

Assessing the exposure of animals to acoustic disturbance: Towards an understanding of the population consequences of disturbance

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Citation: *Proc. Mtgs. Acoust.* **27**, 010027 (2016); doi: 10.1121/2.0000298

View online: <https://doi.org/10.1121/2.0000298>

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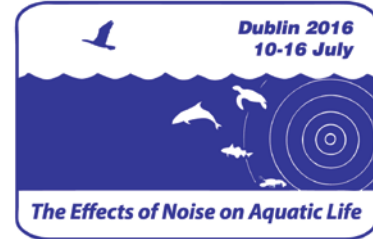
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Fourth International Conference on the Effects of Noise on Aquatic Life

Dublin, Ireland
10-16 July 2016



Assessing the exposure of animals to acoustic disturbance: Towards an understanding of the population consequences of disturbance

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An essential component of risk assessment is identification whether individuals will be exposed to a risk. This requires information on the proportion of the population exposed, for how long, and during what activity (i.e., feeding, migrating, and breeding). Using satellite telemetry data for humpback and blue whales feeding and migratory regions in Antarctica, California, and Bering Sea, we modeled the potential exposure of individuals to an acoustic disturbance. Foraging and transit regions along the tracks were identified and the time spent foraging in each region calculated. A simulated seismic survey was randomly placed (100 iterations) within the habitat of each of species and the amount of time individual animals were exposed determined. A large disturbance (i.e. 100 km) only exposed 6% of the population of humpback whales in Antarctica and 19% blue whales off California. In contrast, humpback whales in the Bering Sea experienced high exposure with only a 5 km disturbance. This approach can be used to develop a framework for estimating the likelihood that a given animal population would be exposed to disturbance and to develop general risk assessment guidelines.



1. INTRODUCTION

Considerable research has been carried out to assess the potential consequences of underwater sound on animals; however much of this effort has focused on the sensitivity of marine animals to underwater noise (Southall *et al.*, 2007; Popper and Hastings, 2009; Tyack *et al.*, 2011; Halvorsen *et al.*, 2012; Goldbogen *et al.*, 2013; Stimpert *et al.*, 2014; Fregosi *et al.*, 2016; Harris *et al.*, 2016; Lucke *et al.*, 2016). While critical to assessing the potential impact of underwater sound, a marine organism's sensitivity to sound only addresses one component of the risk. A complete risk assessment also requires determining the likelihood that an individual will be exposed to the sound, coupled with data on its sensitivity to that sound (Council, 1983; Harwood, 2000; Bittencourt *et al.*, 2014; Voellmy *et al.*, 2014; Nedelec *et al.*, 2015; Farcas *et al.*, 2016; Simpson *et al.*, 2016). Assessment of the likelihood of exposure is a necessary first step to evaluate the population impact of a disturbance, as it provides an estimate of the proportion of the population that would be exposed. These results can then be used to estimate the energetic costs of that disturbance, for example, on an individual's energy budget in terms of energy expended but not acquired, the additional time an individual would have to spend foraging to offset this lost foraging time, and the subsequent effects on offspring growth and survival (NRC, 2005; New *et al.*, 2014; Costa *et al.*, 2016). Figure 1 provides a graphic illustration of this decision process and Figure 2 shows the risk associated with different sensitivities and probabilities of exposure.

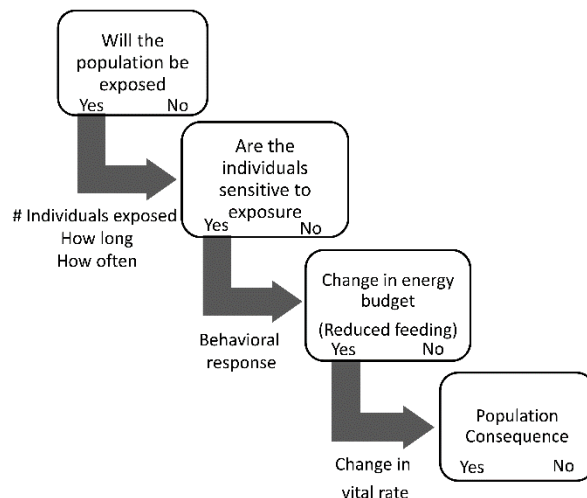


Figure 1. A graphic illustration of the decision path to assess the potential risk associated with exposure to underwater sound where the effect of the disturbance is to reduce the efficiency or time spent foraging.

