

# Assessing the Impact of Underwater Sounds on Fishes and Other Forms of Marine Life

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*Until we gain more information on the impacts of man-made sounds on marine life, interim procedures will have to be developed to assess the risks to fishes.*

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## Effects of Sound on Aquatic Life

Current expansion of offshore industrial activities has led to concern about the impact of man-made sounds upon marine animals (Southall et al., 2007; Hastings, 2008; Popper and Hawkins, 2012, 2014). Offshore oil and gas exploration and developments, wind farm construction and operations, other renewable energy sources, dredging, construction activities, naval sonars, and increases in commercial shipping are all contributing to increased noise in the sea.

While most concerns have been focused on effects on marine mammals, similar issues arise with other marine life including fishes, turtles, and invertebrates.<sup>1</sup> While the basic principles we discuss, however, are applicable to all marine groups, the focus of this paper will be on fishes since that is our particular area of research interest.

Many marine animals use sound during their everyday lives to track prey, avoid predators, navigate, and communicate with one another (e.g., Hawkins and Myrberg, 1983). And even species that do not communicate by sound use the acoustic scene (or soundscape) to learn about and exploit their environment (Fay and Popper, 2000). Thus, anything in the environment that interferes with the ability of a fish to detect and use sounds of biological relevance could have a substantial impact on fitness and survival.

A succession of reports and scientific papers has now emphasized the potential risks to marine animals from exposure to man-made sounds or noise (Southall et al., 2007; Popper and Hawkins, 2012, 2014; Popper et al., 2014). Increasingly, environmental assessments of the impact of offshore developments and other activities have been required to consider the effects of underwater noise.

An environmental assessment essentially evaluates the effects of underwater noise in terms of mortality or any physical injury, impairment to hearing, or behavioral disturbance it might cause to animals in the ocean. The assessment end points are typically aimed at determining whether there is a significant impact on populations of marine animals and on the wider ecosystem. Often a threshold for an adverse effect is sought, and this can lead to conclusions about the likely severity of any impact. This process, referred to as risk assessment, can subsequently be used to construct “what-if” scenarios to evaluate methods for effective prevention, control, or mitigation of impacts, and to provide a reasoned basis for action to reduce risks.

<sup>1</sup> The invertebrate species most susceptible to man-made sounds have yet to be identified. The most likely candidates are those that may detect the kinetic components of sound. Many invertebrates have statocysts that may be involved with sound detection. These species include cephalopods (octopus, squid, and relatives) and crustaceans (crabs, shrimp, lobster, and relatives). For purposes of this discussion, we will refer to invertebrates, but recognize that those affected may form only a small part of this large and diverse group of animals.

## Risk Assessment and Sound Exposure Criteria

As part of the risk assessment process, it is necessary to predict the levels of different types of sounds that may have potential impacts on marine animals, as well as those that are likely to be of no consequence. A criterion is usually provided as a threshold value, expressed in a particular acoustic metric, above which a particular level of damage may take place or behavioral change occur. The precise nature of any effects and the actual metrics that describe the sounds must be specified clearly, although there are many difficulties in achieving this. For example, not only the level of the sound but its frequency range, rise time, duration, repetition rate, and a number of other parameters can also be important in assessing its impact. Understanding the risk to animals therefore becomes more complex than just setting a single threshold value. But in all cases, the fundamental question must lie in understanding how animals respond to various sounds.

The first set of comprehensive sound exposure criteria for marine animals was recommended for marine mammals (Southall, et al. 2007). There is much less information available for other marine animals including turtles, fishes, and invertebrates, although recent guidelines, developed under the auspices of the Standards Group of the Acoustical Society of America (ASA), do provide directions and recommendations for ultimately setting criteria for fishes and turtles (Popper et al., 2014). Currently there is insufficient information to guide the setting of criteria for any invertebrate species.

However, impacts have to be assessed and interim procedures have to be developed until more direct information is available. There are both explicit and subtle pressures to achieve unity and consensus in preparing environmental statements, and because of legal considerations the assessments are not always based on the most recent or best science. There is strong dependence on criteria developed or utilized by government agencies, although these do not always reflect the latest scientific position. For example, the criteria for marine mammals suggested by Southall et al. (2007) have not yet been applied even within the legislative environment that led to that publication. However, many of the recommendations now form the foundation of the recent draft guidance issued by the US regulatory authority, the National Oceanic and Atmospheric Administration, (NOAA, 2013), but at a time when Southall and his colleagues are embarking upon a revision of their original recommendations.

## Challenges in Developing Sound Exposure Criteria

There are a number of difficulties in achieving better sound exposure criteria for effects upon marine life. Chief amongst these is the lack of information on those aspects of underwater sounds that actually cause detrimental effects – whether the result is physical (or physiological) injury, hearing impairment, or changes in behavior. Sounds from various sources differ greatly in their characteristics (see chapters in Popper & Hawkins, 2012, 2014). Some sounds are continuous, such as the sounds from ships, dredging, drilling, operating wind and tidal turbines, and some naval sonar systems. These sounds may be tonal, or they may include a wide range of frequencies. Some may be ‘rougher’ than others, with a high crest factor.

