Organization for Standardization (ISO)*. Methods need to be developed and incorporated into evaluation of pile driving activities that not only look at sound pressure, but also particle motion. As part of this effort, it is important to develop standards for particle motion sensors and sound pressure sensors. In addition, there need to be specific protocols, to be applied throughout the industry, for making particle motion measurements. In addition, regulators and others need to understand and appreciate that particle motion needs to be taken into consideration when planning and regulating pile driving and other activities likely to generate sounds that can potentially affect aquatic organisms.

Finally, there is also a need to consider impacts on fish populations, in addition to the effects on individuals. Effects are the broad range of potentially measurable changes that may be observed in individuals, groups of animals, or even habitats as a result of sound exposure. Impacts are effects that, with some certainty, rise to the level of deleterious ecological significance. Thus, the effect does not indicate the wider significance, whereas the impact deals with the severity, intensity, or duration of the effect on animal populations and ecological communities. It is important to consider effects on individuals in terms of their significance to populations and to ecosystems. The impacts can then be compared with those resulting from other stressors, including chemical pollution, fishing, pathogens, and climate change. However, effects on individuals can be used to determine initial sound exposure criteria.

Key Conclusions

The main objectives of this project were as follows (*see Foreword*):

- *Objective A. Provide a firm scientific, technical basis on which to improve the current understanding of the sound levels that are commonly associated with the very earliest onset of physical injury to fishes (e.g., scale loss), from the exposure to impulsive underwater sound.*

- *Objective B. Provide a firm scientific, technical basis on which to improve the understanding of the earliest onset of temporary threshold shifts to fish from exposure to impulsive underwater sound.*

- *Objective C. Evaluate effectiveness of the existing 2008 interim thresholds for fish protection based on recent research.*
    - *What are the physical and sub-injurious effects to fish commonly associated with sound levels in excess of 206 dB peak?*
    - *What are the sub-injurious effects to fish commonly associated with cumulative SEL values in excess of 183/187 dB?*

- *Objective D. Identify knowledge gaps specific to the onset of physical injury to fish and TTS such that specific research can be proposed and carried out.*

- *Objective E. Provide a summary of other lessons learned.*

The following are key conclusions of this report in relation to the above objectives:

Objective A

- Post-2008 studies demonstrate that the 2008 Interim Criteria are excessively conservative and do not reflect current knowledge of the levels at which there may be an onset of effects from pile driving.

- The 2014 Guidelines (Popper et al. 2014) present updated interim criteria for pile driving (and other anthropogenic sources) that best reflect the post-2008 studies. Therefore, until additional data gaps (Objective D) are filled, it is recommended that the 2014 Guidelines and criteria be adopted as reflecting the best available science.

- At the same time, it is recognized that the criteria in the 2014 Guidelines are based on multiple injuries, whereas the current FESA and CESA requirements are to determine those sound levels that result in the onset of a single injury. Although the criteria in the 2014 Guidelines do not deal with single injuries, the data leading to the criteria can be "mined" to develop levels of onset that can be applied to individual regulatory situations. This data mining should take into consideration different species groupings, as described in the 2014 Guidelines and outlined in Tables 2 and 3 of this report. At the same time, it is clear that the post-2008 data do strongly support the suggestion that the the threshold sound levels that result in a significant impact should be higher than those specified in the 2008 guidelines. The sound levels that result in the onset of a single effect are well above the 2008 levels, and close to those proposed in 2014. Therefore, until more data are available, criteria for onset that are set during consultations should be close to, if not the same as, those proposed in 2014.

Objective B

- As yet, there are insufficient scientific data on which to base criteria for the earliest onset of temporary threshold shifts (TTS) in fishes that have been exposed to impulsive underwater sounds. However, the 2014 Guidelines suggested that a sound level greater than186 dB SEL_{cum} is likely to result in onset of TTS for a wide range of fish types. At the same time, this level is highly conservative and there are many fish species for which TTS onset would only occur at much higher sound levels. This is particularly the case for fishes that do not hear well, such as salmonids, sturgeons, and other ESA listed species.

Objective C

- The effectiveness of the existing 2008 Interim Criteria has been evaluated and it has been concluded that they are excessively conservative and there is no evidence in the literature that those levels would result in onset of any effects on fishes. The more recent data, summarized in Chapter 7, and the 2014 Guidelines provide levels that are closer to those that may result in onset of effects in some, but not all, species. Current consultations should be based on these levels (e.g., Table 3) adjusted for onset to single animals and for animals with different morphologies (Tables 2 and 3).

Objective D

- It has been possible to identify knowledge gaps specific to the onset of physical injury to fish and TTS such that specific research can be proposed and carried out. These knowledge gaps are listed in Chapter 10.

Objective E.

- A summary of the lessons learned is provided at the start of Chapter 10.

Recommendations

- Although it is recognized that current lack of data on responses to, and effects of, particle motion are not sufficient to develop appropriate criteria, the growing international awareness that fishes do possess particle motion receptors means that particle motion must eventually be taken into account in setting future criteria, once appropriate data are available. It is recommended that research on the effects of exposure to particle motion should be carried out within the next 5 to 10 years, and that efforts be funded that will obtain the necessary data on which to base particle motion criteria.

CHAPTER 1: INTRODUCTION

1. Overview of Issues – Purpose of Report

The purpose of this report is to provide a critical literature review and synthesis on the potential effects of underwater anthropogenic (man-made) sound from impact pile driving on fishes. However, the analysis also includes appropriate recent data from studies of effects of seismic air guns and explosions. In addition, data from other anthropogenic sources, such as dredging and shipping, are considered where they can inform and provide guidance.

The review updates a report by Hastings and Popper (2005) (the report was updated and published as Popper and Hastings 2009) using new and relevant data from 2005 to 2018 that considers the effects of impulsive sounds on fishes. The review focuses on:

- Relevant aspects of underwater acoustics (Chapter 3)
- Particle motion (Chapter 4)
- Sound detection by fishes (Chapter 5)
- Behavioral effects of sound on fishes (Chapter 6)
- Physical Impacts on fish tissues (Chapter 7)
- Physiological impacts on fishes (other than TTS) (Chapter 8)
- TTS and damage to the ear of fishes (Chapter 8)
- Effects on various life stages (where data are available) including adult, juvenile, and egg, and larval fish life stages (various locations in text)
- Status of current criteria and guidelines (Chapter 9)
- Lessons Learned: Information gaps and research priorities (Chapter 10)
- Final Conclusions (Chapter 11)

In doing this review, the authors also provide critical background information and analysis of experimental approaches and methodologies so that readers who are less familiar with the topic and the material will be able to better understand and critically evaluate the literature. This background information is particularly important because, in the view of the authors, much of the current literature has weaknesses that impact its value in developing guidelines and criteria. Thus, our goal is to assist regulators and facilitate awareness of these key issues. Accordingly, some of the topics in this report focus on caveats that provide insight into issues that need to be kept in mind when reviewing the literature.

For example, it is difficult to interpret and use data on what fishes can hear without understanding the methodologies used by many investigators and the acoustics of relevance to fishes. It is also impossible to appreciate the value of behavioral experiments on fish responses to anthropogenic sounds without understanding the limitations of doing studies in confined spaces (e.g., tanks or cages) and/or where the sound stimulus to which the fish is exposed is not fully understood.

2. Approach Used in Producing the Report – The Authors

The report has been prepared by team of experts who have been greatly involved in current research and in developing guidelines for potential effects of sound on fishes in the United States and internationally. They have been involved with much of the more recent research on the effects of anthropogenic sound on fishes. They are also international leaders on the overall issue of effects of aquatic noise on marine animals. Two of the authors, Dr. Arthur N. Popper and Dr. Anthony D. Hawkins, are the co-organizers of four international meetings on this topic (e.g., Hawkins et al. 2008; Popper and Hawkins 2012; Popper and Hawkins 2016), while the third author, Dr. Michele Halvorsen, has been part of the organizing team for several additional international meetings. The three authors, along with an international team of colleagues, developed the ANSI-approved 2014 Guidelines for the effects of noise on fishes and turtles (Popper et al. 2014).

Additional technical support for the report preparation has been provided by ICF acoustical engineer David Buehler. Mr. Buehler worked closely with Drs. Hastings and Popper on the 2005 report. The authors are very grateful to their ICF colleagues for their thoughtful guidance and support over the course of this project, and their many contributions to its fulfillment.

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3. Literature Used

The primary source of information used in the report has been the peer-reviewed scientific literature. Virtually all of the potentially relevant literature since 2005 has been critically reviewed and much of it is cited in this report. In particular, the authors have cited in this report those papers and studies that, in their expert opinion, informed the objectives set forth in the Scope of Work in Task Order 1 identified above.

The authors also examined material that was found in reports, on the internet, and provided from other sources (all of these are often referred to as “gray literature”), although, unlike in 2005, the bulk of the newer literature is in peer-reviewed scholarly journals rather than in reports. It is important to also note that, because the new material in reports and on the internet has not undergone the rigors of scientific peer review, the authors have been highly selective in what they used, with the intent of focusing on the best available science. Thus, the authors used gray literature material when the reports met a standard that included appropriate scientific controls, sensible animal handling, and statistical rigor (also see discussion of this point in Popper and Hastings 2009).

4. History of Report[9]

This report is an update of one prepared by Hastings and Popper (2005) for the FHWG. The FHWG was organized in 2004 to help develop information and research on noise effects criteria for fishes as a result of the start of driving 96-inch steel shell piles for the San Francisco-Oakland Bay Bridge by Caltrans (see Foreword for details). The organizers of the FHWG were staff from Caltrans, WSDOT, ODOT, the Federal Highways Administration (FHWA), NOAA Fisheries, CDFW, and USFWS. The FHWG also includes representatives of NOAA Fisheries (Southwest), NOAA Fisheries (Northwest), USFWS and CDFW, and the U.S. Army Corps of Engineers.

"The goal of the Working Group was to reach agreement on:

1. The nature and extent of knowledge about the current scientific basis for underwater noise effects on fish,
2. Interim guidelines for project assessment, mitigation, and monitoring for effects of pile driving noise on fish species, and;
3. Future scientific research needed to satisfactorily resolve uncertainties regarding hydroacoustic impacts on fish species."

The initial role of the FHWG in 2008 was to agree on both peak and accumulative sound exposure thresholds for fish exposed to impact pile driving operations. The development of those guidelines was reviewed by Stadler and Woodbury (Woodbury and Stadler 2008; Stadler and Woodbury 2009) and discussed in several additional documents (Popper et al. 2006; Carlson et al. 2007). In June 2008, a FHWG meeting was convened in Portland, Oregon with the goal of developing an agreement on interim criteria for injury to fish. The outcome of that meeting was the Agreement in Principal for the 2008 Interim Criteria for Injury to Fish from Pile Driving Activities (Fisheries Hydroacoustic Working Group 2008). These criteria are discussed in Chapter 9 of this report (page 96).

5. Overview of Potential Effects

There is a wide range of potential effects on aquatic animals resulting from exposure to anthropogenic sounds. Table 1 (page 18) describes these effects in broad terms, and potential "zones of influence" for these are illustrated in Figure 1 (page 18). These effects are derived from 2014 Guidelines on effects of sound on fishes and turtles (page 99) (Popper et al. 2014). Animals close to a source may show a range of effects from death to behavioral changes. However, as animals are often farther from a source (such as pile driving), the likelihood of the most severe effects (e.g., death, physiological change) decreases but other potential effects (e.g., TTS, masking, behavioral responses) may continue until the animal is sufficiently far from a source that there are no potential effects. It should be noted that even at distances where the animals no longer respond to the source behaviorally, they may still hear the sound, even though it is not of sufficient level to elicit a response (Dooling and Popper 2016).

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[9] Much of the material in this section on the organization of the FHWG is adapted from http://www.dot.ca.gov/env/bio/hydroacoustics.html. In some cases, direct quotes are used from this web page.

Table 1: Potential effects of anthropogenic sound on animals
(Also see Figure 1, page 18)

Effect	Description
Death	Sound exposure leads to instantaneous or delayed mortality.
Physical injury & physiological changes	Physical changes occur that temporarily or permanently impair the structure and functioning of some parts of the body. Physiological changes take place that indicate increased stress or disorientation, perhaps resulting in reduced fitness over time.
Hearing threshold shift	Loss of hearing, temporarily or permanently, leads to inability to respond to biologically relevant sounds.
Masking	The ability of an animal to detect biologically relevant sounds, including the sounds of predators and prey, calls from other animals of the same species, and acoustic cues used for orientation, habitat selection, and settlement is altered. There may be a reduction in "acoustically active space" as a result of a reduction in detection distance.
Behavioral responses	Induced changes in behavior may have adverse effects upon breeding, feeding, or sheltering. Behavioral changes may be ranked, depending on their assumed severity with respect to changes in fitness. Such changes are likely to vary from species to species, and in terms of context, including the location, time of day, time of year, and the condition of the animal. It can be difficult to infer effects on long-term fitness of changes from behavior observed over a limited time period. Some changes in behavior, such as startle reactions, may only be transient.
No obvious behavioral responses	If the signal level is low enough, animals may show transient or no response, even if they detect the sound. Habituation may take place. However, even if there is no response, there is always the possibility that physical injury and physiological changes may take place without the animal showing overt changes in behavior.

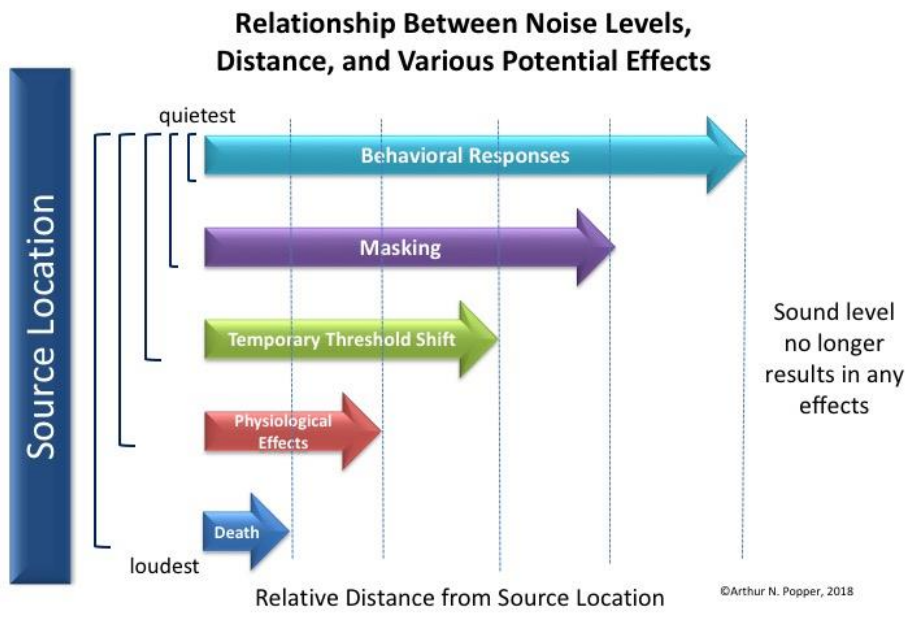

Figure 1: Potential effects on animals at different distances from an anthropogenic source.
All levels and distances are arbitrary. Closest to the source, the potential effects range from death to behavioral responses, but as the animals get farther from the source, and the signal level decreases, the range of potential effects decreases. Figure ©2019 Arthur N. Popper, all rights reserved.

CHAPTER 2: DEFINITION OF TERMS AND CRITICAL POINTS FOR UNDERSTANDING RECENT DATA

This chapter defines terms that are critical to this report. It also presents a number of critical ideas and caveats that need to be understood and considered when evaluating the current literature on effects of anthropogenic sounds on fishes and invertebrates. These ideas are based on data and ideas about fish bioacoustics and effects of sound on animals that have arisen since the original report (Hastings and Popper 2005; Fisheries Hydroacoustics Working Group 2008). Most importantly, many of these ideas influence the interpretation of recent studies.

1. Definition of Terms

It is necessary to first define a number of terms because they are widely found in the literature; however, we have found that different authors provide many of these terms with different meanings or offer no definition. Because the terms are critical for any analysis of potential effects on fishes, these terms do ultimately require an agreed upon set of definitions. We offer definitions here that relate to discussions of criteria and guidelines.

The term *criterion* (plural *criteria*) refers to a numerical principle or standard by which something may be judged or decided. Sound exposure criteria are sound levels, based on acoustic response thresholds, above which sound levels may have adverse effects on specified animals. Some of these criteria are specified for the onset of particular effects, while others are based on dose response relationships and the probability of an effect occurring.

In contrast, the term *guidelines*[10] refers to advice relating to the determination and application of sound exposure criteria. Guidelines provide a critical review of the data available on the effects of sound exposure and consider whether sufficient data exist to establish simplistic, broad criteria, based on specific sound levels that evoke a particular response.

The term *onset* refers to the minimum sound levels above which actual effects on specified animals occur.[11] The response may consist of physical changes (e.g. loss of scales, tissue injury), physiological changes, hearing loss, or behavioral changes. Discussions of criteria often refer to the onset of an effect, whether it be a physical or physiological effect, or a change in behavior (e.g., Woodbury and Stadler 2008) without defining the term. Some authors appear to use it for the start of an effect, even affecting just one animal in a population, while other authors may think of onset in terms of a population effect that is statistically significant. For the purposes of this report, we use *onset* as the start of any effect that is under consideration. It is important that agencies should agree in the future on a definition of the term *onset*. One of the objectives of this

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[10] Note that we differentiate general term guidelines from the very specific Guidelines proposed by Popper et al. 2014 throughout this document. These are referred to in this document as the 2014 Guidelines.

[11] Note, the authors have examined the regulatory and other literature in the United States and in Europe and have spoken with many regulators from various U.S. agencies in an attempt to find a standard, and accepted, definition of "onset." To date we have failed to find such a definition. Indeed, most agency representatives have said that there is no such definition, and have suggested that the term, at least for fishes, needs to be defined. At the same time, the only use of the term in the marine mammal guidelines appears to be a definition of a change in threshold of 6 dB being the onset of TTS for dolphins (NMFS 2018).

review is to improve the current understanding of the sound levels that are commonly associated with the very earliest onset of physical injury to fishes.

At the same time, determination of such a threshold relies on the development of *dose response* relationships. These may involve observations on the changes in effects on the animal caused by differing levels of exposure (or doses) to particular sounds. As the sound level increases, there may be graded or incremental changes in the magnitude of the response. In other cases, there may be a sudden change in the response. In every case it is necessary to seek a particular response level, which may serve as a criterion for defining the acoustic response threshold. Dose response relationships may, in some cases, involve assessment of the number of animals responding in a defined way to a particular sound level.

Under the FESA the term *harm* refers to the killing or injury of a listed species or adverse effects caused by changes to essential behavior such as breeding, feeding, or sheltering. Such an act may also include significant habitat modification or degradation that results in injury of or death to listed species. Harm is one component of the legal definition of *take* under the FESA. There are issues with regard to what actually constitutes an effect that may be significant in terms of its impact on the survival of an animal. The Marine Mammal Protection Act uses *harm* to refer to a wound or other form of physical harm. Signs of injury to a marine mammal include visible blood flow, loss of or damage to an appendage or jaw, inability to use one or more appendages, asymmetry in the shape of the body or body position, noticeable swelling or hemorrhage, laceration, puncture or rupture of eyeball, listless appearance or inability to defend itself, inability to swim or dive upon release from fishing gear, or signs of equilibrium imbalance. However, there are issues with regard to what actually constitutes an effect that may impact survival of an animal.

Although what constitutes harm in the Marine Mammal Protection Act is not applicable to fishes, recent studies on effects on fishes of simulated pile driving sounds provide comparable ratings of injury (Halvorsen et al. 2012c; Halvorsen et al. 2012a), as discussed in detail in Chapter 7. The fish index of trauma (FIT) permits quantification of injuries in fishes. The FIT model applies a mathematical weighting to each injury depending upon its severity, then all weighted injuries are summed to produce a single overall injury value, called the response weighted index (RWI) for each fish. Wherever possible, evidence should be provided that particular levels of injury do impact survival, because fishes can survive some injuries.

It is important to define those sound levels that have significant adverse effects, that is: levels that actually cause harm to individuals. Deciding whether harm has been caused can involve monitoring changes in the response of an individual fish to different sound levels. However, in some cases it may involve monitoring of the proportion of animals that respond in a particular way to exposure to different sound levels. In both cases, we suggest that when there are sufficient data, future criteria should use a statistical approach in defining those sound levels that have significant adverse effects, and that agencies agree on a common definition. For example, “a sound level resulting in significant adverse effects” could refer to when there is a 50% probability of an individual showing a particular adverse effect in response to a specific event, or where 50% of the exposed animals show that effect. We choose 50% because that is a widely accepted threshold for an event, such as for determination of the level of response by an individual animal, or a group of animals, to a sound or other stimulus.

In the ESA text (https://www.fws.gov/endangered/esa-library/pdf/f991108.pdf), NOAA Fisheries interprets the term *harm* as an act that actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation where it actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns, including breeding, spawning, rearing, migrating, feeding or sheltering.

2. Caveats in Data Interpretation

It has become increasingly apparent that there are issues that are critical to consider when reviewing and evaluating data on fish hearing, behavior, and acoustic communication, and on the potential impacts of anthropogenic sounds on fishes. It is imperative that these issues be understood when critically evaluating and using data in the literature. These issues will be mentioned briefly here, and then elaborated upon in other portions of this report.

a. Particle Motion

As discussed in Chapter 4, fishes primarily detect particle motion rather than sound pressure. This affects studies of the hearing abilities of fishes, and how anthropogenic sounds, such as those produced by pile driving, may affect hearing. As argued strongly in several recent and important critical reviews of the literature (Nedelec et al. 2016; Carroll et al. 2017; Hawkins and Popper 2018; Popper and Hawkins 2018), it is very important that particle motion be considered in studies of noise impact on fishes, especially in relation to damage to the hair cells of the inner ear, which essentially respond to particle motion. Although in some instances the particle motion can be estimated from measurements of the sound pressure, under laboratory conditions and in complex field environments it often cannot, and direct measurement is often necessary. It will important in the future to develop protocols for monitoring particle motion, and then to determine those levels of particle motion that have potentially adverse effects on fishes in terms of increased mortality, injury to tissues, changes in physiology, effects on hearing abilities, and changes in behavior. Having such methods and data would then facilitate the development of criteria for exposure to particle motion.

b. Extrapolation between Species

Much of the literature on effects of anthropogenic sound on fishes comes from studies using freshwater species, or on species that are quite different to those species that are ESA listed and of concern on the West Coast of the United States and other areas. Indeed, a major problem, at least in terms of developing guidelines, is that there are more than 33,000 fish species (see www.fishbase.org) that demonstrate great diversity in terms of anatomy, physiology, ecology, and behavior. Thus, the likelihood of a single sound exposure criterion matching the behavior of all species is negligible (Hawkins and Popper 2014; Hawkins et al. 2014; Popper et al. 2014; Hawkins et al. 2015).

A related problem is that many of the behavior experiments have been conducted on species that live well in captivity, often in small tanks, rather than species that are of commercial and ecological importance. Many of these studies involve species such as zebrafish (*Danio rerio*) (Neo et al. 2015; Shafiei Sabet et al. 2015), cichlids (Bruintjes and Radford 2013, 2014), goldfish

(Smith et al. 2006; Smith 2016), or other freshwater species. Such tank-living fishes are used by many labs in attempts to provide the scientific basis for criteria on larger and free-ranging species that are behaviorally and physiologically very different. However, because the data are from "lab species" and the data are from experiments in laboratory settings, the results should be used with utmost caution.

c. Behavioral Studies on Captive and Hatchery Raised Animals

A significant issue in understanding the effects of anthropogenic sound on fishes is that studying their behavioral responses to sounds, especially in the natural environment, can be difficult and costly to perform. Many factors may influence the results, and a careful approach based on well-designed experiments should be adopted.

A number of sound exposure experiments have recently been conducted on fish held in tanks, cages, and large enclosures (e.g., Popper et al. 2007; Sarà et al. 2007; Kastelein et al. 2008; Mueller-Blenkle et al. 2010; Doksaeter et al. 2012; Fewtrell and McCauley 2012; Thomsen et al. 2012; Neo et al. 2016), as have the earlier studies on the effects of pile driving on fishes (e.g., Caltrans 2001; Abbott and Bing-Sawyer 2002; Abbott et al. 2005). Some of these experiments can be valuable in allowing detailed observations of the behavioral responses of fish under controlled conditions. However, it is imperative that observations made on captive fish be supplemented by studies in the wild. Results of studies with free-living fishes likely differ from studies with captive fishes because of the many subtle factors that determine behavior in a natural setting. Moreover, it is well-known that captive animals often do not show the wide range of behavior observed from wild animals (Birkett and Newton-Fisher 2011; Oldfield 2011; Benhaïm et al. 2012), especially if they have been bred in captivity (El Balaa and Blouin-Demers 2011; Petersson et al. 2015).

In addition, fishes reared in captivity may be affected by the circumstances under which they have been maintained. Fish rearing facilities can expose the fish to high noise levels, and fish may become habituated to sound exposure. Fish captured in the wild may be damaged during capture, and the full range of their behavior may not be observed when they are captive. For example, Filiciotto et al. (2013; 2016) demonstrated that offshore aquaculture noise, and in particular the sea soundscape, adversely influenced the oxidative status and the immune function of gilthead sea bream (*Sparus aurata*), creating a mild stress condition. Such conditions may be especially common in captive fish.

Moreover, animals raised in captivity may show atypical physiology, as has been shown by the presence of abnormal otoliths (ear bones involved in hearing – see page 44) within the ears that are associated with abnormal hearing in some hatchery-raised salmonids (Oxman et al. 2007; Reimer et al. 2017). It has also been shown that developmental history, and possibly genetic background, may result in differences in hearing sensitivity in fishes (Popper et al. 2007; Wysocki et al. 2007).

For the purpose of establishing criteria and the associated guidelines, it is clear that behavior should ideally be observed under conditions where the animals are well adapted to a particular location and show normal behavioral patterns (Popper et al. 2014). Although studies in aquaria or in sea cages may provide support for larger scale field programs by providing detailed descriptions of the behavior of individuals (e.g., Sarà et al. 2007), it is unlikely that the animals will exhibit the more complex and context-dependent behavior that they may show when not confined (Birkett and Newton-Fisher 2011; Oldfield 2011).

d. Interpreting Behavioral Changes in Response to Sound Exposure

Work conducted by de Soto (2016) laid out some key concepts to keep in mind when interpreting the results of noise impact studies. The most important point is it should not be automatically assumed that fish will leave a noisy area and thus avoid harmful exposures. Some species are territorial and so may be reluctant to escape. Other fishes cannot move quickly enough to escape the noise. In addition, a typical "fright" response is to freeze, something that has been observed in fish experiencing noise. Animals may respond to noise, as to a predator, by becoming immobile.

Conclusions must not go beyond what the study was designed for and what the results show. For example, if fisheries' catch rates increase after noise exposure, individuals could still have suffered acoustic damage or have been behaviorally impacted by becoming immobile, and thus easier to catch. The point is that noise impacts on catch rates do not allow for wider conclusions about noise impacts on individuals or populations, and a similar approach needs to be taken to the interpretation of other observed impacts.

e. Distinguishing Between Effects and Impacts

The NOAA Roadmap (NOAA 2016) points out that in order to understand how the effects of activities to individual marine animals may or may not impact stocks and populations, it is necessary to understand not only what the likely disturbances are going to be, but how those disturbances or other impacts may affect the reproductive success and survivorship of individuals, and then how those impacts on individuals translate into population changes. It is especially important to examine both the *effects* of sounds and actual *impacts* on populations of fishes (Boehlert and Gill 2010).[12] Effects are the broad range of potentially measurable changes that may be observed in individuals or groups of fish as a result of sound exposure. Such effects are important and can be used to define sound exposure criteria. Impacts are effects that, with some certainty, rise to the level of significant risks in terms of long-term population consequences or changes to ecosystems. However, such impacts can be difficult to determine. Figure 2 (page 24) illustrates the link between effects and impacts.

The "population consequences of acoustic disturbance approach" (National Research Council 2005) recognizes that there may be significant effects from exposure to anthropogenic sounds at individual, population, and ecosystem levels. Similarly, the Productivity Susceptibility Assessment approach (Patrick et al. 2010) has been applied to fish stocks to determine the impact of human activities upon fishes, including fishing. Effects on individuals become significant when they result in altered reproductive success and survival. Changes in behavior that result in alterations in foraging efficiency, changes in energy expenditure, availability of preferred habitat,

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[12] In reviewing the literature, it is clear that the terms "effect" and "impact" are not always used in the same way as used here. Indeed, the two terms are often used interchangeably. This, potentially leads to confusion as to meaning and so we recommend adoption of clear definitions of the terms for all regulatory activities

disruption to migrations, declines in reproductive success, or mortality through predation can be especially significant at the population and ecosystem levels.

The more recent Population Consequences of Disturbance Model (Harwood et al. 2014) allows for the consideration of more data using other disturbance types as surrogates for noise in the case studies. As described in the Population Consequences of Disturbance model, adverse behavioral and physiological changes resulting from disturbance (stimulus or stressor) can either have acute or chronic pathways of affecting vital rates. For example, acute pathways can include changes in behavior or habitat use, or increased stress levels that directly raise the probability of reproductive problems or predation. Chronic effects on vital rates result when behavioral or physiological change has an indirect effect on a vital rate that is mediated through changes in health over a period of time, such as when adverse changes in time/energy budgets affects lipid mass, which then affects vital rates (New et al., 2014).

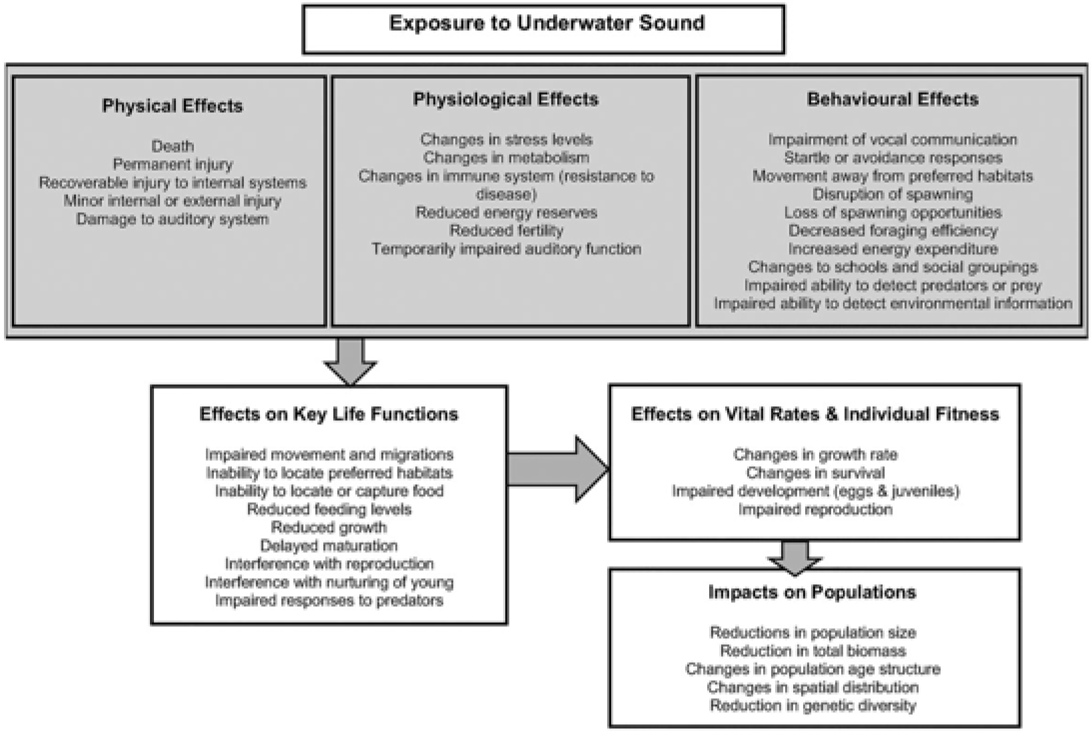

Figure 2: The links between effects and impact.
(from Hawkins and Popper 2016)

An Ecological Risk Assessment has been carried out by Hammar et al. (2014) on a cod population that may have been affected by a wind farm project. They looked at the possible impact of continuous noise from working vessels and operating turbines and exposure to pile driving noise in the construction phase. Their analysis suggests that pile driving was the most hazardous stressor that could pose a serious risk to the cod population. Significant mitigation of this risk could be achieved by avoiding pile driving activity when cod were reproducing.

Purser et al. (2016) demonstrated that the effects of noise varied with the condition of the individual fish. They exposed juvenile European eels (*Anguilla anguilla*) to sound playbacks in a small aquarium tank. The experiments confirmed that short-term exposure to additional noise can increase ventilation rate and reduce anti-predator responses in the fish. Eels in good condition did not respond differently to playbacks of ambient coastal noise and coastal noise with passing ships. However, the additional noise of passing ships caused an increase in ventilation rate and a decrease in startle responses[13] to a looming predatory stimulus in poor condition eels. They concluded that intra-population variation in responses to noise has important implications both for population dynamics and the planning of mitigation measures. Because many fishes, including commercially important species, form huge spawning aggregations, large numbers of individuals in potentially vulnerable condition exist at specific locations at specific times. Noise could, therefore, affect both current and future reproductive success of such populations. Mitigation measures to minimize noise-generating activities during particular periods should consider not just the timing of spawning, but the recovery period for breeding females.

In examining behavioral responses, we will consider those levels of sound that result in the onset of those responses, as well as those that that may affect a larger proportion of the exposed fishes. It is especially important to examine such responses by fishes under appropriate acoustic conditions, and ideally to examine them under field conditions. Actual field data showing the behavioral responses of free-ranging fish combined with adequate measurements of the sound field are required to carry out full impact assessments. The studies of greatest importance are those that investigate long-term changes in behavior and distribution of fish in the wild, including moving from preferred sites for feeding and reproduction, or alteration of migration patterns. Effects on single animals, or small changes in behavior such as a startle response or minor movements, are considered to be less important impacts. This report refers to studies of such effects but attaches much less weight to their findings.

f. Acoustic Problems with Tank Studies

Most studies of fish hearing and fish acoustic behavior have been conducted in tanks that are often small and have thin and non-rigid, walls (see discussion in Rogers et al. 2016). Moreover, the sound sources are sometimes placed in the tank, or perhaps in the air above or below the tank. At times, investigators have lined the inner tank walls with various materials such as foam or "horse hair" with the expectation that these materials cut down reflections and help make a better sound field, but none of these approaches is effective (Rogers et al. 2016).

The critical point is that small tanks, and even many larger tanks or arenas, are acoustically highly complex and provide sound fields that are very difficult (if not impossible) to calibrate. The

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[13] The term *startle response* is often used without definition, and it may have very different meaning to different investigators studying fish hearing (and fish behavior in general). The more classic use of the term refers to the response elicited by stimulation of the Mauthner cell (M-cell) in the brain by various sensory systems including hearing (Eaton and Popper 1995). Stimulation of the M-cell results in a very rapid (millisecond) twist of the animal body in a direction away from the stimulus, followed by the animal potentially swimming in the direction of the turn. Other investigators use the term more loosely as any response of the fish to the onset of a stimulus. There is not necessarily a directionality to the response, and it may not elicit any long-term effect on behavior. Quite often, authors do not indicate what they mean by this term, and so care needs to be taken in determining the relevance of any study that talks about startle unless the term is defined.

tanks do not represent a free field in which fish normally live and hear (e.g., Duncan et al. 2016; Gray et al. 2016b; Rogers et al. 2016), or even the coral reefs, caves or rock formations in which other fishes live.

Indeed, the walls of most tank are usually so thin and flexible that they act as pressure release boundaries (e.g., Rogers et al. 2016). As a consequence, the tank behaves like a “brick” of water surrounded by air. When the acoustic source is in the water, the sound pressure must fall to zero at the walls, bottom, and surface, greatly increasing the levels of particle motion. All six surfaces (four walls, air/water interface, bottom) are nearly perfect sound reflectors. Close to the water surface, the ratio of kinetic energy to potential energy (discussed in Chapters 3 and 4) can be enormous. As a consequence, results from studies in conventional water tanks are often of questionable value. Specialized tanks have been designed to overcome these problems (e.g., Hawkins and MacLennan 1976; Halvorsen et al. 2011; Halvorsen et al. 2012a), but have not generally been utilized because they are both complex to design and use, and very expensive to build.

CHAPTER 3: UNDERWATER ACOUSTICS[14]

1. Sound Waves

To be precise in the use of acoustical terms in this review, it is important to define the key acoustic terminology that is used. A vibrating object or surface produces propagated sound that is composed of both sound pressure and particle motion in the adjacent medium (e.g., air or water) (see Chapter 4 for a discussion of particle motion). *Vibration* is a term that refers to the motion of any object that produces sound, such as a loudspeaker or a driven pile. Once the energy produced by the vibrating object leaves the source and travels through the adjacent medium (e.g., air, water), the energy is considered to be sound. The back and forth motion of the water, sometimes inappropriately termed "vibration," is best described as the particle motion.[15]

Essentially, sound is an alteration in pressure or material displacement propagated via the action of elastic stresses in an elastic medium. Each medium has its own elasticity, which varies with temperature. A sound wave is the realization of sound. The total energy contained in a sound wave consists of the sum of its potential energy (PE) and its kinetic energy (KE). The PE arises from the compression and expansion of the fluid and, hence, is related to the sound pressure, whereas the KE arises from the back and forth particle motion. This review uses the term *sound* to refer to signals that contain both sound pressure and particle motion. The term *vibration* will be limited to the motion of any sources of sounds. There are a number of metrics for describing sounds, all of which apply to sound in any medium, including both air and water. Acoustic terms and metrics are defined in this report according to the most recent international standard published in ISO 18405 (2017). Acoustic terms and metrics applying to pile driving are also published in ISO 18406 (2017).

The total of all the sound in a specific area, at a given time, is termed the *ambient sound*. Some authors exclude sound from anthropogenic sources when describing the ambient sound, but others may include it. The term *soundscape* is used to characterize the ambient sound in terms of its spatial, temporal, and frequency attributes, as well as the types of sources contributing to the sound field. Sounds within a soundscape can be of either natural or anthropogenic origin. It has been demonstrated that the hearing abilities of some fishes are limited by the presence of natural levels of ambient sound, as well as by sounds of anthropogenic origin (Hawkins and Chapman 1975).

Sound is often referred to using terms such as sonic, infrasound, and ultrasound. The range of frequencies to which each term refers is not generally defined and often rather arbitrary. For the purposes of this report, the *sonic* range of hearing for fishes is defined as being from 30 to about 10,000 Hz, whereas *ultrasound* refers to signals above 10,000 Hz (= 10 kHz), and *infrasound* refers to signals below 30 Hz.

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[14] Much of the terminology for underwater acoustics has been recently defined by the International Standards Organization (ISO18405 2017). We follow much of its terminology in this report, but, when reviewing literature published before the standard was in place we have used the terminology employed by the authors for clarity.

[15] For an excellent introduction to underwater sound see www.dosits.org. For a specific description of particle motion, see https://dosits.org/science/sound/what-is-sound/

The term *noise* is often used in the literature to describe unwanted sound that is judged to be unpleasant, loud or disruptive to hearing, or that can hinder detection of a particular signal. In some circumstances *ambient noise* or *background noise* may also be used in the literature to describe ambient sound generated by natural sources that may have such adverse effects. However, not all ambient sounds necessarily have adverse effects

a. Sound Pressure

Sounds are most commonly measured and expressed as the pressure fluctuations in the medium above and below the local hydrostatic pressure. The *sound pressure* is the contribution to total pressure caused by the action of sound. Sound pressure is a function of time, which may be indicated by means of an argument t, as in p(t), where p is sound pressure and t is time. Sound pressure is expressed in SI units (International System of units) of pascals (Pa) or micropascals (µPa). The pascal is a unit of pressure equal to 1 newton per square meter. Sound pressure acts in all directions; it is a scalar quantity that can be described in terms of its magnitude and its temporal and frequency characteristics.

b. Sound Levels and their Metrics

The metrics that are used to describe sounds and the characteristics of their sources must relate to the potential effects on biological receptors. Sounds of differing characteristics (e.g., impulsive vs. continuous; short term vs. long term) potentially have different effects. It is especially important to define those characteristics that are potentially harmful to fishes. For example, when considering potential effects on behavior, or masking by continuous sounds (as from shipping), the critical level might be the root mean square (RMS) sound pressure, measured over a specified time interval, divided by the duration of the time interval, for a specified frequency range.