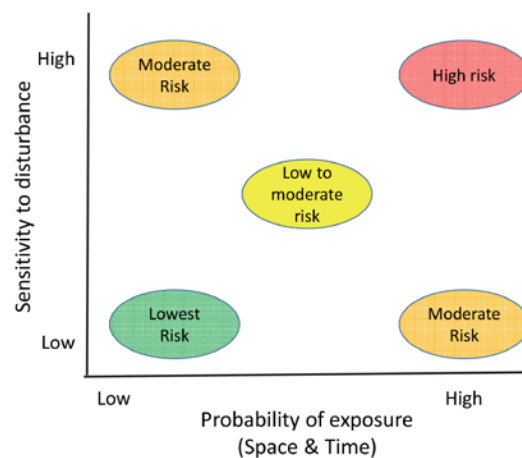


Figure 2. Risk associated with disturbance is a function of both the animal's sensitivity to disturbance and the probability of exposure. An animal with high sensitivity has low risk if there is a low probability of exposure. Similarly, a high probability of exposure has low risk if the animal is not sensitive to the sound. Finally, a high probability of exposure and a high sensitivity result in the highest risk.

For some groups of marine animals there are guidelines concerning their hearing sensitivity that can be used to inform environmental assessments (Southall *et al.*, 2007; Popper and Hastings, 2009). However, there are no guidelines for assessing the likelihood that a given animal would be exposed to underwater sound under natural conditions. The likelihood of

exposure is associated with life history traits, including whether the species is migratory or resident and, if migratory, whether the population migrates to a common feeding ground or a common breeding ground (Figure 3). The simplest scenario is for a resident non-migratory population, as there is no difference in the spatial extent of the various behaviors such as breeding and feeding. However, there is likely to be temporal separation, as breeding behavior in many species is highly seasonal. In contrast, migratory species often separate feeding from breeding both in space and time, and thus animals must transit between these different breeding and feeding areas. Consequently, the timing and sensitivity to disturbance is different across these life history strategies. It follows that the population consequences will also differ if individuals share a common feeding area versus a common breeding area. If individuals share a common feeding area, but return to unique breeding areas, they represent two demographically distinct populations. Alternatively, if they share a common breeding area, but forage in separate areas individuals are still one population, representing one demographic breeding unit.

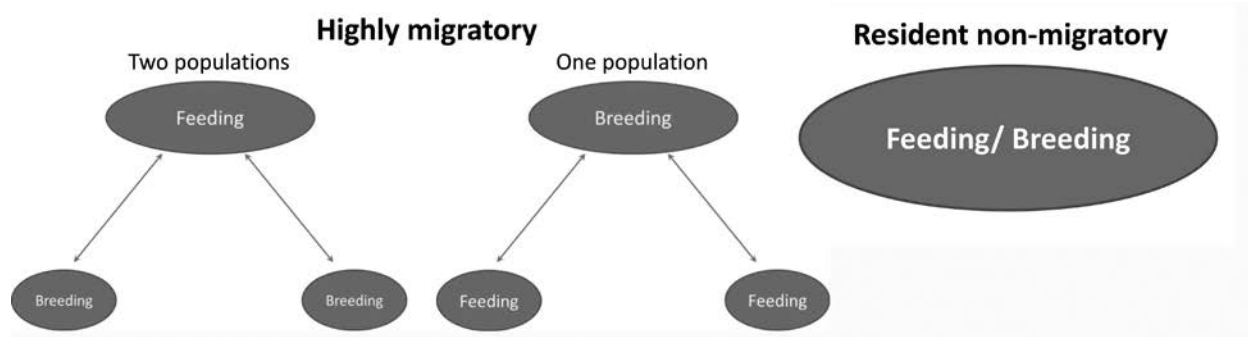


Figure 3. Schematic diagram of the different life history patterns of marine vertebrates. Resident non-migratory species carry out components of their life history within the same region, but may breed seasonally. In contrast highly migratory species migrate from either common breeding ground or common feeding grounds.