Many of the sounds currently being produced in the sea and reaching high sound levels are short-lived or transient. They typically have a sharp rise time, are of brief duration, and may contain a wide range of frequencies. Examples are the sounds from explosions, seismic airguns, and percussive pile driving. Often the impulses are repeated for long periods and thus the duty cycle and total exposure duration need to be taken into account in assessing any effects upon animals. Inevitably, it is necessary to use a range of metrics to describe these sounds fully (Ellison & Frankel, 2012; Ellison et al., 2012).

Adding to the complexity is the diversity of animals, particularly among fishes and invertebrates, which have the potential to be affected by man-made sounds. There are relatively few species of marine mammal to consider (about 125). However, there are over 32,000 extant species of fish ([www.fishbase.org](http://www.fishbase.org)) and tens of thousands of species of marine invertebrates. This diversity, particularly in body type and physiology, is likely to result in substantial interspecific differences in how sound affects different species, as has been shown for studies of effects of naval sonars (Popper et al., 2007; Halvorsen et al., 2012b) and seismic airguns on fishes (McCauley et al., 2003; Popper et al., 2005).

In setting sound exposure criteria there are a number of scientific options. It has been commonplace in the past to specify those sound levels that result in injury to animals, especially if these are likely to result in death. However, this level of damage occurs only very close to very intense sound sources like percussive pile drivers. For marine mammals it has been considered more relevant to estimate the received levels, or thresholds, above which individual marine mam-

mals are predicted to experience changes in their hearing sensitivity (either temporary or permanent) from underwater anthropogenic sound sources. In its most recent guidelines for marine mammals, NOAA (2013) has designated acoustic threshold levels for the onset of both temporary (TTS) and permanent hearing threshold shifts (PTS) for different marine mammal groups, and for both continuous and impulsive sources. However, NOAA has stressed that these acoustic threshold levels do not represent the entirety of an impact assessment. Rather, they provide one of several tools (in addition to behavioral impact thresholds, auditory masking assessments, and other evaluations) to help understand the ultimate effects of any particular type of impact.

With fishes, it is effects on behavior that are considered most relevant in terms of effects upon populations (Popper et al., 2014). These effects can occur at much greater distances from the source than sound levels that can do physical harm, and they almost always involve a lower onset threshold than tissue injury or damage to the auditory system.

### **Metrics**

Because of a general lack of information on the effects of sounds on fishes and other marine animals the sound exposure criteria that have been applied in practice do not always reflect the complexity of the sounds to which animals are being exposed or the hearing capabilities and behavioral responses of the animals themselves. For example, as a conservative measure, the NOAA Fisheries and the US Fish and Wildlife Service (USFWS) have used 150 dB re 1  $\mu$ Pa RMS (Root Mean Square) as the threshold for behavioral effects to fish species that are listed as being threatened or endangered. This criterion has been applied in many biological opinions evaluating percussive pile driving activities. The criterion was selected on the basis that sound pressure levels in excess of 150 dB re 1  $\mu$ Pa RMS could cause temporary behavioral changes (startle and stress) that might decrease a fish's ability to avoid predators (Woodbury and Stadler, 2008; Stadler and Woodbury, 2009). The scientific origin of this value is not known (Hastings, 2008). In addition, species differences have not been taken into consideration in applying this value.

Moreover, sound levels expressed as RMS values may be appropriate for some continuous sounds but they do not adequately describe more complex sounds, as the RMS simply averages out varying sound levels. Sounds that are transient (of short duration and high amplitude) can cause particular damage to tissues, and may also evoke strong behavioral

responses. For impulsive sounds the instantaneous peak level has been used in a number of sound exposure criteria, although this metric does not account for the total energy within the sound and requires a fast sampling rate for effective measurement. The sound exposure level (SEL), which is related to the total acoustic energy, is used as a complementary metric. The SEL takes into account both level and duration of exposure (ANSI, 1994). This metric can be used to normalize a single sound exposure to one second, enabling sounds of differing duration to be compared.

The SEL can also be used to account for accumulated exposure to repeated sound energy over the duration of a repetitive activity such as pile driving, or for continuous activity over a specified period of time. The exposure is then expressed as the cumulative SEL ( $SEL_{cum}$ ) (Popper and Hastings, 2009; Halvorsen et al., 2012a).

The criteria agreed upon by the US Fisheries Hydroacoustic Working Group (FHWG, 2009) for the onset of effects of percussive pile driving activities in terms of injuries to fishes identified the dual criteria of a peak sound pressure level of 206 dB re 1  $\mu$ Pa and an  $SEL_{cum}$  of 187 dB re 1  $\mu$ Pa<sup>2</sup>·s. The additional specification of a peak level recognizes that a cumulative SEL on its own may not be sufficient to account for all potential impacts. However, it is clear that even the use of these dual metrics cannot distinguish fully between a series of sounds that are damaging and those that are not. If the  $SEL_{cum}$  is to be used as a metric for a series of impulses, it is also important to specify the time period over which the SEL is accumulated, the number of impulses, the repetition rate (as there may be recovery between repeated pulses), and the rise time of individual pulses. Recent experimental evidence suggests that the basis for physical injury to fishes from percussive pile driving is a combination of energy in single strikes and the number of strikes, but these two are not related in a linear fashion (Halvorsen et al., 2012a).