Where there is concern about the effects of impulsive sounds, such as from pile driving, on physiology or behavior, then an appropriate metric might be the instantaneous peak level of the impulsive sound. That is, the level of the zero-to-peak sound pressure (Figure 3). The zero-to-peak sound pressure level is the largest absolute value of the instantaneous maximum over-pressure or under-pressure observed during the pulse. In some cases, however, the peak level may also be expressed as the peak-to-peak level, the level of the sum of the maximum over-pressure and the maximum under-pressure. It is important to state what is meant when the term *peak* is applied. In this review the term *peak* generally refers to the zero-to-peak level, rather than the peak-to-peak level.

In some instances, RMS sound pressure may be used to describe the energy within a pulse. For impulses, this is the square root of the average of the squared pressures over the time that comprise that portion of the waveform containing 90% of the sound energy of the impulse.

Alternatively, the total energy within the pulse, especially when the pulse is of complex shape and varies substantially in amplitude, such as that produced during pile driving, may be described by the *sound exposure level* (SEL) (Popper and Hastings 2009). The SEL is the integral, over time, of the squared sound pressure. The SI unit of sound exposure is the product of one pascal squared for one second ($Pa^2 \cdot s$).

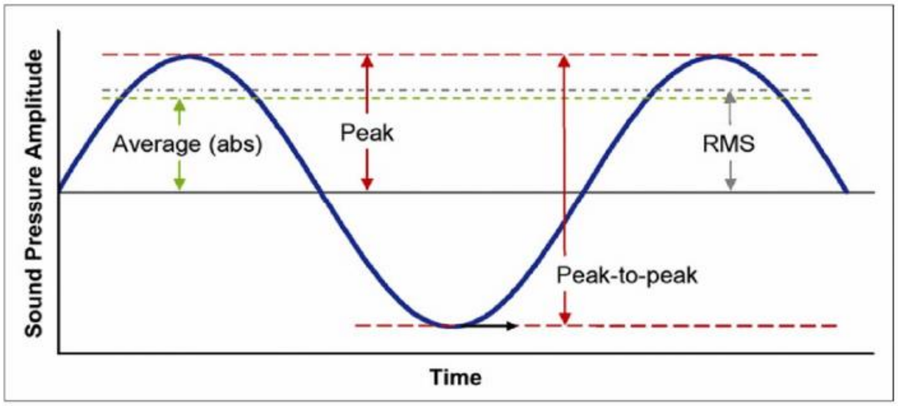

Figure 3: Measurement of sound amplitude.
Illustration of sound pressure showing the Peak, Peak-to-Peak, and RMS sound pressures. The same terminology applies to particle motion. (From Caltrans 2015).

The SEL may be specified for a single impulse or strike (the SEL_{ss}). However, when impulsive sounds are repeated, such as takes place during pile driving, it is appropriate to estimate the *cumulative SEL* (SEL_{cum}) associated with a series of pile strikes. The SEL_{cum} is the total noise energy to which the animal is exposed over a defined time period. It can be difficult to estimate the SEL_{cum} in some circumstances because the animal may be moving relative to the sound source so that the received level of the individual pulses may vary. Moreover, the number of strikes needed to install a pile depends on many factors, such as the size and type of the pile, the type of substrate, and the size of the hammer. In estimating the SEL_{cum} it may be necessary to estimate the total number of strikes that may occur from several piles being driven at the same general location over the same time period.

Sound levels (whether sound pressure or particle motion) are often referred to in terms of dB, which is a logarithmic scale (ISO18405 2017). The level L_F of a field quantity F (e.g., RMS sound pressure or particle motion) is:

$$L_F = 20 \log_{10}(F/F_0) \text{ dB},$$

where F_0 is the reference value of the field quantity. Similarly, the level L_P of a power quantity P (e.g., sound intensity) is:

$$L_P = 10 \log_{10}(P/P_0) \text{ dB}.$$

In water, sound pressure levels are referenced relative to 1 µPa. Particle motion can be expressed in terms of displacement, velocity, or acceleration. Particle displacement is referenced relative to 1 picometer; particle velocity to 1nanometer per second (nm/s); and particle acceleration to 1 micrometer per second squared.

A critical point is that careful consideration must be given to the appropriate metrics for each kind of source. It will ultimately be important agreed upon standards for such measurements so that there is common ground for the description and regulation of each type of sound source. Other important characteristics of impulsive sounds (both pressure and particle motion) include:

- *Rise time*. This is defined as the time interval a signal takes to rise from 10% to 90% of its peak value. The rise time may affect the response of animals, and may be especially important in terms of injury, where short rise times may be especially damaging.

- *Frequency spectrum*. The sound pulse is composed of a range of frequencies, expressed in terms of the level at each frequency measured over a given bandwidth. The bandwidths utilized are generally 1 Hz or 1/3 octave (an octave is a doubling of frequency). It is important to specify the frequency bandwidth. The frequency content is important because different animals respond to different frequency ranges.

c. Sound Propagation at Different Depths

The propagation of low-frequency sounds with long wavelengths may be constrained in shallow water (Rogers and Cox 1988) because the low-frequency sound pressures (including those from pile driving) propagate less well through shallow water than through deeper water. For example, if the water depth is 12 m, then sound pressures at frequencies below about 60 Hz (having a wavelength of greater than 25 m) do not propagate well (Rogers and Cox 1988; Ainslie 2010; Nedelec et al. 2016), although the precise cut-off frequency depends on the speed of propagation through the substrate and its density. A recent study of the propagation of pile driving sounds (Bruns et al. 2016) supports these theoretical analyses.

It is important to note, however, that this constraint does not necessarily apply to particle motion. Close to the water surface, sound pressure may be converted into particle motion as a result of the lower density and greater elasticity of the air above the water, resulting in a higher level of particle motion than would be predicted from simple equations or from results in deeper water. In these circumstances, the particle motion may also have a strong vertical component, especially close to the water surface. Moreover, sounds may also travel within the substrate or accompany waves that are traveling along the interface between the water and the substrate (again something that is likely quite prevalent with pile driving, where much of the energy enters the substrate). Some low-frequency sounds may propagate over considerable distances by way of the substrate/water interface (Bruns et al. 2016; Hazelwood and Macey 2016a).

d. Substrate Transmission

It has recently become apparent that many fishes that live on or close to the bottom are likely to be able to detect the sound that comes into the water after it has propagated through the substrate, both as compression waves and interface waves (Figure 4, page 32) (reviewed in Ballard and Lee 2017; Popper and Hawkins 2018). Interaction between compression waves and shear waves generated within the substrate can give rise to interface waves that travel along the surface of the substrate. These interface waves travel slower than the speed of sound in water and can have strong particle motion components.

In environments where the water depth is comparable to a few acoustic wavelengths, the propagation of waterborne sound is expected to be strongly influenced by the properties of the substrate. However, relatively few acoustical studies have been carried out in such shallow-water environments (Kuperman and Lynch 2004). Most acoustical work in such environments has been conducted at relatively high frequencies and has focused only on the propagation of compressional waves, involving measurements of the sound pressure.

Recently, however, an interesting acoustic propagation experiment was conducted by Ballard et al. (2018) to characterize low-frequency propagation in a very-shallow-water estuarine environment. The emphasis of the work was on understanding the propagation effects induced by the substrate. The measurements were carried out in a shallow estuary with water depths of 2 to 3 m. In monitoring the sounds from an impulsive source, it was noted that the compression wave arrived earliest at the hydrophone locations and included frequencies mainly above 150 Hz. The interface wave, recorded as the vertical particle velocity, arrived later, traveling at a speed of only 85 meters per second (m/s), and consisted mainly of frequencies below 25 Hz. The interface wave was associated with the mud/sand interface within the estuarine bed. Ballard et al. concluded that in an environment with greater water depths, propagating compressional waves from the source as well as from ambient noise sources would likely obscure the low-amplitude arrivals of the interface wave.

These and other results suggest that in some circumstances the signal that comes out of the substrate may be much stronger than the signal from the same source but that has propagated through the water (Popper and Hastings 2009). The motion of the sediment also moves the water close to the seabed and generates evanescent sound pressure and particle motion waves that propagate through the water. It should be noted, however, that the sound levels generated in the water by motion of the substrate may decline steeply vertically above the substrate. Interface waves are also known as Rayleigh and Scholte waves (e.g., Viktorov 1967; Rauch 1986). The Scholte waves traveling along the substrate are typically supported at frequencies less than about 30 Hz and their rate of decay with distance (transmission loss) increases with frequency. The interface waves are often referred to as *ground roll* (Robinson et al. 2014; Hazelwood and Macey 2016b; Hazelwood and Macey 2016a; Chotiros 2017).

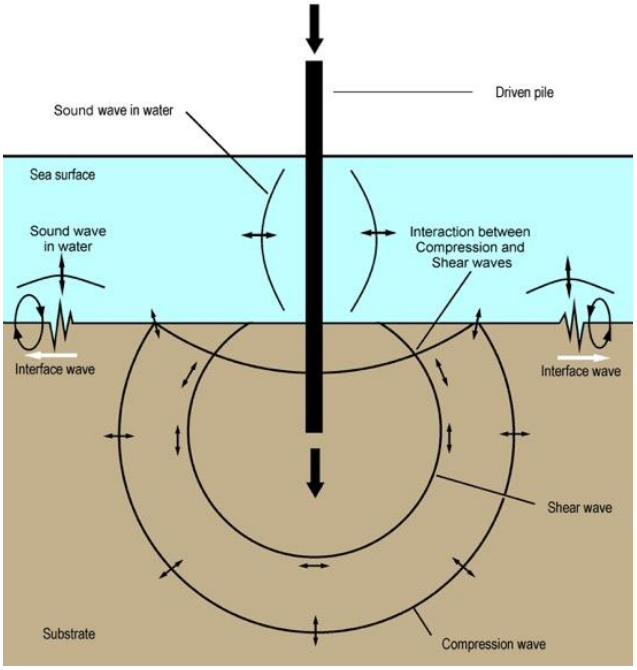

Figure 4: Acoustic signals produced by striking a pile.
Figure ©2019 Anthony D. Hawkins, all rights reserved.

There have been no direct studies of the detection of substrate signals by fishes. However, because fishes are sensitive to particle motion, it is evident that species living on or in the substrate will detect sounds transmitted through and on the substrate, as has been shown for invertebrates (Roberts et al. 2016c; Roberts and Elliott 2017). Because ground roll, generated by seismic airguns and pile drivers, occurs especially at very low frequencies, the sensitivity of animals to infrasound is relevant. In preparing impact assessments for fishes and invertebrates, the presence of substrate transmission and interface waves, and the high levels of particle motion that may be generated, must not be ignored in modeling the propagation of sound from anthropogenic sources. However, in current assessments, these factors often are ignored, possibly leading to erroneous conclusions (Hawkins and Popper 2016).

2. Sound Propagation Models

A critical issue in understanding underwater sound, and the potential impact of the sound on an animal, is how a signal propagates from the source and how the associated signal levels and frequency content of the signals vary at different locations. In midwater in the open ocean, sound propagates great distances with little attenuation, and, given the right circumstances, propagation of low frequencies can occur over tens, hundreds, or thousands of kilometers (Urick 1983). Much effort has gone into developing hydroacoustic models to understand and predict sound propagation.

However, these models have been developed for open-ocean environments, which are essentially boundless systems. These types of deep-water hydroacoustic models are not applicable to shallow-water environments, including bays, rivers, and lakes, where the acoustic signals reflect off the substrate and the air/water interface. As discussed on page 30, the behavior of sound in shallow water is quite different from that in deep water (Urick 1983; Rogers and Cox 1988; Bevelhimer et al. 2016). Significantly, in shallow water there is often strong attenuation of sound pressures at lower frequencies (often those within the hearing range of fishes and invertebrates), depending on parameters such as water depth and substrate composition (Rogers and Cox 1988). This means that even if there is strong low-frequency energy in the sound at the source, the sound might not be detectable at a distance by a fish sensitive only to sound pressure. Propagation of particle motion in shallow water is even more complex. In developing such models, it may be necessary to examine the levels of sound pressure and particle motion generated by different sound sources at various locations within a particular site.

It is often assumed that in shallow water the simple spreading models overestimate sound pressure levels relative to actual measurements, on the basis that the propagation of low-frequency sounds with long wavelengths may be constrained. However, Pine et al. (2014) investigated the propagation of anthropogenic sound in shallow waters (25–45 m deep) from both single and multiple sources. The results from multiple sources are probably inappropriate when considering sound propagation from pile driving. However, the results from a single source suggested that the sound intensities observed over the range 0.1–20 kHz were higher than the estimates derived from standard geometric spreading models by 3–41 dB across all measured distances from the source. The greatest difference from the theoretical spherical spreading model (41 dB) occurred at the site most distant from the sound source (5 kilometers [km]). Geometric spreading models are often used to assess the likely ecological impacts of anthropogenic sound. The models estimate the propagation of underwater sound, but often without *in situ* verification.

With pile drivers, it is also evident that there would be additional energy propagated by way of the substrate, a factor that simple geometric propagation models often ignore. As Hazelwood et al. (2018) point out, seismic interface waves generated by seabed impacts are believed to have biological importance for fishes and other animals living close to the substrate. Their modelling work suggests that the wavelets generated by seabed impacts are propagated by cylindrical spreading, which may result in relatively widespread effects on benthic fishes. The interface waves travelling across saturated sediments are also slower than the compression waves in the adjacent water, affecting the temporal structure of the received impulse. More studies of the propagation of vibration energy from piling activities are needed to estimate the environmental impact of pile driving and other sources impacting the substrate. The effects of substrate borne vibration on benthic life may be quite significant. It is likely that geometric models used by some regulatory bodies may be underestimating the spatial extent to which the energy generated by pile driving may be propagating and creating potential ecological impacts.

It is recognized by Caltrans (2015) that the propagation of pile driving sound underwater is highly complex due to many factors, including the fact that the river or ocean bed and the surface of the water are distinct boundaries that can affect the propagation. In addition, the pile that is driven by an impact driver generates ground vibration in the substrate which can re-radiate sound energy back into the water (Figure 4, page 32).

In practice, it is not always possible to model all of the factors involved in the propagation of sound underwater. For practical reasons, simplified models often are used to predict sound levels at various distances from a pile and the distance at which pile driving sound attenuates to a specific criterion level. Measured transmission loss rates in shallow water typical of pile driving sites have been found to vary considerably from site to site. The rates also vary somewhat between the different measurement metrics: peak SPL, RMS, and SEL (Caltrans, 2015).

NOAA Fisheries developed a spreadsheet model for evaluating underwater sound from pile driving (Woodbury and Stadler 2008; Stadler and Woodbury 2009) that estimates the distance at which pile driving sound attenuates to threshold levels (see discussion on page 98). This spreadsheet applies a simple spreading model with a default attenuation rate of 4.5 dB per doubling of distance that can be used unless site-specific attenuation data are available to support a different rate of attenuation. The simple model was developed to be conservative for most circumstances encountered on the West Coast and as a practical tool for estimating potential impacts. Use of this model can result in an overestimation of sound levels at increasing distance. Although the accuracy of the model is limited, it is considered by the designers to result in a conservative result that is protective of fish. As additional data on sound propagation in shallow water become available, the modeling approach should be improved to address the many complex factors that affect underwater sound propagation, such as salinity, temperature, pressure, water depth, substrate type, and reflections from the water surface and substrate.

At present, while there are a number of propagation models for deep water, there are few peer-reviewed models for very shallow waters, (but see MacGillivray et al. 2011; Lippert et al. 2018). Developing models for shallow water is often very difficult because variables such as water depth and the nature of the substrate have to be taken into account – variable not as important in deep-water models where the bottom is often far from the source and unlikely to have significant effect on the sound waves. It is especially important, when dealing with fishes, to model the particle motion as well as the sound pressure, as the particle motion is what many species are responding to. Such models will subsequently need to be validated by making actual measurements of sound pressure and particle motion at different locations within a site that has been modeled. Currently, although it is clear that complex hydroacoustic models will provide the most accurate results for shallow-water environments, for practical reasons regulatory agencies on the West Coast currently recommend the use of a rather simple but conservative model, developed for use in deep waters, for evaluating the effects of underwater noise on shallow-water fishes.

3. Acoustic Characteristics of Pile Driving

As shown in Figure 4 (page 32), and as discussed in the following paragraphs, pile driving generally produces a complex set of signals, all of which are potentially detectable by animals in the water and on the substrate. A critical outcome from these recent studies is that the sound field associated with pile driving is complex, still poorly understood, and reflects acoustic input from sound propagating through the water, propagating through the substrate, and reflecting off both substrate and surface.

Many measurements have been made of the sound pressure levels generated by both percussive and vibratory pile driving (Nedwell et al. 2003; Thomsen et al. 2006; Urquhart 2006; MacGillivray 2018). Sound pressure levels generated in water by percussive pile driving are high and variable depending on the pile type, the substrate being penetrated, the distance from the source, and depth of water, as well as and whether a bubble curtain or other physical mitigation measures are being employed (Caltrans 2015; Reyff 2016).

Recently, Dahl and Dall'Osto (2017) measured underwater sound pressures from impact pile driving using a vertical line array 120 m from the pile source (water depth 7.5 m) over which bathymetry varied gradually increasing to depth 12.5 m at the line array. Particular emphasis was placed on the arrival of a precursor pulse — a segment of the pressure waveform that had propagated through a higher-speed sediment borne path. The well-defined precursor waveform of characteristic center frequency of around 400 Hz was observed to arrive prior to the main waterborne arrival. The peak absolute value amplitude of the precursor reached within 20 dB of the main waterborne counterpart. It was suggested that the precursor can play a role in establishing a bound on the performance of any underwater noise mitigation strategy unless the sediment extent of the pile is shielded to attenuate this flanking path.

Theobald et al. (2014) pointed to the need for standardization of the measurement of underwater noise radiated from marine pile driving to meet concerns over the potential for impact on marine fauna. A method for the measurement of the underwater noise was described using a combination of fixed autonomous recorders and vessel-based hydrophone deployments.

A study of pile driving sound was carried out during construction at the new Block Island Wind Farm, located 4.5 km from Block Island, Rhode Island, in the Atlantic Ocean (R. Gopu and J. Miller, personal communication, 2018). Evaluation of underwater acoustic monitoring data indicated that pile driving sound was still detectable up to 20 miles from the source in deep water (120 dB re 1 µPa rms), at which point pile driving sound started approaching background sound levels. Based on models calibrated with measured data, the sound levels were a function of water depth, which varied based on direction away from the pile. Overall, the sound levels were quieter in deep waters and louder in shallow waters; the difference between the two could be as large as 10 dB. Sound levels were also shown to be very dependent upon the orientation of the pile (13.3° off vertical), with a 10- or 15-dB difference from one angle to another. The investigators found, in preliminary analysis, that at 500 m from the source, particle motion (which was directly measured) ranged from below 10 Hz to about 800 Hz, and that the levels were above those potentially detectable by several species for which there are particle motion sensitivity thresholds available, including Atlantic salmon (*Salmo salar*) (Hawkins and Johnstone 1978), several flatfishes (Chapman and Johnstone 1974) and Atlantic cod (*Gadus morhua)* (Chapman and Hawkins 1973).

MacGillivray (2018) described the underwater noise from pile driving at a deep-water oil platform. The underwater noise from impact pile driving of 512-m-long conductor casings was measured at a deep-water offshore oil platform in the Santa Barbara Channel. Beam forming measurements, obtained with a vertical array, confirmed that the primary wave front generated by hammering the conductor casing was a Mach cone propagating at an angle of 17.6° below the horizontal. Analysis of the processed array data also revealed the presence of high-frequency secondary waves at angles steeper than 45° below the horizontal. These secondary waves, which appeared to be generated near the sea surface, dominated the acoustic spectrum of the pulses at frequencies above 1 kHz. Shallow hydrophone measurements outside the Mach cone showed clear evidence of a surface shadow zone, which was caused by the strong downward directivity of the source. Although reflected waves, diffraction, and secondary waves still produced sound inside the surface shadow zone, sound levels were 10–15 dB lower in this region. Long-term hydrophone measurements showed that there was little difference (±1 dB) in mean sound levels from impact hammering of different conductors installed at the same platform over three months.

A recent approved international standard (ISO18406 2017) describes the methodologies, procedures, and measurement systems to be used for the measurement of the radiated underwater acoustic sound generated during pile driving using percussive blows with a hammer. However, the standard covers only the measurement of sound pressure and not measurement of sound particle motion. The authors of the standard emphasize that this exclusion does not imply that such measurements are unimportant; indeed, their importance in assessing the impact on aquatic life is recognized. However, at the time of drafting, measurement of these quantities was not yet mature enough for standardization.

CHAPTER 4: PARTICLE MOTION

1. The Importance of Particle Motion

Most assessments of the potential effects of sound on fishes have not paid attention to a key factor, the sensitivity of fishes to the particle motion that accompanies the transmission of sound in water. The intent of this chapter is to bring the issue of particle motion to the forefront, raising the most important issues and then suggesting approaches to ensure that particle motion is better understood and properly taken into account in the future for the evaluation of the potential effects of anthropogenic sound on fishes and invertebrates. At the same time, we do not expect that particle motion will become a component of regulatory activities in the near future. However, an understanding of particle motion and its importance to fishes (and regulation of activities affecting fishes) is important for regulators and others so they are prepared to consider particle motion in future criteria updates.

It is important to note that particle motion was identified as an important stimulus for fishes many years ago, as discussed in several recent reviews (Hawkins and Popper 2018a; Popper and Hawkins 2018). However, in part due to technical difficulties in measuring and using particle motion, as discussed later in this chapter, it has not been the focus of most recent scientific studies. Moreover, particle motion has not been taken into account in setting sound exposure criteria or carrying out environmental impact assessments for anthropogenic sources of sound.

There are reasons that particle motion has not been the focus of study or regulatory activity, despite its being fundamentally important for fishes. Most importantly, there is a significant lack of sufficient scientifically based data about the hearing thresholds and behavioral responses to particle motion by fishes and a lack of scientifically acceptable criteria (e.g., see Popper and Hawkins 2018). In addition, there are no particle motion measurement standards, nor easily used and reasonably priced instrumentation to measure particle motion.

However, because it is internationally recognized that particle motion must ultimately be considered in regulatory activities associated with fishes (e.g., Andersson et al. 2017), developing an understanding of particle motion, and its importance to fishes, is an important first step. In particular, future studies should be designed to include particle motion so that it can eventually be incorporated into regulation. Thus, this chapter provides background on particle motion, its importance to fishes, and an overview of the gaps in our understanding of the potential effects of increased particle motion on fishes.

2. Characterization of Particle Motion

The total energy contained in a sound wave consists of the sum of its PE and its KE. The PE arises from the compression and expansion of the fluid and, hence, is related to the sound pressure, whereas the KE arises from the particle motion. The sound particle is the smallest element of the medium that represents the medium's mean density (ISO18405 2017). Particle motion may be expressed in terms of the particle displacement (SI unit: meter), or its time derivatives: particle velocity (m/s) or particle acceleration (m/s^2). Particle motion, like sound pressure, is a function of time (t).

It is commonplace to characterize a sound by the sound pressure alone (Chapter 3), because sound pressure can easily be measured with conventional hydrophones. It may then be possible to estimate the particle motion, if required, from the sound pressure measurements and a knowledge of the acoustic properties of the medium. In a *free sound field*, which is defined as a location that is distant from acoustic boundaries such as the water surface or bottom, the particle motion and intensity levels can be inferred from measurements of the sound pressure. However, in real acoustic environments, close to the substrate or the water surface, and in the shallow waters that are inhabited by many fishes and invertebrates, the relationship between the particle motion and the sound pressure becomes more complex (Pierce, 1981), and the particle motion is not always correlated with the sound pressure. In these circumstances it is necessary to measure particle motion directly.

The sound intensity is the product of the sound pressure and the particle velocity, for which the SI units are watts/m^2. Particle motion, in contrast to sound pressure, is a vector quantity that can be fully described only by specifying both the magnitude and direction of the motion because the magnitude can be quite different on the x, y, and z axes (Figure 5). Sound intensity is also a vector quantity.

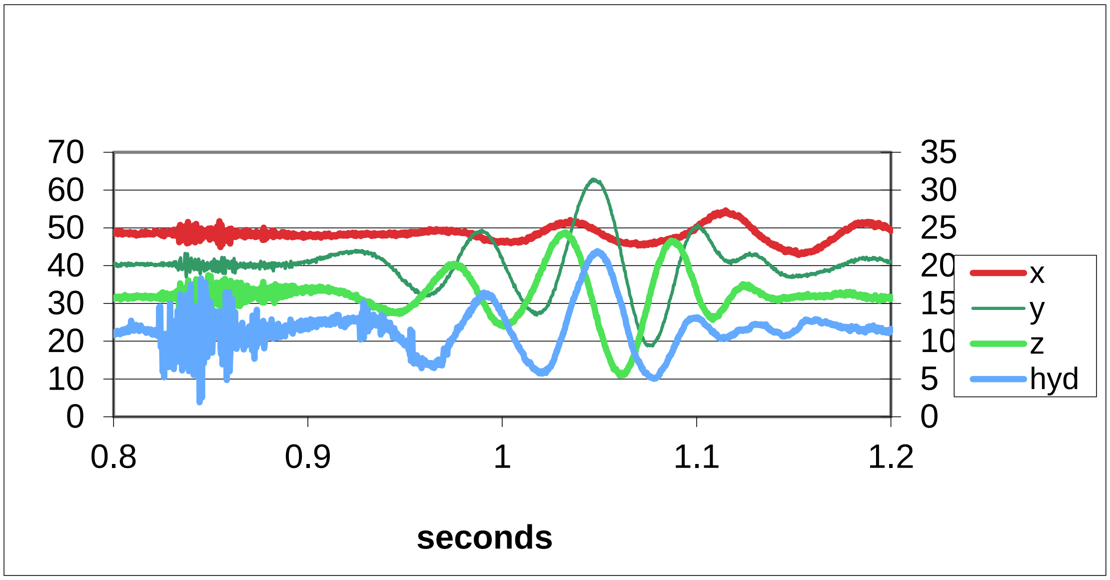

Figure 5: Waveforms of sound pressure and particle velocity.
Data recorded at a distance of around 25 meters when a heavy object was dropped onto the substrate. The particle velocity is monitored in different directions within the substrate (x, y and z), and the sound pressure (hyd) is monitored in the water just above the substrate. The sound pressure (right axis) is expressed in pascals and the particle velocity (left axis) in µm/s. The horizontal and vertical velocity is approximately 40 µm/s at around 16 Hz, while the evanescent pressure amplitude is approximately 16 pascals peak-to-peak.

It is now well understood that particle motion is the critical stimulus for most fish species (Hawkins and Johnstone 1978; Radford et al. 2012; Hawkins et al. 2015; Hawkins and Popper 2016; Nedelec et al. 2016; Andersson et al. 2017; Popper and Hawkins 2018). The significance of this understanding is that much of the data collected on the hearing sensitivity (lowest sound level detectable) and hearing range (range of frequencies detectable) (e.g., Fay 1988; Ladich and Fay 2013) is wrong, because it was based only on measurements of sound pressure and not particle motion. Indeed, in examining all of the data on fish hearing in the literature, it appears that few accurate hearing thresholds have been determined, because only a few studies have considered particle motion in addition to sound pressure (e.g., Chapman and Hawkins 1973; Hawkins and Johnstone 1978) (see Figure 7, page 49).

Thus, any discussion of underwater sound and its effects on fishes must include consideration of particle motion (reviewed in Hawkins and Popper 2016; Nedelec et al. 2016; Popper and Hawkins 2018). It is important to note that fishes, including those sensitive primarily to sound pressure (Chapter 9), must make use of particle motion to determine the direction from which a sound is coming (Hawkins and Popper 2018a). Many of the invertebrate animals that form the food of fish are also sensitive to particle motion (Popper and Hawkins 2018). Any future work exploring the effects of sound on fish and invertebrates, including the determination of detection thresholds, must pay attention to the detection of particle motion signals (Hawkins et al. 2015).

3. The Relationship between Sound Pressure and Particle Motion

In the absence of acoustic boundaries (under free-field conditions, such as in the ocean at some distance from the surface and seabed), the sound pressure radiated from a simple acoustic source falls off as $1/r$, where r is the distance from the source (Harris and van Bergeijk 1962; Ainslie and de Jong 2016). Far from the source (in the acoustic *far field*), the energies associated with acoustic pressure and acoustic particle velocity are equal (KE = PE), and substantially in phase.

Close to the source (in the acoustic *near field*), the particle velocity component of the field contains more energy (KE > PE) and is not of the same phase as the sound pressure. The rate of decline of the particle velocity in the near field depends on the frequency of the signal, with the distance being greater for lower frequencies (van Bergeijk 1964). The rate of decline also depends on the nature of the sound source and its movement pattern (e.g., whether it is a monopole, dipole, or quadrupole (Harris and van Bergeijk 1962)). A monopole is a source which radiates sound equally well in all directions such as a pulsating bubble. Any sound source whose dimensions are much smaller than the wavelength of the sound being radiated will act as a monopole.

A dipole essentially consists of two monopoles of equal source strength, but opposite phase, and separated by a small distance. The water particles surrounding the dipole simply move back and forth between the sources, so that particle motion takes place along a particular directional axis. Sound pressure maxima are aligned with the 0° and 180° axes, with no sound pressure radiated along the 90° and 270° axes. A quadrupole source consists of two identical dipoles, with opposite phase and separated by small distance. Such a source can be resolved into two perpendicular components. A lateral quadrupole is where the orientation of one axis is perpendicular to the other. With a longitudinal quadrupole, the axes are parallel.

4. The Significance of Particle Motion with Regard to Fish Hearing

Most hearing threshold data (the lowest sound levels that a fish can detect 50% of the time at each frequency tested) have been calibrated against sound pressure levels. However, most fishes primarily detect particle motion rather than sound pressure and, as a result, the actual hearing sensitivity of most species is not properly known (Hawkins and Popper 2016; Popper and Hawkins

2018). Many hearing experiments have also been conducted under inappropriate acoustic conditions, and it is not possible retrospectively to convert sound pressure measurements into particle motion. Similar problems exist where damage to hearing from exposure to sound pressure has been examined, but where the damage may have been caused by particle motion. The significance is that it is not yet possible to predict how anthropogenic sounds, such as those produced by pile driving or ship traffic, may affect hearing or if fishes even detect the sounds.

Similarly, it is not yet possible to suggest the levels of sound that are likely to result in behavioral changes because sounds are generally represented in terms of sound pressure even though the majority of fishes are detecting particle motion. Thus, as argued strongly in several recent and important critical reviews of the literature (Nedelec et al. 2016; Carroll et al. 2017; Hawkins and Popper 2018; Popper and Hawkins 2018), it is very important that particle motion be considered in noise impact studies on fishes (and invertebrates), particularly those species lacking a gas-filled bladder (including all elasmobranchs, many teleosts, and marine invertebrates), as these species only detect particle motion. However, even those fishes that detect sound pressure may utilize particle motion to determine sound source direction.

Moreover, it will be important, in the future, to develop protocols for monitoring particle motion, and then to determine those levels of particle motion that could result in potentially adverse effects on fishes, such as increased mortality, injury to tissues, changes in physiology, effects on hearing abilities, and changes in behavior. Determining those levels would facilitate the development of criteria for exposure to particle motion. It is especially likely to be high levels of particle motion that cause the damage to hair cells within the ears, and it is, therefore, important to ensure either that any experiments using sound pressure are carried out under identical, preferable free-field, conditions, where the particle motion can be estimated or that the actual levels of particle motion are measured.

5. Particle Motion Measurement Methods

Measuring sound pressure, especially in midwater in a deep ocean, is well understood (e.g., Caltrans 2015) and it is possible to calculate the accompanying particle motion from the sound pressure measures because there is a clear and known relationship between the two. In contrast, in shallower waters and close to the bottom and to the surface, measurement of particle motion is rather more difficult for a variety of reasons. This difficulty has led to a dearth of data on particle motion and its potential effects on fishes. Moreover, there are far fewer devices available for the detection and analysis of particle motion than to measure pressure (Gray et al. 2016a; Martin et al. 2016; Lumsdon et al. 2018). Special sensors are required to specify the particle motion in terms of the particle displacement or its time derivatives (particle velocity or particle acceleration) in three dimensions.

Popper and Hawkins (2018) describe how particle motion may be monitored using geophones or accelerometers, designed originally to detect motion of the ground (in either air or water). Particle motion hydrophones can be assembled from three of these vector sensors (arranged orthogonally) contained within a neutrally buoyant container (Banner 1973). An alternative approach to determining particle motion is to measure the sound pressure gradient in the water, using a pair of sound pressure hydrophones, and then to derive the particle motion from that gradient, although there are a number of practical considerations to be satisfied when implementing that approach (e.g., see Zeddies et al. 2010, 2012).

Estimations of particle motion based on sound pressure measurements and plane-wave assumptions can lead to substantial errors (Gray et al., 2016b). Sound pressure measurements made under different conditions may be accompanied by very different levels of particle motion. Measurements of particle motion levels made close to the substrate have confirmed that they may be larger than expected. Banner (1968) found that the levels of ambient particle velocity measured in very shallow water were considerably higher than the levels that would accompany the same sound pressure levels under free-field conditions, particularly at low frequencies. More recently, Ceraulo et al. (2016) showed that the particle velocities generated by a pile driver in a shallow-water environment were higher, particularly for the z-axis, with a magnitude of 1 to 10 times (average 3.5) greater than that of the predicted particle velocity for an acoustic plane wave at the same sound pressure.

Moreover, particle motion is a vector quantity. As a consequence, it is necessary to monitor its direction as well as level. The recent development of vector sensing hydrophones, which combine a sound pressure hydrophone with three orthogonal particle motion sensors, may be most useful for future work (Jing et al. 2014; Martin et al. 2016).

6. Particle Motion Levels Generated by Pile Driving

Bruns et al. (2016) installed hydrophones and geophones on the seabed at different distances from a pile driving operation in 19–24 m water depth. The pile itself was also instrumented with sensors. The wave propagation in the pile, water, and soil was investigated. The work was carried out with a 6-m diameter monopile off the coast of Helgoland, Germany. It was clear that the highest sound pressure levels arose very close to the pile and the levels decreased with increasing distance. Moreover, the sound pressure level increased closer to the seabed.

The authors pointed out that, when the hammer hit the pile, a compression wave was induced in the surrounding sea water. In addition, compression waves, shear waves, and Scholte (interface) waves were emitted into the sandy subsoil because of the deformation of the pile shaft and the impact at the foot of the pile. These seismic waves were considered likely to generate additional sound pressure in the water.

The SEL_{ss} was approximately 176 dB re 1 $\mu Pa^2 \cdot s$ at 750 m from the pile. Attenuation of the sound pressure with increasing distance from the pile occurred across the entire frequency band but was more pronounced below 50 Hz and above 1 kHz. However, motion of the seabed was detected by the geophones, mainly at frequencies between 1 and 40 Hz, reaching an SEL of approximately 100 dB re 1 nm^2 at 750 m from the pile. The propagation velocity of the seismic wave was about 250 m/s, which is much slower than the velocity of the compression waves in water. The low-frequency vibration of the seabed was also detected by sound pressure hydrophones near the seabed.

In a study by Dahl and Dall'Osto (2017), the effects associated with a Scholte interface wave were observed as sound pressure data through spectral analysis, with the center frequency of the Scholte wave estimated to be around 9 Hz. It was concluded that Scholte interface waves that propagate along the bottom enable sound transmission at frequencies well below the acoustic cut-off frequency of the underwater waveguide, which in this case was about 65 Hz. Observations with a vertical line array of hydrophones allowed for an exponential fit of the Scholte wave sound pressure amplitude decay with increasing measurement depth above the bottom, and from these results a sediment shear speed of 260 m/s was inferred. It was also shown that the contribution from the Scholte frequency range to both the RMS sound pressure and the sound pressure SEL was negligible. However, no mention was made of the actual particle motion levels, although these are especially important to fishes and invertebrates.

In a recent paper by Hazelwood et al. (2018), the physics of the seismic interface waves generated by seabed impacts, including pile driving, were examined. Hazelwood et al. demonstrated that the particle motion of the water adjacent to interface seismic waves (ground roll) is elliptical and vigorous. The resultant wavelet peaks are propagated through cylindrical spreading, which can result in relatively widespread effects on benthic animals. Hazelwood et a. (2018) suggested that the elliptical pattern of the particle motion within such seismic wavelets provides a mechanism whereby benthic animals may identify the direction of propagation from the source.

7. Research Gaps in Relation to Particle Motion

Virtually all criteria and guidelines for both physical and behavioral effects are described in terms of sound pressure. However, most fishes are primarily detectors of particle motion and relatively few species of fish use sound pressure (Popper et al. 2014; Hawkins and Popper 2016; Popper and Hawkins 2018). Thus, developing criteria and guidelines in terms of sound pressure may have little relevance, particularly with respect to behavioral responses. Moreover, close to a source, in the acoustic near field or close to the substrate, where particle motion is particularly high, physical effects from shaking as a result of this signal has the potential to damage all body tissues, and particularly the ear, which is functionally based on relative motion between the dense otolith and the associated sensory epithelium.

Very little is known about hearing sensitivity to particle motion (Chapter 5) and it is imperative that such data be obtained. Concurrently, it is imperative to measure the signal from pile driving in terms not only of pressure, as now done, but also in terms of particle motion. However, this has been done in relatively few instances (e.g., Kugler et al. 2007; Sigray and Andersson 2011; Sigray and Andersson 2012; Gopu et al. 2018) (also R. Gopu and J. Miller, personal communication, 2018). Finally, behavioral responses need to be recalibrated in terms of particle motion as well as sound pressure for some species (e.g., see Figure 10, page 72).

Additionally, physical effects generated by particle motion need to be examined. The effects of the particle motion generated by pile driving in the acoustic near field or close to the substrate are important, not only because of potential direct effects on the ear, but also because these signals may result in motion of gas bubbles and high-density tissues (most notably the otoliths), potentially causing effects similar to those thought to be produced by sound pressure.

Current criteria and guidelines expressed solely in terms of sound pressure may have little relevance to many fishes with respect to damage to their hearing, physical effects, and their behavior.

CHAPTER 5: SOUND DETECTION - CAPABILITIES AND MECHANISMS

Sound is important in the lives of fishes (e.g., Hawkins 1993; Popper et al. 2001). Fishes may use sound for, among other things, communicating with one another, detecting prey and predators, navigating, and selecting appropriate habitats (e.g., Tavolga 1971; Hawkins and Myrberg 1983; Ladich and Winkler 2017). Moreover, even though many species do not produce sound, all species are likely to glean biologically important information about their environment by detecting and using what is called the "acoustic scene" or soundscape (see definition page 27) (Fay and Popper 2000; Fay 2009; Slabbekoorn 2018). In effect, sound detection provides fishes (as all animals) with three-dimensional information from a larger space around them than is possible using other senses, thereby expanding their sensory world and enabling them to rapidly get important information even in dark and in murky waters. As a consequence, any disruption in the ability of fishes to detect biologically relevant sounds (e.g., those of a predator) can have deleterious effects on the survival of individuals or populations.

1. Sound Detection Mechanisms

Fishes detect sound using an inner ear that is quite similar to those of terrestrial vertebrates, including mammals. Details of ear structure and function have been reviewed extensively elsewhere (e.g., Popper et al. 2003; Cranford et al. 2012; Schulz-Mirbach and Ladich 2016; Hawkins and Popper 2018a; Popper and Hawkins 2018; Schulz-Mirbach et al. 2018) and so will only be discussed briefly here.

The basic structure of the inner ear of fishes is the same as other vertebrates, with three semicircular canals and three otolith organs, the saccule, lagena, and utricle. The saccule is often considered to be the main hearing end organ of fishes, although there is evidence that the utricle and lagena participate in hearing and may even be the main hearing structures in some species (Popper et al. 2003). The ear is innervated by the eighth cranial nerve, which carries both auditory and vestibular responses to the brain. Each otolith organ in teleost fishes (most of the species of greatest concern to regulators) contain a dense calcareous structure, the otolith, which lies in close proximity to a sensory epithelium (or macula). The sensory epithelium contains numerous mechanosensory hair cells.