The bioenergetic consequences of disturbance are easier to assess in highly migratory species that have completely separated feeding from breeding (i.e. capital breeders), as these animals only feed at specific times and places and are not feeding during the migration or while breeding (New *et al.*, 2014; Braithwaite *et al.*, 2015; Villegas-Amtmann *et al.*, 2015; Costa *et al.*, 2016; Wright *et al.*, 2017). Species that are resident and non-migratory tend to be income breeders and are more constrained in space and time than capital breeders (Costa, 1993; Boyd, 1998; Houston *et al.*, 2007).

The species distribution pattern is also important in determining the potential exposure of a population to disturbance. For example, species range can be composed of many small individual home ranges, or a species range can be composed of many individuals whose migratory paths overlap, covering much of the species range (Figure 4). In a migratory species it is more likely that many individuals will travel over most, if not all, of the species range. In this situation, when the disturbance is small relative to the species range, there is a high probability that many of the individuals within the population will be exposed to the disturbance but only for a short period of time. In contrast, for non-migratory species where the species range is composed of individuals with small home ranges, when the disturbance is small, few individuals will be exposed, but they will have a prolonged exposure (Figure 4). Finally, when the disturbance is large relative to the species range, a large proportion of the population (both migratory and non-migratory) will be exposed and for longer periods of time (Figure 4). Not surprisingly, when the disturbance is large relative to the species range the potential for impact is highest irrespective of their migratory

behavior because many individuals will have a high likelihood of exposure. The relationship between species range, number of individuals exposed and area disturbed is shown in Figure 5. Importantly, differences in movement patterns can only be observed when animals are tracked (e.g. photo ID, telemetry) and could not be derived from single point animal sightings that would be obtained from a survey (Harris *et al.*, 2010).

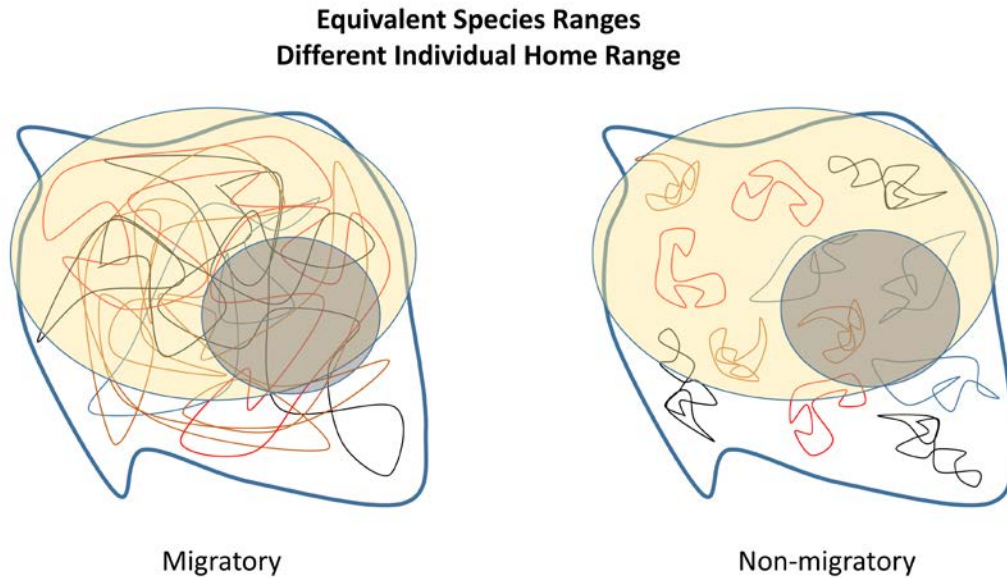


Figure 4. Difference in species range and home range between migratory and non-migratory species. The image on the left shows a hypothetical species distribution (blue line) that is composed of individuals who migrate over most of the species range. The image on the right shows the same species range (blue line), but one that is composed of individuals that are non-migratory with small home ranges. The small gray oval in each image indicates a small disturbance that affects a small proportion of the species range, and the larger peach oval represents a larger disturbance.