Another issue is that in some cases, sound exposure criteria specify the level received by the animal. In others, they specify a level at a particular distance from the source (often neglecting the distributed nature of many real sources, whether they are large ships or extensive seismic airgun arrays). For simplicity, it is often assumed that animals remain at a constant distance from the source, but this may seldom be the case. Where animals are moving, perhaps to avoid the sounds, these movements may later take them outside the range of any effects. Some sources, like seismic airgun arrays and ships, are also moving. It can therefore be difficult



**Sounds we make in the sea may interfere with the ability of fishes to detect and use sounds of biological relevance, and could have a substantial impact on their fitness and survival.**

to model the actual sound levels received by the animals over time and space, or to define precisely the  $SEL_{cum}$  they experience. There is often insufficient information about the complexity of actual animal responses to understand when they will avoid sounds and when they will not.

While, as discussed above, sound levels are commonly expressed in terms of sound pressure, not all fishes can detect sound pressure. Fishes and all invertebrates capable of hearing are essentially sensitive to the kinetic elements of sounds (particle motion). Relatively few species of fish detect sound pressure (Popper and Fay, 2011). But it is still relatively rare to specify and measure sounds in terms of their particle motion levels, despite the importance of kinetic energy to these species.

Moreover, it is not only sound that travels through the water that is of interest, but sound may also be transmitted through the substrate as well, either through direct propagation or via interface waves. Pile driving and seismic airguns, in particular, may result in high levels of ground vibration to which many fishes and invertebrates are especially sensitive. Further complicating the issue is that ground vibrations may re-enter the water at some distance from the source at very high energy levels (Popper and Hastings, 2009), making standard propagation models less than useful in predicting signal levels at an animal distant from a source producing sound that penetrates the substrate.

### **Frequency Weighting**

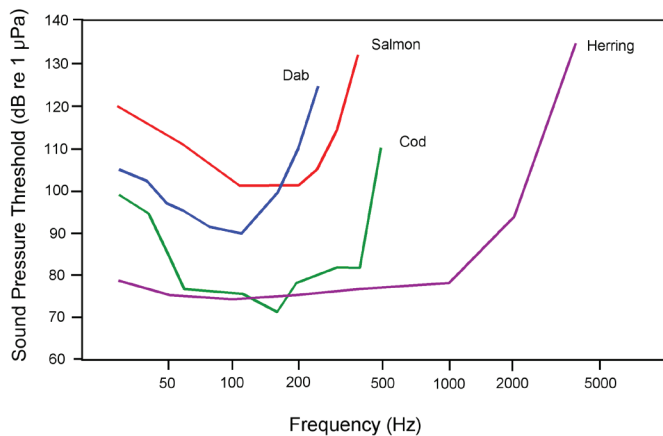
Animals do not hear equally well at all frequencies within their functional hearing range. They are more sensitive to some frequencies than others (Figure 1). Applying frequency weighting to measurements of man-made sounds offers a method for quantitatively compensating for differences in the frequency response of sensory systems. It minimizes the influence of extremely low- and high-frequency sounds sources that may be detected poorly, if at all, by the animal.

For marine mammals, generalized frequency-weighting functions have been derived for different functional hearing groups, distinguishing species that only detect lower frequencies from those that detect ultrasound. Thus, Southall et al. (2007) developed ‘M-weighting’ curves to compare the effects of man-made sounds upon different marine mammals.

The use of weighting curves is especially relevant when effects in terms of behavioral responses of animals are being considered. With tissue injury, or damage to the auditory system, frequencies falling outside the hearing range of the animals may still be important and cannot be eliminated. In this case weighting is not always appropriate. For example, although they may be inaudible, the high frequencies associated with rapid rise-times in impulsive signals may bring about or exacerbate injury. If an animal is subject to seismic airguns or pile driving, the higher frequency components may result in injury even if the animal cannot hear those frequencies. For this reason the latest draft NOAA guidelines for marine mammals do not recommend the use of weighting for measuring peak sound levels (NOAA, 2013).

In evaluating the impact of sounds upon humans, use is made of weighting curves based on equal loudness contours (Suzuki et al., 2004). Observers are asked to match sounds against one another to compare their subjective loudness. Such curves are applied in the evaluation of effects from environmental and industrial noise, and for assessing potential hearing damage and other noise health effects. Thus, the A-weighting curve is derived from the inverse of an idealized equal loudness hearing function across frequencies, standardized to 0 dB at 1 kHz (ANSI, 2006).

Equal-loudness contours are lacking for most marine animals and frequency-weighting functions are instead often based on hearing thresholds at different frequencies. Such weighting is not directly comparable to A-weighting. The hearing threshold (or auditory threshold) is the sound level that is just audible to an animal 50% of the time either under quiet conditions, or in the presence of a specified background noise level. Plotted as a function of frequency these threshold data provide an audiogram (Figure 1). Hearing thresholds are generally determined for pure tones (a single frequency), ideally against a low level of background noise.



**Figure 1:** Audiograms for four species of fish; the dab *Limanda limanda* (Chapman and Sand, 1974); the Atlantic salmon *Salmo salar* (Hawkins and Johnstone (1978); the Atlantic cod *Gadus morhua* (Chapman and Hawkins, 1973); and the Atlantic herring *Clupea harengus* (Enger, 1967). Auditory thresholds for the first three of these species were obtained by behavioral conditioning experiments carried out in the sea. Thresholds for the herring were obtained using auditory evoked potentials from the ear in experiments in the laboratory. Note that the audiograms for the dab and salmon are expressed in terms of sound pressure, for purposes of comparison, although both of these species are actually sensitive to particle motion. The values shown for these two species would only be valid under free-field conditions.