From the perspective of hearing, and of understanding current thinking on potential effects of anthropogenic sounds on fishes, it is critical to understand that the otolith organs of the inner ear are sensitive to particle motion and not sound pressure. This idea is strongly supported by a variety of experimental studies on various species (e.g., Enger and Andersen 1967; Fay and Popper 1974; Fay and Popper 1975; Hawkins and MacLennan 1976). Moreover, the role of the ear as a particle motion detector is also apparent based on anatomy, in that the inner ear of fishes closely resembles that of mass-loaded inertial accelerometers used for measurement of particle motion (e.g., Krysl et al. 2012; Schilt et al. 2012; Schulz-Mirbach and Ladich 2016; Hawkins and Popper 2018b; Hawkins and Popper 2018a; Schulz-Mirbach et al. 2018). Using particle motion, the ear can detect sounds from below 30 Hz to perhaps 500 Hz.

In a free sound field, such as in mid-water in the sea, the particle motion is correlated with the sound pressure (see Chapter 4 on particle motion). However, in shallow-water environments or close to the substrate or sea surface, the relationship between particle motion and sound pressure is more complex, and direct measurement of the particle motion is required.

Particle motion detection by a fish results from the very different densities of the otolith and the rest of the fish body. In fact, the body of a fish is very similar in average density and elasticity to water and, as a consequence, the tissues move back and forth with the acoustic particle motion. The otoliths (or otoconial masses in non-teleost species) are about three times denser than the rest of the body, and so their movements lag the movements of the rest of the body. Thus, the otolith, like the mass of an accelerometer, functions as a stationary mass. This motion stimulates the sensory hair cells of the ear, which then send signals to the brain through the eighth cranial nerve (e.g., Flock 1964; Hudspeth and Corey 1977). Essentially, the otolith organs respond to shaking

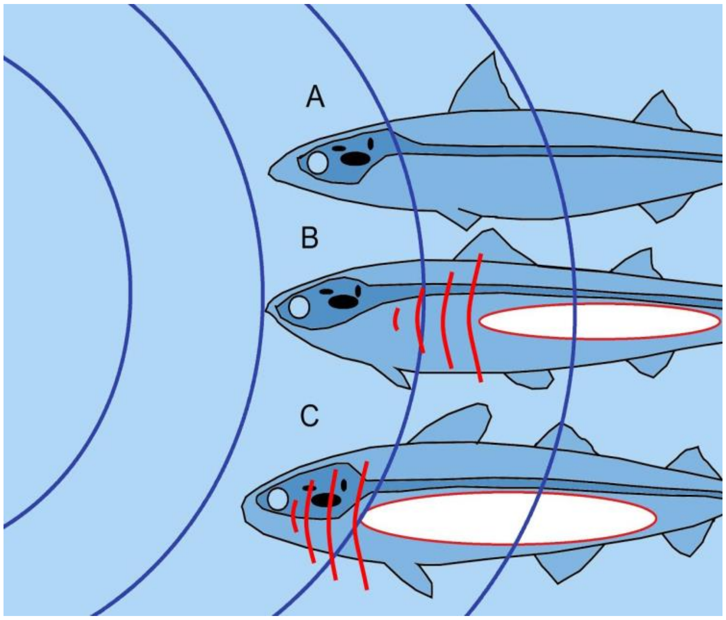

Figure 6: Stimulation of the ear of fishes.
The blue lines from the left represent the signal coming directly from the source while the red lines from the right in the Atlantic salmon (B) and Atlantic cod (C) represent the pressure signal re-radiated from the swim bladder as particle motion. The Atlantic mackerel (A) does not possess a swim bladder. In all three species, the direct particle motion stimulus will stimulate the inner ear. In the Atlantic salmon, the indirect particle motion signal re-radiated from the swim bladder attenuates sufficiently before it reaches the ear so that it (and the pressure signal it represents) cannot stimulate the ear. However, in the Atlantic cod, where the swim bladder extends to close to the ear, the particle motion signal is sufficiently high so that it causes motion of the otolith and thus is detected. As a result, the Atlantic cod is able to detect (indirectly) the sound pressure signal that stimulates the swim bladder, resulting in a wider hearing bandwidth and greater sensitivity than either the Atlantic mackerel or Atlantic salmon possess (see Figure 7). Figure ©2019 Anthony D. Hawkins, all rights reserved.

All teleost fishes and elasmobranchs directly detect the particle motion component of the sound field through the inner ear. However, in some teleost species, the swim bladders and other gas-filled organs act as acoustic transformers, converting sound pressure into particle motion (Figure 6). Incident sound pressures cause the compressible body of gas within the organ to pulsate, generating a much higher amplitude of particle motion than would otherwise be present (e.g., Alexander 1966). The locally produced high particle motion may be coupled directly to the otolith organs of the inner ear, as in the otophysan fishes (goldfish and relatives), as well as other species (e.g., Coombs and Popper 1982; McCormick and Popper 1984; Fletcher and Crawford 2001; Schulz-Mirbach et al. 2013), or may simply propagate through the surrounding tissues to stimulate the otolith organs (Sand and Hawkins 1973; Fay and Popper 1975; Popper et al. 2003).

Thus, in some fishes, in addition to receiving the particle motion directly from the source, parts of the otolith organs also receive indirect stimulation from these gas-filled organs. However, whether the fish actually detects this indirect particle motion depends on the proximity of the ear to the gas bubble, and many species that have a gas bubble still do not receive the indirect stimulation because it attenuates too much before reaching the ear. In those species where the indirect particle motion does stimulate the ear, the fishes hear higher frequency sounds than would be possible from reception of the direct particle motion alone, thereby expanding the frequency range detected and increasing the sensitivity of hearing so that the fish can hear lower intensity sounds (e.g., Fay and Popper 1974; Coombs and Popper 1979; Ladich and Fay 2013).

2. Hearing Capabilities of Fishes

a. The Determination of Auditory Thresholds

The term *auditory threshold*, sometimes termed the *hearing threshold*, refers to the hearing abilities of the animal and specifies the minimum level of sound that must be reached for a particular response to be produced, indicating that the sound has been detected. It is important to define the method of determining and statistically validating the response, whether it is behavioral or physiological. Auditory thresholds are measured by successively lowering and raising the level of the sound in precise steps (e.g., 6 dB) and determining which sound levels the animal can and cannot hear (e.g., Tavolga and Wodinsky 1963; Fay 1969). These values are combined statistically to provide a threshold. It is important to understand that auditory thresholds (as other sensory thresholds) constantly change depending on many factors, internal and external to the animal, and so the actual threshold is considered to be a statistical level, most often where the animal responds to the sound 50% of the time.

Most auditory thresholds are determined using pure tone sounds at different frequencies, although there are also thresholds determined for the minimal ability of an animal to discriminate between frequencies, sound directions, or sound intensities. The resultant graph of hearing threshold levels as a function of frequency is termed the *audiogram* (Figure 7). An *auditory threshold shift* is a change in the animal's auditory threshold level, essentially the difference between auditory thresholds measured before and after exposure to high-intensity acoustic stimuli. PTS is a permanent increase in the auditory threshold, while TTS is a temporary increase in the auditory threshold.

As described in Chapter 2 (page 25), many experiments to determine hearing thresholds in fishes have been conducted in small tanks and under less than satisfactory acoustic conditions. A few studies, however, have been done in highly specialized (and often very complex and expensive) tubular tanks designed so that the relative magnitudes of particle motion and sound pressure can be controlled (Fay and Popper 1974; Hawkins and MacLennan 1976), allowing investigators to vary the ratio of sound pressure to particle motion at the center of the tube.

Other hearing experiments on fishes have been carried out in midwater in the sea (reviewed by Hawkins 2014). There, the effects of reflecting boundaries are largely eliminated and, by changing the distance of the fish from the sound source, the ratio of sound pressure to particle motion can be varied, making it possible to determine whether the fish is sensitive to sound pressure or particle motion.

Because of the complexity of the sound fields in many small tanks, it is virtually impossible to describe the actual acoustic stimuli to which fish are responding, and so great care needs to be taken in interpreting most of the fish hearing data in the literature (Hawkins and Popper 2016; Popper and Hawkins 2018). For example, although the literature contains hearing thresholds identified in tanks for about 100 species (summarized in Ladich and Fay 2013), the vast majority of these thresholds are reported in terms of sound pressure, even when the fish studied primarily detects particle motion. In such experiments, the sound pressure measures are virtually useless for many fish species because it is not possible to predict the particle motion that elicited a response.

There are two general approaches to measuring the hearing capabilities of fishes. The earliest studies were done using behavioral methods in which fishes were “asked” what they could hear (e.g., Poggendorf 1952; Tavolga and Wodinsky 1963; Fay 1969; Hawkins and Chapman 1975), much as humans are asked what they can hear in an audiometric booth. In these studies, fishes were trained to respond in some way (e.g., by moving to a different location in a tank, changing heart rate or respiratory rate, or striking a paddle) whenever they heard a sound. Their behavioral responses reflected not only the sounds that reached the ear, but the processing of sounds by the brain.

An alternative approach involves measurement of the electric potentials in the eighth cranial nerve, which carries information from the ear to the brain, and/or the brainstem (lower portion of the brain) in response to a sound stimulus. The method, called auditory evoked potentials (AEPs), is similar to the tests given to newborn humans to ensure that the ear is functioning properly (Ladich and Fay 2013; Ladich and Schulz-Mirbach 2013).

Investigators have argued that AEP provides a valid indication of both hearing bandwidth (the range of frequencies an animal can detect) and hearing sensitivity threshold at each frequency detected. However, it has been demonstrated that, in fact, there is wide variability in hearing capabilities measured with AEP for fishes, even for the same species. Oftentimes, the largest differences are observed because studies were performed in different experimental setups (Ladich and Fay 2013; Sisneros et al. 2016). Moreover, because hearing capabilities measured with AEP do not include the processing of signals by the brain, such thresholds do not reflect the full hearing capabilities of fishes (Sisneros et al. 2016). Therefore, AEP “thresholds” are not comparable to auditory thresholds determined behaviorally. As a consequence, it is incorrect to use data derived from AEP studies for analyses of fish hearing or fish responses to sound where hearing capacities are being discussed (Hawkins and Popper 2016).

There are circumstances where AEP is a technique of choice for fishes, which include comparative studies where hearing capabilities are compared within a single species and within the same acoustic setup and where determining the lowest levels of hearing are not important (Sisneros et al. 2016). It is important to note, however, that these thresholds are not absolute, they are relative. An example would be in studies of potential hearing loss resulting from sound exposure, where the question is whether there has been a change in sensitivity before and after exposure to a sound, such as from pile driving (e.g., Popper et al. 2007; Halvorsen et al. 2009; Smith et al. 2011).

b. Hearing Capabilities

There has been a large number of studies on fish hearing capabilities, but many have been conducted under unsuitable acoustic conditions. Those experiments conducted under suitable conditions show that most fishes can detect sounds from below 50 Hz to over 500 Hz (e.g., Atlantic cod (Chapman and Hawkins 1973) (Figure 7), while additional species, like the Atlantic herring (*Clupea harengus*) (Enger 1967) can detect sounds to over 1,000 Hz (reviewed in Fay 1988; Ladich and Fay 2013). A few species, such as the otophysan fishes (e.g., goldfish and catfishes), with special adaptations peripheral to the inner ear that closely couple the ear to the swim bladder, can detect sounds to over 3 kHz and have better hearing sensitivity than other species. Other species, like the dab (Chapman and Sand 1974), and Atlantic salmon (Hawkins and Johnstone 1978) show more limited sensitivity and detect sounds over a narrower frequency range. These two species respond only to particle motion. Thus, there is substantial variability in hearing range and sensitivity among fishes that may reflect differences in the structure of their ears, the environment in which they live, and many other as yet unknown factors (e.g., Ladich 2014). The most important difference is between those species that detect sound pressure, and those that only detect particle motion (Popper and Hawkins, 2018).

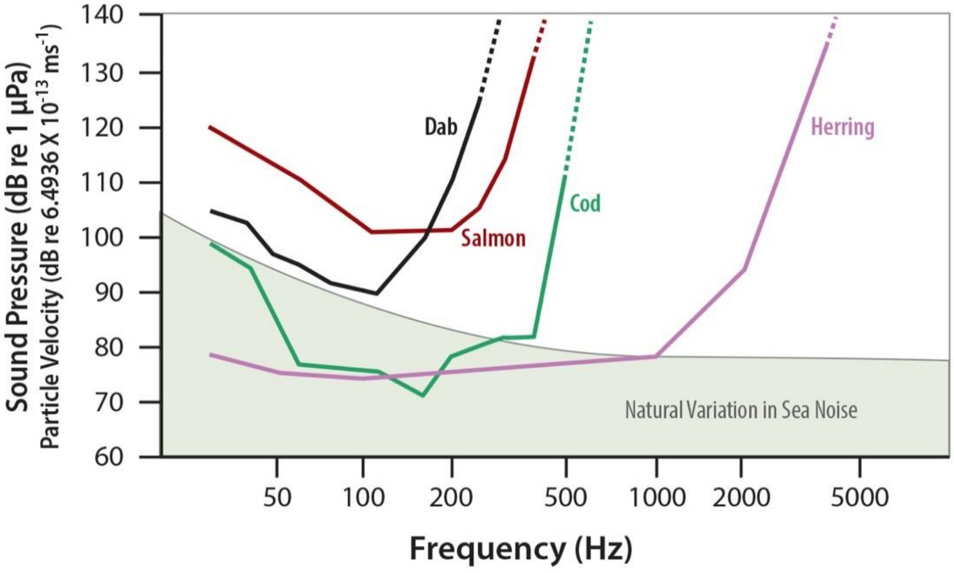

Figure 7: Fish audiograms obtained under open sea, free-field conditions.
The sound hearing thresholds were determined in response to pure tone stimuli at different frequencies. The lower the thresholds, the more sensitive the fish are to the sounds. Note that the absolute thresholds in the Atlantic cod and Atlantic herring may be below the natural ambient noise levels, especially at their most sensitive frequencies. However, when the fish are in the presence of such noise the thresholds are raised (see Figure 8, page 51). The Atlantic cod and Atlantic herring are sensitive to both sound pressure and particle motion, whereas the dab and Atlantic salmon are only sensitive to particle motion. The reference level for the particle velocity is based on the level that exists in a free sound field for the given sound pressure level.[16] Figure ©2019 Anthony D. Hawkins, all rights reserved.

However, many experiments that have examined the hearing of fishes, and their behavioral responses to sound, have been flawed, because the experiments were performed in small tanks, and some have used the AEP technique, which has the problems discussed above. The most valuable studies are those that were carried out under conditions where the sound fields were modeled and accurate measurements or estimates of both sound pressure and particle motion were made.

c. Generalizing Hearing Capabilities

Because it is impossible to determine hearing sensitivity for all of the more than 33,000 fish species (or even all of those that are listed in one place or another), one approach to understand hearing has been to distinguish fish groups on the basis of differences in their anatomy and what is known about hearing in other species with comparable anatomy. For example, in the 2014 Guidelines, Popper et al. (2014), suggested five groups, as shown in Table 2.

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[16] Note, for the particle velocity levels in this figure to match the sound pressure levels in a free sound field it is necessary to calculate an appropriate particle velocity reference level. If the standard reference levels are used, then the curves will not match one another and so they are not included here to keep the figure relatively simple.

Table 2: Grouping of Fishes as per 2014 Guidelines

1. Fishes lacking swim bladders that are sensitive only to sound particle motion and show sensitivity to only a narrow band of frequencies (e.g., flatfishes - Pleuronectiformes; and sharks skates and rays - Chondrichthyes).
2. Fishes with a swim bladder where that organ does not appear to play a role in hearing. These fish are sensitive only to particle motion and show sensitivity to only a narrow band of frequencies. This group includes salmonids (Salmonidae) and some tunas and mackerel (Scombridae), but many other species are likely to fit into this category as well.
3. Fishes with swim bladders that are close, but not intimately connected, to the ear. These fishes are sensitive to both particle motion and sound pressure, and show a more extended frequency range than groups 1 or 2, extending up to about 500 Hz. This group includes codfishes (Gadidae), eels (Anguillidae), some drums and croakers (Sciaenidae), and perhaps other fishes.
4. Fishes that have special structures mechanically linking the swim bladder to the ear. These fishes are primarily sensitive to sound pressure, although they also detect particle motion. They have a wider frequency range, extending to several kHz and generally show higher sensitivity to sound pressure than do fishes in groups 1, 2, or 3. The group includes some of the squirrelfishes (Holocentridae), drums and croakers (Sciaenidae), herrings (Clupeidae), and the large group of Otophysan fishes.
5. Eggs and larvae.

It cannot be assumed that fishes without swim bladders (Group 1), which only detect particle motion, are completely insensitive to sounds. Many elasmobranch species clearly detect and respond to underwater sounds (e.g., Myrberg 2001; Casper et al. 2012a). Indeed, there are circumstances in which the magnitudes of particle motion are much greater for a given sound pressure; for example, close to the water surface and in shallow water. As a consequence, it is important to take into account the acoustical habitats that fishes are occupying, and the possible conversion of sound pressure into particle motion, when assessing whether fishes can detect sounds from a particular source.

d. Masking and the Implications of Detecting Anthropogenic Sounds

Ambient sound (ambient noise) levels, including sounds from natural and anthropogenic sources, can affect the ability of any animal (or human) to detect biologically relevant sounds, including important parts of the acoustic scene. This interference with the detection of one sound (referred to as the signal) by another sound (referred to as the masker) is called *masking*, (see Fay and Megela Simmons 1999).

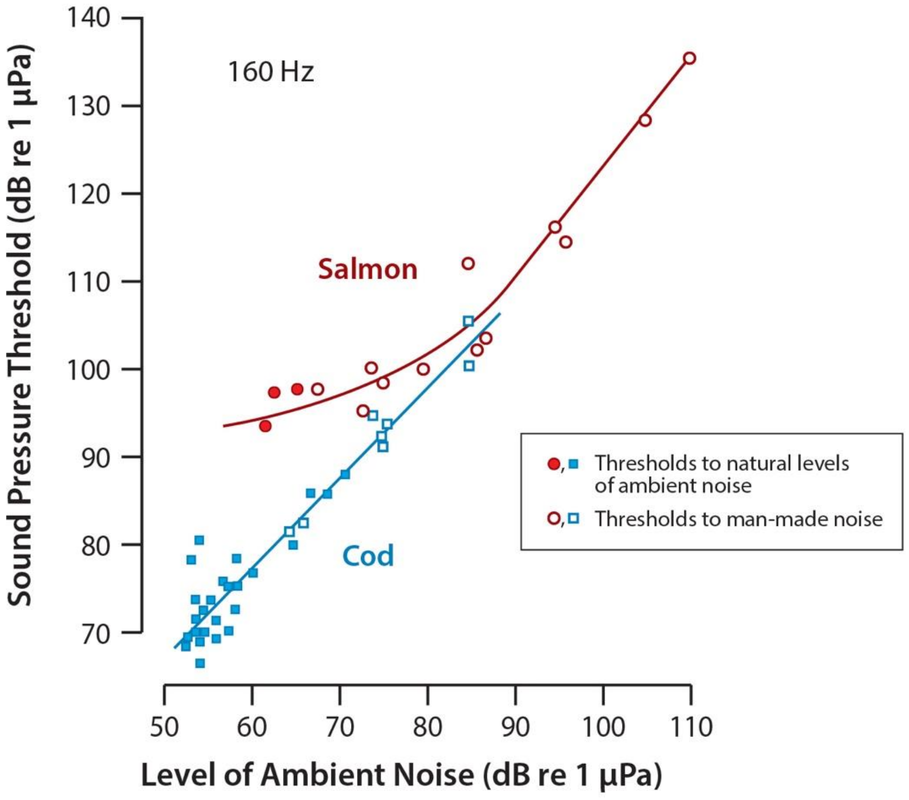

Figure 8: Masking hearing thresholds.
Masking in the Atlantic cod and Atlantic salmon by ambient noise. The thresholds were determined using a pure tone signal at a frequency of 160 Hz. The ambient noise (natural sea noise, augmented by white noise from a loudspeaker) is expressed as the spectrum level at that same frequency (dB re 1 µPa/Hz). The filled dots are thresholds to natural levels of ambient noise, while the open dots are thresholds to anthropogenic noise. Note that the thresholds in the Atlantic salmon were only influenced by high noise levels, above the natural ambient levels of noise. Figure ©2019 Anthony D. Hawkins, all rights reserved.

Fishes are certainly adapted to detect biologically important signals in the presence of natural ambient sounds (as are humans). At some frequencies, the lowest sound levels detectable by fishes are limited by background or ambient noise levels (Figures 7 and 8) (Hawkins and Chapman 1975). Any increase in the level of ambient sea noise results in a decline in sensitivity (i.e., poorer hearing). Thus, there is considerable concern about how much the presence of anthropogenic sounds (often termed noise), may mask sounds of importance to fishes (including the acoustic scene) and decrease hearing sensitivity (e.g., Popper et al. 2014; Hawkins et al. 2015). Figure 8 shows the effects of variations in the noise level on the detection of a 160 Hz pure tone stimulus for the Atlantic cod (Hawkins and Chapman 1975) and the Atlantic salmon (Hawkins and Johnstone 1978). As the spectrum level of the noise increases, the hearing thresholds increase (and sensitivity decreases) for both species. However, although the thresholds of the cod can be affected by variations in the natural levels of ambient noise, the thresholds in the salmon are rather higher, and tend to be affected only by levels of man-made noise (except under very noisy natural conditions).

Moreover, there is evidence that increases in ambient sound, in addition to generally impacting the detection of biologically relevant sounds, also affects the detection distances for sounds by a fish like the Atlantic cod. In effect, the presence of anthropogenic sound has the potential to affect the range at which sounds can be detected (Dooling and Popper 2016). If the range is decreased, the likelihood of detection of biologically important sounds decreases, preventing fishes from, for example, hearing an oncoming predator or finding distant prey.

e. Auditory Scene Analysis

One of the most important roles for sound detection in fishes (as in all animals), and a driving force in the evolution of hearing is for detection and analysis of what is referred to as the *auditory scene* (Bregman 1990; Fay and Popper 2000). The term refers to the perceptual scene, made up of all the sources that produce or scatter sounds that are detectable by an animal, including those produced by normal environmental conditions such as movement of water currents (abiotic sounds), those produced by other animals (biotic sounds), and those produced by anthropogenic sources. In addition, fishes, as all other vertebrate species, have the ability to segregate in perception many of the different, simultaneous sound sources that impinge on them to form this perceptual scene (Fay 2000; Fay and Popper 2000). It is assumed that animals perceiving these individual acoustic sources have a greater chance of behaving appropriately with respect to them, and, thus, a greater chance of survival. Part of the auditory scene is sounds of predators and of prey. These sounds enable animals to know the presence and position of the predator to permit escape and enable animals to locate food sources.

In addition, by performing auditory scene analysis, an animal can potentially perceive the soundscape of an environment as a collection of sources, thereby giving the animal critical information for environmental assessment, orientation, and navigation (e.g., Clark et al. 2009; Fay 2009; Slabbekoorn et al. 2010). Most importantly, any sounds that disrupt the ability of an animal (e.g., a fish) to detect and use the auditory scene has the potential of impacting the survival of the animal (Fay 2009).

f. Directional Hearing

Determination of the direction of a sound source, or sound source localization, is one of the most important aspects of hearing for all animals, including fishes (e.g., Fay and Popper 2000; Fay 2005; Heffner and Heffner 2016). Sound source localization enables a fish to move toward a food source or mate, or away from a potential predator by providing the ability to know the location of a source as well as its presence (Fay 2005). The mechanism for localization is based on the detection of particle motion signals that enable fishes to detect sounds in three dimensions, based on the orientation of the sensory hair cells of the inner ear (e.g., Sand and Bleckmann 2008; Hawkins and Popper 2018a; Popper and Hawkins 2018). Most importantly, as with scene analysis and masking, the presence of anthropogenic sound in the environment has the potential to impair detection of sounds, and, thus, the ability to perform sound source localization.

3. Hearing Loss in Fishes

a. Permanent Threshold Shift

Hearing loss in humans and other animals is often described as being a permanent or a temporary change in hearing threshold. PTS is a loss of hearing that never recovers. Most often, PTS is associated with the death of sensory hair cells in the ear and/or damage to the nerves innervating the ear (Liberman 2016). To date, there is no evidence of PTS in fishes, and it is considered unlikely to occur because fishes can replace any lost or damaged hair cells, precluding any permanent hearing loss (e.g., Smith 2016; Smith and Monroe 2016). It is also possible, however, that damage to the swim bladder or other organs involved in the detection of sounds might result in permanent changes to the hearing abilities of some fishes, although this would not be called PTS.

b. Temporary Threshold Shift

In contrast to PTS, TTS is a relatively short-lived reduction in hearing sensitivity due to changes in the sensory cells of the ear generally resulting from exposure to intense sounds for short periods of time, or somewhat longer exposures to lower sound levels sounds (e.g., Rabinowitz 2012; Finneran 2015). TTS has been demonstrated in some fish species, and the extent of TTS is of variable duration and magnitude (reviewed in Smith and Monroe 2016). Termination of a TTS-inducing sound leads to the return of normal hearing ability. The recovery period is variable and can range from less than 24 hours up to two weeks, depending on many factors, including the intensity and duration of exposure, the species involved, and the type of sound. (e.g., Popper and Clarke 1976; Scholik and Yan 2001; Scholik and Yan 2002b, 2002a; Amoser and Ladich 2003; Smith 2004; Smith et al. 2004b; Popper et al. 2005; Smith et al. 2006; Smith et al. 2011; Smith 2016; Smith and Monroe 2016).

Despite these data, there are substantial issues with regard to TTS in fishes (reviewed in Smith and Monroe 2016). Importantly, the majority of studies that have demonstrated TTS have involved species that have an adaption that enhances hearing sensitivity (Group 4 of Table 2, page 50) (e.g., Scholik and Yan 2002b; Amoser and Ladich 2003; Smith et al. 2004a; Popper et al. 2005). Studies on species without specializations (Groups 1–3 in Table 2) generally showed less hearing loss than fishes with specializations, or no hearing loss at all, even when fishes were exposed to very high sound pressure levels such as those from high-intensity, low-frequency sonars (Scholik and Yan 2002a; Popper et al. 2007; Halvorsen et al. 2013), seismic air guns (Popper et al. 2005), or lower level sounds imposed continuously over several months (Wysocki et al. 2007).

It is difficult to make broad generalizations about TTS for fishes because of the limited number of studies and because these studies used different species and sound sources. However, goldfish, a species that hears well and is in Group 4 of Table 2, were the subject of a number of the studies, and so Smith and Monroe (2016) were able to derive a number of generalizations on TTS based for that species. Although it is not known whether the same generalizations would apply to other species, the results for goldfish follow the same general trends in TTS found in the far more extensive studies on TTS in mammals (reviewed in Le Prell et al. 2012; Rabinowitz 2012), and so it is likely that such generalizations apply to all fish species.

Smith and Monroe (2016) point out that the level of TTS, measured in decibels (a log scale), increases with the duration of exposure to the same noise level, and the time for recovery increases as the duration of exposure increases. There is also evidence that TTS increases as the sound level of the stimulus increases (Smith et al. 2004a) and that the greatest level of TTS occurs when the noise source is at the frequency of best hearing in the goldfish (Smith et al. 2004a; Smith et al. 2006; Smith et al. 2011).

These and other results from goldfish studies led Smith and colleagues (Smith et al. 2004b; Smith 2012, 2016; Smith and Monroe 2016) to suggest that TTS in fishes only occurs when the potentially damaging sound is at least 60 dB above the auditory threshold at the frequency of exposure for an extended period of time. As a consequence, they propose that the level of sound needed to induce TTS in a species is related to the auditory threshold of that species.

Interestingly, in the one study with very intense simulated pile driving sounds, Casper et al. (2013b) demonstrated that damage to sensory hair cells, the precursor to temporary hearing loss (Smith et al. 2006; Smith et al. 2011), only occurred for sound exposure levels that were substantially above the levels that resulted in damage to other body tissues in hybrid striped bass. However, there was no effect on the hair cells in hybrid striped bass and only minor damage to the sensory cells of the Mozambique tilapia (Chapter 7, page 81).

One critical issue in evaluating the potential for TTS on wild animals is the stimulus that produces TTS. For some fishes that have enhanced sensitivity, such as the goldfish, the causal factor for TTS may be related to sound pressure. But, for those species that are primarily particle motion detectors, TTS is likely related to the direct stimulation of the ear by the particle motion. Thus, any suggestion that a particular sound pressure is related to TTS for all fishes is likely erroneous for some species.

The question that also arises is whether fishes that are primarily particle motion detectors will show TTS (or inner ear damage) in the wild. Much depends upon the level of particle motion at the position of the fish, the time over which the fish is exposed to the particle motion, and the level of particle motion above hearing threshold needed to produce TTS. Based on the data from Smith et al. (2006; 2011) for fishes that detect sound pressure, a signal needs to be at least 60 dB above the auditory threshold for an extended period of time (hours) to induce TTS. Although it is important to extrapolate with caution, one might suggest that the received particle motion signal must be well above the particle motion threshold for particle motion to induce TTS in fishes that only detect that component of the sound signal.

At the same time, most TTS data for fishes are for signals that are of longer duration than those produced by pile driving. The only data for TTS and impulsive sounds are from one study of effects on hearing resulting from exposure to seismic air guns (Popper et al. 2005). This study, which exposed several species to 5 or 20 seismic shots, induced TTS in the lake chub, a species that has hearing specializations similar to those of the goldfish and in adult (but not young-of-the-year) northern pike. There was no TTS in a salmonid, nor the broad whitefish. Moreover, in both species that showed TTS, full recovery took less than 24 hours, and there was no damage to the tissues of the inner ears (Song et al. 2008).

c. Onset of Masking and TTS

In developing the 2014 Guidelines for the effects of sound on fishes, Popper et al. (2014) defined onset of TTS in fishes as any reduction in hearing of 6 dB (change of 50%) or greater that persists for 24 hours or longer. The intent of the definition was to recognize that changes in hearing levels less than 6 dB are generally difficult to differentiate because of the experimental approaches used to measure hearing sensitivity. At the same time, the 2014 Guidelines defined masking in fishes to be the impairment of the ability to detect sounds, including the auditory scene, by greater than 6 dB and lasting more than 30 seconds. This level was chosen because it is unlikely that a change of less than 6 dB, or a brief episode of masking, results in a significant effect. As an aside, recent NOAA Fisheries guidelines for marine mammals also uses a change of hearing sensitivity of 6 dB as the onset level for TTS (NMFS 2018).

CHAPTER 6: EFFECTS ON BEHAVIOR

1. Fish Behavior that May be Affected by Anthropogenic Sounds

Before considering specifics about behavioral responses to anthropogenic sound, broad issues that have a substantial impact on understanding of behavior need to be raised. For example, Hawkins and Popper (2018a) reviewed the effects of anthropogenic sounds on fishes and pointed out that sound provides fishes with a means of rapid, directional, and long-distance communication. It also provides animals with a gestalt view of their environment by giving an acoustic image of the world that extends far beyond what is available from other senses. Many fishes use sound to: communicate with one another; identify other individual fish; gather together in shoals; detect prey and predators; orient themselves with respect to the local environment; navigate from one place to another; and select appropriate habitats. Any interference from the presence of other, anthropogenic, sounds with fishes' ability to use natural sounds has the potential to compromise the fitness of fishes (Slabbekoorn et al. 2010; Hawkins and Popper 2014).

As NOAA (2016) points out in its *Ocean Noise Strategy Roadmap,* studies on fishes have focused more on characterizing the physical effects than on potential impacts on behavior. However, physical effects are likely to result only when fishes are very close to a source, whereas behavioral effects may occur even when fishes are at far greater distances from a sound source. Thus, far more animals may show behavioral effects than show physical or physiological effects as a result of exposure to sounds. Fish may show distinct changes in their behavior as a result of exposure to any audible sounds.

A significant issue that has to be kept in mind in assessing potential effects on behavior is that different behavioral responses may occur, depending on the level of the sound, the level of ambient background sound, what the fish are doing at the time of the sound, and their previous experience with the same and other sounds. Whether or not a fish responds may also depend on its condition, motivational state, and the presence of other animals including predators (De Robertis and Handegard 2013). Lucke et al. (2016) emphasized that the internal state, motivation, context, and previous experience of animals affected their behavioral responses and also influenced the long-term and cumulative effects of sound exposure. Thus, if we are to better understand the sensitivity of fishes to sound, there is a particular need for studies that examine variation in levels of behavioral response in parallel with detailed characterization of the sound fields, ideally using a variety of different sound measurement metrics to ascertain which aspects of the sounds are most important.

A very important consideration is that whatever guidelines are set for behavior, the wide variation in fish behavior and how animals respond at a particular time and under particular conditions make it truly impossible for there to be one signal level that indicates onset of behavioral effect. Indeed, a signal level that initiates a behavioral response in one species at one point in time may not produce the same response in the same species (or other species) at another time or place.

Moreover, it is also important to realize that, although not a focus of this report, the invertebrate prey of fishes may also be affected by underwater sounds. Invertebrates play a central role in marine food webs and in providing ecosystem services (Morley et al. 2014). Playback studies on invertebrates have shown that the effects of anthropogenic sound fields on functionally important species in lower trophic levels have the potential to be substantial (Solan et al. 2016). The exclusion of invertebrates from impact assessments is likely to lead to an under-appreciation of the effects of anthropogenic sound on offshore marine environments and fishes.

Finally, it is also important to note that although much of the focus of concern is on marine species, the same issues arise for freshwater fishes, although there are far fewer studies (Linke et al. 2018; Mickle and Higgs 2018). The acoustic landscape of marine vs. freshwater environments differs quite markedly because of differences in water depth and closeness of surface and bottom. Freshwater environments may be less efficient at sound transmission than marine environments; however, freshwater environments harbor a wide range of species that are important from a conservation perspective. Thus, it is important to assess the effects of pile driving on the behavior of freshwater fishes as well as marine fishes.

a. Levels of Sound Perception

Of particular interest in considering the impact of noise on aquatic animals is the effect of anthropogenic sounds on their ability to detect natural sounds that are biologically important to them. In these situations, unlike masking *per se*, animals not only need to be able to detect sounds (something that is affected by the presence of masking sounds), but they also need to be able to hear the sounds at levels that make the sounds meaningful to the animal. Thus, Dooling and colleagues have defined several levels of sound perception (Dooling and Blumenrath 2016). Although the initial concept was developed for birds, the ideas are applicable across all animals and need to be taken into consideration when trying to evaluate effects of sound on behavior (Dooling et al. 2015; Dooling and Popper 2016). In the following list, the sound of interest is successively louder at the receiving animal.

- **Detection**: the sound of interest is barely audible about the background noise. The animal may or may not respond to this sound because it is barely detectable.

- **Discrimination**: a level that is above that for just detecting a change (often referred to as the Just Noticeable Difference – JND), and there is sufficient information in the sound so that the animal can discriminate between sounds, such as those from predators and prey.

- **Localization**: the animal is able to determine the direction from which the sounds are coming and perhaps its distance from the source, both of which may be important for responding to prey or predators and for orientation and navigation.

- **Recognition:** the animal can identify the sound and decide how significant it might be; that is, the animal can recognize the nature and context of the sound.

2. Recent Studies on Behavioral Responses to Sound

There is a wide range of potential effects of anthropogenic sounds on fishes, just as there are on terrestrial animals (Slabbekoorn et al. 2010). For example, Weilgart (2017) reviewed the impact of effects of anthropogenic sound on fishes and invertebrates. She concluded that behaviorally the animals showed alarm responses, increased aggression, hiding, and flight reactions; and decreased anti-predator defense, nest digging, nest care, courtship calls, spawning, egg clutches, and feeding. The schooling of fishes became uncoordinated, disaggregated, and unstructured due to noise.

In the following sections, we review recent literature that has the potential to provide insight into fish responses to sound. At the same time, it is important to note that none of the results to date, with perhaps the exception of a field study by Hawkins et al. (2014), provides sufficient data to inform future guidelines. Most of the other work gives insights into fish responses to sound that are qualitatively instructive, and which may provide ideas for future studies that have the potential to inform guidance. Additional studies are described in the Appendix (page **Error! Bookmark not defined.**).

a. Responses to Anthropogenic Sounds

It is generally assumed that fishes that are more sensitive to sounds, fishes with better hearing abilities, are more likely to show behavioral responses to sounds than less sensitive species. However, a number of studies have suggested that this may not always be the case. Shafiei Sabet et al. (2016) compared laboratory responses of zebrafish and Lake Victoria cichlids (*Haplochromis piceatus*) and found that zebrafish have better hearing sensitivity (lower auditory thresholds) and a wider frequency range than the cichlid. Both species exhibited a significant reduction in swimming speed in the first minute of exposure. The results showed that sound exposure could cause both similar and species-specific responses in two fish species and that the responses were not obviously related to differences in their hearing abilities.

This finding was confirmed by Hawkins et al. (2014), who observed the behavior of wild schools of sprat (*Sprattus sprattus*) and mackerel (*Scomber scombrus*) to sound playback. The sound pressure levels to which the fish schools responded were remarkably similar for both species, despite major differences in their hearing abilities.

b. Foraging Behavior

While searching for food, fish may listen for the incidental sounds made by their prey. There is also evidence that at least one species of marine catfish (*Arius felis*) uses a form of *echolocation* to identify objects in its environment by producing low-frequency sounds and listening to their reflections from objects (Tavolga 1977). The presence of anthropogenic sounds may interfere with foraging behavior, either by masking the relevant sounds or by resembling the sounds that the prey may generate. In addition, a small number of laboratory experimental studies has investigated how foraging behavior is affected by exposure to increased noise levels (Purser and Radford 2011). These data lead to the idea that additional noise in the underwater environment can lead to reduced food consumption, although the effects are likely to be species specific (for more details of these publications, see the Appendix). Unfortunately, these studies were all conducted on captive fish under restricted conditions, and there is a need for similar work on wild fishes under more natural conditions.

c. Predator Avoidance

Fishes may avoid predators by listening for the sounds that the predators produce, either deliberately or inadvertently. Some fishes can detect the specific acoustic signals of marine mammals and have also been shown to react to the playback of such signals by reducing the production of their own sounds (Luczkovich et al. 2000; Remage-Healey and Bass 2006). Noise playback can also affect the responses of fishes to visual predatory stimuli, although elevated noise levels affect the anti-predator behavior of different species in different ways (Voellmy et al. 2014b). These experimental studies (for details, see the Appendix) have shown that elevated sound levels, including intermittent or pulsed sounds, may affect predator prey interactions. Again, however, the experiments cited above were all carried out on captive fish in enclosed environments; there is a need to examine the behavior of wild fishes under more natural conditions.

Boat noise also could affect the predator responses of wild fishes. Jain-Schlaepfer et al. (2018) showed that noise produced from small motorboats impacted the behavior of juvenile damselfish by affecting the way they assessed risk and their ability to detect and avoid a predator strike. The investigators suggested that this would have a marked impact on individual fitness and survival. Holmes et al. (2017) investigated the effect of playback of boat noise on fish cognition (the ability of individuals to learn and remember information). Fish exposed to boat noise playback failed to subsequently respond to a predator, while fish exposed to ambient reef noise responded appropriately. It is evident from these studies that anthropogenic sound can affect predator avoidance by wild coral reef fishes.

d. Migrations and Home Ranges

Many fishes migrate to feeding areas, or spawning grounds, and may subsequently return to other locations. During passage through the sea, fishes like salmon may use a variety of cues to orientate and navigate, including natural soundscapes, the Earth’s magnetic field, water currents, and the presence of chemical cues. Acoustic cues are generated naturally by wind, sediment transport, rainfall, surf, long-period waves, geological and meteorological processes, and probably from turbulence associated with currents (e.g., Tonolla et al. 2010; Tonolla et al. 2011). Marine animals may navigate during their migrations using the surrounding soundscape.

Alterations to these cues may have adverse effects on migratory fish. High level sounds may result in avoidance responses, deflecting fish away from their migration routes. Studies have suggested other possible negative impacts on migrating fishes. Bagočius (2015) suggested that migrating Atlantic salmon would be adversely affected by pile driving noise. Montgomery et al. (2006) suggested that the ability of larval reef fishes and decapod crustaceans to locate their home reefs by responding to their characteristic sounds might be affected by changes in the noise level from anthropogenic sources. Stanley et al. (2012) pointed out that sound is a useful cue for guiding the orientation of larvae because it travels long distances underwater. Sound also has the potential to convey valuable information about the quality and type of the habitat at the source. There are significant differences in the spectral and temporal composition of the ambient sound associated with different coastal habitat types (reviewed by Radford et al. (2010). Gordon et al. (2018) have recently pointed out that changes in habitats may negatively affect the auditory settlement behavior of coral reef fishes. Acoustic cues guide the orientation, habitat selection, and settlement of many fishes, but these processes may be impaired if degradation alters reef soundscapes.