To examine how the above concepts can be applied we chose two species of highly migratory marine mammals the humpback, *Megaptera novaeangliae*, and blue whale, *Balaenoptera musculus*. These two species were chosen because of the concern about the effects of seismic surveys (Nowacek *et al.*, 2015), and the detailed information that is available on their movement patterns (Zerbini *et al.*, 2006; Dalla Rosa *et al.*, 2008; Bailey *et al.*, 2009). Further, migratory species they are the simplest to model for an initial analysis. Using empirical tracking data, we examined how the likelihood of exposure varied across these three populations in response to a simulated 3D seismic survey.

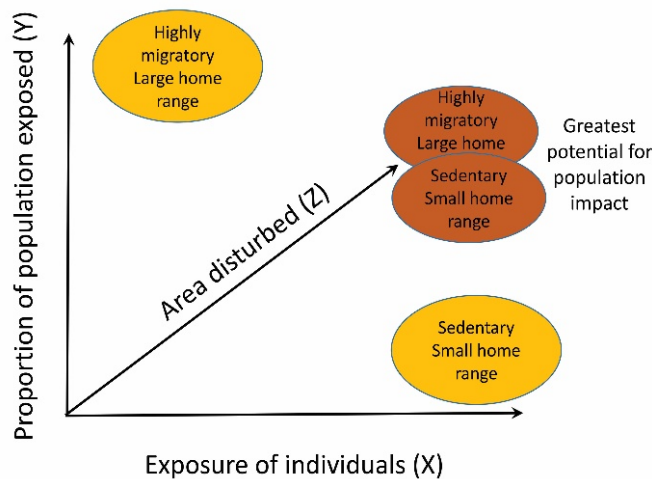


Figure 5. Relationship between the species range and the home range of individuals within that population. The yellow ovals (upper left and lower right) show the effects of a small disturbance relative to the species range. Brown ovals (upper right) show the effect of a large disturbance relative to the species range. The brown ovals have the greatest potential impact on both highly migratory and resident species.

2. MATERIALS AND METHODS

2.1 Animal movement

Existing Argos satellite tracking data were collated for blue whales in the California Current System ($n = 117$) and humpbacks whales in the Bering Sea ($n = 10$) and Western Antarctica Peninsula (wAP, $n = 24$). Tracks were processed using a Bayesian State-Space animal movement model (package *bsam* in R, (Jonsen *et al.*, 2005; Jonsen, 2016)) to filter the data and estimate behavioral states (Area Restricted Search ARS, and Transit, Figures 6 and 7). The tagging methods have been reported elsewhere (Zerbini *et al.*, 2006; Mate *et al.*, 2007; Dalla Rosa *et al.*, 2008; Bailey *et al.*, 2009). As data were collected in different years, all tracks were converted to day of the year (1 – 365 days) for analysis and were then used to calculate the proportion of the population that was exposed, the duration of the exposure for each individual, and the probability of individuals to be exposed during ARS or transit behavior. ARS has been used extensively as a proxy to derive foraging behavior from animal tracking data (Fauchald and Tveraa, 2006; Breed *et al.*, 2009; Costa *et al.*, 2012), and while it overestimates foraging behavior, it provides an index of when and where animals are attempting to forage (Robinson *et al.*, 2012). Alternatively, transiting behavior likely includes little or no foraging.

2.2 Disturbance

To address how different blue and humpback whales would be affected by seismic surveys, we chose to recreate generalized 3D surveys (~80 transect lines, ~50 x 50 km). The area of exposure moved along these transect lines at a speed of 9 km h^{-1} , simulating the movement of a seismic ship during a survey. The propagation of sound depends on the local characteristics of the water column and bathymetry thus making direct comparison across areas problematic. We therefore chose a simple approach that allows comparison between the different sites, by selecting circular areas of disturbance at different radii (5, 25, 50, and 100 km). We thus obtained a range of scenarios that incorporate the probable range of acoustic sensitivities. These scenarios ranged from 5 km, which might be associated with TTS (Temporary Threshold Shift)(Finneran, 2015) to 100 km, a range where the animal might experience a minor change in behavior (Southall *et al.*, 2007). The advantage of this approach is that it allows comparison across populations and species, highlighting differences associated with the animal's ecology and biology without the confounding variables associated with site specific acoustic propagation. When the site is known, the modeling approach detailed here could be applied with appropriate site specific input parameters.