Although measuring equal loudness levels in human listeners is relatively straightforward, it is much more difficult to examine loudness matching or perform loudness comparisons with aquatic animals. Nevertheless, some have aspired to determine such weighting curves for marine mammals. Thus, Finneran and Schlundt (2011) have relied upon objective measurements, such as response latency, to estimate equal loudness contours for the bottlenose dolphin (*Tursiops truncatus*). From their experiments they derived auditory weighting functions, which they have suggested could be used to predict the frequency-dependent effects of noise on odontocetes (toothed whales, including dolphins).

In the latest provisional NOAA guidance on sound exposure criteria (NOAA, 2013) marine mammals are divided into functional hearing groups (low-, mid-, and high-frequency cetaceans, and otariid and phocid pinnipeds). Marine mammal auditory weighting functions are incorporated into the setting of threshold criteria in the  $SEL_{cum}$  metric. Use is made of “representative” or surrogate individuals/species for establishing acoustic threshold levels for species where little or no

data exist. This is done as a matter of practicality, as there are insufficient adequate data for all marine mammal species found worldwide.

### Hearing Sensitivity Measures for Fishes

It is of course important that the audiograms on which weighting curves are based are obtained under acoustic conditions that give reliable and repeatable measures that truly reflect the performance of the animal’s hearing system. Of the many extant species of fishes, very few have had their audiograms measured (e.g., Ladich and Fay, 2013), and of these only a very few have been measured under acoustic conditions that provided a calibrated acoustic field with valid measurement techniques. Hawkins (2014) and Rogers et al. (2014) have recently reviewed experiments aimed at providing those conditions. Most studies have been done without regard to the kinetic element of the sound field. Moreover, they have often been done in small chambers (often with glass or plastic walls) where the sound fields are highly complex and where it is almost impossible to set up and calibrate a reliable kinetic field (Parvulescu, 1964; Rogers et al., 2014).

In addition, there are methodological problems associated with the determination of hearing thresholds and the preparation of audiograms. Significantly, experiments to determine audiograms for fishes are often carried out in noisy aquaria where the thresholds determined to a particular sound may be greatly affected by the level of man-made background noise. Indeed, even when determined against natural aquatic noise backgrounds, detection of the stimuli may be masked, as shown by Hawkins and Chapman (1975). Here, the audiogram may parallel ambient noise levels, especially at the lower frequencies, where ambient noise is higher. Masking by noise may influence the curve used for weighting.

The techniques that have been used to obtain the actual thresholds from fishes also vary by investigator and laboratory (i.e., there is no standard protocol). Some thresholds have been determined using behavioral conditioning techniques. Here, the animal is trained to show a distinctive behavioral response when exposed to sounds (e.g., Tavolga and Wodinsky, 1963). Such threshold determinations require a significant investment of time in training each animal to respond, but they do provide a true measure of the best hearing capability to the sound (Sisneros et al., 2014). Behaviorally derived thresholds reflect the abilities of the animal to detect and process the sound, and give an indication of the lowest sound level to which an animal may give a behavioral response.

Many investigators have used physiological measures such as the auditory evoked potentials (AEP) to determine audiograms in fishes (reviewed by Ladich and Fay, 2013). The AEP is measured with electrodes close to the central nervous system or ear in response to short tone bursts. With repeated signal presentation and averaging of the response, the summed electrical activity in the vicinity of the electrode can be discriminated against the electrical background noise, and the “threshold” can be estimated as the minimum sound pressure required to elicit an electrical response visible to the investigator (or of a specified criterion magnitude).

It is important to recognize, however, that AEP measures of the audiogram can differ significantly from those derived using behavioral conditioning techniques (Sisneros et al., 2014). AEPs only reflect the responses to sound at the level of the hair cells of the ear, or the responses of particular groups of auditory nerve fibers, or in some cases the summed responses of cells within the central nervous system. Such physiological measures are very useful for comparing hearing mechanisms, or for determining differences before and after some intervention, such as exposure to loud sounds (e.g., Popper et al., 2005; Halvorsen et al., 2012b). But they are much less useful for determining the hearing capabilities of a particular species, or comparing the behavioral responses of different species.

### Weighting and Sound Exposure Criteria for Fishes

The audiogram does not give a full indication of those sounds that will evoke behavioral responses or the magnitude of these responses in wild unconstrained animals. Nor does it provide information on the performance of the animal in more complex auditory tasks. As Liberman (2014) has pointed out, criteria for risk to the auditory system are usually constructed assuming that the audiogram is the gold standard functional test, and therefore that an exposure which causes only a temporary threshold shift is essentially benign. Liberman has shown that this assumption is not true for mice and guinea pigs, and we would predict that this would also be the case for aquatic vertebrates.

There are more than 32,000 extant species of fishes to be considered, and the choice of appropriate surrogate species or the definition of functional hearing groups is especially problematic. One solution is to divide fishes into several different categories based on the structures associated with hearing and then develop generalized guidelines that, at least

for now, do not depend on the audiograms (Popper et al., 2014). The functional groups include:

- fishes without a swim bladder (these can only detect kinetic energy – e.g., sharks, gobies, flounder, some tuna including Atlantic mackerel);
- fishes with a swim bladder that is far from the ear and thus not likely to contribute to pressure reception, so the fishes are primarily kinetic detectors (e.g., salmon, cichlids); and
- fishes where the swim bladder or other air bubble is close to the ear and enables sound pressure to be detected, broadening the hearing range and increasing hearing sensitivity (e.g., goldfish, herring, sprat, catfish, cod).