Some demersal fishes and other aquatic animals spend much of their time within particular areas of the seabed, termed home ranges, where they are familiar with the presence of prey, and may be able to shelter from predators. The presence of high levels of anthropogenic sound close to their home ranges may result in the fish being displaced, with detrimental effects with respect to foraging and predator avoidance, and perhaps also with respect to their reproductive behavior.

e. Sound Production by Fishes

Many fishes make sounds, especially during their reproductive behavior (e.g., Hawkins and Myrberg 1983; Bass and Ladich 2008; Radford et al. 2014; Ladich and Winkler 2017). Sounds usually have most of their energy below 1,000 Hz and are characterized by a pattern of amplitude modulation. Additional information on the sounds made by fishes is provided in the Appendix.

Many commercially important fish species that NOAA Fisheries is charged with managing produce sounds or are known to use sounds during critical life stages (NMFS 2016). These include Atlantic cod and haddock (*Melanogrammus aeglefinus*) (Family Gadidae), members of the snapper-grouper complex (e.g., families Serranidae and Lutjanidae), grunts (family Haemulidae), and croakers and drums (family Sciaenidae), among other species (Normandeau 2012; Hawkins et al. 2015). Interference with detection of these sounds by anthropogenic sources has the potential to compromise reproductive activities and, thus, species fitness.

f. Sounds and Spawning

Sounds are also important for many fish species for spawning. In particular, any interference with detection of such sounds can have a significant impact on reproductive success of a population. Further details of fish spawning behavior, and the role of sound production is provided in the Appendix.

By way of example, male haddock, make repetitive low-frequency sounds during their reproductive behavior. Observations by Casaretto et al. (2015) showed that male haddock were territorial and that visits to their territories by females, induced by the sounds of males, triggered courtship behavior leading to the spawning embrace. Similarly, vocal aggregations of male Atlantic cod have been recorded off the Norwegian Lofoten Islands (Nordeide and Kjellsby 1999). Disruption of spawning by anthropogenic sounds might lead to additional time and effort by males to re-establish territories and by females to repeat the process of choosing their mates. Females might have to travel farther or search longer to find males. The overall effect might be to reduce reproductive success. It has been suggested by de Jong et al. (2017) that acoustic communication often plays a crucial role in reproductive interactions, and they point out that more than 800 species of fish have been found to communicate acoustically.

Soniferous coral reef fishes may aggregate at particular locations (e.g., McWilliam et al. (2017), and such fishes may be very susceptible to changes in environmental conditions, including changes to the ambient noise induced by anthropogenic sounds. Anthropogenic sounds may also interfere with soniferous coral reef fishes' mating behavior, either by masking the sounds made by the fish or by resembling those sounds.

g. Interference with Fish Communication

Fish vocalizations are an important component of the marine soundscape, and they provide valuable information regarding the behavior of the signaler in a variety of different contexts, such as general interactions, territorial displays, feeding, contact vocalization, and courtship interactions. Some species vocalize at key life stages or while foraging, and disruption to the acoustic habitat at these times could lead to adverse consequences at the population level (Putland et al. 2018).

Lugli (2010) pointed out that the habitat ambient noise may exert an important selective pressure on frequencies used in acoustic communication by animals. The analysis of the ambient noise/sound spectrum relationships showed the sound frequencies of the fish calls matched the frequency band of the quiet window in the ambient noise typical of their own habitat. Any generation of anthropogenic noise within these quiet windows may greatly reduce the ability of fish to communicate with one another. Vasconcelos and Ladich (2008) investigated the effects of ship noise on the detectability of communication signals in the Lusitanian toadfish (*Halobatrachus didactylus*). It was concluded, albeit from experiments in a laboratory tank, that acoustic communication, which is essential during agonistic encounters and mate attraction, might be restricted in coastal environments that are altered by human activities.

A study by Stanley et al. (2017) investigated the alteration of estimated effective communication spaces at three spawning locations for populations of commercially and ecologically important fishes. They pointed out that high levels of low-frequency noise could reduce the communication space at spawning sites during times of high vocalization activity.

Fuller details of the above studies are provided in the Appendix.

h. The Masking of Biologically Significant Sounds

As discussed in Chapter 5, any increase in the level of ambient sea noise results in a decline of fish sensitivity to sounds (also see the Appendix).

In terms of the behavioral impact of masking, it is likely that high levels of low-frequency sound generated by ships, pile drivers, air-guns, low-frequency sonars, and other sources can interfere with fishes' detection of the overall acoustic scene (or soundscape) and affect communication by means of sound in fish, both through masking biologically significant sounds that overlap in time and frequency and through the generation of signals that are similar to those produced by the fish themselves (e.g., Kaplan et al. 2015; Kaplan et al. 2016; Pine et al. 2016). Anthropogenic sounds can disrupt their lives significantly by preventing them from hearing approaching predators such a seals, dolphins, and otters, by preventing them from detecting the sounds of their prey, or by generating sounds that the fish may mistakenly believe are calls from other fish, thereby disrupting mating behavior. Many fish calls are repeated low-frequency pulses, similar to those generated at a distance by sources like pile drivers and seismic airguns.

In addition to impact on hearing sensitivity *per se*, there is also limited evidence that anthropogenic sounds will result in fishes altering their own sounds to avoid masking. For example, Radford et al. (2014) reviewed how acoustic signaling fish might respond to the presence of anthropogenic noise and concluded that there was evidence that some species of fish might have the potential to compete with anthropogenic noise by making changes in their calls. Similarly, Holt and Johnstone (2014, 2015) investigated what effects elevated noise levels had upon a sound-producing freshwater fish, the blacktail shiner (*Cyprinella venusta*) in tanks. When elevated levels of natural river noise were played back to the fish, it was found that several acoustic features of the fish calls were altered under noisy conditions. Most notable the spectral composition of the calls was altered by the fish (termed the Lombard effect).

3. Observed Effects on the Behavior of Fishes from Impulsive Sound Sources

Many anthropogenic sound sources are continuous, such as the sounds generated by ships. However, also important are the sounds produced by impulsive sources. These produce sounds that are typically transient, brief (less than 1 second[17]), broadband, and consist of high peak sound pressure (SPL_{peak}) with rapid rise time and rapid decay.

In considering effects of impulsive sounds on the behavior of fishes it is also important to distinguish between fishes that live close to the seabed, that is, demersal fishes, and those that occupy mid-water habitats, the pelagic fishes. This distinction between them is based on the fact that fishes on or close to the seabed are likely to be influenced by particle motion transmitted by way of the substrate, and particularly sounds propagated as interface waves or ground roll (Hazelwood 2012; Bruns et al. 2016; Hazelwood and Macey 2016a). Impulsive sources such as pile drivers and seismic airguns can generate substrate waves that may travel great distances, especially at very low frequencies. Transmission of sound through the seabed and at the water/ground interface may especially affect bottom-living fishes and their invertebrate prey. Pelagic fishes are much less likely to be influenced by substrate-borne waves, and, for such species, measurements of sound pressure in midwater may be sufficient to predict the particle motion levels.

a. Pile Driving

There have been several recent experimental studies on the behavioral responses of fishes to pile driving. Many of them have been conducted on captive fish maintained in confined spaces, but a few studies have also been conducted on fishes in the wild.

i. Pelagic Species

Hawkins et al. (2014) observed the behavior of wild, pelagic fish in response to sound playback using a sonar/echo sounder. Schools of sprat and mackerel were examined in mid water

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[17] There is no agreement as to the actual duration of an impulsive sound in the literature. NMFS (2016) suggests less than 1 second, but other sources suggest as little as 220 milliseconds and still others up to 10 second. For purposes of this report we use 1 second, with the suggestion that at some point there needs to be a standard definition for impulsive underwater sounds.

at a quiet coastal location. As shown by Blaxter et al. (1981), the sprat, as other members of the same taxonomic family, have inner ear specializations enabling it to detect sound pressure. In contrast, the Atlantic mackerel lacks a swim bladder and so it is likely that the mackerel is only sensitive to particle motion (see Figure 6, page 49).

The fish were exposed to short sequences of repeated impulsive sounds, simulating the strikes from a pile driver, at different sound levels. The incidence of behavioral responses increased with increasing sound level. Sprat schools were more likely to disperse or change density (Figure 9, page 63), and mackerel schools more likely to change depth with some scattering of individual fish. The sound pressure levels to which the fish schools responded on 50% of presentations were 163.2 and 163.3 dB re 1 µPa peak-to-peak measured at the location of the schools, and the single strike sound exposure levels ($SEL_{ss)}$ were 135.0 and 142.0 dB re 1 $\mu Pa^{2\cdot}$ s, for sprat and mackerel, respectively, estimated from dose response curves.

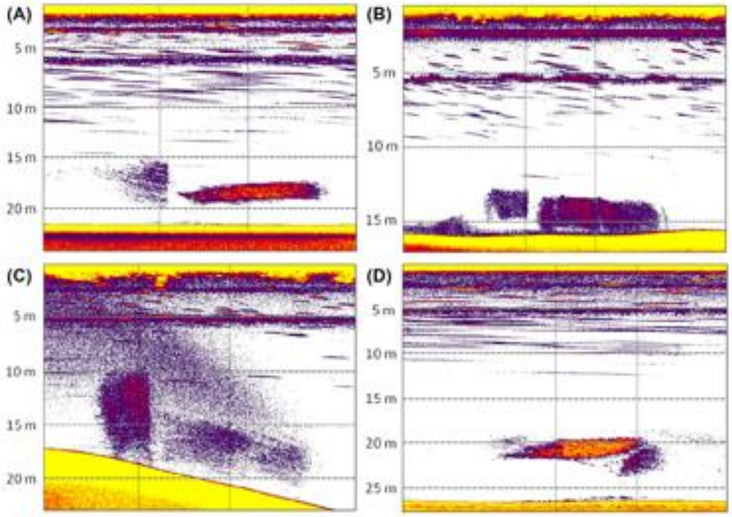

Figure 9: Echograms, showing typical responses of sprat schools to sound exposure.
(A) A medium sized sprat school cut off abruptly after the beginning of the sound and reappearing a few seconds later as a denser school slightly closer to the seabed. (B) A medium sized sprat school cut off at the onset of the sound and reappearing seconds later slightly closer to the seabed. (C) A large sprat school cut off at the onset of the sound and reappearing at a greater depth at lower density. (D) A small sprat school increasing in density in response to sound exposure. A vertical line indicates the beginning and end of each pile driving sound (from Hawkins et al. 2014)

Because the mackerel is likely to respond to particle motion, rather than sound pressure, the sound levels to which 50% of mackerel schools responded was reached at a peak-to-peak particle velocity level of -80.4 dB re 1 m/s.[18]

The response levels reported by Hawkins et al. (2014) suggested that sprats, mackerel, and zooplankton would show changes in their behavior at considerable distances — many kilometers — from a pile driving operation. However, the responses of sprat at night were very different to

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[18] The velocity level corresponds to 96 dB re 1 nm/s, the current standard reference for particle velocity in water

those shown during the day. Sprat schools break up at night, and the individual fish did not respond to the playback of pile driving sounds at that time.

Despite major differences in their hearing abilities the sprat and mackerel responded to impulsive sounds at similar sound levels during the daytime playback experiments. This response may be the result of mackerel being readier to respond to any stimulus — observations suggested that they were perhaps “flightier” than the sprat. It is also interesting that aggregations of zooplankton responded to similar sound levels, although they showed only limited short-lived changes in depth. In addition, when the sounds were presented to dispersed sprats, at night, no response was observed. The results suggest that the motivational state of the animals, and their actual activities, may be just as important in determining whether they respond as the level of sound they receive.

It was evident in observing the responses to sound by zooplankton layers and by sprat and mackerel schools that the responses occurred soon after onset of the sound, after a brief latent period. Essentially the animals were responding to receipt of the initial sound pulses. It was not clear, however, which of the characteristics of the impulsive sounds were especially important in evoking the responses. The peak excursion in sound level, the rise time, and the total energy in the pulse may all have played a role. Further experiments are required to elucidate the relative importance of these different parameters in evoking responses from fish. It is also important to determine whether the fish habituate to repeated sounds.

ii. Demersal Species

There have been a number of studies of the response of captive demersal species to pile driving sounds. However, some studies have been carried out with very poor controls, while others were carried out under poor acoustic conditions. Some studies are reported briefly here but others are outlined in the Appendix.

Studies on captive demersal fishes were carried out by Neo et al. (2014; 2015; 2016, 2018). In addition to showing that that intermittent sounds may yield longer-lasting behavioral effects than do continuous sounds (Neo et al. 2014), there was also an indication that amplitude fluctuations and pulse rate intervals might have subtle effects on the kind and intensity of a response (Neo et al. 2015). Moreover, ramp-up procedures do not necessarily lead to mitigation (Neo et al. 2016). At the same time, it must be kept in mind that these studies were done in enclosures that did not resemble natural acoustic environments and that many of the studies were done with zebrafish, a species that is small, thrives in small tanks, and which hears far better than most (if not all) species likely to be encountered in a pile driving operation.

In a study of the response of European sea bass (*Dicentrarchus labrax*) to impulsive sounds, Neo et al. (2018) confirmed that the potential effects of anthropogenic noise may be context-dependent and vary with the time of the day. Groups of sea bass in an outdoor pen were exposed to a series of eight repeated impulsive sounds over the course of two days at variable times of day and night. Their findings suggest that the impact of impulsive anthropogenic noise may be stronger at night than during the day for some fishes. Moreover, their results also suggest that habituation should be taken into account for sound impact assessments and potential mitigating measures.

Kastelein et al. (2015, 2017) determined acoustic dose response relationships for behavioral responses to the playback of pile driving sounds by the European sea bass in a netting enclosure within a shallow pool. Initial responses included sudden, short-lived changes in swimming speed and direction. Sustained responses including changes in school cohesion, swimming depth, and speed were also observed and quantified. The 50% initial response threshold occurred at an SEL_{ss} of 131 dB re 1 $\mu\text{Pa}^2\cdot\text{s}$ for 31-centimeter (cm) fish and 141 dB re 1 $\mu\text{Pa}^2\cdot\text{s}$ for 44 cm fish. Thus, the small fish were acoustically more sensitive than were the large fish. Kastelein et al. concluded that, if wild sea bass were exposed to pile driving sounds at the levels used in the study, there were unlikely to be any adverse effects on their ecology, because their initial responses were short-lived. However, the experiments were carried out on fish that had spent their whole lives in captivity within a very shallow rectangular pool, where the sound field was nothing like that in the wild.

Bruintjes et al. (2016a) showed that European sea bass in both laboratory and open-water conditions increased ventilation rate, as a measure of oxygen consumption, during playback of pile driving sounds. In the open-water experiments, the fish was placed in a glass jar (said to be acoustically transparent for low-frequency wavelengths). The ventilation rate of sea bass was significantly affected by the playback of pile driving sounds. The fish showed rapid recovery following acute short-term exposure once the acoustic disturbance ceased. In the open-water experiments during piling-noise playbacks, the peak values measured were around 200 dB re 1μPa.

Bruintjes et al. (2016b) also investigated the impact of pile driving on the oxygen uptake (a secondary stress response) of black sea bream and European plaice (*Pleuronectes platessa*) using an experimental pile driver set up in a flooded ship-building dock. Simulated impact pile driving was produced by a post driver with a 200-kilogram hammer striking a steel pipe. The hammer struck the post 10 times per minute. During 30-minute pile driving exposure, individuals were subjected to a SEL_{cum} of 184.41 dB re 1 $\mu\text{Pa}^2\cdot\text{s}$. During 30-minute ambient control conditions, the fish were exposed to a SEL_{cum} of 159.33 dB re 1 $\mu\text{Pa}^2\cdot\text{s}$. During pile driving, black sea bream increased oxygen uptake compared with the ambient control conditions. Plaice did not show differences in oxygen consumption between the pile driving and ambient sound treatment. The results underlined the importance of collecting species-specific data.

In a more detailed series of subsequent experiments on laboratory-bred European sea bass, Radford et al. (2016a) exposed juvenile fish to playbacks of pile driving sounds and seismic sounds in laboratory-based studies intended to examine how an initial impact of different sound types potentially changes over time. Only sound pressure measurements were made. The fish ventilation rates, which were thought to be an indicator of stress, were measured as the opercular beat rate (OBR). Pile driving sounds were played back at a peak level of 163.31 dB re 1 μPa (and an SEL_{ss} level of 147.40 dB re 1 $\mu\text{Pa}^2\cdot\text{s}$). Naïve fish showed elevated ventilation rates, indicating heightened stress, in response to impulsive additional noise (playbacks of recordings of pile-driving and seismic surveys), but not to a more continuous additional noise source (playbacks of recordings of ship noise). However, fish exposed to playbacks of pile-driving or seismic noise for 12 weeks no longer responded with an elevated ventilation rate to the same noise type. Fish exposed long-term to playback of pile-driving noise also no longer responded to short-term playback of seismic noise. Based on the results, the authors suggested that the lessened response after repeated exposure to pile driving sound was due to increased tolerance (habituation), or a shift in hearing threshold (TTS or PTS) following initial exposure. Habituation is a learned reduction in response to a stimulus as the organisms realize that it does not have detrimental consequences, whereas a shift in hearing threshold may indicate damage to the auditory system. It was concluded that considering how responses to anthropogenic noise change with repeated exposure is important both when assessing likely fitness consequences and the need for mitigation measures.

It is important to note that there are both behavioral and acoustic limitations to such tank-based playback experiments, including that the speakers do not generate sound in the lowest frequency ranges, that experiments are conducted in the near field, and that the sound field will differ from that in open-water conditions.

Debusschere et al. (2014) found no mortality in juvenile European sea bass following exposure to pile driving sounds in a series of *in situ* experiments. They then examined stress responses in young European sea bass (Debusschere et al. 2016a; Debusschere et al. 2016b), as discussed in Chapter 8.

Iafrate et al. (2016) used acoustic telemetry to assess the movement, residency, and survival of free-living sheepshead (*Archosargus probatocephalus*) and grey snapper (*Lutjanus griseus*) in response to 35 days of pile driving at a wharf complex during an actual pile driving operation. These results are far more valuable than those produced under laboratory conditions with sound playback. Received sound pressure levels from pile strikes on the interior of the wharf, where reef fish primarily occurred, were on average 152–157 dB re 1μPa (peak). No obvious signs of mortality or injury to tagged fish were evident. Sheepshead showed higher site fidelity to the wharf within which they lived prior to pile driving than did grey snapper. No significant decrease in sheepshead daytime residency was observed during pile driving within the central portion of the wharf and area of highest sound exposure, and no major indicators of displacement from the exposure wharf with the onset of pile driving were observed. Of the 13 sheepshead present at the start of pile driving, only one fish left the area after 10 days of pile driving. However, with grey snapper there was evidence of potential displacement from the exposure wharf that coincided with the start of pile driving, together with a decrease in daytime residency for a subset of this species with high site fidelity prior to the event. Results indicated that snapper may be more likely to depart an area of pile driving disturbance than sheepshead but were thought to be less at risk for behavioral impact given their lower site fidelity.

The authors concluded that additional data need to be collected to determine the range of possible exposure values from coastal pile driving activities and that measurements of particle motion should be included, particularly in shallow, complex environments. Nonetheless, they remarked that their study provided a minimum exposure threshold for sound pressure levels within a few hundred meters of pile driving, below which no clear behavioral response was observed for fish exposed to pile driving in a natural environment for a duration of several weeks.

Krebs et al. (2016) monitored the movements of Atlantic sturgeon (*Acipenser oxyrhinchus*), using acoustic telemetry, during a pile driving operation at a new bridge on the Hudson River in New York. Fewer sturgeon were detected during pile driving and the fish remained in the vicinity of the pile driving for a shorter time than during silent control periods. Moreover, the short time spent by sturgeon near pile driving suggested that they were unlikely to have reached the current criterion of 187 dB re 1 μPa^2·s cumulative sound exposure level. The authors concluded that Atlantic sturgeon in the Hudson River were likely to avoid underwater noise associated with impact pile driving and were not likely to remain in the vicinity long enough to experience noise levels that would result in the onset of physiological effects. The study provides empirical evidence that the 206 dB re 1 μPa SPL_{peak} level for the onset of physiological effects was the appropriate metric for assessing the potential impacts of pile driving noise on sturgeon.

Roberts et al. (2016b; 2016a) examined the responses of a number of wild demersal species to the playback of pile driving sounds, observing their behavior using an underwater video system in a sea lough on the coast of Ireland. The results indicated that impulsive sounds within a received sound pressure level range of 163–167 dB re 1μPa (peak-to-peak) elicited behavioral responses including startle responses and directional avoidance. The exposure levels were similar to the 50% response levels determined by Hawkins et al. (2014) for schools of sprat and mackerel using the same sound projector array. However, Roberts et al. (2016a) emphasized that although the water-borne component of the sound was accurately reproduced by the sound projectors, the projectors were not able to replicate the additional substrate-borne vibrations that pile drivers produce.

Following a series of laboratory experiments exposing marine invertebrates to substrate vibration (Roberts et al. 2016c), Roberts and Elliott (2017) reviewed the impacts of anthropogenic vibration on the marine epibenthos, which forms the food of many demersal fishes. They pointed out that anthropogenic activities on the seabed, including pile driving, may produce high amplitude vibrations. Such anthropogenic vibrations may elicit behavioral, physical, or physiological changes, and it is important that noise impact assessments must consider the role of seabed vibration, in tandem with sound, upon benthic organisms.

Using what they termed an open field test, Spiga et al. (2017) investigated the effects of recordings of piling and drilling noise on the anti-predator behavior of captive juvenile European sea bass in response to a visual stimulus (a predatory mimic). However, the fish were sourced from a commercial hatchery and were maintained in captivity in a shallow holding tank, and so it is possible that behavioral responses were affected by hatchery, and not real-world, experiences. The sounds were recorded and were played back to the fish using a sound projector with a limited low-frequency response. The impulsive nature of piling noise triggered a reflexive startle response, which contrasted with the behavior elicited by the continuous drilling noise. When presented with the predatory mimic, fish exposed to both piling and drilling noise explored the experimental arena more extensively than control fish exposed to ambient noise. Fish under drilling and piling conditions also exhibited reduced predator inspection behavior. It was concluded that the additional noise (piling and drilling playbacks) affected the kinematic component of the anti-predator response (swimming path and velocity, including turning) more than the behavioral component (responsiveness and response latency). Piling and drilling noise also induced stress as measured by ventilation rate. Spiga et al. concluded that exposure to elevated noise levels significantly affected the behavior and physiology of European sea bass.

Herbert-Read et al. (2017) examined the swimming trajectories of individual juvenile European seabass in groups under controlled laboratory conditions. Groups were exposed to playbacks of either ambient background sound recorded in their natural habitat, or playbacks of pile driving. The pile driving playback affected the structure and dynamics of the fish shoals significantly more than did the ambient sound playback. Compared with the ambient sound playback, groups experiencing the pile driving playback became less cohesive, and less directionally ordered, and were less correlated in speed and directional changes. In effect, the pile driving sounds disrupted the abilities of individuals to coordinate their movements with one another. The investigators suggested that the results highlighted the potential for noise pollution from pile driving to disrupt the collective dynamics of fish shoals, which could have implications for the functional benefits of a group's collective behavior.

b. Seismic Airguns

Airguns used for seismic surveys produce high-intensity impulses at rather lower levels and longer intervals than pile driving impulses, but with similar spectral characteristics. However, just like pile driving, the nature of the impulse varies depends upon the specific source (Popper et al. 2014; Gisiner 2016). The impulse levels from the airguns may be similar to those at greater distances from a pile driver.

The behavioral and physiological effects of exposure to airguns have been reviewed by a number of investigators (McCauley et al. 2000; Slabbekoorn et al. 2010; Normandeau 2012; Popper et al. 2014; Radford et al. 2014; Hawkins et al. 2015; Carroll et al. 2017). Although signals from seismic air guns can be similar to those of pile driving in terms of frequency range of major energy, duration of impulse, and rapid rise time, there are few data from seismic studies that inform development of guidelines because most of the studies have not measured the sounds received by the animals and most were done with animals confined to cages where behavior is aberrant. And, even when there are such data, there is substantial variability in results depending on species and study methods. Additional details of these studies of behavioral responses to airguns are provided in the Appendix.

c. Explosions

Almost nothing is known about effects of explosions on the behavior of wild fishes. One likely behavioral effect is a startle reaction if the received signal is of sufficient magnitude. Such a response lasts less than one second and does not necessarily result in significant changes in subsequent behavior. Koschinski (2011) reviewed the underwater noise pollution from munitions clearance and disposal and has considered the possible effects on marine vertebrates. He pointed out that underwater detonations represented the loudest anthropogenic point sources of noise in the oceans and that they have the potential for serious injury to aquatic vertebrates. However, the effects on the behavior of fishes and other aquatic vertebrates were not considered.

4. Responses to Other Sound Stimuli

A range of responses has been observed when the behavior of wild fishes has been examined in the presence of anthropogenic sounds. Some fishes have shown changes in swimming behavior and orientation, including startle reactions. In some cases, the response may habituate with repeated presentations of the same sound. Sound can particularly cause changes in schooling patterns and distribution (Hawkins et al. 2014). For example, the horizontal and vertical distributions of both pelagic and demersal fishes have been shown to change during and after airgun operations (e.g., Løkkeborg et al. 2012 a, b). In some circumstances, fish react to approaching ships, leading to concern by fisheries scientists that vessel avoidance will bias stock-assessment surveys by research vessels. Fish can respond to approaching vessels by diving towards the seafloor or by moving horizontally out of the vessel's path, with reactions often initiated well before the vessel reaches the fish (Ona et al. 2007). However, the stimuli to which these fishes are responding are not always clear (Sand et al. 2008).

Doksaeter et al. (2009) investigated the behavioral reactions of overwintering Atlantic herring to sonar signals of two different frequency ranges 1–2 and 6–7 kHz, and to playback of killer whale feeding sounds. The sonar signals were frequency modulated sweeps, whereas the feeding sounds played back included calls, echolocation clicks, and tail-slaps. The experiments were carried out in controlled exposure experiments in Vestfjorden, Norway. A vessel towing an operational naval sonar source approached and passed over the fish in a block design setup. No significant escape reactions, either vertically or horizontally, were detected in response to sonar transmissions. However, killer whale feeding sounds induced vertical and horizontal movements of herring. Doksaeter et al. concluded that military sonars of such frequencies and source levels may be operated in areas of overwintering herring without substantially affecting herring behavior or herring fishery. The avoidance during playback of killer whale sounds involved a reduction in herring density almost immediately after the start of playback, demonstrating that herring may react to impulsive sounds. The sounds that the herring reacted to were mainly in a similar frequency range as the sonar signals.

A subsequent study (Doksaeter et al., 2012) examined herring reactions to sonar signals and other stimuli when kept in captivity, using detailed acoustic and video monitoring. Throughout the experiments, spanning three seasons of a year, the fish did not react significantly to sonar signals from a passing frigate, at received signals up to 168 dB re 1µPa (RMS). In contrast, the fish did exhibit a significant diving reaction when exposed to other sounds, including a much lower sound level from a two-stroke boat engine. The lack of herring reaction to the sonar signals was consistent with the earlier *in situ* behavioral studies. The differences observed in the behavioral reactions of the fish underlines the key importance of the characteristics of the actual stimuli presented to fish.

5. Overall Conclusions Regarding the Effects of Pile Driving on Fish Behavior

In their original examination of the effects of sound on fish, Hastings and Popper (2005) concluded that the studies available at that time provided only a preliminary indication of the potential impact of pile driving on fishes. At that time, the available data provided only a very preliminary indication of the kinds of behavioral effects that might be encountered as a result of pile driving. They remarked that there was a need for new studies of behavioral responses of fish to pile driving. In particular, there was a need to determine if there were longer-term behavioral effects from pile driving that might alter the movement patterns of fish schools, and affect feeding behavior, responses to predators, and mating and reproductive behavior.

A few additional studies of the effects of pile driving and other intermittent sound stimuli upon the behavior of fishes have since been completed. As we have seen, many of these studies have been carried out upon captive fish, often under inappropriate acoustic conditions. However, it is possible to draw some general conclusions, as discussed below.

a. The Relevance of Behavioral Responses

Behavioral changes in response to sounds are often considered to be more relevant than injuries in assessing effects, because a larger part of the population experiences behavioral changes and fish are affected over a much wider area. However, different behavioral responses may occur, depending on the level of the sound, the level of ambient background sound, what the fish are doing at the time, and their previous experience of the same and other sounds (Hawkins and Popper 2014; Hawkins and Popper 2016). Whether a fish responds may also depend on its condition, motivational state, and the presence of other animals including predators.

It is generally assumed that fishes with better hearing abilities are more likely to respond to sounds than are less sensitive species. However, a number of studies have suggested that this may not always be the case. For example, the study by Hawkins et al. (2014) on sprat and mackerel showed that the sound pressure levels to which fish schools responded, estimated from dose response curves, were remarkably similar for both species, despite major differences in their hearing abilities.

b. Behavioral Responses Shown by Fishes to Anthropogenic Sounds

Studies have demonstrated that fishes exposed to pile driving sounds may show alarm responses. They may increase their swimming speeds (often showing a directional response), change their ventilation and heart rates, and show startle responses. Such transient escape reflexes are unlikely to result in adverse impacts, as the fish may rapidly return to their normal behavior. However, stronger more sustained responses may generate oxygen debt and place an energetic load on the fish.

The schooling of fishes — their gathering into shoals — is often an important aspect of their behavior. Playback of pile driving sounds to pelagic fishes has been shown to cause both the break-up of fish schools, and the consolidation of schools, the latter of which may have adverse effects through a reduction in oxygen levels or the accumulation of waste material. Such sounds also disrupt the collective dynamics of shoals of juvenile sea bass, which could have implications for the functional benefits of a group's collective behavior.

The presence of anthropogenic sounds may interfere with foraging behavior either by masking the relevant sounds or by resembling the sounds that the prey may generate. Sound exposure can also result in food-handling errors. The majority of studies so far have been conducted in laboratory tanks, but they have indicated that exposure to noise can result in decreased feeding efficiency by fishes. Additional noise in the environment can lead to reduced food consumption, although the effects are likely to be species specific.

Elevated noise levels have the potential to affect anti-predator behavior of different species in different ways. The presence of noise may interfere with the detection, location, and identification of predator sounds by fishes, perhaps rendering them more likely to be captured. It has been shown that noise exposure decreases eel anti-predator responses. However, results showed that those effects quickly dissipated. Once the noise was removed, there was complete recovery in the case of eel anti-predator startle responses. In other studies, prey were captured more readily by their natural predators during exposure to motorboat noise. More than twice as many prey were consumed by a predator in field experiments when vessels were passing. It is evident that elevated sound levels, and especially intermittent sounds, may affect predator/prey interactions.

Many fishes use sound to communicate with one another; identify other individual fish; gather together in shoals; detect prey and predators; orient themselves with respect to the local environment; navigate from one place to another; synchronize spawning; and select appropriate habitats. Masking of natural sounds may reduce the distances over which fishes may communicate with one another and may also result in their failing to detect environmental sounds that are important to them. Masking may also interfere with the ability of fishes to discriminate between different sounds, locate the source of the sound, and identify particular sources. Any interference with the ability of fish to detect and analyze key signals has potential consequences for the fitness and survival of individuals, populations, and species. Anthropogenic sounds that are similar to natural sounds may also cause misleading information to be relayed to fishes.

During their migrations fishes need to have access to the various environmental cues that they use to position and orientate themselves relative to the geography of the area. During passage through the sea, fishes like the salmon may use a variety of cues to orientate and navigate, including natural soundscapes. Larval reef fishes may locate their home reefs by responding to their characteristic sounds. Sound also has the potential to convey valuable information about the quality of habitats. Alterations to acoustic cues as a result of developments in the sea, estuaries, rivers, and lakes may have adverse effects on migratory fishes and fishes seeking particular habitats. High level sounds may result in avoidance responses, deflecting fishes away from their migration routes. The presence of high levels of anthropogenic sound may also result in fishes being displaced from their preferred habitats.

Some fishes migrate to particular locations, often at a particular time of the year, to engage in reproductive behavior. Other fishes may remain at particular locations within a preferred habitat and reproduce there. Reproduction is often preceded by elaborate behavior patterns that may involve the male fish competing with one another for particular territories or even for individual females that are ready to spawn. Both territorial defense and the courtship of females may involve sound production by the fish. Selection of individual males by females may be based on the characteristics of the sounds made by different males. The actual act of spawning, where eggs and sperm are brought into contact, requires synchronization, which may also involve the production of sounds. It is evident that anthropogenic sounds have the potential to prevent fish calls being detected and may also affect the discrimination of sounds and the identification of particular individuals. The localization of fish that are ready to spawn by other fish may also be adversely affected.

c. Findings that May Be Useful for Estimating Response Distances

NOAA Fisheries currently uses a criterion for behavioral response of 150 dB RMS re 1 µPa (page 97). However, there is some uncertainty about the origins of this criterion, and the data that led to it (Hastings 2008). It is now clear that with behavioral response criteria it may be difficult to choose a single value for all fishes, because species can vary greatly in their behavioral responses. Information that may assist in the setting of behavioral criteria is reviewed in Chapter 9.

However, the Hawkins et al. (2014) study on the behavioral responses of wild free-living pelagic sprat and mackerel to the playback of pile driving sounds yielded dose response curves, enabling the sound level that resulted in clear responses on 50% of presentations to be determined (Hawkins et al. 2014). The sound pressure levels were 163.2 and 163.3 dB re 1 µPa peak-to-peak, and SEL_{ss} were 135.0 and 142.0 dB re 1 $\mu Pa^{2\cdot}s$, for sprat and mackerel, respectively, estimated from dose response curves.

It is interesting to consider how far away from a pile driving operation such sound levels extend. Results from field observations during a wind turbine foundation installation at the Block Island Wind Farm, Rhode Island (R. Gopu and J. Miller, personal communication, 2018) show peak-to-peak received sound pressure levels estimated for all of the hammer strikes from three different pile driving operations plotted against distance of the receiver from the pile (Figure 10). The figure shows a sound pressure level of 163.3 dB re 1 µPa peak-to-peak (green line) for the behavioral response of the fish. Therefore, it can be expected that responses from sprat and mackerel might, take place at distances ranging from 4,000 meters to more than 6,000 meters (Figure 10). However, in some circumstances, the signal may be masked by the presence of other anthropogenic sounds (e.g., commercial vessels, pleasure boats), and even natural ambient noise levels, and, as a consequence of this masking, the behavioral responses may be limited to shorter distances.

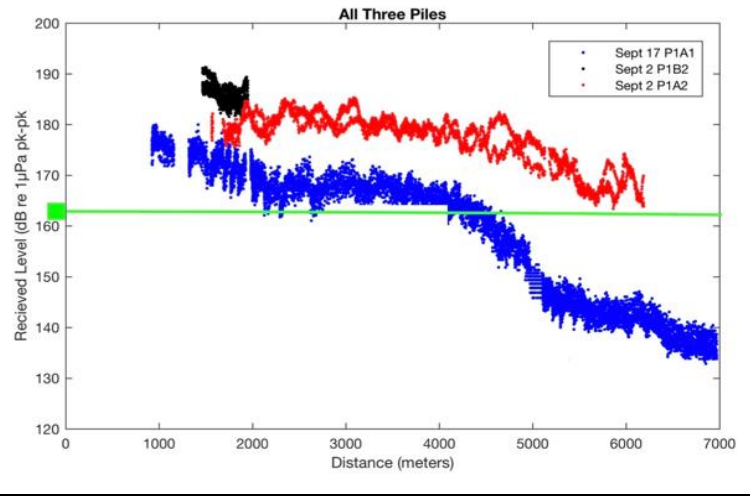

Figure 10: Peak-to-peak received sound pressure level for pile driving operation.
Data calculated for all of the hammer strikes from three different pile driving operations during the construction of an offshore wind farm at Block Island, Rhode Island, plotted against distance of the receiver from the pile. The green line shows the level that caused strong behavioral responses in wild sprat and mackerel (Hawkins et al. 2014). The responses from sprat and mackerel might be expected to take place at distances ranging from 4,000 meters to more than 6,000 meters, depending on the pile being driven. (From Gopu et al. 2018)

CHAPTER 7: PHYSICAL EFFECTS

1. Overview of Chapter

This chapter focuses on physical effects of high-intensity impulsive sounds on fishes. Much of this discussion is based upon recent studies that investigated the effects of pile driving sounds on fishes. These findings are the basis for the 2014 Guidelines for onset of potential effects of pile driving and other impulsive sources (Popper et al. 2014).

There are also studies that have focused on other intense sound sources, including seismic airguns and sonar. These studies are reviewed only briefly because they do little to inform knowledge of the effects of pile driving sounds.

2. Pile Driving

Exposure to pile driving sounds has the potential to kill fishes, damage tissues, cause delayed mortality, or reduce fitness (Hastings and Popper 2005; Popper and Hastings 2009; Halvorsen et al. 2012a). However, data on direct mortality due to exposure to pile driving have come from only a few field studies, most notably from a demonstration project for construction of the San Francisco–Oakland Bay Bridge east span (e.g., Caltrans 2001). This study demonstrated a high stunning and mortality of fishes (several different species) that were within 10 m (33–39 f) of the 96-inch shell piles that were driven. The report documented dead fish rising to the surface around the pile, and there may have been additional mortality that was not observed because many moribund fishes sink from a ruptured swim bladder. Damage reported was to the liver and kidneys, regions around the swim bladder, was reported in more recently lab studies.

A major concern regarding most potential effects on fishes exposed to pile driving is that effects were discussed in gray literature that was not subject to rigorous peer review (e.g., Caltrans 2001; Abbott and Bing-Sawyer 2002; Abbott 2004; Abbott et al. 2005). Furthermore, analysis of these studies raised concerns regarding experimental design, including the need for controls (Hastings and Popper 2005; Popper and Hastings 2009).

More recently, however, studies have investigated the physical effects of pile driving on fishes, all of which have been carefully designed with appropriate controls and statistical power. These studies took place in the U.S. (Halvorsen et al. 2011; Casper et al. 2012b; Halvorsen et al. 2012c; Halvorsen et al. 2012a; Casper et al. 2013a; Casper et al. 2013b; Casper et al. 2017) and in Europe (Bolle et al. 2012; Debusschere et al. 2014; Bolle et al. 2016; Debusschere et al. 2016b). The work done in the United States was a direct outgrowth of the 2008 Interim Criteria set forth by the FHWG. The studies provide greater insight into potential effects from exposure to pile driving, as well as providing material for updating the current guidance.

This chapter will focus on recent findings on physical effects from high-intensity impulsive sounds on fishes, and primarily on the work done in the United States because it provides the most comprehensive insight into physical effects. Other potential effects of pile driving and other impulsive sounds are also discussed in Chapter 5 (Hearing), Chapter 6 (Behavior), and Chapter 8 (Physiology).

3. General Mechanisms of Injury Response

There are two potential physical mechanisms by which fishes could be affected by pile driving sounds. One, about which very little is known, is from direct impact of the particle motion component of the signal on the fish that may result in intense shaking of the animal (Chapter 3) (Popper and Hawkins 2018).

The second physical mechanism that can cause damage is barotrauma, which is injury to tissues caused by a rapid change in pressure (or pressure difference) across an anatomical structure. Such pressure changes can result from impulsive sources such as pile driving, seismic airguns, and explosive events. Barotrauma can be induced by compression or by decompression aspects (discussed in detail below) of an impulsive signal.

Gas-filled spaces within fishes may be especially important in relation to barotrauma. As a result of the rapid pressure changes associated with the impulsive pile driving sound, the gas within such gas bodies (e.g., swim bladder, auditory capsules, free gas bubbles in the body) may quickly and repeatedly expand and contract (Sand and Hawkins 1973; Blaxter 1981). The walls of the swim bladder or other gas bodies may then move with sufficient magnitude and rapidity to cause damage to surrounding organs and tissues as well as to the swim bladder itself. In particular, because the swim bladder is situated in the abdominal cavity and often just below the spinal column, it lies very close to many key internal organs and tissues (e.g., kidney, liver, gut).

Barotrauma injury patterns and severity of injuries from underwater sound are influenced both by fish morphology and the acoustic characteristics. Specifically, impulsive signals created by pile driving, airguns, and explosions carry the highest potential for causing barotrauma because of their short rise time and high amplitude.