To incorporate spatial variability, we randomly placed 100 3D surveys within the foraging range for each species (Bering Sea, California Current and Western Antarctica Peninsula). Temporal variability was considered in our simulations by starting the surveys at different weeks of the year (1 through 52), allowing us to define the time of year during when individual whales were likely to be exposed to a hypothetical seismic survey. For each iteration (100 surveys, 52 weeks, 4 radii of exposure) we calculated the proportion of the population (based on empirical movement data for that week of the year) that was exposed to the disturbance, the proportion of time that individuals were exposed, and the behavioral state that was affected by calculating the overlap between the whale tracks and the moving circle of disturbance.

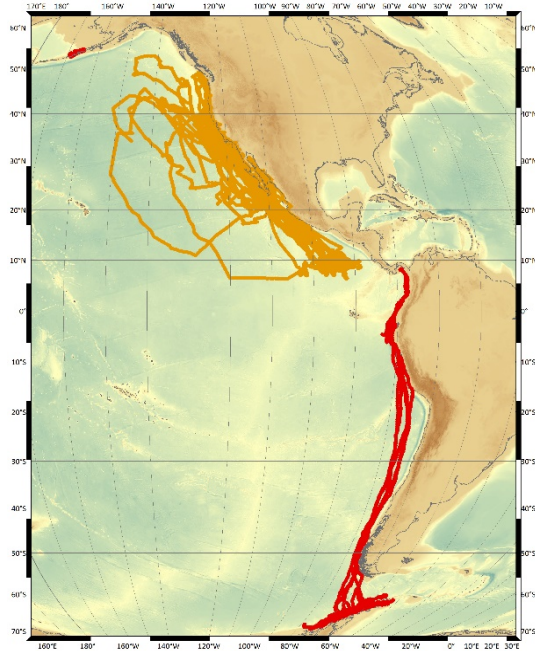


Figure 6. Map showing tracks of humpback whales (red lines) tagged in the Bering Sea and in Western Antarctic Peninsula. Tracks of blue whales (orange lines) tagged off California.

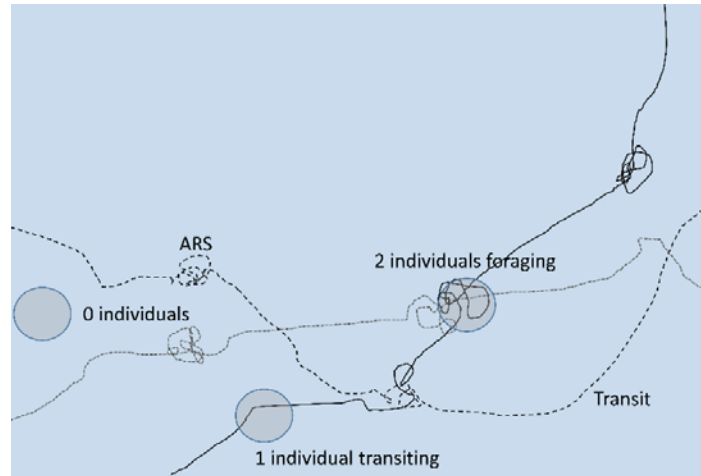


Figure 7. Tracks of 3 animals shown with their respective regions of putative foraging shown as regions of Area Restricted Search (ARS) and regions of more directed movement usually associated with transiting. Gray circles represent the simulated disturbance caused by a seismic survey. In this example disturbances occurred where no animals were present, where one animal was transiting and where two animals were foraging.

3. RESULTS

Since blue and humpback whales are migratory, it is not unexpected that there were long periods when none of the individuals were exposed to a simulated seismic survey (Figures 8 - 10). Their seasonal presence in different areas also resulted in pronounced peaks when individual whales were more likely to be exposed. As expected, the time and proportion of individuals exposed increased as the size of the disturbance increased. Humpback whales in the Bering Sea had the greatest potential for exposure compared to humpback whales in the western Antarctic Peninsula and blue whales.