Popper et al. (2014) considered that fishes showing sound pressure sensitivity are more likely to be affected by any increase in man-made noise since the sound levels are more likely to be well above their hearing thresholds than will be the case for fishes in the other groups.

Nedwell et al. (2007) proposed a systematic weighting approach for application to aquatic animals using a metric known as the  $dB_{ht}(\text{Species})^2$  as a tool for quantifying the level of sound experienced by individual marine species (including marine mammals). The  $dB_{ht}$  metric takes into account each species’ hearing ability by referencing the sound to the hearing thresholds for that species. Since any given sound will be detected at different levels by different species (as they have differing hearing abilities) the species name is appended when specifying a level. For instance, the same sound may have a level of 70  $dB_{ht}$  for the Atlantic cod (*Gadus morhua*) and 110  $dB_{ht}$  for a common seal (*Phoca vitulina*). The  $dB_{ht}$  is said by the originators to be similar to the A-weighting that is used for human sound exposure in air. Actually, it is not strictly analogous to A-weighting as the  $dB_{ht}$  is based on the audiogram, whereas the A-weighting is based on subjective equal loudness contours.

The level of a man-made sound expressed as  $dB_{ht}(\text{Species})$  is usually much lower than the un-weighted sound level, because the latter contains energy at frequencies that the species cannot detect. The weighting eliminates this energy. Where the energy within the received sound falls mainly within the hearing range of the animal, then the weighted level may be similar to the un-weighted level.

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<sup>2</sup> Strictly the use of such attachments to the dB is incorrect, and strongly deprecated by standards authorities. The use of such attachments is necessary here to enable us to refer to the work of others but we would emphasize that the symbol dB indicates a non-dimensional ratio and is neither a quantity symbol nor an abbreviation for level.

Essentially, the  $dB_{ht}$  (Species) metric is a frequency-dependent, non-dimensional ratio of measured sound level to the hearing threshold of an animal. The weighting is not just applying a frequency filter; it is providing a level that is weighted by the actual sensitivity of the animal to sound, as indicated by the audiogram. The level of a sound expressed in  $dB_{ht}$  will be higher for an animal with greater hearing sensitivity.

It is of course critical that the  $dB_{ht}$  (Species) be based upon accurate behavioral threshold determinations. Values based on AEP thresholds are often employed, although these thresholds rarely provide valid hearing measures for fishes, as cautioned earlier.

Moreover, as we have pointed out above, not all fishes and perhaps no invertebrates, respond to sound pressure. Many are sensitive to particle motion. In theory a  $dB_{ht}$  value can be determined for particle motion. However, the value is more commonly expressed in terms of sound pressure, even for animals that are known to be sensitive to particle motion. Particular care must be taken in doing this as the values will not be appropriate under all acoustical conditions, especially for low frequency sounds. Indeed, there are many circumstances where it will be inappropriate; for example close to a sound source, close to the sea surface, and in shallow water.

Despite the lack of high quality audiograms for the majority of marine animals, the  $dB_{ht}$  (Species) has often been utilized within the United Kingdom for assessing the effects of man-made sounds upon these animals, and it appears to have the tacit approval of some regulatory agencies. In particular, the  $dB_{ht}$  (Species) has been used to evaluate the likelihood of fishes responding behaviorally to sound exposure.

Nedwell et al. (2007) suggested that strong avoidance responses by fishes start at a level about 90 dB above the  $dB_{ht}$  (Species) thresholds, while different proportions of fishes respond at lower weighted levels. Mild reactions in a minority of individuals may occur at levels between 0 and 50 dB above the hearing threshold, and stronger reactions may occur in a majority of individuals at levels between 50 and 90 dB above the hearing threshold.

It must be noted, however, that these recommended levels are largely derived from the proportion of fishes reacting to sounds in only a very few studies on a few species of fish in very particular environments (Maes et al., 2004; Nedwell et al., 2007). There are very few other field data derived from wild fishes under different conditions to support the assump-

tions about the sound levels at which fishes will react. Moreover, the initial observations by Nedwell and his colleagues were based on fishes exposed to swept tonal sounds; sounds that are rather different from the sounds generated by, for example ships or percussive pile drivers. Clearly, substantial caution must be exercised in applying the  $dB_{ht}$  measure. Indeed, defining response criteria applicable to all species may be too simplistic an approach to evaluating behavior.

### **Behavioral Measures of the Responses of Fishes to Sound**

A major problem in assessing magnitude of effect is how to interpret expressions used in the  $dB_{ht}$  approach such as “*strong avoidance reaction by virtually all individuals*” in terms of the effects on the behavior of particular fishes engaged in different activities. Avoidance reactions by cod, perhaps gathered in an area at a particular time of year for spawning, must be assessed differently to avoidance responses within a routine feeding area by dab (a species without a swim bladder).

Similarly, interruption of the annual coastal return migrations of a species may need especially careful consideration. Environmental statements often deal with these difficulties by constructing short verbal scenarios for the fishes concerned, outlining any effects upon animal populations and the wider ecosystem. So far, however, these scenarios have been mainly anecdotal and speculative, with a minimum of actual evidence being presented. Thus, the most important part of the risk assessment is often the least supported by quantitative data.