4. The Swim Bladder

In considering potential physical effects of pile driving on fish, it is necessary to take account of variations associated with the swim bladder. One issue to consider is that there is substantial inter-specific variation in swim bladder morphology and location in the abdominal cavity, and this variation could result in inter-specific differences in the effects of pile driving. Swim bladders range from simple single ellipsoid-shaped chambers to multi-chambered structures. In some cases, the front, or rostral, end of the swim bladder ends some distance from the fish's skull, whereas in other species it may terminate close to, or in contact with, the skull or even the inner ear (Helfman et al. 2009).

There appear to be differences in the effects of impulsive sounds on fishes based on the presence or absence of the swim bladder, the structure and conformation of the swim bladder, and the mechanisms by which gas is added to, and removed from, the swim bladder. Moreover, there is evidence, discussed below, that fishes without a swim bladder, such as flatfishes, may be less likely to be impacted by barotrauma caused by impulsive sounds because there is no gas-filled organ to affect neighboring tissues (Goertner et al. 1994; Halvorsen et al. 2012c).

Fishes with swim bladders can be divided into two (non-taxonomic) categories. Fish in one category, referred to as physostomes, have the swim bladder connected to the gut via a pneumatic duct.[19] Air, gulped by the fish at the surface, goes into the gut through the pneumatic duct and fills the swim bladder. Gas release is the reverse pattern. By being able to expel air from the swim bladder, physostomes are able to voluntarily respond to the pressures of the physical sound field and decrease the tension on the swim bladder, thereby potentially decreasing the severity of tissue injuries caused by impulsive signals (Casper et al. 2012b; Halvorsen et al. 2012c; Halvorsen et al. 2012a). Physostomes include salmonids and sturgeon, as well as a variety of other species.

The second category of fishes, the physoclists, have a closed swim bladder system. In these species, a gas gland uses a process of diffusion to move gas into the swim bladder from the blood circulatory system, thereby adjusting the fish's buoyancy (e.g., Helfman et al. 2009), while removal of gas is by a separate set of blood capillaries in close contact with the swim bladder. Filling or removing gas from the swim bladder is much slower in physoclists than in physostomes, but physoclists do not have to go to the water surface to get the gas needed for buoyancy control. Because the mechanism for changing the volume of gas in the physoclist swim bladder is relatively slow, the swim bladder and surrounding tissues are at higher risk of injury from rapid pressure changes caused by impulsive sounds (e.g., Goertner et al. 1994; Halvorsen et al. 2012c). Physoclists include families such as bass, perch, codfish, and rockfish.

5. Physical Effects from Impulsive Sounds

Depending upon the sound source and distance from a source, barotrauma may result from compression or decompression. In effect, compression can be considered as squeezing a fish, and decompression can be considered a rapid release of all squeezing (pressure) on a fish.

a. Compression and Decompression

Injuries can result from high amplitude positive overpressures, especially from a sound pressure pulse characterized by an initial positive pressure increase with a rapid rise time and high amplitude peak pressure, such as might occur from an explosion or when a fish is very near a source of impulsive sounds (Cole 1948). There is also the suggestion that sources that can cause compression injuries at short distances have a high positive overpressure (Cole 1948). Negative sound pressures within the pulses from pile driving are sufficient to cause decompression injuries (Halvorsen et al. 2012c; Halvorsen et al. 2012a).

Decompression injuries are caused by a rapid release of pressure, which is observed, for example, in instances when physoclist fishes are quickly brought to the surface by anglers. Decompression injuries to fish may occur through two different mechanisms, one involves any gas bladder (e.g., swim bladder or bubble or gas) and the other involves dissolved gases in the blood and tissues of any fish. Impulsive signals occur in repeated succession and have an inherent “pulsing” characteristic, which means that compression and decompression are repeated in rapid succession on the fish’s body.

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[19] Some physostomes also have a rete mirabile and gas gland which are less developed than physoclists but they do function (Helfman et al. 2009).

These two forces generated by impulsive signals, compression and decompression, induce barotrauma injury, which occurs through different mechanistic pathways. Decompression causes gas to come out of solution, which forms bubbles in the blood, which may rupture blood vessels, veins, and organs (Brown et al. 2009) causing lethal hemorrhaging (Brown et al. 2009; Brown et al. 2012; Halvorsen et al. 2012c; Halvorsen et al. 2012a). Bubbles in the tissues show up often in the gills and block oxygen exchange, causing suffocation. Such bubbles can also lacerate organs (Govoni et al. 2003; Schreer et al. 2009). Barotrauma injuries are expressed externally and internally, and range in severity from minor to mortal. A few examples of barotrauma caused by pressure changes include bulging eyes, intestinal eversion (i.e., stomach protruding out of the mouth), and ruptured swim bladder (Gaspin 1975; Rummer and Bennett 2005; Brown et al. 2009; Brown et al. 2012; Halvorsen et al. 2012c; Halvorsen et al. 2012a). During decompression, the combination of mechanisms, pulsing the fish swim bladder (and extension through to rupture) and gas coming out of blood, has the potential to cause lethal injuries in fishes.

b. Severity of Injuries from Impulsive Sounds

Halvorsen et al. (2011; 2012a) described barotrauma injuries in fishes exposed to simulated pile driving sounds and presented a physiologically based injury classification approach called the FIT that permits quantification of such injuries in fishes. The FIT model applied a mathematical weighting to each injury dependent upon its severity (5, 3, or 1), then all weighted injuries were summed to produce a single overall injury value, called the RWI for each fish (for details see Halvorsen et al. 2011; 2012a).

Weightings of injuries were determined based on the severity of each type of injury (e.g., Woodley and Halvorsen, personal observations; Gaspin 1975; Yelverton et al. 1975; McKinstry et al. 2007). As an example, an exposed fish with three injuries, one mortal injury (weighted 5) and two mild injuries (weighted 1), resulted in an RWI of 7, the overall injury value, shown by the following equation:

$$\text{RWI} = \sum (\text{injury x weighting score})$$

Because different injury patterns can yield the same RWI score, the FIT model provided a way to convolve complex data into a single value. Three injury categories were used, mortal, moderate, and mild, a few example injuries were:

- mortal injuries include ruptured swim bladder and organ hemorrhage;
- moderate injuries include organ bruising; and
- mild injuries include fin bruising.

6. Effects of Pile Driving Sounds on Fishes

Following the publication of the FHWG 2008 Interim Criteria, the National Transportation Research Board of the U.S. National Academies of Science put out a request for proposal for studies to experimentally examine effects of pile driving sounds on fishes that was awarded to Dr.

Popper's laboratory at the University of Maryland.[20] The project resulted in an initial report (Halvorsen et al. 2011) followed by a series of six peer-reviewed papers that provided a quantitative approach to assessing onset of physical damage to fishes (Casper et al. 2012b; Halvorsen et al. 2012c; Halvorsen et al. 2012a; Casper et al. 2013a; Casper et al. 2013b; Casper et al. 2017). The following sections describe the results of this research, which provided the basis for guidance regarding onset of physical injuries from sound to fishes. These studies defined onset of injury as the threshold at which a maximum of two mild injuries appeared.

It is important to note that the onset thresholds used in these studies were not those currently used by regulatory agencies to set acoustic criteria but were based on the results observed during the experiments. Indeed, it is acknowledged by the agencies that, although some fish may recover from injuries in captivity, it is not always apparent when and where recovery of wild fishes is more or less likely. It is possible in some circumstances that a single mild injury may have detrimental effects.

a. Brief Methodological Overview

The experimental approach used in the studies is described in detail in the two initial papers (Halvorsen et al. 2011; Halvorsen et al. 2012a). The studies were done in the laboratory, thereby mitigating many of the problems that are inherent in doing field studies and allowing for well controlled exposure experiments (discussed in Popper and Hastings 2009; Halvorsen et al. 2011). As discussed below, the experimental chamber and other aspects of the study were designed so that they did not have the problems of most lab-based tank studies described in earlier parts of this report.

To bring the studies into the laboratory, it was necessary to develop an approach that generated nearly identical signals to those produced during field pile driving at 10 m from the source in terms of amplitudes, sound spectra, rise times, and energy levels. This was done by the development of an apparatus that allowed for controlled exposures. The apparatus, the High Intensity Controlled Impedance – Fluid filled wave Tube (HICI-FT), consisted of a stainless-steel tube with 3.81-cm thick walls and having an electromagnetic shaker on each end.

The pile driving signals were recorded in the field from steel pile installations (MacGillivray and Racca 2005). The shakers were controlled independently, which allowed for the generation of plane-wave pressure and velocity fields that nearly identically simulated free-field impulsive pile driving signals at 10 m from the source. The HICI-FT also provided control over variables that cannot be controlled in the field where the source is pile driving during a construction project, including pulse rate, signal amplitudes, exposure duration, number of impulses, and temperature, (Halvorsen et al. 2011; Halvorsen et al. 2012a).

7. Summary of the Research Series

The reporting of all relevant acoustic metrics is important to understanding and comparing results across studies. To help read the results, this report uses a shorthand form. The SEL (SEL_{cum}

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[20] Additional funding to the investigators' laboratory was also provided by BOEM and Caltrans. Co-investigators with Dr. Popper were Dr. Thomas Carlson and Dr. Michele Halvorsen. Dr. Hawkins was advisor on the project.

and SEL_{ss}) metric is reported in terms of dB re 1 µPa^2·s. However, for reporting of results herein, the *re 1 µPa^2·s* will not be used at each mention of SEL.

The SEL_{cum} is calculated based upon the SEL_{ss} and number of impulses. For example, an SEL_{cum} of 180 dB re 1 µPa^2·s may be generated by an SEL_{ss} of 177 dB re 1 µPa^2·s from 1920 impulses, which will be written in shorthand as: SEL_{cum} 180 dB [SEL_{ss} 177 dB, # 1920]. Reporting of SPL will include units and specificity of form (peak or RMS).

a. Impulse Effects on Chinook Salmon

In the initial study, Halvorsen et al. (2012a) investigated the onset of tissue injury from exposure to impulsive pile driving signals on a physostomous fish, juvenile chinook salmon (*Oncorhynchus tshawytscha*) and examined the metrics most relevant to determine severity of impacts on fish. The FIT model was applied to the findings and used as a framework to report results. The study also tested the validity of the "equal energy hypothesis," which suggests that regardless of how acoustic energy is accumulated (e.g., combination of SEL_{ss} and number of impulses), the impacts on the organism will be the same (see also Woodbury and Stadler 2008; Stadler and Woodbury 2009).

The results provided a dose response curve showing an injury onset threshold based on an RWI of 2, indicating that a maximum of two minor injuries could be sustained by a fish. The impulsive signals that generated an RWI of 2 were an SEL_{cum} 210 dB [SEL_{ss} 180 dB, # 960] and [SEL_{ss} 177 dB, # 1,920]. The SEL_{cum} 210 dB [SEL_{ss} 180 dB, # 960] threshold was confirmed in a later study on the same species (Casper et al. 2012b).

The study also showed that the relationship between the number of injuries and their severity (collectively RWI) increased as the SEL_{ss} and SEL_{cum} increased. There were no reported mortalities immediately after exposure, but it was suggested that fishes with an RWI ranking greater than 5 would ultimately die of the injuries. The most common mortal injuries were ruptured swim bladder and hemorrhaging kidney.

This first study showed that the metrics of each impulse are important for understanding how the energy impact is cumulated by the animal. The SEL_{cum} values were held constant while altering the SEL_{ss} and the number of impulses, thus testing the equal energy hypothesis. By holding the SEL_{cum} constant while varying the number of impulses (960 or 1,920) and SEL_{ss}, different injury severities were induced. It was demonstrated that that the 960 impulses produced more severe overall impacts than 1,920 impulses on salmon, thereby demonstrating that the equal energy hypothesis is not applicable to fishes, at least for impulsive sources. This finding was replicated in additional studies (Casper et al. 2013a; Casper et al. 2017).

Lastly, the investigators concluded that a single acoustic metric to determine acoustic impacts on fishes, specifically, the SEL_{cum}, is insufficient as a predictor. Rather, at least three or four acoustic metrics are necessary to properly determine impacts. The recommended metrics include the number of impulses, the SPL_{peak} or the SEL_{ss} (which are correlated values) and the SEL_{cum} values (Halvorsen et al. 2012a).

b. Recovery from Injury

FESA and CESA consider that injury, even if it is recoverable, is still likely to cause “harm.” However, it is not clear whether harm would still be considered to exist once the animal recovers. As part of this issue, there is the question as to whether, after injury is sustained from sound exposure, injuries worsen or whether there is recovery. Studies using the same experimental paradigm as described above found that animals kept in the lab recovered from many injuries, and there did not appear to be further manifestation of injuries after exposure (Casper et al. 2012b; Casper et al. 2013a). The investigators were careful to point out, however, that recovery occurred in a laboratory environment where the fish were not subject to predation. Thus, caution should be applied when extrapolating the recovery results to the field where animals with barotrauma injuries may have lower fitness and, therefore, may be less likely to avoid predators or disease. It has to be assumed that any injury suffered by individual fish in the wild, even a single mild injury, could possibly affect survival. A critical finding from the study, however, is that all injuries are sustained during sound exposure and that additional injuries (perhaps arising from cellular changes during exposure) do not seem to take place.

Casper et al. (2012b) investigated the ability of juvenile Chinook salmon to recover from exposure to pile driving sounds in the laboratory. Because this study looked at healing rather than injury severity *per se*, the FIT model was not applied. Instead an injury index was used, which was a ratio of frequency of injury occurrence over time; starting on the day of exposure out to 10 days post-exposure. Results (Figure 11) showed that the injury index increased with higher SEL_{cum} and decreased (i.e., showed recovery) over the 10 days. The results further supported the suggestion that the minor injuries that had an onset at SEL_{cum} 210 dB [SEL_{ss} 180 dB, # 960] are minor and survivable. At the same time, it is important to note, as stated above, that the recovering fish were in the laboratory where they were not exposed to the stressors of being in the wild. It is important, therefore, to ultimately examine recovery of animals in the wild, or at least under laboratory conditions where the fish are not cared for and there are some external stressors.

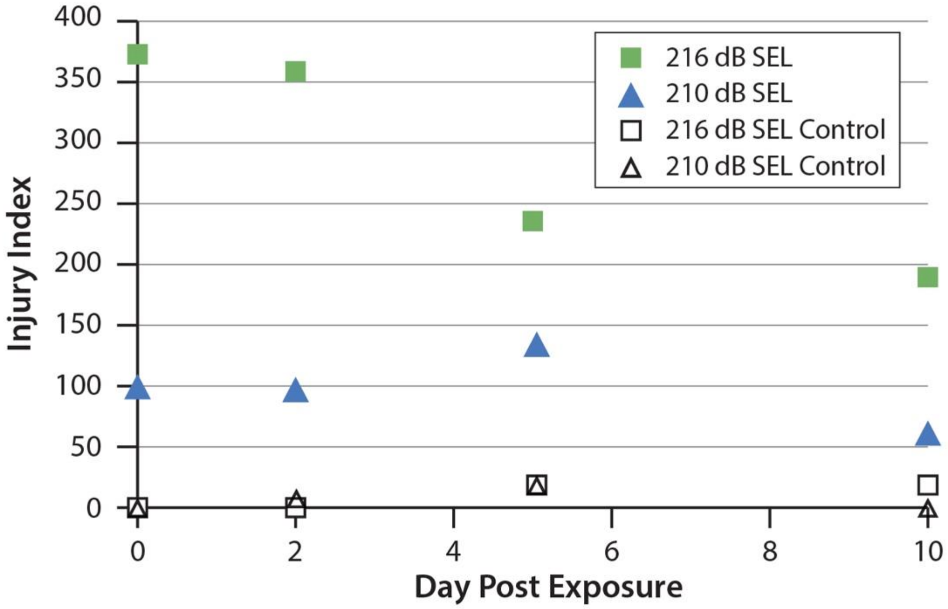

Figure 11: Healing following pile driving exposure.
The injury occurrence for two exposures (solid symbols) and their respective controls (open symbols), shows trend of healing (decrease in injury occurrence) over 10 days post-exposure. (From Casper et al. 2012b)

c. Impulsive Effects on Different Species

Additional fishes were tested to determine the applicability of the results to other species. Over a series of several studies, five more species were investigated, three in Halvorsen et al. (2012c) and two in Casper et al. (2013a; 2017).

Halvorsen et al. (2012c) examined the effects of exposure to pile driving sounds on the lake sturgeon (*Asipenser fulvescens*), a physostome, Nile tilapia (*Oreochromis niloticus*), a physoclist, and the hogchoker (*Trinectes maculatus*), a flatfish without a swim bladder. Results are summarized in Figure 12.

It was found that hogchoker exposed to an $\mathrm{SEL_{cum}}$ 216 dB [$\mathrm{SEL_{ss}}$ 186 dB, # 960] had no observed barotrauma injuries. In comparison, this identical sound exposure paradigm was capable of inducing mortal injuries in Chinook salmon.

The lake sturgeon and Nile tilapia were exposed to $\mathrm{SEL_{cum}}$ from 204 to 216 dB [$\mathrm{SEL_{ss}}$ 174 to 186 dB, # 960]. After exposure, fish were examined for barotrauma and the FIT model was applied to calculate an RWI for each fish (Figure 12).

The maximum reported RWI values were for the physoclist (Nile tilapia), while for the physostomes, the lake sturgeon had higher RWI values than did the Chinook salmon, but both had lower RWI values than the physoclist had (Figure 12). The injury onset threshold of $\mathrm{SEL_{cum}}$ 210 dB [180 dB, 960] continued to be applicable for physostomes, but less so for the physoclists. Based on the results (Figure 12), the onset of injury for physoclists is an $\mathrm{SEL_{cum}}$ 207 dB [$\mathrm{SEL_{ss}}$ 177 dB,

960], which is 3 dB lower (a halving of energy) than the physostome group's threshold. For the non-swim bladder fish (hogchoker), the injury onset threshold is greater than SEL_{cum} 216 dB [SEL_{ss} 186 dB, #960]. The conclusion was that physoclists, which cannot quickly remove gas from the swim bladder, are more sensitive to barotrauma injury than are physostomes, which can quickly remove gas from the swim bladder. Furthermore, it appears that non-swim bladder fishes are less likely to be affected by exposure to impulsive signals than are fishes with a swim bladder.

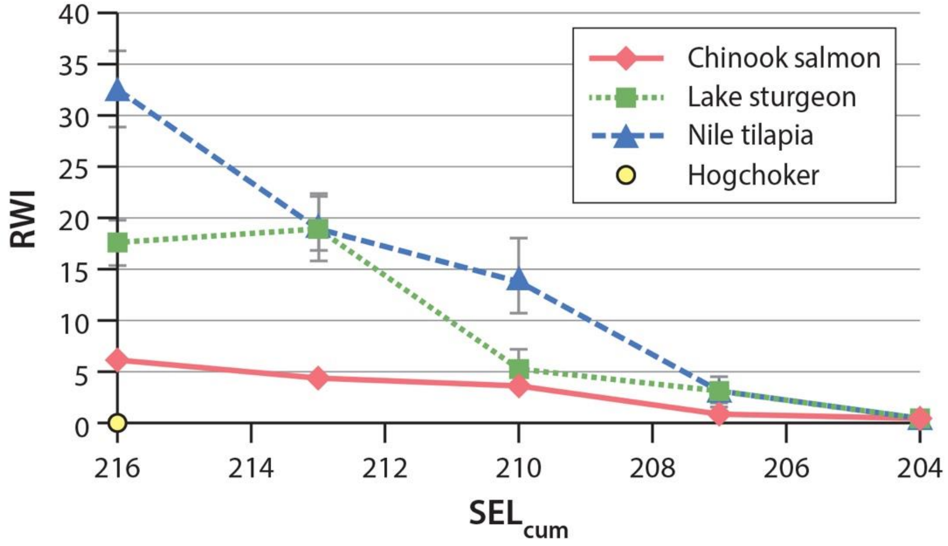

Figure 12: RWI values for four species.
Comparison of RWI values for four species from a range of SEL_{cum} exposure values. SEL_{cum} on x-axis, RWI values on y-axis. (From Halvorsen et al. 2012c)

d. Impulsive Effects on the Inner Ear

A critical question is whether impulsive sound results in temporary loss of hearing (referred to as TTS; Chapter 5). Although testing for hearing abilities was not included in the studies using the HICI-FT, a correlate to TTS is damage to the sensory hair cells of the inner ear (Chapter 5). Casper et al. (2013b) examined inner ear tissues (as well as other tissue damage) in two physoclists, hybrid striped bass (white bass [*Morone chrysops*] × striped bass [*Morone saxatilis*]) and Mozambique tilapia (*Oreochromis mossambicus*), after exposure to impulsive signals in the HICI-FT. The maximum exposure level was SEL_{cum} 216 dB [SEL_{ss} 186 dB, # 960]. This is a level reported to cause mortal injury in both the physostomes and physoclists, and, as expected, both species showed barotrauma injuries. Significantly, at this sound level no damage to the sensory hair cells of the inner ear was observed in hybrid striped bass, and only minor damage to the sensory hair cells of the inner ear was observed in the Mozambique tilapia.

The results from this study are important in two ways. First, they further support the levels of onset of injury reported in the earlier studies on physoclists. Second, they suggest that onset of damage to inner ear sensory cells occurs at higher sound levels than onset of damage to other tissues, leading to a tentative conclusion that hair cell damage in fishes exposed to impulsive sounds may not occur until fishes show mortal tissue injuries.

Still, it is very possible that the basis for this lack of damage to sensory cells is that in both species there is no air bubble near the ear to move and damage the inner ear tissues (e.g., Halvorsen et al. 2012c). Another reason for minimal effects on the ear may be that the signal in the experimental tank replicates a free-field condition where particle motion, the major stimulus of the inner ear, is relatively low. The question arises, therefore, whether inner ear tissues would be more readily damaged in species that have rostral extensions of the swim bladder that bring the air chamber close to (or in contact with) the ear, or if the fish is closer to the source (or to the bottom) where the particle motion is higher and directly impacts the ear.

e. Impulsive Effects on Fish of Different Sizes

Part of the current FHWG guidelines is the idea that onset of injury occurs at lower $\mathrm{SEL}_{\mathrm{cum}}$ levels in small fishes than in larger fishes. This idea was derived from studies of Yelverton et al. (1975) in which fishes of different sizes (in a poorly controlled study) were exposed to explosions (also see discussion in Hastings and Popper 2005; Popper and Hastings 2009). The assumption was that the effects of explosions would be similar to those from pile driving or other intense impulsive sources.

Casper et al. (2013a) tested this idea with exposure to impulsive pile driving signals on two sizes (17.2 grams and 1.3 grams) of hybrid striped bass (Figure 13). The maximum exposure level was $\mathrm{SEL}_{\mathrm{cum}}$ 213 dB [$\mathrm{SEL}_{\mathrm{ss}}$ 183 dB, # 960]. After exposure, fish were examined for barotrauma, and the FIT model was applied to the results to determine an RWI for each fish. The larger hybrid striped bass had higher RWI values (more total injuries and more severe injuries) than the smaller fish (Figure 11). Thus, at the maximum $\mathrm{SEL}_{\mathrm{cum}}$ 213 dB [$\mathrm{SEL}_{\mathrm{ss}}$ 183 dB, # 960] there was a difference of 20 RWI units between the two sizes. The large and small bass groups were also monitored for recovery from induced barotrauma injuries and it was found that most injuries to the larger bass were healed around 10 days post-exposure, while the smaller bass overall showed healing but with greater variability.

The reported RWI from these results were compared with the RWI results from two of the other studies (Halvorsen 2012, 2012) (Figure 14) and showed an interesting finding. The hybrid striped bass, a physoclist, had a maximum RWI of 35 compared with the RWI of 20 for Nile tilapia, a physoclist, for the same treatment, showing variability within the physoclists. However, the number of injuries in the striped bass was only slightly elevated as compared with those in the Nile tilapia. What is difficult to determine is the injury onset threshold for the hybrid striped bass, because at $\mathrm{SEL}_{\mathrm{cum}}$ 204 dB [$\mathrm{SEL}_{\mathrm{ss}}$ 174 dB, # 960] the RWI was 5. An RWI of 5 could, for example, represent a single mortal injury, or it could represent five minor injuries. Therefore, the bass threshold might be as low as $\mathrm{SEL}_{\mathrm{cum}}$ 201 dB [$\mathrm{SEL}_{\mathrm{ss}}$ 171 dB, # 960] where the RWI was 0.

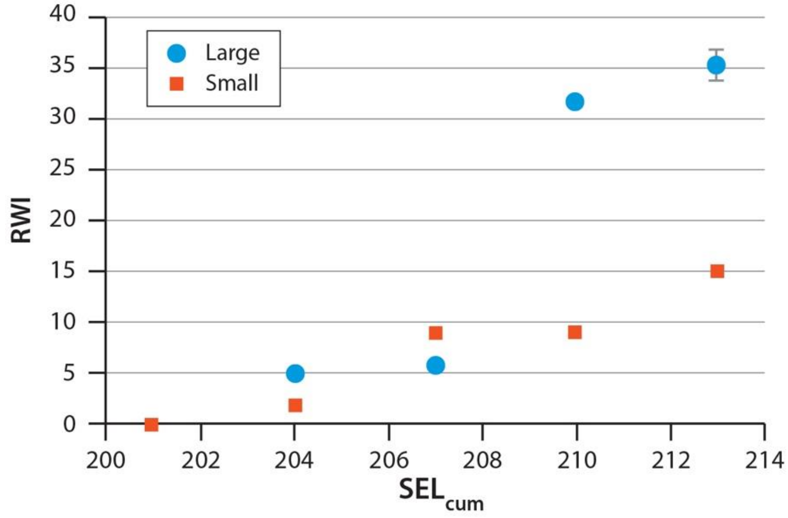

Figure 13: Effects of pile driving on fish of different sizes.
Comparison of two hybrid striped bass sizes (17.2 grams and 1.3 grams). X-axis is the SEL_{cum} exposures, y-axis is the RWI values. Higher SEL_{cum} have a higher impact on larger bass. (From Casper et al. 2013a)

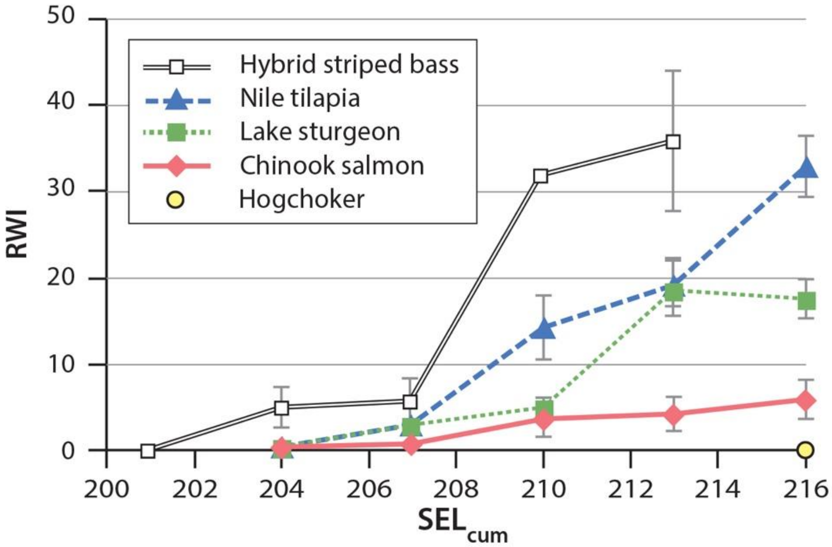

Figure 14: Comparison of five species' RWI values.
RWI values (y-axis) for five species from exposure to impulsive signals from the range of SEL_{cum} values (x-axis). Two physoclists (red), two physostomes (blue), one non-swim bladder (yellow dot). (From Casper et al. 2013a)

It is evident from these studies that there are variabilities in the RWI values for the three morphological swim bladder categories of physoclist, physostome, and non-swim bladder fish. The physoclist fishes are more sensitive to impulsive signals, and, when exposed, are at a higher risk of severe barotrauma injuries than are physostomes and non-swim bladder fish.

f. Effects from Different Numbers of Impulses

To understand how the metric of number of impulses might influence injury onset, Casper et al. (2017), following the research series protocols, exposed hybrid striped bass, a physoclist, to different numbers of impulses. Treatment exposures varied the number of strikes and held the SEL_{ss} metric constant to produce a range of SEL_{cum} values. The range of exposures from three SEL_{ss} of 177, 180, and 183 dB using 8 to 384 impulses produced SEL_{cum} of 191 to 209 dB. After exposure, fish were examined for the number of barotrauma injuries. An RWI was not calculated. Instead a probability of $\geq$ 1 injury was used for the analysis.

Figure 15 presents data from the study to illustrate the probability of injury relative to the three SEL_{ss} exposure treatments. The highest SEL_{ss} treatment (183 dB) caused mortal injuries in fish exposed to as few as eight impulses. Furthermore, the 183 dB SEL_{ss} treatment group shows a higher risk of injury than do the 180 and 177 dB SEL_{ss} treatment groups. As the number of impulses decreased, the risk of injury also decreased, but at different rates.

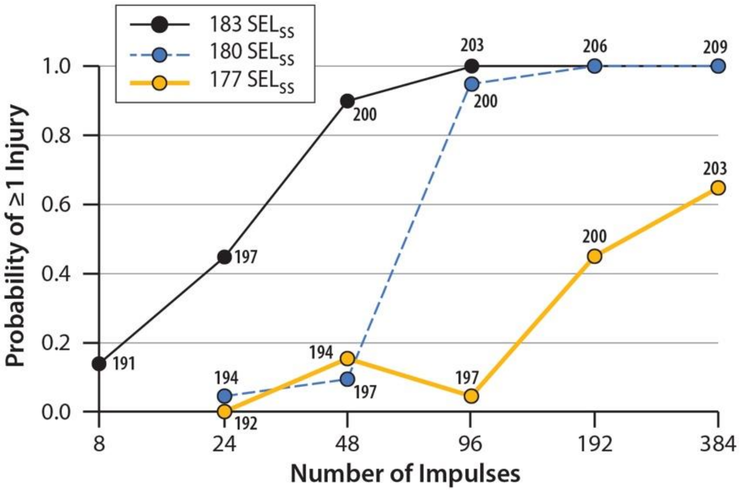

Figure 15: Injury as related to number of pile driving impulses.
Comparison of injury probability over number of impulses (log scaled) for each SEL_{ss} treatment. The 183 dB SEL_{ss} shows a higher risk of injury than do the 180 and 177 dB SEL_{ss} treatments. (Data from Casper et al. 2017)

Figure 16 helps clarify the results by focusing on a single SEL_{cum} grouping of 200 dB; a pink oval encompasses three data points. The way the SEL_{cum} was achieved is shown in each box, in order of decreasing risk of injury: 1) SEL_{cum} 200 dB [SEL_{ss} 183 dB, # 96]; 2) SEL_{cum} 200 dB [SEL_{ss} 180 dB, # 48]; 3) SEL_{cum} 200 dB [SEL_{ss} 177 dB, # 192]. The SEL_{ss} of 180 and 183 dB have comparable injury risks. However, the 177 dB SEL_{ss} treatment scenario indicates that more than double the impulses can occur with a much lower risk of injury; this can also be seen in Figure 15.

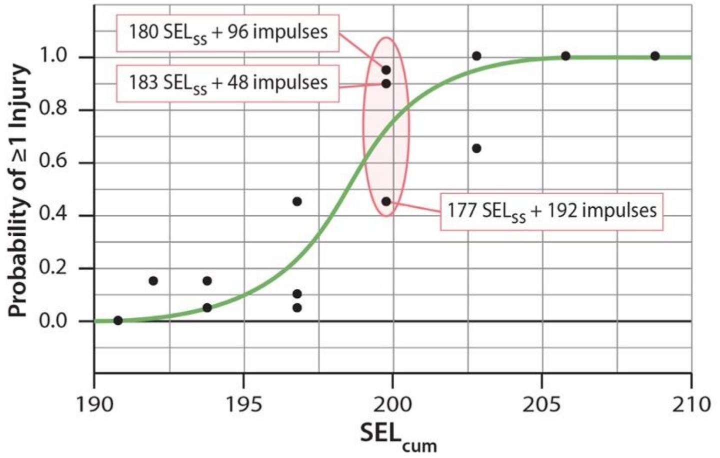

Figure 16: Probability of injury from pile driving.
Probability of injury as a function of SEL_{cum} (green curve). A pink oval encompasses three data points at 200 dB SEL_{cum} which shows that lower SEL_{ss} values can be presented with a higher number of impulses and result in a lower risk of injury. Each box depicts the SEL_{ss} and number of impulses for the data point it is nearest. (Modified from Casper et al. 2017)

Figure 17 shows the four-dimensional data in an easy way to visualize. All four parameters are on one graph, the SEL_{cum}, the SEL_{ss}, the probability of ≥ 1 injury, and the number of impulses. The call out on each bar is the SEL_{cum} value which was plotted in a diagonal arrangement for comparison. Using the example from Figure 16 and focusing on the SEL_{cum} of 200 dB based on different SEL_{ss} values and number of impulses, it is clear that at the higher SEL_{ss} values, a lower number of impulses causes a higher risk of injury.

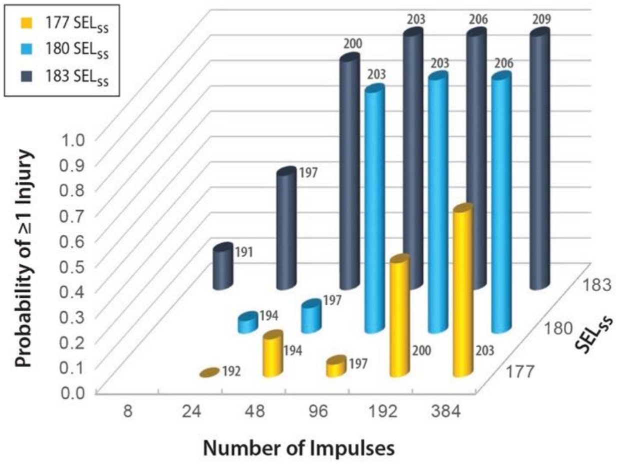

Figure 17: Comparison of probability of injury versus number of impulses and SEL_{ss}.
Data shown for each SEL_{cum}. The x axis is the number of impulses; y-axis is the SEL_{ss}; and z-axis is the probability of > 1 injury. Each bar is the SEL_{cum} value as identified by the ‘call out.’ The SEL_{cum} is organized in a diagonal comparison. (Based on data from Casper et al. 2017)

In summary, these data indicate that the $\mathrm{SEL}_{\mathrm{ss}}$ value drives the risk of injury, such that a fish could receive more impulses at a lower $\mathrm{SEL}_{\mathrm{ss}}$ (e.g. an $\mathrm{SEL}_{\mathrm{ss}}$ of 177 dB) and be at a much lower risk of injury than if those impulses were at a higher $\mathrm{SEL}_{\mathrm{ss}}$ (e.g., an $\mathrm{SEL}_{\mathrm{ss}}$ of 183 dB). In addition, all three metrics should be reported for comparability and understanding acoustic impacts on fishes across studies.

g. Summary of Pile Driving Studies

The studies that examined the onset of injury in fishes using the HICI-FT provided the first dose response curve for any fish, and the first quantified data on potential physical effects and recovery from pile driving sounds for several fish species. The results of the studies demonstrated the exposure levels that resulted in the onset of barotrauma and hair cell damage. Furthermore, it was shown that the number and severity of injuries increase with higher $\mathrm{SEL}_{\mathrm{ss}}$ values. The results also demonstrated the complexity of the effects with regard to the acoustic metrics, including the $\mathrm{SEL}_{\mathrm{ss}}$, how sound energy is cumulated, and number of pulses, and that physiological effects will likely never be described by a single metric or a simple formula of the relationship between the number of injuries and sound parameters.

In addition, the studies provided a good estimate of the sound pressure levels that result in onset of even minor injuries, and further demonstrated that onset levels vary by species. Although not certain, it is likely that major variables for injury onset are the swim bladder and how it is filled, and the size of the individual animal. Other morphological swim bladder characteristics that might be relevant include placement of the swim bladder and the rigidity of tissues surrounding the swim bladder. Moreover, the likelihood of injury seems to be lower in fishes without a swim bladder, which may extend to sharks and rays, even though only teleosts have been tested.

At the same time, it is recognized that there are other issues that need to be examined before there is a more complete understanding of potential physical effects of pile driving, most of which could be tested in a future iteration of the HICI-FT. The HICI-FT sound field generated plane-wave conditions comparable to 10 m from the pile source where the particle motion is correlated to the sound pressure. However, it is still not clear what would happen were fishes closer to the source in the acoustic near field with higher particle motion where the fish (and the associated otolith) receives a greater shaking. Under such circumstances, would there be damage to the sensory hair cells of the ear and to other tissues (Popper and Hawkins 2018)? Also for consideration is the coupling of the pile to the sea floor which causes substrate vibration (ground roll); would fishes near or in contact with the sea floor suffer from tissue damage as a result of the higher levels of particle motion (Hazelwood and Macey 2016b; Hazelwood and Macey 2016a)?

8. Other Pile Driving Studies

Other studies have investigated the potential of impulsive pile driving sound exposure to cause mortality or sub-lethal injury in fishes. However, these other studies were not as extensive, systematic, or controlled as those discussed above, nor did they use a variety of species.

Using an apparatus that was somewhat similar to the HICI-FT, Bolle et al. (2012; 2016) investigated mortality in larvae of common sole (*Solea solea*), a flatfish without a swim bladder, after exposure to impulsive pile driving signals at various SEL_{cum} values. The maximum exposure was SEL_{cum} 206 dB [SEL_{ss} 186 dB, # 100]. The investigators found no mortality and no tissue damage. This finding is not surprising when compared with the Halvorsen et al. (2012c) study showing the hogchoker had no observable barotrauma injury after exposure to SEL_{cum} 216 dB [SEL_{ss} 186 dB, # 960], much higher energy levels.

More recent studies with the same chamber were conducted by Bolle et al. (studies described in Andersson et al. 2017 but not yet published)[21] with larvae of sea bass and herring (both having swim bladders). Several different life stages were tested, but none of the species showed a difference in mortality between control and exposed animals. The sea bass were exposed to levels of up to SEL_{cum} 216 dB re 1 $\mu Pa^{2} \cdot s$ and SPL_{peak} 217 dB re 1 μPa, while herring were exposed to levels of up to SEL_{cum} 212 dB re 1 $\mu Pa^{2} \cdot s$ and SPL_{peak} 207 dB re 1 μPa. The results were compared with the earlier study in which the larvae of common sole were exposed to impulsive sounds (Bolle et al. 2012; Bolle et al. 2016). No change occurred during the 7 days (for the sole) or 10 days (for sea bass and herring) after the exposure. Together, the tested larvae represented the entire range of swim bladder shape types described by Popper et al. (2014). Thus, no distinction between the presence or absence of a swim bladder was observed, nor between fish with swim bladders connected to the esophagus and those without. The larvae of sea bass may be an appropriate surrogate for other larvae species, including the listed rockfish larvae; this study would be relevant to work being carried out in areas like Puget Sound.

The report on the work of Bolle et al. also discussed the effects of swim bladder resonance. This was something they suspected does not occur when their chamber is used. Instead, they made use of a theoretical model in which they considered the swim bladder as a gas bubble. From the theoretical study, they showed that the resonance effect was insignificant for swim bladders smaller than 2 millimeters (mm) (radius of the gas bubble). The swim bladders of fish larvae that were tested in the pressure chamber were considerably smaller than that. However, the relevance of the swim bladder's resonance was expected to be higher for bigger swim bladders or at higher levels of high frequencies, but this assumption was not tested experimentally.

In addition to the lab-based pile driving studies, a few field-based studies have examined the physical effects of actual pile driving operations on caged fishes (e.g., Caltrans 2001; Abbott and Bing-Sawyer 2002; Maes et al. 2004; Ruggerone et al. 2008; California Department of Transportation 2010a; California Department of Transportation 2010b). Most of these studies were reviewed by Popper and Hastings (2009) and add nothing to inform ideas about potential revision to the Popper et al. 2014 Guidelines. Indeed, as pointed out by Popper and Hastings (2009), these and other similar studies had methodological problems that may have (depending on the study)

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[21] The authors of this report have not been able to review this work, but, because it comes from a highly reputable group, we thought it important to discuss it here. The work is cited as:
Bolle, L.J., Blom, E., Halvorsen, M.B., Woodley, C.M., de Jong, C.A.F., Wessels, P.W., van Damme, C.J.G., Hoek, R., Winter, H.V., Woodley, C.M. Inskickat manuskript, a. Barotrauma injuries in European sea bass due to exposure to pile-driving sounds.
Bolle, L.J., de Jong, C.A.F., Blom, E., Wessels, P.W., van Damme, C.J.G., Winter, H.V. Inskickat manuskript, b. Do Pile-driving Sounds Cause Mortality in Fish Larvae?

included issues with controls, whether fish were properly acclimated to depth or had under-filled swim bladders, number of animals, signal parameters, animal handing, and necropsy methodology. Indeed, these issues, many of which are derived from the difficulty of doing field studies, prompted the funding that resulted in lab-based studies to examine the critical issue of onset of injury.