The mean peak values presented in Figures 8, 9, and 10 for each population and each radii of exposure are provided in Table 1. Using these data, we calculated the relative amount of foraging time during the periods that there were foraging which, on average, individuals within each region would be exposed to the simulated seismic survey (Table 2). The potential effect on foraging behavior was far greater for humpback whales in the Bering Sea than for either humpback whales in the western Antarctic Peninsula or blue whales (Table 2).

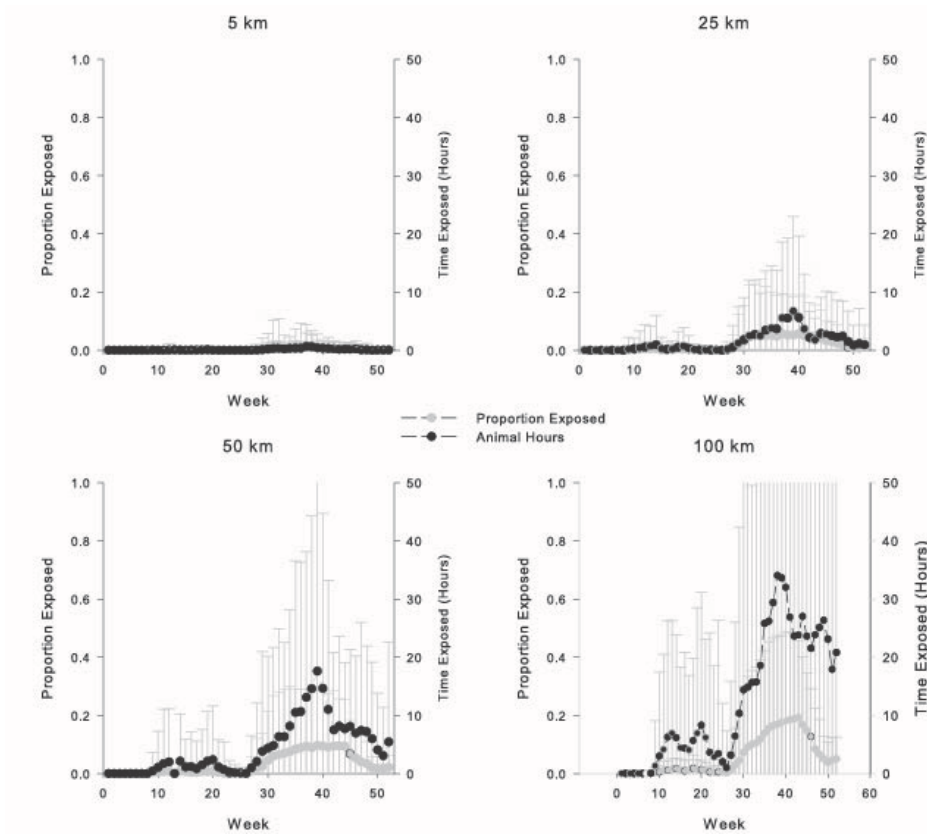


Figure 8. The proportion of blue whales tagged off California that would be exposed to a simulated seismic survey (light grey), and the length of time of their exposure (black) at disturbance radii of 5, 25, 50 and 100 km.