Application of a weighted and formulaic approach to impact assessment has the virtue of being relatively easy to use. The  $dB_{ht}$  (Species), properly applied and based upon a legitimate audiogram, does permit the distance at which a sound is detected to be estimated. However, the use of this metric to forecast the level of response is too simplistic. In practice, very few studies have been carried out to investigate the levels of sound at which behavioral responses occur for the key species at risk. Those experiments that have been carried out have not always defined the actual responses of the fish in any detail. The assumption that particular levels of response occur at specific  $dB_{ht}$  levels for all species of fish requires validation if it is to be routinely applied in risk assessments. It is apparent from experiments in the field that the behavior of fishes and other animals can be greatly affected by a wide range of factors, including their previous



experience of sound exposure, seasonal changes, day/night differences, and the very nature and condition of the animals themselves (e.g., their motivation). Where sounds are well above their hearing thresholds animals will not necessarily be constrained in their behavioral responses by their hearing abilities. As the NOAA (2013) guidelines point out, auditory weighting functions best reflect an animal's ability to hear a sound. These functions may not necessarily reflect how an animal will perceive and react behaviorally to that sound.

Based on this discussion, it is evident that there are major procedural difficulties in bridging the gaps between setting sound exposure criteria, estimating the distance and time over which specified effects upon behavior might occur, and then evaluating the actual risk to fish populations.

### Behavioral Studies of Fishes

Supporting the need to better understand the actual behavioral responses of fishes is a recent series of experiments on the behavior of wild, pelagic fishes in response to sounds. These studies indicate that fishes can show strong behavioral reactions to impulsive sounds (Hawkins et al., 2014). The experiments also showed that the responses of a particular species to sounds can differ greatly from day to night (Figure 2).

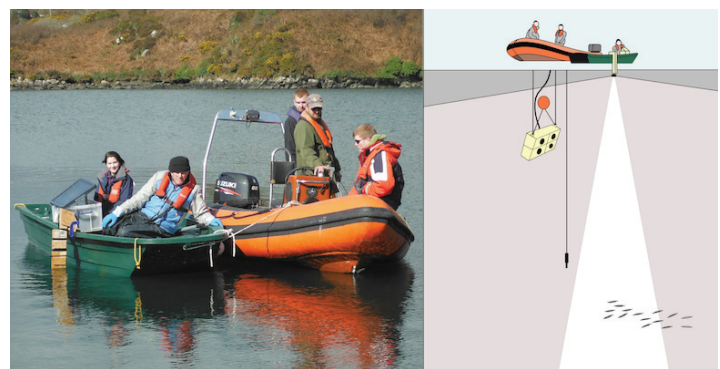
In these experiments, two species of fish, the sprat *Sprattus sprattus* (related to herring) and Atlantic mackerel *Scomber scombrus* (related to tuna) (Figure 2) were examined at the same coastal location. Schools of both sprat and mackerel were exposed to short sequences of repeated impulsive sounds, simulating the strikes from a percussive pile driver, at different sound pressure levels. The sound exposure experiments were carried out in a quiet area where fishes were not accustomed to heavy disturbance from shipping and other intense sound sources. Two small boats, tethered together, were allowed to drift silently over fish schools, and sounds, as well as silent control trials, were presented from an array of custom-built sound projectors (Figure 3).

Behavioral responses included the break up of fish schools and changes in the depth of the schools (Figure 4).

The incidence of responses increased with increasing sound levels, with sprat schools being more likely to disperse and mackerel schools more likely to change depth. The sound pressure levels to which the fish schools responded on 50% of presentations were 163.2 & 163.3 dB re 1  $\mu\text{Pa}$  peak-to-peak, and the single strike sound exposure levels were 135.0 & 142.0 dB re 1  $\mu\text{Pa}^2\cdot\text{s}$ , for sprat and mackerel respectively,

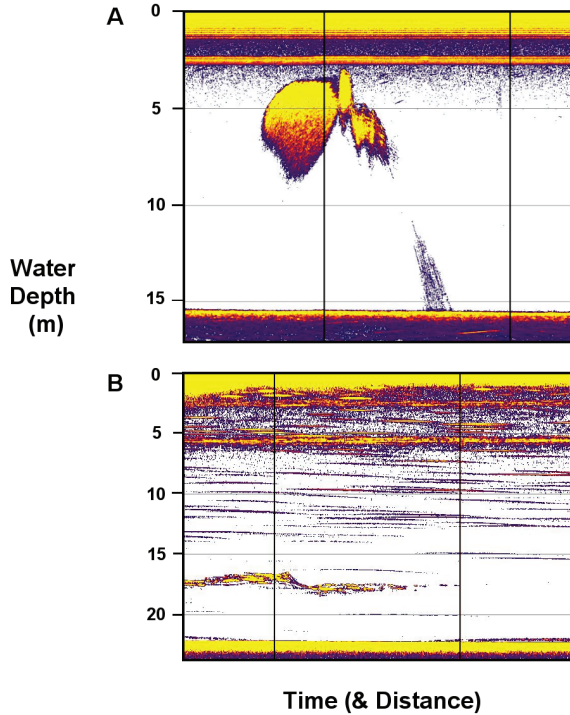


**Figure 2:** Atlantic mackerel *Scomber scombrus* (top) and European sprat *Sprattus sprattus* (bottom). Both species may gather in large schools in the sea.