9. Physical Effects of Other Types of Sound Sources

a. Seismic Airguns

Seismic airgun arrays are most often used for offshore gas and oil exploration. Airguns produce a high-intensity impulsive signals that are of a similar frequency range to that of pile driving, although with a somewhat different rise time and a lower repetition rate (Laws and Hedgeland 2008; Gisiner 2016). However, the specific characteristics of seismic airguns vary depending on volume of the airgun and number of airguns in an array, as do the characteristics of pile driving sounds (Reyff 2012; Caltrans 2015; Reyff 2016).

Several studies have examined the potential physical effects of seismic airguns on caged fishes. In one study, McCauley et al. (2003) exposed pink snapper (*Pagrus auratus*) to several passes of a small (20 cubic inch [in^3]) airgun. All fish survived the exposure and they were then kept for about 58 days, after which their inner ears were examined for sensory hair cell damage. The fish showed significant damage to the saccules of the ears, suggesting potential hearing loss. No other tissues were examined.

In another study, the northern pike (*Esox lucius*), broad whitefish (*Coregonus nasus*), and lake chub (*Couesius plumbeus*) were exposed to 5 or 20 shots from a 730 in^3 airgun in the Mackenzie River Delta (Popper et al. 2005). No tissue damage was found (Popper et al. 2005) nor was there any damage to the inner ear sensory cells (Song et al. 2008) even with $\mathrm{SEL_{ss}}$ that averaged 177.7 dB re 1 µPa^2·s and $\mathrm{SPL_{peak}}$ of 207.3 dB re 1 µPa. However, both lake chub and adult northern pike showed TTS, which mostly recovered within 18 to 24 hours post-exposure.

Most recently, a study was conducted to examine the effects of exposure to a single airgun shot on endangered pallid sturgeon (*Scaphirhynchus albus*) and on paddlefish (*Polyodon spathula*) in Lake Sakakawea (North Dakota, United States) (Popper et al. 2016). Animals were in cages at different distances from the source. Maximum negative $\mathrm{SPL_{peak}}$ reached 231 dB re 1 µPa ($\mathrm{SEL_{ss}}$ of 205 dB). No tissue damage was found in any of the exposed animals.

b. Sonar

The only recent studies on high-intensity sources have been of the effects of low- and mid-frequency active sonars (Popper et al. 2007; Kane et al. 2010; Halvorsen et al. 2012b; Halvorsen et al. 2013). Low-frequency active (LFA) sonars are within the hearing range of most fish species, while mid-frequency active (MFA) are within the hearing ranges of fishes that hear above about 2 kHz. A very important difference between the low-frequency sonars and the impulsive sources is that the low-frequency sonars do not have short (impulsive) onsets and they are of longer duration (often several seconds).

Several different species including rainbow trout (*Oncorhynchus mykiss*) and channel catfish (*Ictalurus punctatus*) were exposed to LFA sonar for up to 648 seconds at an approximate SPL_{rms} of 193 dB re 1 µPa and an SEL_{cum} of 188.5 dB without any physical damage to either body tissues or inner ear sensory hair cells (Popper et al. 2007; Kane et al. 2010), although some fishes showed up to 20 dB TTS. Fishes exposed to mid-frequency sonar of SPL_{rms} 210 dB re 1 µPa and SEL_{cum} of 220 dB also showed no physical effects, although some of the channel catfish showed a small amount of TTS (Halvorsen et al. 2012b).

c. Explosions

Very little is known about potential effects of explosions on fishes. Most of the earlier studies were performed with poor controls and often had other problems in experimental design (Popper and Hastings 2009). Govoni et al. (2008) conducted a field study that examined the effects of shock waves from underwater explosions on the larvae of spot (*Leiostomus xanthurus*, 18.0–20.1 mm, with swim bladder) and pinfish (*Lagodon rhomboides*, 15.9–17.2 mm, with swim bladder). For the exposed spot, mortality increased by 100% at SEL_{ss} of 182–187 dB re 1 $\mu Pa^{2}\cdot s$ and SPL_{peak} of 229–236 dB re 1 μPa. In pinfish that were exposed to the noise levels SEL_{ss} 183–186 dB re 1 $\mu Pa^{2}\cdot s$ and SPL_{peak} 235– 239 dB re 1 μPa, mortality increased by 33–100%. The SEL_{ss} levels are comparable with those generated by pile driving, whereas the SPL_{peak} were higher than those normally generated by pile driving (Bolle et al. 2012). The study indicated that larvae are more susceptible to shock waves from an underwater explosion than are larger juveniles and adult individuals (Govoni et al. 2008).

10. Conclusions

In the years since the initial development of the 2008 Interim Criteria for effects of pile driving on fishes (Fisheries Hydroacoustic Working Group 2008), studies described in this Chapter have examined the potential for physical injury to fishes exposed to high-intensity impulsive sounds from pile driving. These studies clearly demonstrate that the onset of physical injury from pile driving signals begin at sound exposures that are at least 16 dB, and potentially 23 dB, higher than in the 2008 Interim Criteria.[22] Put another way, evidence using highly controlled studies since 2008 suggests that sound at the interim criteria levels will have no physical impact on fishes and is below the reported levels that document onset of physical impact. Moreover, although not directly applicable to pile driving, studies using other high-intensity sound sources, both impulsive and otherwise, support the idea that the onset of physical effects occurs at higher sound levels than those established by the 2008 Interim Criteria.

There is also strong evidence from studies with a number of different impulsive sources that the 206 dB peak level used in conjunction with, or as an alternative to, SEL measurements is likely well below the levels that will produce onset of physical effects (Popper et al. 2005; Bolle et al. 2012; Bolle et al. 2016; Popper et al. 2016). Thus, it is clear that, in terms of peak energy, onset of physical effects is only likely to occur at an SPL_{peak} that exceeds 206 dB peak.

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[22] It should be noted that even in 2008, suggestions were made, but rejected, for higher criteria than ultimately approved by the FHWG, even though the data available then were not as strong as those obtained since 2008 (Hastings and Popper 2005; Popper and Hastings 2009).

In addition, the recent studies from both the United States and Europe demonstrate that the equal energy hypothesis does not apply to impulsive pile driving signals (see page 78). There is also strong evidence that, unlike in the 2008 Interim Criteria, larger fishes may be at a slightly higher risk of injury from impulsive pile driving exposure than are smaller animals, although this finding needs further investigation with additional species that represent wider size differential and age classes.

There is also evidence that hair cell damage does not occur until the sound exposure is at higher levels than those that cause onset of damage to other tissues. However, how this observation relates to TTS is not yet fully determined. It is possible that TTS results from a minor form of impact on the cilia of hair cells, not necessarily hair cell death. This topic needs to be re-examined to clarify the underlying mechanisms and the study replicated using different species. The studies should investigate fishes with and without swim bladders or other air bubbles that come close to the ear and which may directly damage inner ear tissues.

Furthermore, studies performed to date have primarily focused on signals that are found at some distance from the source (the acoustic far field) where sound pressure is the dominant component of the sound field. Closer to the source, however, in the acoustic near field, particle motion dominates, and this shaking may have different effects on the fish. Indeed, considering that the inner ears of fishes are particle motion detectors, it is possible that impulsive stimulation where particle motion is greatest may directly cause severe damage to the ears, therefore having a substantial effect on hearing. Moreover, particle motion associated with the substrate vibrations from a pile coupled to the sea floor will occur at greater distances from the source; therefore, particle motion remains a high priority concern for animals that live in close proximity to the substrate.

Finally, a critical caveat in performing studies on the effects of sound sources on fishes is that the fishes must be physiologically acclimated to the experimental depth so that the swim bladder is normally inflated, (i.e., the fish is neutrally buoyant). Studies with overinflated or underinflated swim bladders do not duplicate the normal state of fishes, and the degree of buoyancy will influence how the swim bladder walls move, and the severity of the resultant damage to surrounding tissues.

CHAPTER 8: PHYSIOLOGICAL EFFECTS

1. Overview of Physiological Effects

Beyond the issues of physical effects (Chapter 7), behavior (Chapter 5), and hearing (Chapter 5), there are concerns of potential effects of anthropogenic sounds on other aspects of physiology that could have long-term consequences for fitness (e.g., Wright et al. 2007; Slabbekoorn et al. 2010; Kight and Swaddle 2011; Weilgart 2017). These effects are generally referred to as *stress*. For purposes of this report, *stress* can be defined as a state of biological strain or tension resulting from adverse circumstances. In such circumstances, animals, including fishes, may show hormonal, autonomic, immune, and behavioral responses that may initially allow them to adapt to adverse conditions. However, some stressors may change the state of physiological processes and affect homeostasis, thus having an adverse effect on the animals' health and well-being.

Indeed, the addition of anthropogenic noise to the environment is well known to result in a variety of stress and other health effects in humans and other animals, including deficits in sleep (e.g., Gourévitch et al. 2014; Tennessen et al. 2016; Foraster et al. 2017; Murphy 2017; Weilgart 2017). There is every reason, therefore, to assume that the same kinds of effects seen in other vertebrates will also be encountered in fishes. For example, Weilgart (2017) suggested that non-hearing effects of noise on marine animals, such as stress, may be as, or more, severe than hearing effects (also see de Soto 2016). Moreover, even temporary exposures to stressors in early life stages can have health and reproductive consequences later on in life (Donaldson et al. 2011; Kight and Swaddle 2011).

Although there is a limited number of studies of these physiological effects of sound on fishes, most have been done with exposure to continuous sounds (e.g., Wysocki et al. 2007; Buscaino et al. 2010; Bruintjes and Radford 2014; Nedelec et al. 2015; Sierra-Flores et al. 2015) and only rarely with impulsive noise (e.g., Debusschere et al. 2016b).

Studies on captive fishes exposed to relatively short-term continuous white noise or simulated boat sounds have shown an increase in secretion of cortisol, a stress hormone, as well as other physiological effects (e.g., Smith et al. 2004a; Wysocki et al. 2006; Buscaino et al. 2010; Anderson et al. 2011; Crovo et al. 2015; Sierra-Flores et al. 2015; Simpson et al. 2015; Celi et al. 2016). In all cases reported, stress hormone levels returned to normal after cessation of the sound. Moreover, one study showed cortisol changes after short exposure but none after somewhat longer exposure (Spiga et al. 2012).

The European eel has been the subject of several laboratory-based studies. Eels exposed to the playback of ship noise in an aquarium tank showed elevated ventilation and metabolic rates (indicators of stress) compared with control individuals (Simpson et al. 2015). Simpson et al. (2015) suggested that acoustic disturbance could have important physiological and behavioral impacts on animals, compromising life-or-death responses. In later laboratory experiments, Bruintjes et al. (2016a) found that exposure to ship noise decreased European eel anti-predator responses, increased startle latency, and increased ventilation rates relative to ambient-noise-exposed controls. However, those effects quickly dissipated. Indeed, European eels showed rapid recovery of startle responses and startle latency, and rapid, albeit incomplete, recovery of ventilation rate in the two minutes after noise cessation.

Purser et al. (2016) demonstrated that the effects of noise varied with the condition of the individual European eel. They showed that additional continuous noise caused an increase in ventilation rate and a decrease in startle responses in poor condition eels. They concluded that intra-population variation in responses to noise has important implications both for population dynamics and the planning of mitigation measures.

Even though most of the studies to date show some physiological changes (at least after relatively short exposures), there are still so few data that it is not possible to come to any general conclusions regarding real or potential effects of anthropogenic sound of any kind on fish stress levels. Moreover, there are significant issues for all of the studies showing effects, starting with the fact that the studies were performed in enclosed areas where the acoustics were not properly calibrated or measured, and where the acoustics had no relationship to the sounds that a fish would normally encounter in the wild, as discussed on page 25 (Popper and Hawkins 2018).

As a consequence, one has to be very careful in extrapolating to free-swimming wild animals that are potentially capable of moving away, especially when compared with more lab-oriented species such as zebrafish and goldfish. In addition, measuring of stress levels through blood assays needs to be done with considerable care, as handling the fish and other factors have the potentially to significantly affect results (Lawrence et al. 2018; Sadoul and Geffroy 2019).

2. Long-Term Exposure to Sounds

Other potential noise-induced effects of sound may, like other environmental stressors, have an effect on fish reproduction and growth (Pickering 1981) but there are very few recent experimental data to test this idea. One such study looked at growth, health, and hearing over a long period in fish and was conducted in an aquaculture facility on the rainbow trout, a salmonid (Wysocki et al. 2007). In this study, fish were continuously exposed to sounds typically found in an aquaculture facility for nine months as the fish were raised from newly hatched larvae. The study, in which noise-exposed fish were compared with control animals treated identically but which were in tanks that were quietened using a variety of methods (Davidson et al. 2007), showed that the sounds had no impact on growth, survival, or susceptibility to disease (a very sensitive measure of stress), even over nine months of exposure.

A study by Sierra-Flores et al. (2015) examined the effects of noise exposure on adult Atlantic cod in an aquaculture facility. The investigators found that short-term noise exposure can trigger physiological stress responses that included significant increases in plasma cortisol levels within 10 to 20 minutes of the start of noise exposure, although cortisol levels were not nearly as high as produced in the same species from things like capture and confinement. The same investigators also looked at the effects of exposure to six one-hour-long noise events daily over several months on Atlantic cod brood stock. The investigators found that noise-exposed fish had a shorter spawning period than did controls that did not receive the sound, although they produced the same total egg volume. However, while the number of eggs spawned were the same between experimental and control animals, fewer eggs from the noise-exposed animals were fertile, suggesting less courtship as a result of the sound exposure.

In contrast, a study on the daffodil cichlid (*Neolamprologus pulcher*) showed no effect of four weeks of playback of motorboat noise on hatching success of eggs, fry survival, or growth and development as compared with controls from the same genetic stock (Bruintjes and Radford 2014). Yet another species, the lined seahorse (*Hippocampus erectus*), subject to chronic noise exposure for one month, did show substantial effects on weight and growth as well as changes in plasma cortisol levels and other stress effects (Anderson et al. 2011).

Comparison with the aforementioned trout and cichlid studies is impossible because there were so many differences in variables, from species, to the size of tanks, to the ways sounds were presented and the sound field in the tanks. The significance of comparing the studies highlights the difficulties in doing physiological studies and the need to do more of them under similar conditions in order to really understand stress effects resulting from noise (See also Bruintjes and Radford 2014).

3. Impulsive and Intermittent Sounds

One study has examined the effects of impulsive sounds on fish physiology (see Chapter 7 on physical effects). Debusschere et al. (2016b) examined effects of pile driving during offshore wind farm construction in young European sea bass. Fish were placed in glass 500 milliliter vials and submerged to about 2.5 m depth at a distance of 45 m from the pile being driven. During several different trials, fish were exposed to sounds for 1.5 hours to from 1,739 to 3,067 strikes, with SEL_{ss} from 181 to 188 dB re 1 µPa^2·s and an SEL_{cum} ranging from 215 to 222 dB re 1 µPa^2·s. The fish were then returned to the ship for study.

The investigators found no change in cortisol levels, but there were some changes in whole-body lactate levels and oxygen consumption. Although the investigators concluded that exposure to pile driving may have had some physiological effects on fish, and suggested decrease in fitness as a result, there is a number of issues that lead to the need for replication of the work.

One concern with the study was the location of the fish and the fact that the animals were restrained and could not, as might normally be expected of an animal exposed to such high sound levels, move away. Moreover, the fishes were at a depth shallower than at which they would normally live, and this could have resulted in stress effects. However, the authors only examined the animals in terms of sound pressure and not particle motion. This is of concern because the studies were done with the animals close enough to the surface and sufficiently within the acoustic near field of the signal that particle motion would account for a very significant part of the sound as opposed to what might happen had the fish been farther from the source.

With a different type of sound, a biochemical stress response was exhibited in caged European sea bass when a seismic survey (2,500 in^3) passed by at distances from 180 m to 6,500 m (Santulli et al. 1999). Cortisol in the plasma, muscle, and liver all increased significantly after exposure to seismic airgun noise. Other biochemical measures (e.g., glucose, lactate) also showed a primary (e.g. plasma cortisol) and secondary (e.g. blood glucose and other blood measures) stress response even at distances of 2 km from the seismic survey. Most biochemical values returned to pre-exposure levels after 72 hours.

In a lab-based study, Radford et al. (2016a) exposed juvenile European sea bass to playbacks of pile driving sounds and seismic sounds in a series intended to examine how an initial impact of different sound types potentially changes over time. European sea bass not previously exposed to high sound levels, showed elevated ventilation rates, indicating heightened stress. Short-term exposure to pile driving noise resulted in a significantly greater increase in ventilation than did short-term exposure to either ambient noise or ship noise.

Nichols et al. (2015) examined effects of both continuous and intermittent boat noise on the giant kelpfish (*Heterostichus rostratus*). Although the study was done in tanks, with all the associated acoustics issues, the results showed increased cortisol levels only in fishes exposed to intermittent noise and not to those exposed to continuous sounds of the same levels (or to controls without any sound). The investigators also found that the change in cortisol levels increased linearly as the sound level of the random intermittent noise was increased over a range of about 16 dB. However, because each exposure lasted for 60 minutes, with the intermittent sound being on for about 40% of the time, it is not known whether animals would have acclimated to sounds over a long time period, as found by Spiga et al. (2012).

4. CONCLUSIONS

The results of studies on physiology, as measured by various stress parameters, are highly variable and are not particularly instructive with regard to how exposure to pile driving or other intermittent sounds might impact fishes. In particular, all of the studies, including both long- and short-term exposures, were done on captive animals in enclosed areas where the fish could not avoid the sounds. Thus, the acoustics were different than the acoustics an animal would encounter in the wild, and the fish could not move away from the disturbing sound; therefore, it is possible that it was not the sound itself that resulted in the stress response, but the inability of the animals to move away from the sound. Indeed, although all of the studies included a quiet control, it is possible that an additional control for constraint of movement from any noxious stimulus would be needed to determine if it is the sound, and not the general nature of the experiment, that is the causal factor.

Similarly, the results from the study on pile driving sounds (Debusschere et al. 2016b) are inconclusive because the work was only on one species that was constrained in tiny chambers. How these results would compare to those from free-swimming animals, and even animals that stay in place but are capable of free swimming, as opposed to animals whose normal response is to swim away, is not clear. Moreover, the work was with juvenile fish and it is not, at this point, possible to extrapolate results to any other life stages where stress responses will, in general, differ considerably from those in juveniles.

In conclusion, the few results to date suggest that, perhaps, animals in the wild exposed to sounds they cannot escape, such as increased background noise levels in a harbor or in the vicinity of a coral reef where fish do not leave the reef, may have some impact on stress levels. However, direct studies on such animals are needed. Currently, nothing is known about the potential impact on stress of short-term continuous or intermittent sounds, especially when fish can move away.

At the same time, it is well known that other animals and humans show stress responses to noise, and that the effects are greater as a result of intermittent sounds than for continuous noise (e.g., Campeau and Watson 1997; Gourévitch et al. 2014). Thus, if fishes follow the patterns found in other vertebrates, then it is likely that anthropogenic sounds added to the aquatic environment could have negative impacts on the physiology of fishes.

CHAPTER 9: CURRENT STATUS OF CRITERIA AND GUIDELINES

Interim sound exposure criteria for the onset of physiological effects on fishes, for use on the United States West Coast, were proposed by the FHWG (2008)[23]. Some of the discussions and history of the project are commented upon in several papers (e.g., Popper et al. 2006; Carlson et al. 2007; Woodbury and Stadler 2008; Stadler and Woodbury 2009). More recently, a new set of interim criteria and associated Guidelines were proposed (Popper et al. 2014) based on much stronger research studies (Chapter 7). These Guidelines raised the effective onset-of-effects sound levels, at least for physical effects, substantially. In this chapter, we first briefly review the 2008 Interim Criteria and provide some insight into the issues associated with their continued use. We then discuss the newer set of suggested interim criteria, as well as criteria and guidelines from European sources. Following this discussion are thoughts about potential updated interim criteria and guidelines based on the most recent data, as discussed in the first seven chapters of this report.

1. Current Interim Criteria

a. Onset of Physical Effects

The interim sound exposure criteria, which are still in use (e.g., Caltrans 2015)[24], were based on a recommendation of dual criteria of SPL_{peak} and SEL_{cum} (Popper et al. 2006; Carlson et al. 2007; Popper and Hastings 2009).

The rationale for dual criteria was that it was sometimes difficult to determine one or the other measure when trying to set a signal level for onset of an effect, and having alternative approaches provides a more conservative guideline for the protection of the animals. The SEL_{cum} was suggested because animals are often exposed to many pile driving strikes in succession, and any effect would likely come from an accumulation of energy from the multiple strikes.

Carlson et al. (2007) argued that, because it was unlikely that the effects of sound on fishes would follow the equal energy hypothesis (see page 78), it is important to include both SPL_{peak} and SEL_{ss} in setting any interim criteria. In fact, more recent experimental work strongly supports the rejection of the equal energy hypothesis (e.g., Halvorsen et al. 2012a; Casper et al. 2017) (also see page 78).

In 2008, the FHWG adopted the interim dual-criteria model for sound exposure that comprised:

SPL_{peak}: 206 dB re 1 µPa;
SEL_{cum}: 187 dB re 1µPa^2·s for fishes above 2 grams; and
SEL_{cum}: 183 dB re 1µPa^2·s for fishes below 2 grams.

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[23] http://www.dot.ca.gov/hq/env/bio/fisheries_bioacoustics.htm

[24] http://www.dot.ca.gov/hq/env/bio/files/bio_tech_guidance_hydroacoustic_effects_110215.pdf

At the time these criteria were developed, there was very limited scientific research on the topic of pile driving effects on fishes. The criteria were intentionally called “interim” because it was understood by all parties that the criteria were based on limited scientific information and would need to be updated as new research emerges. The interim criteria have been criticized because they were based on limited experimental data that were not rigorously obtained, and studies often were not peer-reviewed and frequently did not include proper controls (e.g., Carlson et al. 2007; Popper and Hastings 2009).

A major issue with the 2008 Interim Criteria, as discussed here, is that they are based on early studies and do not take account of more recent studies discussed in the 2014 Guidelines paper. Indeed, the 2008 Interim Criteria levels are at least 16 dB below the most recent onset of effects data (Chapter 7 and Table 3, page 100).

The only similarity between the 2008 Interim Criteria and the 2014 Guidelines is for onset for TTS – 187 dB SEL_{cum}. But even here, the potential for TTS at this level is very low because there is likelihood of considerable variation in TTS onset levels depending on many physiological factors, including the hearing sensitivity of the species of interest. Thus, the 2014 Guidelines provide conservative levels of sound that might result in TTS in several species and suggest that change in hearing sensitivity should be a minimum of 6 decibels (dB) to be considered TTS. However, even the levels for onset of TTS in the 2014 Guidelines must be taken as extremely tentative because they are based on data from only three species, while TTS could not be induced in other species even with very high sound levels.

b. Onset of Behavioral Effects

NOAA Fisheries currently uses 150 dB re 1 µPa as the RMS sound pressure level that may result in onset of behavioral effects (Caltrans 2015). The NOAA Fisheries Hydroacoustics Biological Assessment Guidance document[25] considers that sound pressure above the 150 dB SPL_{rms} level are expected to cause temporary changes in behavior, which might include startle response (although startle is not defined and has broad meaning to fish biologists), feeding disruption, and area avoidance (see footnote on page 25).

There are substantial problems with the 150 dB SPL_{rms} criterion. First, the origin of this criterion is unknown (Hastings 2008). That is, NOAA Fisheries and other agencies use the value but never document the scientific basis for this, or any other, value for the onset of potential behavioral effects.

Second, the criterion is based on sound pressure detection while, as documented in Chapter 3 of this Report, most of the species of interest, including salmonids, are primarily detectors of particle motion (see also Popper and Hawkins 2018). Thus, any behavioral criteria should be based on the acoustic signals that the fish can actually detect and respond to. It is important that new studies be designed to examine behavioral responses to different levels of particle motion, especially studies for species that only detect particle motion.

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[25] http://www.dot.ca.gov/hq/env/bio/files/hydroacoustic_bio_assessment_guidance_revised_2017.pdf

Finally, and perhaps most importantly, a single criterion value for the onset of effects on behavior does not take into consideration the very substantial species differences in hearing sensitivity or behavior, nor does it take into consideration response changes with animal age, season, or even motivational state (see also Neo et al. 2014). Despite these caveats, it is important to establish provisional or interim criteria for behavioral effects. Recent studies of the effects of pile driving and other impulsive sound stimuli on the behavior of fishes, described in Chapter 6, may be relevant to the setting of criteria. Figure 10 in that chapter utilized a sound pressure level of 163.3 dB re 1 µPa peak-to-peak for the behavioral response of fishes, based on dose/response relationships derived from field experiments on sprat and mackerel. The chosen level was based on the level to which 50% of the fish schools responded. Interim criteria for behavioral effects could initially be based on these findings until new scientific data are available from field studies on a range of species.

c. Pile Driving Calculator

Currently, NOAA Fisheries relies on a set of equations and an Excel-based calculator[26] to determine the exposure of fishes to pile driving (described in Stadler and Woodbury 2009). This calculator is used to determine the distance from a pile where the SPL_{peak} and the SEL_{cum} drop below the current threshold values and to determine the area around the pile that is ensonified above those levels. The assumptions for the calculator are discussed by Stadler and Woodbury (2009) and include the SEL_{ss}, size of the pile being driven, number of strikes, and distance from the source. The calculations assume that all strikes have the same SEL_{ss}. Because the model (Woodbury and Stadler 2008) also assumes that fishes are stationary, the model does not account for any change in their actual exposure during a pile driving operation (e.g., Krebs et al. 2016). In addition, the model does not consider potential recovery from effects during the time between strikes.

An important problem with the NOAA Fisheries Pile Driving Calculator is its approach to modeling sound propagation, and thus the determination of the ensonified area in which fish are exposed to sound levels that exceed the interim criteria. Although recognizing that propagation is complex and depends upon things like water depth and substrate. (Stadler and Woodbury 2009), these issues are not considered in the distance part of the equation, and the calculator uses a default attenuation rate of 4.5 dB per doubling of distance, although no basis for the use of this default attenuation is provided. Indeed, a recent analysis of a number of propagation models for pile driving suggests that the use of this constant is not correct (Lippert et al. 2018).

Moreover, Stadler and Woodbury claim that use of this constant will tend to overestimate the area being ensonified. Indeed, as discussed in a more recent modeling of sound propagation from pile driving on the Hudson River, the extent of sound propagation, and the attenuation over distance from the source, can vary not only in different directions from the source, but also as the sound travels in any one direction, with water depth and substrate parameters affecting propagation (MacGillivray et al. 2011; Martin et al. 2012). Clearly the modeling of underwater noise propagation from pile driving activities is far more complex than can be represented with a simple calculator. The nature of the bathymetry and bottom characteristics play a major role in actual results, although Lippert et al. (2018) demonstrated that the propagation simplifies close to the

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[26] Available at: http://www.dot.ca.gov/hq/env/bio/files/NMFS%20Pile%20Driving%20Calculations.xls

pile, where damped cylindrical spreading occurs. The simple NOAA Fisheries model was designed to be conservative in an attempt to account for many complex factors that a simple model cannot address. Pile driving propagation may be too complex to be dealt with by a single model to be used over a wide geographic range.

Moreover, although the importance of sound emanating from the substrate is recognized in the papers discussing the modeling, the calculator does not take substrate transmission into consideration (Stadler and Woodbury 2009). It is clear that the substrate characteristics are very critical for the assessment and prediction of propagation in shallow waters (see page 30) (MacGillivray et al. 2011; Hazelwood 2012; Hazelwood and Macey 2016b; Hazelwood and Macey 2016a). Indeed, as shown in the Hudson River, other factors such as the presence of vessels associated with construction can also significantly affect sound propagation from a pile (Martin et al. 2012).

At the same time, it is understood that the current calculator is simple, and, as such, it can quickly be applied to projects in varying locations and site conditions. Given the complexity of sound propagation, a simple calculator is an important tool for ESA biologists (including those with NOAA Fisheries and USFWS) who typically are not acoustic experts. Accordingly, it would be appropriate to engage experts in acoustic propagation and modeling to examine the calculator, which was developed more than 11 years ago, to see if there is a way to incorporate new information and knowledge while retaining the calculator's ease of use.

2. Recent Criteria and Guidelines

In 2009, NOAA Fisheries recognized that additional criteria were needed for both fishes and turtles, particularly based on new peer-reviewed data in the literature. NOAA Fisheries convened a panel of international experts on effects of sound on fishes, turtles, and marine mammals co-chaired by Drs. Arthur N. Popper and Richard R. Fay. This group, later supported by the Office of Naval Research, the National Science Foundation, the Oil and Gas Producers Association, and other agencies, examined the complete literature up to late 2013 and developed a set of guidelines for potential effects of anthropogenic sound on fishes and turtles, based on the best available science at that time (Popper et al. 2014).[27]

Significantly, these 2014 Guidelines, which are still considered interim because it is expected that new data will develop subsequent to the report, was produced under the auspices of ANSI through the Acoustical Society of America. The 2014 Guidelines were extensively peer-reviewed and were published as an ANSI accredited report.

In developing the 2014 Guidelines, the authors realized that it was not possible to define sound exposure criteria for every possible sound source or type of response to the sound, or to do an analysis for every fish species (or even all of those potentially listed in various locales). Instead, the authors developed an approach that focused on fish groups based on morphology of auditory apparatus (Table 2, page 50), on major sound types (e.g., pile driving, shipping), and major potential effects (Table 1, page 18).

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[27] Disclosure: Drs. Halvorsen, Hawkins, and Popper were members of this group and co-authors of the published guidelines.

Using these differentiations, the authors then analyzed each type of sound and their potential effects, based on the most recent data, on the various fish “types.” The authors concluded that the 2014 Guidelines for pile driving were the strongest, because the 2014 Guidelines were based upon recent research studies on effects of pile driving on several different fish species including Chinook salmon (Table 3), as reviewed in Chapter 7.

Table 3: Pile driving criteria and guidelines.

Suggested criteria and 2014 Guidelines (from Popper et al. 2014) for the onset of each of the potential affects. Guidelines are based on the best available data until 2014. Data for mortality and the onset of injuries that are recoverable are from papers cited in Chapter 7 and are based on 960 sound events at 1.2 s intervals. TTS based on Popper et al. (2005). See text for details. See text for discussion of “recoverable injury.”

Note that the same peak levels are used both for mortality (A) and recoverable injury (B) because the same SEL$_{ss}$ was used throughout the pile driving studies. Notes: peak and RMS sound pressure levels dB re 1 µPa; SEL dB re 1 µPa^{2}·s. All criteria are presented as sound pressure even for fish without swim bladders. Relative risk (high, moderate, low) is given for animals at three distances from the source defined in relative terms as near (N), intermediate (I), and far (F). Note that “near” might be considered to be within tens of meters of the source, “intermediate” within hundreds of meters, and “far” within thousands of meters.

Type of Fish	Mortality and potential mortal injury	Impairment			Behavior
		Recoverable injury	TTS	Masking	
Fish: no swim bladder (particle motion detection)	>219 dB SEL$_{cum}$ or >213 dB peak	>216 dB SEL$_{cum}$ or >213 dB peak	>>186 dB SEL$_{cum}$	(N) Moderate (I) Low (F) Low	(N) High (I) Moderate (F) Low
Fish: swim bladder is not involved in hearing (particle motion detection)	210 dB SEL$_{cum}$ or >207 dB peak	203 dB SEL$_{cum}$ or >207 dB peak	>186 dB SEL$_{cum}$	(N) Moderate (I) Low (F) Low	(N) High (I) Moderate (F) Low
Fish: swim bladder involved in hearing (primarily pressure detection)	207 dB SEL$_{cum}$ or >207 dB peak	203 dB SEL$_{cum}$ or >207 dB peak	186 dB SEL$_{cum}$	(N) High (I) High (F) Moderate	(N) High (I) High (F) Moderate
Eggs and larvae	>210 dB SEL$_{cum}$ or >207 dB peak	(N) Moderate (I) Low (F) Low	(N)Moderate (I) Low (F) Low	(N) Moderate (I) Low (F) Low	(N) Moderate (I) Low (F) Low

Wherever possible, the table uses numerical values for onset of the effect, expressed in appropriate metrics. However, numerical criteria were only proposed where acceptable data on received sound levels were available. Where there were insufficient data to set even interim criteria, the authors chose to give subjective estimates of the likelihood of there being effects at different distances from the source for different types of fishes. The authors recognized, however, that these estimates were arbitrary, and very much depended upon the source level. New information is needed for the setting of valid criteria. The authors also pointed out that these criteria were still interim and that the criteria would need to be re-examined and refined as gaps in knowledge were filled (Popper et al. 2014; Hawkins et al. 2015).

Furthermore, it is important to note that setting thresholds for a response by the animal, whether that response consists of physical changes (e.g. tissue injury), physiological changes[28], hearing loss, or behavioral changes, relies on the determination of dose response relationships. Setting the thresholds may involve observations on the changes in effects on individual animals caused by differing levels of exposure (or doses) to particular sounds. They may also involve monitoring the proportion of animals that respond in a specific way to different doses. As the sound level increases, there may be graded or incremental change in the magnitude of the individual response, or the proportion of animals that respond. In other cases, there may be a sudden change in the response or the numbers responding. In every case it is necessary to seek a particular dose level, which may serve as a criterion for defining a response threshold (Dunlop et al. 2012; Williams et al. 2014). There is currently a lack of dose response data for behavioral or stress related effects resulting from exposure of fishes and invertebrates to noise, perhaps because so few species are protected by statute.

a. Application of 2014 Criteria by Agencies

A critical point of understanding in Table 3 (page 100) is the nature of *recoverable injury* as defined by the authors of the 2014 Guidelines (their Table 7.1). That definition states that recoverable injuries are: “injuries, including hair cell damage, minor internal or external hematoma, etc. None of these injuries are likely to result in mortality.” The authors of the 2014 Guidelines and the papers cited in Chapter 7 do not define onset as anything but the first occurrence of an injury. However, as pointed out by the papers cited in Chapter 7 (e.g., Casper et al. 2012b; Casper et al. 2013a), the levels proposed for recoverable injury are for onset of two recoverable injuries — a level that the authors deemed acceptable for these very minor effects (e.g., minor hematoma at the base of the fins).

It seems, however, that currently agencies believe onset to be the start of a single injury, something not examined in the work to date. At the same time, the data presented in Chapter 7 provide insight and show that no injury whatsoever result at levels a few dB below the criteria for onset of injuries recognized in the 2014 Guidelines. However, these levels vary based on species, fish size, and whether there is a swim bladder present. Thus, although it may not be acceptable to agencies to use the specific criteria recommended in the 2014 Guidelines (Table 3, page 100), these interim criteria can serve as a very good guide, particularly when combined with data in Chapter 7, for determining the onset of any effects. At the same time, it is also important to recognize that there is inter-species variability in the onset of effects, as discussed in Chapter 7. We suggest that a sensible approach might be for each consultation to use onset criteria adjusted for the species differences shown in Table 3 (page 100) and discussed in the 2014 Guidelines and in Table 2 (page 50). With such an approach, if a species of concern has no swim bladder then the onset criterion for a consultation might be a few dB higher than the criterion for a species that has a swim bladder that is involved in sound detection. At the same time, it is very clear that any

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[28] It is often hard to distinguish between physiological and physical effects because they may be intertwined. For example, a physical effect on the kidney may result in physiological changes as well, whereas a physiological effect on the kidney may result in physical effects elsewhere in the body. In the literature on effects of noise on aquatic animals, the terms “physical” and “physiological” are often used interchangeably (see also Hawkins and Popper 2018b).

recommendations for onset of effects must be far closer to those recommended in the 2014 Guidelines than to the 2008 Interim Criteria.

b. Additions to the 2014 Guidelines

More recently, it has been pointed out acoustic impact assessments carried out on fishes should take into consideration not only sound pressure, but the potentially high levels of particle motion (Hawkins et al. 2015; Hawkins and Popper 2016; Nedelec et al. 2016; Popper and Hawkins 2018). In addition, there is growing international awareness that fishes do possess particle motion receptors, and that this must be taken into account in setting future criteria, once appropriate data are available.

However, several things stand in the way of incorporating particle motion into future studies. For example, currently there are no particle motion measurement standards, the cost of instrumentation to measure particle motion is relatively high, and there is a lack of sound exposure criteria for particle motion, as discussed in Chapters 2 and 3 (see e.g., Popper and Hawkins 2018).

Thus, it is important to develop protocols for monitoring particle motion, and then to determine those levels of particle motion that have potentially adverse effects in terms of increased mortality, injury to tissues, effects on hearing abilities, or changes in behavior and physiology. It should be noted that although development of the protocols is relatively straight-forward, protocols should be developed under the guidance of an international standards body (e.g., ISO). Developing the actual data upon which to base criteria for particle motion will be time consuming. Such studies will also require funding for research specifically directed at the relevant questions.

3. International Guidelines for Fishes

a. European Commission

The monitoring of underwater noise is included in the European Commission's Marine Strategy Framework Directive (MSFD)[29], which is concerned with ensuring Good Environmental Status (GES) of European waters (Tasker et al. 2010; Tasker et al. 2012; Van der Graaf et al. 2012; Dekeling et al. 2016; Andersson et al. 2017). The MSFD requires that the introduction of energy, including underwater noise, must be at levels that do not adversely affect the marine environment, and that this requirement must be implemented by 2020. Noise is listed as one of 11 descriptors for the assessment of GES within European waters because noise is considered a pollutant that, depending on source and intensity, may degrade habitat and have adverse effects on marine life, ranging from disturbance of communication or group cohesion to injury and mortality. No specific criteria are provided. Instead, indicators for achieving GES are specified.

A task group was established to provide scientific and technical support to the European Commission for the conception, development, implementation, and monitoring of noise and other forms of energy (Tasker et al. 2010). A distinction was made between loud impulsive sounds and

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[29] http://ec.europa.eu/environment/marine/eu-coast-and-marine-policy/marine-strategy-framework-directive/index_en.htm

ambient (continuous) noise. The task group recommended that high amplitude impulsive anthropogenic sounds within a frequency band between 10 Hz and 10 kHz should be assessed using either sound energy over time (SEL) or peak sound level of the sound source. Such sounds would include those from pile driving and seismic surveys. The indicator recommended for achieving GES for such impulsive sounds was based on the proportion of days within a calendar year, over a specified area of the sea in which anthropogenic sound sources exceed either of two levels, 183 dB re 1 $\mu Pa^2 \cdot s$ SEL or 224 dB re 1 μPa SPL_{peak} when extrapolated to one meter, measured over the frequency band 10 Hz to 10 kHz. This indicator would be based on reports of occurrence by those undertaking activities likely to generate these sounds, rather than on direct independent measurements.

In contrast, the indicator recommended by Tasker et al. (2010) for continuous noise was that noise within the 1/3 octave bands 63 and 125 Hz (center frequency) should not exceed the baseline value of 100 dB (re 1 μPa RMS; the average noise level in these octave bands over a year). This indicator would be based on direct independent measurements.

The task group emphasized that these indicators were only initial suggestions. The group identified a number of further needs for research and development to make the indicators for inputs of energy more attuned to the needs of the marine environment (Tasker et al. 2010).

Van der Graaf et al. (2012) provided additional technical advice and options for the implementation of MSFD Descriptor 11 on Underwater Noise and other forms of Energy. However, the paper is not an official opinion of the European Commission, nor of the participating Institutions and European Union Member States. The document points out that the impact that is addressed by the Impulsive Noise Indicator is “considerable” displacement of animals. This means the displacement of a significant proportion of individuals for a relevant time period and spatial scale. The indicator also addresses the cumulative impact of sound-generating activities and possible associated displacement, rather than that of individual projects. The initial purpose of this indicator would be to assess the pressure upon animals, which was meant to provide an overview of all loud impulsive low- and mid-frequency sound sources throughout the year and through areas.