**Figure 3:** Sound playback experiments were carried out from two boats, tethered together, drifting silently over fish schools. An array of four low frequency sound projectors was suspended from one boat, and a sonar system on the second boat used to observe fish schools. A short sequence of impulsive sounds (simulating percussive pile driving sounds) was transmitted and the subsequent responses of the fish followed on a combined echo sounder and side-scan sonar. A hydrophone was subsequently deployed at different depths to measure the received sound levels.

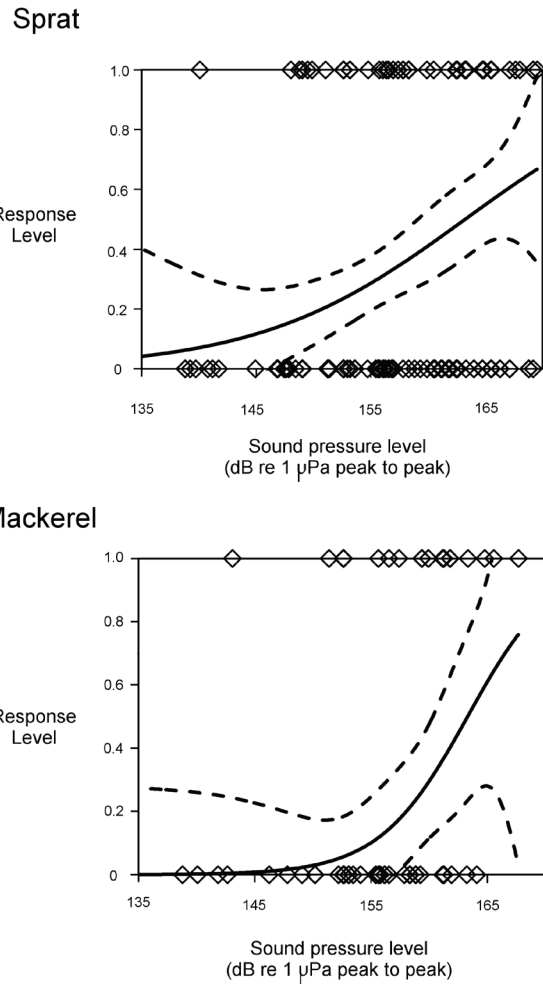




**Figure 4:** Responses of a sprat school (A, at 5 m depth) and a mackerel school (B, at 17 m depth) to sound playback, observed on the echo sounder. The sound was a 20 s sequence of 10 impulses, occurring between the two vertical lines. After a short latency the sprat school breaks up and the fish disperse. The mackerel school changes depth.

estimated from dose response curves (Figure 5). These levels were remarkably similar for sprat and mackerel. It was of particular interest, however, that the fish responded strongly to sound playback during daytime when they were aggregated into schools, but did not respond at night, when the schools had already broken up and the individual fish were dispersed.

Mackerel and sprat are very different species. The mackerel is a fast-moving predator, able to move rapidly from one depth to another. It lacks a gas-filled swim bladder, an organ which serves as an accessory hearing organ in many other fishes and which enables them to detect sound pressure (Popper and Fay, 2011). The hearing abilities of mackerel appear to be relatively poor. Iversen (1969) examined hearing in a closely related scombrid fish lacking a swim bladder, the mackerel tuna *Euthynnus affinis*, and found that it was much less sensitive to sound than other scombrid fishes with swim bladders. The mackerel audiogram is likely to be similar to that shown for the dab in Figure 1.



**Figure 5:** Dose response curves showing the responses of sprat and mackerel schools to sounds at different levels. Each data point is derived from a single school exposed to sound, with a response designated 1, and a lack of response designated 0. The response level effectively represents the proportion of occasions on which the fish responded. In each figure the solid line represents the non-linear regression fit to the data, and the dashed lines are the 95 % confidence intervals. In both cases, the received sound pressure is expressed as peak-to-peak level. Results were also examined in terms of the single strike sound exposure level and showed the same trends. Note that the mackerel is likely to be sensitive to particle motion.

In contrast, the sprat is a small forage fish, forming large dense schools. Clupeid fishes including the sprat are thought to be especially sensitive to sounds by virtue of specialized gas-filled bullae in the head, associated with the ear, that enables them to detect sound pressure (Enger, 1967; Blaxter et al., 1981). The sprat audiogram is likely to be similar to that shown for the herring in Figure 1.

During the day, when sprat aggregated in schools as a defense against predation, they were especially sensitive to sounds. At dusk, when attacks from visual predators were greatly re-

duced, the sprat schools broke up and the individual sprat dispersed, perhaps allowing them to forage and feed more effectively (Hawkins et al., 2012). At night the individual sprat no longer responded to the playback of pile driving sounds. During daytime, responses by both sprat and mackerel to impulsive sounds occurred at similar and relatively low sound pressures, corresponding in level to those recorded at tens of kilometers from an operating pile driver. We would stress, however, that it would be premature to use these data to define sound exposure criteria for sprat and mackerel. Other schools of the same species, under different conditions, might respond differently. Moreover, although the response levels were provided in terms of sound pressure it is likely that the mackerel responds to particle motion.

The next step must be to assess the implications of the behavior observed from these fishes. Does the break up of sprat and mackerel schools result in lasting damage to their populations? To answer this question it will be necessary to examine the effects of repeated exposure of the same fish aggregations to sound over time, and to evaluate the energetic and other fitness consequences of their responses.

### **Examining the Impact and Significance of the Observed Changes in Behavior**

More detailed studies of the behavior of the key species that are at risk are required to establish whether the responses observed are likely to result in adverse effects upon fish populations. Attempts have been made to model fish responses in the absence of direct information on their behavior when exposed to noise. Thus, Rossington et al. (2013) used an individual based model to predict the impacts on Atlantic cod from noise generated during a pile-driving event at an offshore wind farm in Liverpool Bay, UK. The model tracked individual “fish” within the population. Each “fish” was represented as a particle that was subject to advection by the tides and also had a set of behavioral rules, which governed their responses.

Compared with the “non-hearing” fish, the “hearing” fish were delayed in reaching their destination in the estuary as a result of the assumed behavioral changes. However, what significance can be attached to this finding? The assumption that cod were swimming towards a particular destination may not apply in practice. There are no available data on cod movements in the area concerned, and their movements may vary with season, time of day, and other factors. Moreover, the assumptions made on the responses of cod may not have

been realistic representations of what would happen if cod were actually exposed to pile driving noise. Indeed, it is very likely that context plays an important role in determining the behavior of fish including their responses to sounds, as it does for marine mammals (Ellison et al. 2012). The significance of behavioral responses will vary, depending on whether animals are feeding, migrating, seeking particular habitats, spawning, or engaged in other activities.

### **The Way Forward**

There is a need to examine more closely those sound exposure response patterns that give rise to significant detrimental effects on fish populations before a more complete risk assessment approach can be developed and incorporated into environmental statements. The development of specific sound exposure criteria, whether weighted or unweighted, is only the first step in performing risk assessment. It is important to specify in greater detail the characteristics of those sounds that result in effects. It is also necessary to describe the behavioural responses of the animals in greater detail and to assess the implications of those responses in terms of risks to populations. Significant changes in behavior might include abandonment of spawning behavior or spawning sites, movement away from preferred habitats, disruption of feeding, increased energy consumption, and diversion or delay of migrations.

There is also a need to reappraise the use of weighting curves, especially where these are used to assess the likelihood of behavioral responses from fishes (and other animals) to sound exposure. The use of the audiogram for weighting behavior, and its use to assess and compare responses at much higher sound levels is itself open to question. There is currently insufficient evidence to justify the establishment of a scale of weighted values, specifying the level of response for all species, to be routinely applied to environmental assessments.

Currently, our ability to model the levels of sounds from particular sources over space and time, although still imperfect, is improving. It is becoming possible to map the areas over which animals might experience effects and to assess the level of exposure of marine animals to sound under a range of circumstances. Predicting the effects of that exposure in terms of physical injury to fishes is now possible for some sound sources as a result of recent laboratory experiments (e.g., Halvorsen et al., 2012a; Casper et al., 2013). However, predicting effects upon behavior is much more difficult. Modeling behavior may offer scope for improving the objectivity

of risk assessment, rather than relying on simple anecdotal scenarios for describing the impact of behavioral changes. However, what we really need are more behavioral observations and experiments on the behavior of wild fishes. Only if we know how and when they react, and are able to assess the significance of those reactions, can we estimate the risks to which they will be exposed in an objective and valid way.

## Biosketches



**Professor Anthony (Tony) Hawkins** is a Director of Loughne, a small company carrying out marine research for government departments and other customers. Dr Hawkins' PhD thesis was on sound production by fish and resulted in a lead article in the journal *Nature* on the spawning behavior of haddock, an important food fish. He

subsequently received the A B Wood Gold Medal and Prize from the British Acoustical Society for his work on the hearing abilities of fish. After carrying out further research on fish migrations, fish energetics and the behavior of fish in response to fishing gears, Dr Hawkins was appointed Director of Fisheries Research for Scotland in 1987 and served in that post until 2002. He is currently working with fishermen and other stakeholders to provide advice to the European Commission on the management of fisheries in the North Sea – one of the most intensively fished areas in the World. Dr Hawkins is a fellow of the Royal Society of Edinburgh and was appointed a Commander of the British Empire by Queen Elizabeth for his work on fisheries.

**Arthur Popper** is professor of Biology at the University of Maryland and editor of *Acoustics Today*. His research interests over his career have focused on hearing by fishes, and this has more recently evolved into a focus on applied issues on effects of man-made sound on aquatic animals. Dr. Popper is also the editor of the Springer Handbook of Auditory Research (SHAR), a series of over 50 books that are very widely used in the auditory community.



Dr. Popper has also organized three international meetings on effects of noise on aquatic life (with Dr. Hawkins) and serves as consultant on the topic for government agencies and industry both in the U.S. and abroad. He is a fellow of the American Association for the Advancement of Science and of the Acoustical Society of America. Dr. Popper will formally “retire” from the University of Maryland at the end of June, 2014 but will continue his editorial and research activities as professor emeritus. And, in addition to his science-related activities, he will continue in his role as co-director of Terapin Teachers ([www.tt.umd.edu](http://www.tt.umd.edu)), a program that is focused on increasing the number, and improving the discipline-focused education, of college students who go on to become science and math teachers in U.S. high schools.

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- See also [http://www.dot.ca.gov/hq/env/bio/files/fhwgcriteria\\_agree.pdf](http://www.dot.ca.gov/hq/env/bio/files/fhwgcriteria_agree.pdf)
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