A necessary follow-up in future years would be to evaluate effects on biota and set sound exposure criteria. Van der Graaf et al. (2012) also stressed that any noise modeling should ideally be done together with *in-situ* measurements. The use of modeling would strengthen analysis by overcoming bias introduced by changes in human activities or by the natural variability of the environment and would extend the monitoring to poorly or uncovered areas. The document noted that Member States were required to review their marine strategies by 2018, six years following the initial establishment of the MSFD. It was suggested that impulsive sounds should be given priority for the development of indicators of GES.

b. Swedish Environmental Protection Agency

The Swedish Environmental Protection Agency has recently published “A framework for regulating underwater noise during pile driving” (Andersson et al. 2017). The framework is based on reviews of research papers and provides a sound basis for environmental impact assessments. The framework aims to define the noise levels that can cause injury and other negative effects and, on this basis, recommends noise levels that can be used to establish guidance values for regulating underwater noise for Swedish waters and species. It considers effects (injury and behavioral effects such as flight, but not subtle effects) on representative fish species such as the Atlantic cod and Atlantic herring, and on fish larvae and eggs. It also considers effects on the harbour porpoise (*Phocoena phocoena*). The units used include the sound pressure level SPL_{peak}, which it considers as having a high relevance for behavioral effects, and the sound exposure level, which it considers to be the metric most related to hearing impairing effects. It uses both the SEL_{ss} and the SEL_{cum}. The report considers the noise levels that have been found to be harmful to Swedish species and marine areas. It also contains examples that illustrate sound propagation in Swedish waters, and how the local acoustic environment affects sound levels from a pile driving activity as a function of distance in four study areas around the Swedish coastline.

The framework concludes that the sound pressure levels at which fish are at risk of death or sustaining serious injury to internal organs are: SPL_{peak} 207 dB re 1 μPa, SEL_{ss} 174 dB re 1 $\mu Pa^{2\cdot}s$ and SEL_{cum} 204 dB re 1 $\mu Pa^{2\cdot}s$. The proposed SPL value are taken from the 2014 Guidelines in Popper et al. (2014) (see page 99). The framework notes that for injury in fish, the cumulative sound exposure level has higher relevance than the single-strike level, because the cited studies found injuries after a certain time period of exposure. The thresholds for fish larvae and eggs are based on the fact that no negative effects were observed at exposures of up to SPL_{peak} 217 dB re 1 μPa, SEL_{ss} 187 dB re 1 $\mu Pa^{2\cdot}s$ and SEL_{cum} 207 dB re 1 $\mu Pa^{2\cdot}s$. However, the paper notes that there are relatively few studies on the early life stages of fish.

The framework does not propose sound levels for flight behavior or TTS in fishes. It states that these levels have been omitted because, unlike physiological damage to internal organs, both flight behavior and hearing damage are linked to the species' specific sensitivity to frequency and sound intensity. And using the existing literature, it is not possible to assess whether flight behavior negatively affects the species at the population level or whether the effect of the impact is related to the area and period of time. However, the paper mentions that during exposure to pile driving noise, behavioral changes were observed in Atlantic cod in large-scale experiments in a pen in the sea at SPL_{peak} 140 to 160 dB re 1 μPa (Mueller-Blenkle et al. 2010)[30] and in European sprat (similar hearing to herring) at $SPL_{peak\text{-}peak}$ 163 dB re 1 μPa and SEL_{ss} 135 dB re 1 $\mu Pa^{2\cdot}s$ (Hawkins et al. 2014) (see page 58). The results showed that the fish reacted to pile driving noise. The structure of fish schools may be affected, and fish may swim away from the noise source, as has also been shown for sturgeon exposed to pile driving in the Hudson River (Krebs et al. 2016).

The Swedish paper underlines the importance of particle motion to fishes. It also points out that much energy is transmitted down the length of the pile, and out into the water column and into the bottom, and that this contributes to the noise that is generated (see discussion of this issue on page 30 of this Report). Much of the energy that is transmitted into the bottom can pass from the bottom into the water, especially if there is a stiff bottom. The range (or the lower propagation losses) of the low-frequency noise components in the bottom sediment, however, is much larger than for higher frequencies. Because the bottom, the pile, water, and air interact when noise is

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[30] It is important to note that the Mueller-Blenkle et al. study has not been published in peer-reviewed literature, and in the opinion of the authors of this report, the paper has serious experimental issues that preclude acceptance of the results. The paper is only mentioned here because it is treated (erroneously in our view) in the Andersson et al. (2017) report.

generated, it is important to take all the components into consideration at the source, especially when choosing noise-mitigation methods.

c. United Kingdom

In the United Kingdom, Nedwell et al. (2007) proposed a set of guidelines for behavioral responses utilizing what they referred to as the dB$_{ht}$ (Species) concept. The dBht (Species) metric purports to take into account each species’ hearing ability by referencing the sound to the hearing thresholds for that species. Animals do not hear equally well at all frequencies within their hearing range. Frequency weighting is therefore applied in assessing the effects of sounds upon particular fish species. Such weighting minimizes the influence of low- and high-frequency sounds that may be detected poorly, if at all, by the animal. Using the dB$_{ht}$ weighting metric, Nedwell et al. suggested that specific dB$_{ht}$ levels above the hearing threshold of a fish elicited particular responses: 0–50 dB elicits a mild, but unsustained, reaction in a minority of individuals; 50–90 dB elicits a stronger reaction by the majority of individuals, but habituation may limit effects; 90 dB and above elicits a strong avoidance reaction by virtually all individuals; above 110 dB is the tolerance limit of sound.

This approach has been utilized within the United Kingdom for assessing the effects of anthropogenic sounds on fishes and it appears to have the tacit approval of some UK regulatory agencies. However, as discussed below and elsewhere the dB$_{ht}$ approach *has serious flaws* that make it unacceptable (Hawkins and Popper 2014; Hawkins and Popper 2016).

The levels above threshold that result in behavioral changes of various types were largely derived from controlled exposure experiments with a few species at a fish avoidance system at a power-plant water intake, supplemented by observations from the testing of a fish guidance system in shallow raceways (Nedwell et al. 2007). Few independent experiments have been carried out to confirm these values for other species or in other circumstances, especially at sufficiently high sound levels to determine how fish respond at 90 dB or more above their hearing thresholds.

Indeed, it has become evident that this approach should not be applied because of the inherent flaws (Hawkins and Popper 2014; Hawkins and Popper 2016). One problem is that, to be useful, the dB$_{ht}$ (Species) must be based upon accurate behavioral threshold determinations rather than the thresholds based on AEP that are often employed, because such thresholds do not provide valid measures for fishes and may differ significantly from those derived using behavioral conditioning techniques, as discussed on page 47 (see also Ladich and Fay 2013; Sisneros et al. 2016). Moreover, in many instances hearing thresholds have been determined under inappropriate acoustic conditions, often within small aquarium tanks. Finally, as discussed in Chapter 6, many fishes are sensitive to particle motion rather than sound pressure, and for them the use of the dB$_{ht}$, which is based on sound pressure thresholds, is inappropriate.

d. United Nations

A report on anthropogenic underwater noise has recently been submitted to the Assembly of the United Nations, and to those States party to the United Nations Convention on the Law of the Sea (UNEP 2017). The report describes the various sources of underwater noise, and their potential effects, and addresses further needs with regard to cooperation and coordination in addressing anthropogenic underwater noise. The issue of whether noise pollution should be included in the Convention on the Law of the Sea is to be considered by Member States. If it is included, then States would be required to take all measures necessary to prevent, reduce, and control noise pollution of the marine environment. State laws and regulations would then have to take into account any internationally agreed rules, standards, and recommended practices and procedures.

The report points out that the ISO has already published a number of international standards related to measurement of underwater noise radiating from ships and pile driving, as well as terminology related to underwater acoustics. However, the report comments that there are significant data and knowledge gaps in relation to anthropogenic underwater sound and its impacts on the marine environment. Most of the research so far has focused on marine mammals, with very few studies on fishes and invertebrates. There is also an incomplete understanding of particle motion and the sensitivity of fish and invertebrates. The report adds that many sound sources, such as pile driving and shipping, are not fully understood, including with regard to sound levels and fields emitted.

The report concludes that the development of adequate noise criteria and restrictions depends on further research and understanding concerning hearing sensitivity of more animal groups, the appropriate metrics to use based on functional hearing groups, and the impacts of noise on marine species. It points out that cooperation and coordination within and across different sectors representing sound-generating activities or impacted sectors can facilitate awareness-raising, the sharing of information on the sources and impacts of anthropogenic underwater noise, and the development and sharing of best practices for minimizing such impacts and addressing cumulative impacts.

4. Guidelines for Marine Mammals as Models for Fish Guidelines

Although not directly applicable to fishes because of major differences in hearing capabilities, behavior, or physiology, recent efforts to develop guidelines for marine mammals may provide lessons (but not actual threshold values) that can be applied to further development of the guidelines for fishes. Initial Scientific Recommendations for Marine Mammal Noise Exposure Criteria were developed by Southall et al. (2007).

Only two categories of effects have been considered for marine mammals by Southall et al. (2007): (1) injury and (2) behavioral disturbance. Frequency-weighting has been utilized for assessing the effects of relatively intense sounds on hearing in marine mammals to ensure that extremely low- and high-frequency sound sources that are detected poorly, if at all, do not affect the criteria that are selected. As we have seen, Nedwell et al. (2007) have suggested that weighted curves should also be utilized for fishes. Marine mammals have been divided into functional hearing groups for the setting of criteria, based on similarities in their hearing. Although fishes can be divided into functional hearing groups based on their auditory anatomy (Table 2, page 50) it is most unlikely that all the fishes within each of these groups have similar hearing abilities, because of their high diversity.

Dual metrics are used for each of the five functional marine mammal hearing groups, using both sound pressure and SEL. The minimum exposure criteria for injury in marine mammals are defined as the levels at which a single exposure is estimated to cause onset of PTS. Injury has to be defined differently for fishes.

As with fishes, one challenge in developing behavioral criteria for marine mammals has been to distinguish significant behavioral responses from insignificant, momentary alterations in behavior. The startle response to a brief, transient event is unlikely to persist long enough to constitute significant disturbance. Even strong behavioral responses to single pulses, other than those that may secondarily result in injury or death (e.g., stampeding), are expected to dissipate rapidly enough as to have limited long-term consequence. Consequently, upon exposure to a single pulse, the onset of significant behavioral disturbance is proposed to occur at the lowest level of noise exposure that has a measurable transient effect on hearing (i.e., TTS-onset). Southall et al. (2007) recognized that this was not a behavioral effect *per se*, but they decided to use this auditory effect as a *de facto* behavioral threshold until better measures were identified. It would not be appropriate to define behavioral responses for fishes in this way, because of the lack of information on TTS-onset, and the major differences in the hearing abilities of fish species.

It was also concluded for marine mammals that insufficient information existed to assess the use of SEL as a relevant metric in the context of marine mammal behavioral disturbance for anything other than a single pulse exposure. It was decided that future noise exposure criteria for behavioral disturbance might distinguish between SPL and SEL exposure criteria for additional conditions, but it was concluded that for most sound types (the exception being single pulses), the available data were best assessed in relation to the SPL. Thus, SEL thresholds were recommended for a single pulse, but not for multiple pulses.

5. Developing Future Criteria

This section considers future criteria and is based on the information reviewed in this report. However, as pointed out elsewhere in this report, there is a dearth of high-quality data for many questions. To help remedy this situation, Chapter 10 proposes a set of studies that will have the highest impact in helping to update and refine the 2008 Interim Criteria.

a. Onset of Physical Injury and Death

Data on direct mortality due to exposure to pile driving has been shown in only a few instances (e.g., Caltrans 2001), and only when fish have been very close to the pile being driven. Because immediate mortality is not well documented for fishes and is likely to occur only when fish are very close to the source, the greater concern is onset of physical injuries that could lower fitness and potentially result in long-term mortality.

The majority of earlier studies on physical effects from pile driving, upon which the 2008 Interim Criteria are largely based, appeared in grey literature (e.g., Caltrans 2001; Abbott and Bing-Sawyer 2002; Abbott 2004; Abbott et al. 2005). Moreover, concerns have been expressed regarding the experimental design of these earlier studies, including the lack of controls for such studies (Hastings and Popper 2005; Popper and Hastings 2009).

In contrast, recent studies on the effects of simulated pile driving on fishes, discussed in Chapter 7, have demonstrated that onset of physical effects to a variety of species occurs when the $\mathrm{SEL}_{\mathrm{cum}}$ is at least 16 dB above the 2008 Interim Criteria, and probably more than 23 dB higher. These studies also show that specifying a single acoustic metric to determine impacts on fishes, and especially the $\mathrm{SEL}_{\mathrm{cum}}$ metric, is insufficient as a predictor. The recommended metrics now include the number of impulses, the peak sound pressure level (the $\mathrm{SPL}_{\mathrm{peak}}$) or the $\mathrm{SEL}_{\mathrm{ss}}$ (which are correlated values), and $\mathrm{SEL}_{\mathrm{cum}}$ values.

Because the 2008 Interim Criteria are set well below levels that actually result in onset of physical effects for a number of ESA listed and other species (Chapter 7), it is recommended that the updated levels proposed by Popper et al. (2014) and shown in Table 3 (page 100) of this report be adopted by transportation and resource agencies that manage the effects of pile driving noise on fishes. These levels are still conservative, in that they are below those that result in onset of physical effects in the 2014 Guidelines. Most importantly, unlike the 2008 Interim Criteria, the new (still interim) criteria in the 2014 Guidelines are based on an extensive set of controlled and peer-reviewed studies that have provided quantitative dose response relationships between sound levels and particular effects.

Another critical issue is that, *in the future*, criteria must include potential effects of particle motion on fishes, for reasons discussed in Chapter 4. Particle motion is a critical factor related to physical effects on fish, especially for animals near the bottom or surface, or relatively near the source. In particular, although both sound pressure and particle motion impact the movement of the swim bladder walls, and affect the body tissues surrounding the swim bladder, particle motion is of particular concern because it is directly detected by, and can affect, the inner ear. Any such effects will potentially be detrimental to hearing, thereby affecting fitness of animals at least until the ear repairs itself (as in TTS), if it does that after exposure to particle motion.

b. Physiological Effects

The two most important questions are: whether there is a correlation between physical and physiological effects; and whether it will be possible to define thresholds for the onset of such effects. There are significant issues that have yet to be defined, and for which there are few data for developing criteria. For example, little is known about the onset of physiological effects such as stress responses, changes in hormone levels, effects on reproductive physiology, changes in heart rate, impact on sleep, and all of the other things that are likely to occur in fishes under the presence of loud sounds, just as they occur in humans and other terrestrial vertebrates (see page 91) (Slabbekoorn et al. 2010; Weilgart 2017).

Moreover, determination of levels for the onset of physiological effects will likely be far more difficult to actually measure than for the onset of physical effects. Monitoring of physiological effects often requires sophisticated tools and methodologies, whereas it is often easy to actually see most physical effects. There is also the issue that how an animal is affected physiologically will vary significantly depending on numerous factors that could include things like age, time of year, reproductive state, time of day, and even water temperature.

Indeed, it may be appropriate to conclude that it is not possible at this time to set onset levels for sound exposure that would result in physiological effects. If it proves possible to successfully define onset levels for the more observable physical and behavioral effects, setting onset levels for physiological effects may be unnecessary.

c. Hearing (including TTS and Masking)

Setting criteria for potential TTS or masking, as discussed in Chapter 5 (see page 53), is very difficult (Andersson et al. 2017). It has also been very difficult to define such criteria for marine mammals (Finneran 2015; NMFS 2016, 2018). There are many reasons for this problem, particularly because the levels of hearing loss vary based on many factors, perhaps most important of which is the hearing sensitivity of fish, as well as the characteristics of the anthropogenic sound such as duration, intensity, rise time, and spectrum.

Moreover, species that hear well are more likely to be masked by lower anthropogenic sound levels than fishes that have poorer hearing. Similarly, for TTS to occur, the level of the anthropogenic sound must be at least 60 dB above the hearing threshold of a fish (page 53). Thus, fishes that hear well (none of which are ESA listed) have the potential to develop TTS at much lower noise levels than do fishes that do not hear as well.

In developing the 2014 Guidelines, the authors also asked what level of TTS and masking may potentially impact fishes. They concluded that there must be a minimum of 6 dB hearing loss for 24 hours in order to suggest an effect. This decision was based on the methods by which hearing loss is measured, and the extensive variability in results even within a single animal. Moreover, because most TTS recovery is quite fast, the likelihood of having an impact on fitness and survival is relatively low.

In conclusion, at this point, the only guidance comes from data in Table 3 (page 100) (Popper et al. 2014) which identifies conservative lower levels of sound that might result in onset of TTS in several species. However, even those levels must be taken as extremely tentative, as they are based on data from a single study on three species (Popper et al. 2005). TTS was not found in one of the species that was investigated (a salmonid fish). Moreover, as emphasized elsewhere, TTS data based on particle motion is needed because particle motion is the primary stimulus to the ear. In regions of high particle motion, fishes may actually experience TTS even when the sound pressure is relatively low. Moreover, onset of TTS in all animals is a function of the level of the sound above the threshold of detection of that sound. Thus, an animal that has poor hearing sensitivity (e.g., hears poorly, such as salmonids) will only show TTS to sounds that are far higher in intensity than an animal that has low good hearing sensitivity (e.g., low thresholds, such as goldfish).

Masking of key signals by some anthropogenic sound is likely to be of greater importance in terms of potential effects upon some fishes than TTS. Most anthropogenic sounds will not be sufficiently intense to produce TTS, but these same sounds may have the potential to decrease the ability of fishes to detect natural sounds of importance to them. In developing the 2014 Guidelines, however, the authors pointed out that masking is likely only effective if it decreases hearing sensitivity by at least 6 dB for 30 minutes. Due to lack of data, the authors were not able to consider whether intermittent sounds, such as pile driving, have any real masking impact. It is possible that intermittent sounds may result in less masking because the signal only occurs for a very short period of time (milliseconds), followed by a longer period (one second or more) of silence.

Accordingly, because masking in fishes by intermittent sounds has yet to be investigated, it is likely not possible at this time to develop criteria for masking beyond the initial suggestions in the 2014 Guidelines.

d. Behavior

There are particular problems, discussed on page 97, in setting criteria for behavioral effects from pile driving, or any other anthropogenic source. As NOAA (2016) points out in its *Ocean Noise Strategy Roadmap*, studies on fishes have focused more on characterizing the physical effects such as hearing impairment, barotrauma, and death, although behavioral effects such as changes in direction, speed, or schooling patterns as well as changes in stress hormones have been documented (see Chapters 5 and 7). Behavioral responses by fishes may take place at greater distances from a sound source than do physical or physiological effects, and far more animals may show behavioral effects than physical or physiological effects as a result of sounds. *Indeed, it is the opinion of the authors of this document, as discussed in Chapter 10, that behavior is the most critical issue in understanding effects of anthropogenic sound (including pile driving) on fishes*. And it is the issue for which there is the fewest investigations and poorest understanding.

As discussed on page 97, the current NOAA Fisheries criteria use SPL_{rms} 150 dB re 1 µPa as the level that may result in behavioral effects (Caltrans 2015). However, the origin of this criterion is not known, and it is not clear if it has any scientific validity (Hastings 2008). Moreover, the criterion does not specify a particular behavior, but simply assumes there is the potential to experience a behavioral response. Caltrans (2015) goes on to state that the SPL_{rms} 150 dB guideline for potential behavioral effects should be considered in some consultations depending on an activity's location and the time of year. However, the document also notes that more research and discussions are needed to get a better understanding of the behavioral component of the thresholds.

The problem with the current behavior criterion applied by NOAA Fisheries is that it is far too simplistic to actually provide guidance on potential behavioral effects of pile driving, or any other, sound on all fishes. This criterion does not take into consideration a number of factors, including:

- Differences in hearing capabilities between different species;
- Whether a species can detect a particular sound (e.g., is the level of the sound of concern above the fish's threshold of hearing?);
- How different species, the same species at different times of the year, and animals of different ages, sizes, or sexes respond to a particular stimulus;
- The motivational state of the animal (e.g., is it feeding, mating, moving around a home range);
- Whether the behavioral response is at all significant or whether it is momentary and does not alter the life of the fish; and

- Whether fishes habituate after hearing a sound a few times and then no longer show any behavioral response.

Moreover, in trying to develop criteria for behavior, there are a number of other broader issue that must be considered. These include:

- A wide range of behavioral responses is shown by fishes to sounds. Alarm and startle responses are generated, and there may be changes in schooling behavior, interference with feeding behavior, greater vulnerability to predation, and the masking of key signals — which may especially interfere with courtship and spawning behavior. Which of these are of biological significance and which are not?

- The circumstances under which sounds are presented to fishes in experiments, both in the laboratory and field, are critical in determining behavioral responses: whether the animals have previously experienced such sounds, and whether they resemble natural sounds of interest to them. In many sound playback experiments, the stimulus and background noise fields are very poorly described, if they are described at all (e.g., Popper and Schilt 2008; Hawkins and Popper 2018b).

- Particle motion levels are rarely specified. However, because most fishes only, or primarily, respond to particle motion, defining criteria in terms of sound pressure, especially in shallow waters, where there is no way to predict particle motion from sound pressure, may be inappropriate.

- Whether or not a fish responds may depend on its condition, motivational state, and the presence of other animals including predators (De Robertis and Handegard 2013). It has been emphasized that the internal state, motivation, context, and previous experience of animals may affect their behavioral responses and may also influence the long-term and cumulative effects of sound exposure (Lucke et al. 2016). Thus, if we are to better understand the sensitivity of marine animals to sound there is a particular need for studies that examine variation in levels of behavioral response in parallel with detailed characterization of the sound fields, ideally using a variety of different sound measurement metrics to ascertain which aspects of the sounds are most important.

Finally, it is critical to decide the kind of behaviors that should define a behavioral response that is at a criterion level. Is the level one that elicits a simple startle response that is a singular event and results in no change in other behaviors or fish movement (and is likely not of any real behavioral consequence), or is the level one that results in fishes permanently leaving an area, or something in between?

Because of the complexity described above, developing behavioral guidelines is far more difficult than developing guidelines for physical or physiological effects. Carrying out appropriate experiments that take into consideration factors that range from species differences to motivational state of an animal is very complex. This complexity underlines the importance of studying the responses of wild fishes to sounds within their natural environment. Animals in tanks or even in large enclosures show very different responses to behavioral stimuli than do wild animals (e.g., Oldfield 2011). Studies on captive animals, such as most of those described in Chapter 6, are suitable for gaining physiological information such as hearing sensitivity, but not for understanding how a wild animal will respond behaviorally to a sound stimulus.

Perhaps the most useful insight into setting behavioral criteria, recognizing that these results still suffer from many of the issues discussed throughout this report, are recent behavioral studies on the effects of simulated pile driving sounds carried out on wild free-swimming fishes. These studies provide the first useful data on the sound pressure levels that generate behavioral responses (Hawkins et al. 2014; Iafrate et al. 2016; Roberts et al. 2016b). As discussed in Chapter 6, all of these studies were on wild animals and examined the behavioral responses to different received sound levels. In the studies by Hawkins et al. (2014), dose response curves were prepared that showed the proportion of the exposed fish that responded in a particular way to sounds at different levels. Interestingly, the behavioral responses shown by 50% of those exposed to sound, for each case species, were to sound levels within the $SPL_{peak\text{-}peak}$ range of 152 to 167 dB re 1 µPa. It should be noted that it is inappropriate to express such criteria in terms of RMS levels for impulsive sounds. It is more appropriate to use SPL_{peak} or SEL_{ss}.

It is remarkable how similar the sound pressure levels that evoked behavioral responses were in these three key experiments. It is notable that they are only slightly higher than the SPL_{rms} 150 dB re 1 µPa specified by NOAA Fisheries for behavioral effects (Caltrans 2015). At the same time, these studies did not evaluate many critical aspects of behavioral responses including how long animals responded to the sound sources, whether their responses habituated over time, and whether animals would have moved away from the source had the sounds continued. Moreover, responses were not reported in terms of the particle motion levels (except in the case of Hawkins et al. 2014, where the particle velocity levels were estimated for the mackerel – a species sensitive to particle motion). However, it may be important to define interim criteria for behavioral effects until more scientific information is available, and a received level of $SPL_{peak\text{-}peak}$ level of 163 dB re 1 µPa might be appropriate initially.

CHAPTER 10: LESSONS LEARNED - RESEARCH GAPS AND RESEARCH RECOMMENDATIONS NEEDED FOR FUTURE CRITERIA AND GUIDELINES

1. Overview

a. Lessons Learned

It is apparent from the lessons learned from the research discussed in this report and from the 2014 Guidelines (Popper et al., 2014), that we now have improved our understanding of the effects of pile driving and other anthropogenic sound on fishes since the promulgation of the 2008 Interim Criteria.

Perhaps the most telling outcome (and *first lesson learned*) of this extensive body of research is that the 2008 Interim Criteria for onset of physical effects are well below the levels of sound that may result in onset of physical effects. There is a strong case for updating the 2008 Interim Criteria, but before this can happen, there will need to be agreement with the appropriate resource agencies on how onset is defined, otherwise applying new criteria could substantially delay ESA consultation. Indeed, *onset* is one of a set of terms and methods (e.g., for measurement of thresholds) used in the United States and internationally that would benefit greatly from the development of a set of standards that have formal approval such as via ANSI (as the 2014 Guidelines) or ISO.

Until onset is defined, and there are additional data on the sound levels that result in onset of effects, it is important to have an understanding of those sound levels that might result in onset even of a single mild injury in an individual animal. Because this level is likely to be variable and based on such factors as fish species and size, it is reasonable to suggest that agencies develop onset criteria for each consultation based, in part, on the anatomy and behavior of the species in question, as described in the 2014 Guidelines (i.e., Table 2 of this report) and in part on the onset levels proposed in those Guidelines (i.e., Table 3 of this report). Moreover. development of such criteria should strongly take into consideration that the onset of two minor injuries, as outlined in the 2014 Guidelines, is well above the levels in the 2008 Interim Criteria, and even onset of a single effect will be far closer to the 2014 levels than the 2008 levels.

A *second critical lesson learned* is that the major area of concern is effects of pile driving (and other anthropogenic sounds) on behavior because fishes will detect, and possibly be affected by, sounds at distances far beyond those where any physical effects would take place. However, as discussed in Chapter 6, and in Section 10.3 below, there are still major gaps in the behavior literature.

There are few lessons learned that help revise behavioral criteria, although it is clear that several of the “rules” upon which current behavior criteria are based are incorrect. The two most important assumptions that should be revisited are: (a) the same criteria level can be used for all species or even the same species under different conditions, and (b) fishes do not move away from pile driving for the duration of the operation.

Thus, even though there are not sufficient data to provide concrete new guidelines, there must be flexibility in the behavior guidelines that reflect what fishes are likely to do when exposed to the sounds from sources like pile driving (e.g., swim away or avoid the area), using the few levels that are discussed in Chapter 6 and based on well-designed studies. Swimming away will, of course, reduce the level of sound to which the fishes are exposed.

A *third lesson learned* is the substantial increase in understanding of particle motion in the lives of fishes. Although it is not possible to apply particle motion within the regulatory regime at this time, and perhaps not for the next five or ten years, researchers and regulators should keep this issue in mind and work to incorporate particle motion in future studies and regulatory activities because it, far more than sound pressure, has an impact on both physical and behavioral responses to all anthropogenic sounds.

b. Lessons to be Learned

To continue to improve the recommended 2014 Guidelines and to develop a better set of criteria and guidelines with respect to potentially protecting fishes from the effects of pile driving, it will be critical to fill the many knowledge gaps that remain. These have been identified in previous chapters in this report and all relate to our understanding of the potential impacts of pile driving. It should also be recognized that, because the number of knowledge gaps is substantial, it is likely not possible to answer all of the associated research questions in a reasonable amount of time or at reasonable cost. Thus, this chapter focuses on the knowledge gaps and the necessary research, that are the most important for increasing our knowledge in areas that are most likely to provide the greatest increases in our ability to update criteria and guidelines.

In considering the research to be done, it is necessary to repeat a point made by Popper and Hawkins (2018): they argued that rather than have multiple investigators go in a variety of directions in their research, there be some mechanism that guides (perhaps through directed funding) the research so that it is focused on the critical questions that will best inform industry and regulators. Indeed, this was, in essence, the approach taken by the National Highway Transportation Board of the National Academies of Science when it put out a request for proposals for studies of effects of pile driving on fishes in 2006, resulting in the body of work discussed in Chapter 7 and which, in the view of the authors of this report and those of the 2014 Guidelines, should serve as the basis for updated interim criteria and guidelines.

2. Major Research Gaps and Research Recommendations

There are many research gaps discussed in previous chapters. A fuller discussion of the broad range of research gaps was provided in a conference promoted by BOEM in March 2012 that was organized by Hawkins and Popper (Normandeau 2012).[31] These research gaps, and some recommendations for research priorities, were updated and organized in a paper by Hawkins et al. (2015). That paper provides a broad overview of the most critical issues that apply to all fishes (and invertebrates), including those exposed to pile driving. The research gaps and recommendations in this chapter and report, however, focus on the key questions and data needed

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[31] The complete report, with appendices, can be found at: https://www.cbd.int/doc/meetings/mar/mcbem-2014-01/other/mcbem-2014-01-submission-boem-04-en.pdf

to specifically inform understanding and regulating the effects of pile driving on fishes. *Note that the research gaps are* ***not*** *listed in order of priority, and all the research gaps that are listed need to be filled.*

a. Selection of Species

Before discussing research gaps *per se,* it is important to make the point that in most all cases, it is going to be imperative to obtain data from multiple species and from a range of sizes and ages of fish within each species. This diversity is necessary because there is likely to be substantial variation in potential effects depending on differences in species' anatomy, physiology, and behavioral responses to various stimuli. These differences are not only between species, but likely to be a function of changes that fishes undergo as they age. Thus, the behavioral reactions of larval fish may be very different from that of sexually mature animals. Likewise, there are very likely differences in behavioral responses depending on whether a fish is feeding, mating, or only swimming around.

Therefore, in deciding on which studies to do, and then in designing appropriate experiments, these factors must be taken into consideration, and every effort needs to be made to have samples of animals that span the diversity of fishes, or at least the diversity of fishes of particular interest. However, considering that there are more than 33,000 extant fish species, selection of species is difficult.

To deal with this issue, at least in terms of physical effects, the 2014 Guidelines suggested dividing fishes into several morphological groups that relate to the presence/absence and configuration of the swim bladder (see Table 2 on page 50). This concept is discussed in detail in Chapter 7 (page 74). Although this approach should perhaps be modified if concerns are only on pile driving, having a representative set of species or animal types will be critical for future work.

Moreover, in addition to different morphologies, there is some need to consider not only marine species, but to recognize that there have been relatively few studies that have examined the effects of pile driving sounds (or sounds from any other anthropogenic sources) on freshwater fishes (Mickle and Higgs 2018). Freshwater environments may be slightly less efficient at sound transmission than marine environments; however, rivers, estuaries, and lakes harbor a wide range of species that are important from a conservation perspective.

In the following listing of research gaps and recommendations, it should be understood that the comparative approach discussed in this section will be required.

b. Behavioral Responses to Pile Driving Sounds

The most important issues to be addressed are the effects of anthropogenic sound on fish behavior. Although questions about physical effects are important, the distance around the source that includes sounds of sufficient level to physically harm an animal is relatively small compared with the much greater area that is potentially ensonified by the sound and heard by the fish. Following on from this, far fewer animals are likely to be directly harmed through injuries generated by the sound compared with the number of animals that may show changes in behavior due to the presence of the sound. Although many of these behavioral effects are likely to be minimal and have little or no impact on fish fitness and survival, any anthropogenic sounds that alter the ability of animals to hear natural sounds that are important to them (e.g., as a result of masking), cause temporary loss of hearing sensitivity (TTS), or frighten the fish away from preferred locales or from migration routes, could have substantial short and long-term impacts.

There are numerous behavioral issues that need to be examined, from the sound levels that are likely to elicit behavioral responses (e.g., based on hearing studies, and studies of hearing in the presence of maskers) to actual responses to sound pressure vs. responses to particle motion (an area in which we have very substantial needs for better understanding and few data) (Popper and Hawkins 2018). Data are needed on general behavioral responses to sounds at different sound levels and on how these responses change during the course of a pile driving operation, perhaps as fishes habituate to the sounds or temporarily lose hearing because of the presence of persistent sounds. One significant issue is what fishes do when they are exposed to pile driving sound — do they move away or stay in place (Hawkins et al. 2014; Heilprint et al. 2015; Krebs et al. 2016). Moreover, do the responses of fishes differ when they are at different distances from the source and in the acoustic near field vs. the acoustic far field? A particularly significant issue is whether there are consequences for fish populations.

In addition, long-term, realistic field studies are needed on the effects of pile driving on the behavior of fishes, taking account of cumulative and synergistic effects, along with stress indicators. If we are to better understand the sensitivity of marine fishes to sound, there is a particular need for studies that examine variation in levels of behavioral response in parallel with detailed characterization of the sound fields, ideally using a variety of different sound measurement metrics to ascertain which aspects of the sounds are most important.

There are two critical caveats in doing any study on fish behavior. As pointed out by Hawkins and Popper (2016), studies of fish behavior must be done in the wild, because studies done in the laboratory or on enclosed fishes must be suspect in terms of the behavioral responses (see also Chapter 6, page 62). The fundamental issue is that animals in enclosures are not likely to respond to a stimulus in the same way as they do in the wild where there are no constraints on their movement.

In addition, it is almost impossible to duplicate the acoustic characteristics of a sound produced in the wild with a sound in a tank or shallow enclosure (Chapter 5, page 25) (Duncan et al. 2016; Gray et al. 2016b; Rogers et al. 2016). Animals in a tank may be exposed and respond to a particular sound in the laboratory, but the same source in the wild may actually “sound” very different.

c. Effects of Particle Motion

As discussed in Chapters 2 and 8, virtually all criteria and guidelines for both physical and behavioral effects are described in terms of sound pressure. However, most fishes are primarily detectors of particle motion, and relatively fewer species of fish use sound pressure (Popper et al. 2014; Hawkins and Popper 2016; Popper and Hawkins 2018). It will be important in the future to determine criteria based on responses to particle motion (see Chapter 4 on particle motion).

d. Development of Dose Response Data

Studies on physical effects of pile driving signals on fishes (Chapter 7) were in terms of SEL_{cum}. However, due to the constraints of time and funding, the studies were not done with a full evaluation that could lead to understanding dose response relationships of different sound parameters such as signal intensity, number of strikes, and inter-strike interval. Indeed, as discussed on page 84, a recent study (Casper et al. 2017) suggests that the dose response relationship is more complex than previously thought. At the same time, these studies further eliminate any support for an equal energy hypothesis (page 78) (Halvorsen et al. 2012a).

Studies on dose response relationships will provide insight not only for understanding the onset of physical effects or behavioral effects, but also for determining, using statistical techniques, those levels above the onset level at which potentially harmful effects start to occur. Such information will enable regulators and others to be able to make better decisions on criteria, particularly if they are willing to accept the idea that a small effect may not have any impact on the fitness of the animal.

e. Preparation for Future: Measurement and Use of Particle Motion in Regulatory Activities

In anticipation of the time in 5 to 10 years when potential effects of particle motion on fishes and invertebrates is better understood, and when criteria can be reasonably developed, methods need to be developed and incorporated into evaluation of pile driving activities that not only look at sound pressure, but also particle motion. Moreover, as new tools become available for measurement of both sound pressure and particle motion simultaneously, these tools need to be employed on a regular basis both in measurement of pile driving signals and also in determination of effectiveness of mitigation measures such as air bubble curtains.

As a critical part of this effort, it is most important to support organizations such as ANSI and ISO, of which ANSI is a member, to develop standards for particle motion sensors as well as sound pressure sensors, and to develop specific protocols for making particle motion measurements that would be employed throughout the industry.

In addition to obtaining data related to particle motion, regulators and others need to understand and appreciate that particle motion needs to be taken into consideration when planning and regulating pile driving and other activities likely to generate sound that can potentially affect aquatic organisms.

f. Hearing

Although there is a body of literature on the hearing of perhaps 100 fish species (Chapter 5, page 47), most data were obtained using sound pressure measures and/or a technique known as AEP in small tanks. The AEP thresholds are often not well correlated with the reported sound pressure levels that have been applied in many experiments because of a lack of testing protocols and the acoustics of the small tanks in which most of the studies were performed cannot reliably be calibrated to the actual signal levels to which the fish were responding (Rogers et al. 2016). Additionally, most fishes primarily detect particle motion (Popper and Hawkins 2018), which has not been measured or examined in most hearing studies carried out on fishes. Unfortunately, this lack of oversight results in most of the data on hearing in the literature being of little value. Finally, it is worth noting that AEP is not a reflection of higher brain processing of acoustic information and cannot be used to understand how an animal might respond behaviorally.

There is a clear need for more data on the hearing sensitivity of fishes of interest, in order to correlate with sound levels (sound pressure and particle motion) for consideration of issues related to behavioral responses, masking, and TTS. Such studies need to be done behaviorally to produce reliable hearing sensitivity data. The studies need to be done in environments, such as open water, where the sound fields can be calibrated for both pressure and particle motion, and where hearing can be measured in terms of both types of signal (see Figure 7, page 49). Such studies need to determine hearing thresholds not only under quiet conditions but also in the presence of masking signals, to determine the ability of the fish to discriminate signals of particular interest to them in the presence of anthropogenic noise, including the impulsive sounds generated by pile driving.

g. Modeling of Sound Fields

Once the effects of sounds on fishes have been defined, it is necessary to estimate the extent of those geographic areas over which those effects might take place. It is, of course, possible to make extensive measurements around a source to determine the potential zones of influence within which the sound levels are above values that result in unacceptable physical or behavioral effects. Alternatively, it may be possible to determine how close to a protected species or habitat a particular noise-making activity can take place without adverse impact.

However, making such measurements is often difficult and time-consuming, and, in many cases, modeling of a sound field may be more efficient and effective in providing the information needed for regulatory purposes. Most, but not all, of the models in current use were developed for work in free-field environments where there are no boundaries. Accordingly, many of these models may lack accuracy in shallow-water environments where the sound interacts with both the water surface and the substrate, as discussed in Chapter 3. Moreover, while particle motion levels can be estimated from pressure measurements in free sound fields, such estimates are less valid for shallow water conditions, and so models for areas in which pile driving takes place must include consideration of particle motion as well as sound pressure propagation (MacGillivray et al. 2011; Pangerc and Theobald 2015).

Thus, it is necessary to continue to develop models that can be used to adequately predict both sound pressure and particle motion levels at locations around any pile driving operation. These models need to take into account the various features of shallow-water environments, including variance in substrates, because these factors have a significant effect on sound propagation. At the same time, to ensure that the predictions of propagation models are correct, it is necessary to validate such models by making field measurements of the sound pressure and particle motion levels at different locations (Farcas et al. 2016).

3. Other Research Gaps and Recommendations

The fact is that knowledge of fish bioacoustics, and particularly of the effects of anthropogenic sounds on fishes, is far less understood than similar issues with marine mammals (Southall et al. 2007; NMFS 2016). Moreover, virtually nothing is known about effects of anthropogenic sound on marine and freshwater sharks (Casper et al. 2012a), but, considering their importance in the marine ecosystem and the number of species that are ESA listed, future studies are warranted.

The difference in knowledge between marine mammals and other aquatic species arises because of the far greater human interest in marine mammals (“charismatic megafauna”) and the heavy legal protection with which they are provided, as well as far greater support for studies on these mammals than for other aquatic animals. In addition, it is much easier to study marine mammals than fishes in the field if, for no other reason, that marine mammals come to the surface frequently and they are often sufficiently large to enable long-range tracking devices to be placed upon them.

To fully advance understanding of potential effects of sounds on fishes there need to be two approaches. One involves investigating the basic biology of hearing and acoustic behavior of fishes, so it is possible to understand normal behavior and then use this as a baseline for understanding potential effects. Following from this, far more information is needed about the general effects of anthropogenic sounds on these animals, much as outlined by Hawkins et al. (2015). The studies described earlier in this chapter are, in fact, only a start in the information needed. At the same time, those types of studies are, in our view, those that are most crucially needed in order to develop and implement future criteria for effects of pile driving (and other impulsive sounds) on fishes.

There is also a need to consider impacts on fish populations, in addition to the effects on individuals. Effects are the broad range of potentially measurable changes that may be observed in individuals, groups of animals, or even habitats as a result of sound exposure. Impacts are effects that, with some certainty, rise to the level of deleterious ecological significance (Boehlert and Gill, 2010). Thus, the effect does not indicate the significance, whereas the impact deals with the severity, intensity, or duration of the effect on animal populations and ecological communities. Such impacts can then be compared with those resulting from other stressors, including chemical pollution, fishing, pathogens, and climate change.

CHAPTER 11: CONCLUSIONS AND RECOMMENDATIONS

The main objectives of this project were as follows (*from Foreword*):

- *A. Provide a firm scientific, technical basis on which to improve the current understanding of the sound levels that are commonly associated with the very earliest onset of physical injury to fishes (e.g., scale loss), from the exposure to impulsive underwater sound.*

- *B. Provide a firm scientific, technical basis on which to improve the understanding of the earliest onset of temporary threshold shifts to fish from exposure to impulsive underwater sound.*

- *C. Evaluate effectiveness of the existing 2008 interim thresholds for fish protection based on recent research.*
  - *What are the physical and sub-injurious effects to fish commonly associated with sound levels in excess of 206 dB peak?*
  - *What are the sub-injurious effects to fish commonly associated with cumulative SEL values in excess of 183/187 dB?*

- *D. Identify knowledge gaps specific to the onset of physical injury to fish and TTS such that specific research can be proposed and carried out.*

- *E. Provide a summary of other lessons learned.*

The following are key conclusions of this report in relation to the above objectives:

Objective A

- Post-2008 studies demonstrate that the 2008 Interim Criteria are excessively conservative and do not reflect current knowledge of the levels at which there may be an onset of effects from pile driving.

- The 2014 Guidelines (Popper et al. 2014) present updated interim criteria for pile driving (and other anthropogenic sources) that best reflect the post-2008 studies. Therefore, until additional data gaps (Objective D) are filled, it is recommended that the 2014 Guidelines and criteria be adopted as reflecting the best available science.

- At the same time, it is recognized that the criteria in the 2014 Guidelines are based on multiple injuries, whereas the current FESA and CESA requirements are to determine those sound levels that result in the onset of a single injury. Although the criteria in the 2014 Guidelines do not deal with single injuries, the data leading to the criteria can be “mined” to develop levels of onset that can be applied to individual regulatory situations. This data mining should take into consideration different species groupings, as described in the 2014 Guidelines and outlined in Tables 2 and 3 of this report. At the same time, it is clear that the post-2008 data do strongly support the suggestion that the the threshold sound levels that result in a significant impact should be higher than those specified in the 2008 guidelines. The sound levels that result in the onset of a single effect are well above the 2008 levels, and close to those proposed in 2014. Therefore, until more data are available, criteria for onset that are set during consultations should be close to, if not the same as, those proposed in 2014.

Objective B

- As yet, there are insufficient scientific data on which to base criteria for the earliest onset of temporary threshold shifts (TTS) in fishes that have been exposed to impulsive underwater sounds. However, the 2014 Guidelines suggested that a sound level greater than186 dB SEL_{cum} is likely to result in onset of TTS for a wide range of fish types. At the same time, this level is highly conservative and there are many fish species for which TTS onset would only occur at much higher sound levels. This is particularly the case for fishes that do not hear well, such as salmonids, sturgeons, and other ESA listed species.

Objective C

- The effectiveness of the existing 2008 Interim Criteria has been evaluated and it has been concluded that they are excessively conservative and there is no evidence in the literature that those levels would result in onset of any effects on fishes. The more recent data, summarized in Chapter 7, and the 2014 Guidelines provide levels that are closer to those that may result in onset of effects in some, but not all, species. Current consultations should be based on these levels (e.g., Table 3) adjusted for onset to single animals and for animals with different morphologies (Tables 2 and 3).

Objective D

- It has been possible to identify knowledge gaps specific to the onset of physical injury to fish and TTS such that specific research can be proposed and carried out. These knowledge gaps are listed in Chapter 10.

Objective E.

- A summary of the lessons learned is provided at the start of Chapter 10.

Recommendations

- Although it is recognized that current lack of data on responses to, and effects of, particle motion are not sufficient to develop appropriate criteria, the growing international awareness that fishes do possess particle motion receptors means that particle motion must eventually be taken into account in setting future criteria, once appropriate data are available. It is recommended that research on the effects of exposure to particle motion should be carried out within the next 5 to 10 years, and that efforts be funded that will obtain the necessary data on which to base particle motion criteria.

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APPENDIX: EFFECTS ON FISH BEHAVIOR

Foraging Behavior

Only a few experimental studies have investigated how foraging behavior is affected by exposure to increased noise levels. Purser and Radford (2011) exposed captive three-spined sticklebacks (*Gasterosteus aculeatus*) to brief and prolonged noise to investigate how foraging performance was affected by the addition of acoustic noise to an otherwise quiet environment. The authors said that the addition of noise induced only mild fear-related behaviors — there was an increase in startle responses, but no change in the time spent freezing or hiding compared with a silent control — and, thus, had no significant impact on the total amount of food eaten. However, there was strong evidence that the addition of noise increased food-handling errors and reduced discrimination between food and non-food items, results that were consistent with a shift in attention. Consequently, noise resulted in decreased foraging efficiency, with more attacks needed to consume the same number of prey items. It was suggested that acoustic noise has the potential to influence a whole host of everyday activities through effects on attention, and that even very brief noise exposure can cause functionally significant impacts, emphasizing the threat posed by ever-increasing levels of anthropogenic noise in the environment.

Wale et al. (2013) conducted a series of tank-based experiments to consider how playback of ship noise affected foraging behavior in the shore crab (*Carcinus maenas*). Ship noise playback was more likely than ambient-noise playback to disrupt feeding, although crabs experiencing the two sound treatments did not differ in their likelihood of, or speed at, finding a food source in the first place. It was suggested that anthropogenic noise had the potential to increase the risks of starvation, and the authors suggested that the behavior of invertebrates, and not only vertebrates, is susceptible to the impact of underwater noise.

Voellmy et al. (2014b) examined how exposure to playback of noise originally recorded from ships affected the feeding behavior of two sympatric fish species: the three-spined stickleback (*Gasterosteus aculeatus*) and the European minnow (*Phoxinus phoxinus*). The fish were maintained within glass aquarium tanks. Both species consumed significantly fewer live prey, and showed startle responses significantly more often, during playback of anthropogenic sounds than during control conditions. Minnows showed a qualitative shift in activity away from foraging behavior (with greater inactivity and more social behavior, perhaps indicating stress) under increased noise conditions. Sticklebacks maintained foraging effort but made more mistakes, perhaps resulting from the impact of noise on cognition involved in food detection, classification, and decision making. Voellmy et al. concluded that additional noise in the environment can lead to reduced food consumption, but that the effects of elevated noise are likely to be species specific.

Predator Avoidance

Remage-Healey et al. (2006) showed that Gulf toadfish (*Opsanus beta*) can detect the acoustic signals of bottlenose dolphins. Using underwater playbacks to toadfish in their natural environment, they found that low-frequency dolphin sounds (“pops”) within the toadfish's range of hearing dramatically reduced toadfish calling rates by 50%. Predator sound playbacks also had consequences for circulating stress hormones, as cortisol levels were significantly elevated in male toadfish exposed to dolphin pops. The presence of anthropogenic sound may interfere with the detection, location, and identification of predator sounds by fishes, perhaps rendering them more likely to be captured.

Wale et al. (2013) examined the effects of noise exposure on the predatory responses of the shore crab. A wooden stick was plunged by hand into the water and straight out again (only the dowel touched the water), directly in front of the crab; this was used to simulate the action of a bird attempting to catch the crab. The initial reaction of the crab to the predator stimulus was recorded as either running (movement away from the stimulus) or freezing (cessation of all movement). After the dowel was removed, the time taken for the crab to return to the shelter was recorded. The results showed that although captive shore crabs exposed to ship noise playback were just as likely as ambient-noise controls to detect and respond to a simulated predatory attack, they were slower to retreat to shelter. Crabs placed on their backs and exposed to ship noise playback also righted themselves faster than those experiencing ambient noise Wale et al. (2013) argued that from a functional perspective, faster righting could be perceived as beneficial, with the crabs able to escape predation quicker, spending shorter periods of time on their back with their weak undersides exposed. However, because remaining motionless may reduce the likelihood of further predatory attack, such behavior may reduce the likelihood of further predatory attention.

In a laboratory study, Voellmy et al. (2014a) examined how additional noise (playback of field recordings of a ship passing through a harbor), compared with control conditions (playback of recordings from the same harbors without ship noise), affected responses to a visual predatory stimulus. They compared the anti-predator behavior of the same two sympatric fish species, the three-spined stickleback and the European minnow, where they had previously examined the effects of sound on their feeding behavior. Unlike sticklebacks, minnows do not possess body armor, which is likely to influence their relative levels of risk-taking behavior. The anti-predator behavior consisted of a response to an overhead visual stimulus (a seagull model that moved over the top of the tank) when fish were exposed to additional noise. Effects of additional-noise playbacks differed between the two species: sticklebacks responded significantly more quickly to the visual predatory stimulus during additional-noise playbacks than during control conditions, while minnows exhibited no significant change in their response latency. It was suggested that elevated noise levels have the potential to affect anti-predator behavior of different species in different ways.

Simpson et al. (2015) subsequently showed that captive juvenile European eels (*Anguilla anguilla*), exposed to additional noise (playback of ship noise) in an aquarium tank, performed less well in two simulated predation paradigms. Eels were 50% less likely and 25% slower to startle in response to an ambush predator and were caught more than twice as quickly by a pursuit predator. Furthermore, eels experiencing additional noise showed diminished spatial performance and elevated ventilation and metabolic rates (indicators of stress) compared with control individuals. The authors suggested that acoustic disturbance could have important physiological and behavioral impacts on animals, compromising life-or-death responses.

In later experiments the same group (Bruintjes et al., 2016a) examined how eel anti-predator behavior was affected, both during short-term (two-minute) exposure to playback of recordings of ship noise and in the immediate aftermath of noise exposure. As previously found, noise exposure decreased eel anti-predator responses, increased startle latency and increased ventilation rates relative to ambient-noise-exposed controls. However, the results showed for the first time that those effects quickly dissipated; eels showed rapid recovery of startle responses and startle latency, and rapid albeit incomplete recovery of ventilation rate in the two minutes after noise cessation. There was complete recovery in the case of eel anti-predator startle responses. The authors concluded that if recovery from short-term noise exposure is rapid, then the fitness consequences may be lessened; survival may only be compromised during actual periods of noise pollution. Noise exposure may have little lasting influence if there is an opportunity to compensate in quieter periods, especially if there is rapid recovery from any initial impact.

Simpson et al. (2016) examined the effects of motorboat noise on post-settlement survival and physiology of a prey fish species and its performance when exposed to predators. Both playback of motorboat noise and direct disturbance by motorboats elevated metabolic rates in the damselfish (*Pomacentrus amboinensis*), which when stressed by motorboat noise responded less often and less rapidly to simulated predatory strikes. Prey were captured more readily by their natural predator (dusky dottyback [*Pseudochromis fuscus*]) during exposure to motorboat noise compared with ambient conditions, and more than twice as many prey were consumed by the predator in field experiments when motorboats were passing. The study suggested that noise in the marine environment has the potential to impact fish demography, highlighting the need to include anthropogenic noise in management plans.

Shafiei Sabet et al. (2015) investigated the impact on both predator and prey for zebrafish (*Danio rerio*) preying on water fleas (*Daphnia magna*). They experimentally raised ambient sound levels in an aquarium and tested four sound conditions that varied in temporal pattern: continuous, fast and slow regular intermittent and irregular intermittent, which were compared with ambient sound levels with no extra exposure. They found no effects on water flea swimming speed or depth, but a number of individual zebrafish showed increased startle responses, especially to the intermittent sound treatments, which was also reflected in a significant increase in zebrafish swimming speed, but not in any change in zebrafish swimming depth. Discrimination in attacking edible water fleas or inedible duckweed particles was low for the zebrafish and unaffected by sound exposure, but foraging was affected in two ways: intermittent sounds delayed the initial acceleration response and all treatments caused a rise in handling error. The experiments confirmed that elevated sound levels, and especially intermittent sounds, may affect predator prey interactions.

Spiga et al. (2017) suggested that piling and drilling sounds affected the kinematic component of the anti-predator response of European sea bass (*Dicentrarchus labrax*) (affecting their swimming path and velocity, including turning), rather than the behavioral component (responsiveness and response latency).

Little of the effect of anthropogenic noise on marine organisms within a natural setting. However, a recent study of the impact of real boat noise on wild juvenile damselfish (*Pomacentrus wardi*) by McCormick et al. (2018) has shown that noise from 30-horsepower 2-stroke outboard motors reduced boldness and activity of fish on habitat patches compared with ambient reef-sound controls. Fish also no longer responded to alarm odors with an antipredator response, instead increasing activity and space use, and fewer fish responded appropriately to a looming threat. In contrast, although there was a minor influence of noise from a 30-horsepower 4-stroke outboard on space use, there was no influence on their ability to respond to alarm odors, and no impact on their escape response. The evidence suggested that anthropogenic noise impacts the way juvenile fish assess risk, which reduces individual fitness and survival; however, not all engine types caused major effects.

Ferrari et al. (2018) investigated the effect of playback of boat noise on fish cognition (the ability of individuals to learn and remember information). Fish exposed to boat noise playback failed to subsequently respond to a predator, although their reef noise counterparts responded appropriately. Further experiments indicated that these results were likely due to failed learning, as opposed to stress effects from the sound exposure. Neither playbacks nor real boat noise affected survival in the absence of predator training. The results indicated that boat noise has the potential to cause latent effects on learning long after the stressor has gone.

Migrations and Home Ranges

Alterations to migratory cues, including sound, as a result of developments in the sea, estuaries, rivers and lakes may have adverse effects on migratory fish. High level sounds may also result in avoidance responses, deflecting fish away from their migration routes. Bagočius (2015) assessed the possible negative impacts on migrating Atlantic salmon (*Salmo salar*) caused by pile driving noise. Measurements made in the Baltic Sea showed that pile driving into the bottom of a lagoon generated sound pulses with an SEL of 218 dB re µPa$^{2\cdot}$ s @1 m and it was concluded that this level posed a risk to the migrating fish. It was said that the noise reached the levels likely to cause behavioral reactions at a distance of 40–100 m from the pile. However, the risk to the migrating fish was assessed using rather poor salmon hearing threshold data and by applying the rather doubtful response criteria suggested by Nedwell et al. (2007), which have been commented upon in Chapter 9.

Montgomery et al. (2006) suggested that larval reef fishes and decapod crustaceans may locate their home reefs by responding to their characteristic sounds. Many reef species tend to be associated with particular sites as adults but have a pelagic phase at an early part of their life history cycle. This pelagic period in the water column is considered to be a dispersal phase. These animals are capable swimmers and later decide to settle on a reef. They can locate reefs from hundreds of meters if not kilometers away. The authors considered the physics of underwater sound, how it is produced, and how it propagates.

Radford et al. (2010) identified marked differences in the characteristics of ambient sound in different types of coastal habitat, including a macro algal dominated reef, a sea urchin dominated reef, and a sandy beach. Their study provided evidence that there are significant differences in the spectral and temporal composition of ambient sound associated with different coastal habitat types over relatively short spatial scales. An acoustic cue that conveys both directional and habitat quality information that is transmitted considerable distances offshore would have the potential to be of immense value to the pelagic larval stage of a coastal organism attempting to remotely locate a suitable habitat in which to settle. They subsequently showed through field measurement and modeling that the spatially extended sound source of a reef creates a surrounding zone, which extends for a distance offshore equal to the length of the reef, within which there is almost no loss in the sound level (Radford et al. 2011).

In a more recent study, Gordon et al. (2018) pointed out that factors like climate change are causing widespread damage to the world's tropical coral reefs. The recruitment of juvenile fish is influenced by acoustic cues that guide larval orientation, habitat selection, and settlement to reefs. Their recordings of Australia's Great Barrier Reef before and after recent severe degradation demonstrated major changes to natural reef sound. In field experiments using those recordings, they demonstrated the potential impact of such acoustic changes. Following environmental degradation, the reef sounds were less attractive than their pre-degradation equivalents to young fishes. Reductions in fish settlement, caused by acoustic changes, may threaten the recovery potential of degraded coral reefs.

Earlier independent studies (Leis and Carson-Ewart 2003; Simpson et al. 2004) had shown that attraction to sound occurs with a very wide range of tropical reef fish species. It would appear that sound is used as an orientation and settlement cue for the late larval stages of reef fishes and crustaceans. Holles et al. (2013) subsequently carried out laboratory sound playback experiments with coral reef larvae, using continuous noise. They concluded that anthropogenic noise could have a disruptive effect on the response of fish larvae to natural reef sounds, with implications for their settlement and population dynamics.

Stanley et al. (2012) pointed out that although sound is a useful cue for guiding the orientation of larvae because it travels long distances underwater, it also has the potential to convey valuable information about the quality and type of the habitat at the source. They provided evidence that settlement-stage coastal crab species could interpret and show a strong settlement and metamorphosis response to habitat-related differences in natural underwater sound. Laboratory and field-based experiments demonstrated that the time to metamorphosis in the settlement-stage larvae of common coastal crab species varied in response to different underwater sound signatures produced by different habitat types. The megalopae of five species of both temperate and tropical crabs showed a significant decrease in time to metamorphosis, when exposed to sound from their optimal settlement habitat type compared with other habitat types. The results demonstrated that it is the frequency and temporal composition of underwater sound rather than the overall sound level per se that is the important characteristic of sound that mediates settlement and metamorphosis in the crab megalopae, which makes it clear that the metrics used to describe sounds are of critical importance. The results also indicated that sounds emanating from specific underwater habitats may play a major role in determining spatial patterns of recruitment in coastal crab species.

Pine et al. (2012) investigated how the sound emitted from an underwater tidal turbine and an offshore wind turbine would influence the settlement and metamorphosis of the pelagic larvae of estuarine brachyuran crabs. In a laboratory experiment the median time to metamorphosis (TTM) for the megalopae of the crabs *Austrohelice crassa* and *Hemigrapsus crenulatus* was significantly increased by at least 18 h when exposed to either tidal turbine or sea-based wind turbine sound, compared with silent control treatments. Contrastingly, when either species were subjected to natural habitat sound, observed the median TTM decreased by approximately 21–31% compared with silent control treatments, 38–47% compared with tidal turbine sound treatments, and 46–60% compared with wind turbine sound treatments. A lack of difference in median TTM in *A. crassa* between two different source levels of tidal turbine sound suggested that the frequency composition of turbine sound was more relevant in explaining such responses rather than sound intensity. It was concluded that sound mediates natural metamorphosis behavior in two common species of estuarine crabs, and that exposure to continuous turbine sound interferes with this natural process. These results raised particular concerns about the potential ecological impacts of sound generated by renewable energy generation systems placed in the nearshore environment.

Parmentier et al. (2015)examined the swimming behavior of coral-reef fish larvae from 20 species of 10 different families under natural and artificial sound conditions. Overall, their results highlighted two settlement strategies: a direct selection of habitats using sound (45% of the species), or a by-default selection by avoidance of certain sound habitats (35%). These results clearly demonstrated the need to analyze the influence of sounds at the species-specific level because congeneric and confamilial species expressed different behaviors when exposed to the same sounds.

Simpson et al. (2016) tested which components of reef noise evoke behavioral responses in larval fish. They used light traps to measure the responses of a diverse range of settlement-stage fish to the filtered high-frequency (570-2000 Hz) and low-frequency (<570 Hz) components of reef noise and compared these catches with those from control silent traps. Of seven fish families investigated, four (Pomacentridae, Apogonidae, Lethrinidae and Gobiidae) were caught in significantly greater numbers in the high-frequency traps than in either the low-frequency or the silent traps. The Syngnathidae preferred high to low-frequency traps, while the Blenniidae preferred high-frequency to silent traps. Only the Siganidae showed no preference between any of the sound treatments. Although some species-level variation in response was found, the general trend was a preference for high-frequency traps. The study suggested that most settlement-stage fishes select the higher-frequency audible components of reef sound, which arise mainly from the sounds made by marine invertebrates, as a means of selectively orienting towards suitable settlement habitats.

Sound Production by Fishes

Within a family of fishes, the sounds of different species can often be distinguished by their different temporal patterns (Hawkins and Rasmussen 1978a). The great majority of sounds emitted by fish is produced in a social context and involve interactions between individuals (Myrberg 1981). Sounds are produced when an individual is disturbed by a predator or subjected to a noxious stimulus. They are also produced during aggressive displays between fish of the same species. The most common context of sound production is during reproductive activity, where often the male fish is the dominant sound producer. Here, the sounds produced may serve to attract females, drive away competing males, and synchronize mating activities and the release of eggs and sperm (Casaretto et al. 2015).

Radford et al. (2015) Radford et al. (2015) used both field and laboratory experiments to describe the sound production of a nocturnal planktivore, the New Zealand bigeye (*Pempheris adspersa*), and provided calculations for the potential effective distance of the sound for intraspecific communication. The vocalizations were as popping sounds, with individual pops of short duration (7.9±0.3 ms) and a peak frequency of 405±12 Hz. Sound production varied during a 24-hour period, with peak vocalization activity occurring during the night, when the fish were most active. The source level of the bigeye vocalization was 115.8±0.2 dB re 1 μPa m. The effective calling range, or active space, depended on both season and lunar phase, with a maximum calling distance of 31.6 m and a minimum of 0.6 m. It was suggested that the fish vocalizations functioned effectively as contact calls for maintaining school cohesion in darkness.

Radford et al. (2016b) recently examined sound production by captive bluefin gurnard (*Chelidonichthys kumu*). Four types of sound were produced and characterized, twice as many as previously reported in this species. These sounds fitted two aural categories; grunt and growl, the mean peak frequencies for which ranged between 129 to 215 Hz. This species vocalized throughout the 24-hour period at an average rate of 18.5±2.0 sounds per hour, with an increase in vocalization rate at dawn and dusk.

Sounds and Spawning

Many fishes make sounds during spawning. For example, male haddock (*Melanogrammus aeglefinus*), make repetitive low-frequency sounds during their reproductive behavior. Observations by Casaretto et al. (2015) showed that male haddock were territorial and that visits to their territories by females, induced by the sounds of males, triggered courtship behavior, leading to spawning. The observations were consistent with the behavior expected from lekking species. Lekking is said to occur when non–resource-based aggregations of males are visited by females for the purpose of mating.

Haddock like other widely distributed gadoids, gather together in large concentrations to spawn at particular locations on the continental shelf. Within these areas the male haddock produce a diversity of sounds over the spawning season, with distinctive sounds associated with particular behavioral acts (Hawkins and Amorim 2000). The dominant male fish engage in their territorial displays, which are accompanied by almost continuous production of regularly repeated, low-frequency sounds. Detailed studies in the aquarium have shown that haddock reproductive behavior involves extensive acoustic and visual displays (Hawkins et al. 1967; Hawkins and Rasmussen 1978b). Male haddock produce a long series of separate “knocks” that are repeated at a faster rate as a female approaches and courtship proceeds, culminating in a continuous hum. Courtship ends in a sexual embrace, where both fish move upwards through the water, simultaneously releasing eggs and sperm (Hawkins et al. 1967). In contrast to the visual components of the displays, sounds are more likely to be effective over a distance in the sea (Hawkins et al. 1967), indicating the location of male aggregations and their readiness to spawn. Then later, following courtship, the sounds may help to synchronize the release of eggs and sperm into the water.

Casaretto et al. (2014) showed that it was possible to locate haddock aggregations within a Norwegian fjord by means of their sounds. In particular, at night, the individual sounds merge into a continuous low-frequency rumble, with many male haddock producing sounds simultaneously. Similarly, vocal aggregations of male Atlantic cod (*Gadus morhua*) have been recorded off the Norwegian Lofoten Islands (Nordeide and Kjellsby, 1999). It has been suggested for haddock by Casaretto et al. (2014) that the mature female fish, ready for spawning, may detect the male sounds at a distance, visiting the areas occupied by the males and then selecting particular males, perhaps on the basis of the distinctive sounds that they produce. It is evident that the calls of individual males differ, and that these differences may convey information about the fitness of the males, enabling females to choose between them (Casaretto et al. 2015). Disruption of spawning by anthropogenic sounds might lead to additional time and effort by males to re-establish territories and by females to repeat the process of choosing their mates. Females might have to travel farther or search longer to find males. The overall effect might be to reduce reproductive success.

Particular locations are also important for aggregation of soniferous coral reef fishes. McWilliam et al. (2017) examined the soundscapes at a number of field sites around Lizard Island in the Great Barrier Reef, Australia. Six individual fish choruses were identified where each chorus displayed distinct acoustic characteristics. Choruses exhibited diurnal activity and some field sites displayed consistently higher diversity of choruses and levels than others. Several of the choruses displayed site fidelity, indicating that particular sites may represent important habitat for fish species, such as fish spawning aggregations sites. Coral reefs, and perhaps other habitats, are finely balanced, highly sensitive ecosystems, with the inhabitants showing a high degree of territorial behavior that may be very susceptible to changes in environmental conditions, including changes to the ambient noise induced by anthropogenic sounds. Anthropogenic sounds may also interfere with the mating behavior itself, either by masking the sounds made by the fish or by resembling those sounds.

De Jong et al. (2017) declared that acoustic communication often plays a crucial role in reproductive interactions and more than 800 species of fish have been found to communicate acoustically. In their study they tested the effect of a low-frequency continuous multi-tonal sound on courtship behavior in two closely related marine fishes: the two-spotted goby (*Gobiusculus flavescens*) and the painted goby (*Pomatoschistus pictus*) in aquarium experiments. Both species used visual and acoustic signals during courtship. In the two-spotted goby a repeated-measures experiment tested the same individuals in the noise and the control treatment, in alternating order. For the painted goby they allowed females to spawn and tested the effect of noise on female spawning decisions. Males of both species reduced acoustic courtship, but only painted gobies also showed less visual courtship in the noise treatment compared with the control. Female painted gobies were less likely to spawn when exposed to noise. They concluded that their results provided experimental evidence for negative effects of noise on spawning success. They pointed out that spawning is a crucial component of reproduction and although their laboratory results should not be extrapolated directly to field populations, their results suggested that reproductive success may be sensitive to noise pollution, potentially reducing fitness.

Interference with Fish Communication

Fish vocalizations are an important component of the marine soundscape, and they provide valuable information regarding the behavior of the signaler in a variety of different contexts, such as general interactions, territorial displays, feeding, contact vocalization, and courtship interactions. Some species vocalize at key life stages or while foraging, and disruption to the acoustic habitat at these times could lead to adverse consequences at the population level (Putland et al., 2017).

Lugli (2010) pointed out that the habitat ambient noise may exert an important selective pressure on frequencies used in acoustic communication by animals. Lugli examined the spectral features of ambient noise in very shallow freshwater, brackish and marine habitats and correlated them with the range of dominant frequencies of sounds used by nine species of Mediterranean gobies reproducing in these environments. Ambient noise spectra of these habitats featured a low-frequency quiet window centered at 100 Hz (stream, sandy/rocky seashore), or at 200 Hz (spring, brackish lagoon). The analysis of the ambient noise/sound spectrum relationships showed the sound frequencies of the fish calls matched the frequency band of the quiet window in the ambient noise typical of their own habitat. This adaptation improves signal detection and recognition, thereby increasing the active space of acoustic communication. Thus, any generation of anthropogenic noise within these quiet windows may greatly reduce the ability of fish to communicate with one another.

Vasconcelos and Ladich (2008) investigated the effects of ship noise on the detectability of communication signals in the Lusitanian toadfish (*Halobatrachus didactylus*). Ambient and ferry-boat sounds were recorded in the Tagus River estuary (Portugal), as well as toadfish sounds, and their sound pressure levels determined. Hearing sensitivities were measured under quiet lab conditions and in the presence of these masking noises at levels encountered in the field, using the AEP recording technique. The Lusitanian toadfish was described as a hearing generalist, with best hearing sensitivity at low frequencies between 50 and 200 Hz (thresholds below 100 dB re 1µPa). Under ambient noise conditions, hearing was only slightly masked at lower frequencies by ambient sound levels. In the presence of ship noise, auditory thresholds increased considerably, by up to 36 dB, at most frequencies tested. Comparisons were made between masked audiograms and sound spectra of the toadfish's mating and agonistic vocalizations and it was suggested that ship noise decreased the ability of the fish to detect conspecific acoustic signals. Vasconcelos and Ladich concluded that acoustic communication, which is essential during agonistic encounters and mate attraction, might be restricted in coastal environments altered by human activities.

A study by Stanley et al. (2017) investigated the alteration of estimated effective communication spaces at three spawning locations for populations of the commercially and ecologically important fishes. As we have seen, both Atlantic cod and haddock make extensive use of sound, especially during spawning. Both the "grunt" and "knock" vocalizations emitted by Atlantic cod and haddock occupy the same frequency range as many underwater anthropogenic sound sources, with the peak of acoustic energy in the 50–260 Hz frequency band. Stanley et al. (2017) pointed out that high levels of low-frequency noise could reduce the communication space at spawning sites during times of high vocalization activity. It should also be noted that the calls from such species tend to be repetitive, and pile driving sounds, at a considerable distance from the source, would resemble the sounds of these fish and might be interpreted by the fish as being fish calls. This finding raises concerns that communication between conspecifics may be compromised by exposure to underwater noise during critical biological periods.

Kunc et al. (2014) pointed out that although increasing noise levels were specifically a problem for species using acoustic signals (i.e., species relying on signals that use the same sensory modality as anthropogenic noise), many species used other sensory modalities, such as visual and olfactory signals, to communicate. They examined the effects of anthropogenic noise (underwater engine noise) on the common cuttlefish (*Sepia officinalis*) which uses highly complex visual signals. They showed that cuttlefish adjusted their visual displays by changing their color more frequently during a playback of anthropogenic noise, compared with before and after the playback. They concluded that their experimental results provided evidence that anthropogenic noise also has a marked effect on the behavior of species that are not reliant on acoustic communication. Interference in the acoustic sensory channel may affect signals in other sensory channels.

It is important to assess the risk to spawning aggregations of sound producing fish from the effects of anthropogenic noise (including pile driving). It would also be sensible to locate the regular spawning locations favored by sound producing species like the haddock and cod, and then to consider whether seasonal protection of these areas from noise-making activities is needed to ensure that the fish remain within these areas and that their spawning behavior is not disrupted.

The Masking of Biologically Significant Sounds

The fact that fishes can hear shows that sound is important to them. Indeed, as we have seen, many fishes themselves produce sounds, and many are acutely sensitive to sounds. At some frequencies, some species, including the cod, are not limited by their absolute sensitivity but by the masking of low level sounds by the background or ambient noise, even under relatively quiet sea conditions (Hawkins and Chapman, 1975). Any increase in the level of sea noise results in a decline in sensitivity. Where the detection of a sound is impaired in the presence of ambient noise, or is masked by anthropogenic sound, not all frequencies contained within the background noise are equally effective at masking (Hawkins and Chapman, 1975). Experiments with cod and salmon (Hawkins and Johnstone, 1978) have shown that only the frequencies in a narrow band on either side of a masked tone (the critical band) contribute to the masking. Thus, both cod and salmon employ an auditory filter to remove the masking effect of frequencies away from the sound being listened to. The filter in the cod is rather narrower than the filter in the salmon. In practical terms, to determine the range over which a sound can be detected by a fish it is necessary to know the source level and frequency characteristics of the sound, the threshold of the fish to those frequencies, losses during transmission, the prevailing level of ambient noise, and the width of the critical band for the fish. Some authors (for example Nedwell et al., 2007) have ignored the importance of the critical band and have suggested that the audiogram itself can be used to determine the detection range. In contrast, those working with cetaceans have recognized to need to take account of the critical band (see for example Southall et al., 2007).

High levels of low-frequency sound generated by ships, pile drivers, air-guns, low-frequency sonars and other sources can interfere with detection of the overall acoustic scene (or soundscape) as well as affecting communication by means of sound in fish, both through masking biologically significant sounds that overlap in time and frequency and through the generation of signals that are similar to those produced by the fish themselves. Anthropogenic sounds can disrupt their lives significantly by preventing them from hearing approaching predators such a seals, dolphins and otters, or by preventing them from detecting the sounds of their prey, or by generating sounds that the fish may mistakenly believe are calls from other fish, thereby disrupting mating behavior. Many fish calls are repeated low-frequency pulses, similar to those generated at a distance by sources like pile drivers and seismic airguns.

Radford et al. (2014) reviewed how acoustic signaling fish might respond to the presence of anthropogenic noise. They concluded that there was evidence that some species of fish might have the potential to compete with anthropogenic noise by making changes in their calls.

Kaplan and Mooney (2015) examined the noise levels generated by small boats at three coral reefs in the U.S. Virgin Islands National Park. They emphasized that because increased noise may affect the behavior and physiology of marine organisms, detailed assessments of noise levels in many habitats were needed to better understand the extent to which animals may be exposed to increasing noise. They showed that there was substantial overlap between vessel noise and the relevant frequency bands for fish communication and hearing. The abundance of boat noise on these reefs reflected the prevalence of noise as a potential stressor. The frequency overlap with vessel noise could result in masking of sounds vital to reproduction, feeding, and territorial defense. In a subsequent paper, Kaplan et al. (2016) emphasized the need for long-term acoustic monitoring alongside more targeted studies investigating the potential for effects of noise on aquatic organisms.

Holt and Johnston (2014, 2015) investigated what effects elevated noise levels had on a sound-producing freshwater fish, the blacktail shiner (*Cyprinella venusta*). Sounds were played back to the fish in a shallow aquarium tank. Elevated levels of natural river noise were played back to the fish and the effects on courtship and aggressive signals examined. Several acoustic features of the fish calls were altered under noisy conditions. Most notable the spectral composition of the calls was altered by the fish (termed the Lombard effect). However, inter-fish distances remained unchanged between quiet and noisy conditions. In a later study (Holt and Johnston, 2015) it was shown that the calls of the fish were likely to be masked by noise from traffic on a bridge across a river. The distance at fish calls might emerge from the traffic noise was estimated using a simple propagation model. The noise was shown to diminish the normal active area of both growls and knocks to some degree, depending on the frequency of concern and the distance from the noise source. Growls showed the potential to be more severely affected than knocks, due to their lower amplitude and greater spectral overlap with anthropogenic noise. Holt and Johnston concluded that entire watersheds may be impacted by noise pollution from traffic crossings. The noise measurements made within the river showed that it was possible for sounds to propagate into the substrate where they could re-emerge into the water column within the river.

Pine et al. (2016) carried out a comprehensive survey of underwater vessel sound and then estimated the likely impact distances for acoustic masking within the Hauraki Gulf (near Auckland New Zealand). Their results suggested that the sound emanating from both recreational and commercial vessels would significantly raise background sound levels and was likely to have a wide-ranging masking impact on marine life.

The Effects of Seismic Airguns

The behavioral and physiological effects of exposure to airguns has been reviewed by a number of investigators (McCauley et al., 2000; Slabbekoorn et al., 2010; Normandeau, 2012b; Popper et al., 2014; Radford et al., 2014; Hawkins et al., 2015; Carroll et al., 2017). Although signals from seismic air guns are similar to those of pile driving in terms of frequency range of major energy, duration of impulse, and having a rapid rise time, there are few data from seismic studies that inform development of guidelines because most of the studies have not measured the sound levels received by the animals. And, even when there are such data, there is substantial variability in results depending on species and study methods.

A number of studies have examined the behavior of fish exposed directly to a seismic survey. McCauley et al. (2000) found that captive fishes showed a generic fish alarm response whereby the fish swam faster, swam towards the bottom, tightened school structure, or showed all three responses, at an estimated 2–5 km from a seismic source.

Several other studies showed various responses of fishes to air guns. For example, Slotte et al. (2004), using sonar, observed that fish (Atlantic herring and blue whiting (*Micromesistius poutassou*) in an area where seismic shooting occurred, moved out of the area or to deeper waters (10–50 m deeper). Wardle et al. (2001) observed small changes in the position of pollack (*Pollachius pollachius*) in response to the firing of a single airgun. Wilson and Dill (2002) found that herring dropped in the water column when exposed to predator (simulated odontocete) sounds. Finally, studies on caged sandeels (*Ammodytes marinus*), in the North Sea, revealed distinct but minor reactions to seismic shootings (Hassel et al., 2004).

In one of the few studies that measured sound levels received by the fish, Fewtrell and McCauley (2012) exposed various species of caged marine fish (the trevally, family Carangidae; and the pink snapper [*Pagrus auratus*]) and also one species of squid, to the noise from a single air gun. The behavior of the animals was observed using underwater cameras. Sound levels received by the animals ranged between 120 and 184 dB re 1 $\mu Pa^2 \bullet s$ (SELss). Results indicated that as air gun sound levels increased, the fish responded by moving to the bottom of the water column and swimming faster in more tightly cohesive groups. Significant increases in alarm responses were observed in fish and squid to air gun noise exceeding 147–151 dB re 1 $\mu Pa^2 \bullet s$ (SELss). The authors pointed out that precise responses to air gun and seismic survey sounds are species specific and dependent on the actual sound exposure regime.

Early Norwegian studies on the impact of seismic surveys on fish (summarized in Dalen et al., 2007) suggest that impact ranges of airguns are several meters for injury but that behavioral changes in some fish species at distances of tens of kilometers may affect catch rates in fisheries. Handegard (2010) summarized the results of two surveys of the effects of seismic shooting on fish catches in Norway and compared the difference in sound exposure (see also Handegard et al., 2013). In an earlier study, trawl and long-line catches of cod and haddock declined by about 50% following 5 days of shooting. In more recent experiments, long-line catches also declined, but gillnet catches increased. The number of airgun emissions were far higher in later experiments carried out by Løkkeborg et al. (2012 a, b), but were distributed over a larger area and for a longer duration. In a central position within both areas (assuming cylindrical spreading), the total accumulated sound exposure levels (SEL_{cum}) were similar over the duration of the two experiments, but the daily levels were approximately 10 dB higher in the earlier Engås et al. (1996) experiments. These data, however, do not inform the setting of exposure criteria because the measured sound exposure levels were over long periods and received levels were not determined.

Peña et al. (2013) described the real-time behavior of herring schools exposed to a full-scale 3D seismic survey, observed using sonar. No changes were observed in swimming speed, swimming direction, or school size that could be attributed to a transmitting seismic vessel as it approached from a distance of 27 km to 2 km, over a 6-hour period. The unexpected lack of a response to the seismic survey was interpreted as a combination of a strong motivation for feeding by the fish, a lack of suddenness of the airgun stimulus, and an increased level of tolerance to seismic shooting.

A number of more recent studies have demonstrated that exposure to seismic airguns has an impact on fish catches, presumably as a result of changes in fish behavior and distribution during and after sound exposure (e.g., Løkkeborg et al. 2012 a, b; Streever et al. 2016). Reductions in fish catches have actually been observed in commercial line and trawl fisheries during and after seismic surveys (Løkkeborg et al. 2012 a, b). In some studies, catch rates in static gillnets increased, and this result was attributed to a rise in swimming activity by the fish in response to airgun sounds, thus making the fish more vulnerable to capture by gillnets (Løkkeborg et al. 2012a, b).

Streever et al. (2016) carried out long-term monitoring of fish catches using four fyke nets allowed assessment of changes in catch rates during a 2014 seismic survey in Prudhoe Bay, Alaska. Fyke net locations were instrumented with both conventional hydrophones and vector sensors. Catch rates were generally within the range of those found in 27 previous sampling seasons. The effect of air guns on eight species was assessed using a modified Before-After/Control-Impact analysis, with historical data and 2014 data as the Before-After components of the analysis and days without and with air gun activity as the Control-Impact components. Results showed significant changes associated with air guns in catch rates at one or more nets at $p < 0.1$ for all eight species and at $p < 0.05$ for seven of the eight. Changes included both increased and decreased catch rates, perhaps reflecting displacement of fish in response to air gun sounds throughout the study area. Measured sound pressure levels associated with air gun pulses were low and usually undetectable close to the fyke nets, reflecting the loss of low frequencies in shallow water (approximately 1.5 m). Attempts to measure particle velocities failed when wind-driven surface waves overwhelmed vector sensors. However, Streever et al. concluded that fish responses may have been related to changes in particle motion associated with the air gun sounds.

In conclusion, it is evident that behavioral reactions can occur to seismic airguns, but at this stage there are few data that can be applied to develop guidelines. Carroll et al. (2017), in a critical review of the potential impacts of marine seismic surveys on fishes and invertebrates, concluded that the main challenges of seismic impact research are the translation of laboratory results to field populations over a range of sound exposure scenarios and the lack of sound exposure standardization which hinders the identification of response thresholds. An integrated multidisciplinary approach to manipulative and *in situ* studies is the most effective way to establish impact thresholds in the context of realistic exposure levels, but if an integrated multidisciplinary approach is not practical, the limitations of each approach must be carefully considered.