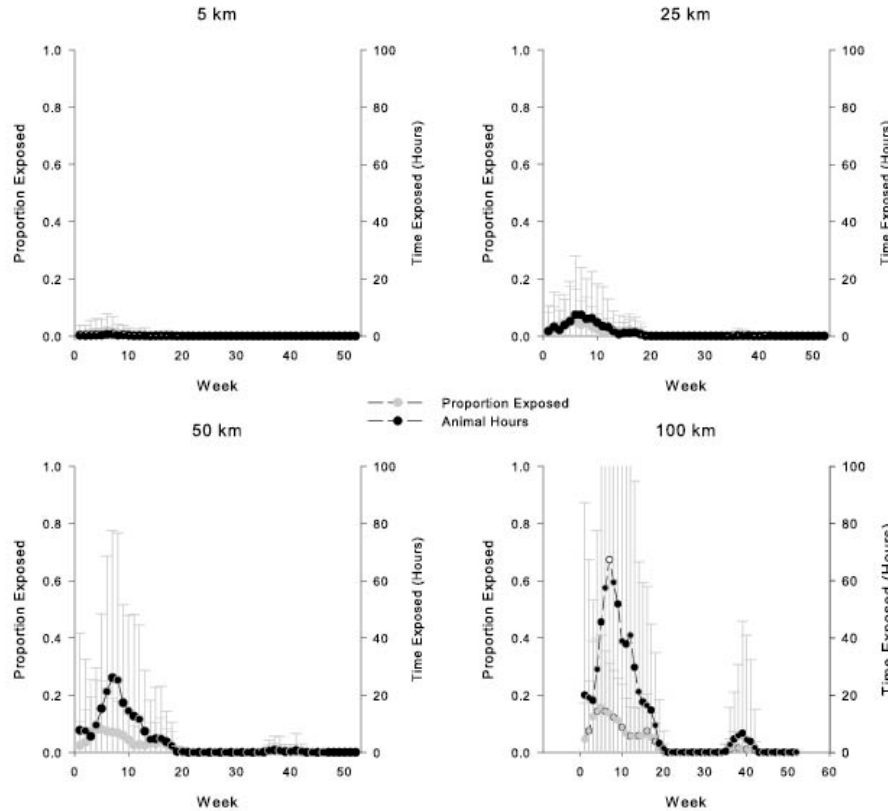


Figure 9. The proportion of humpback whales tagged in the Western Antarctic Peninsula exposed and the time they would be exposed to a simulated seismic survey with a disturbance range of 5, 25, 50 and 100 km.

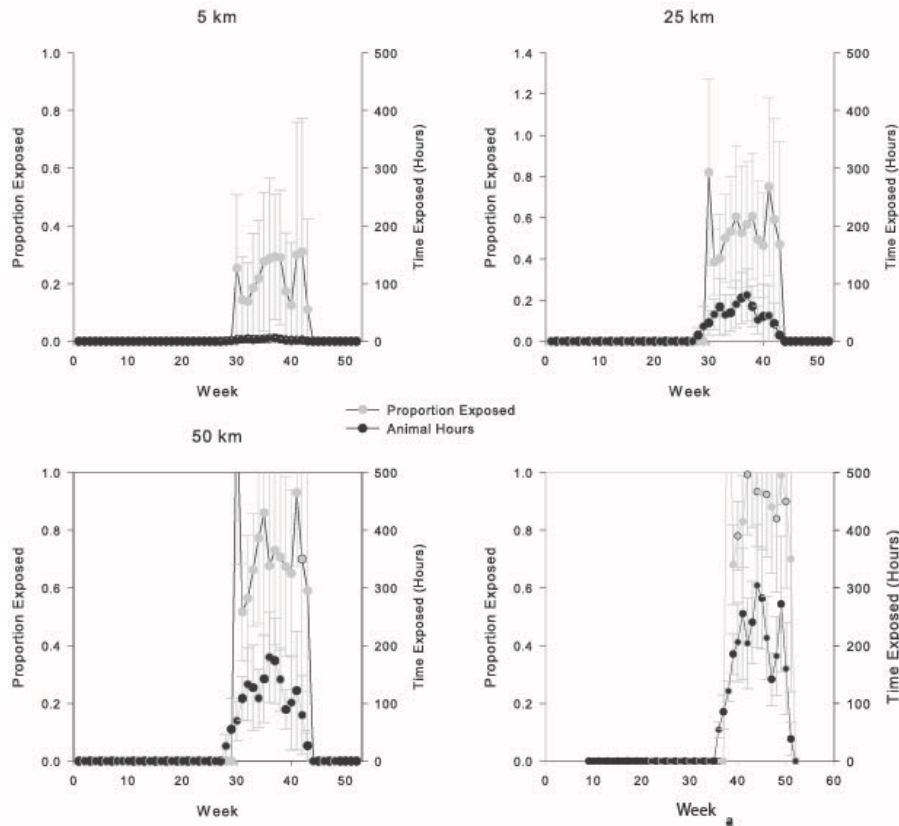


Figure 10. The proportion of humpback whales tagged in the Bering Sea exposed and the time they would be exposed to a simulated seismic survey with a disturbance range of 5, 25, 50 and 100 km.

Table 1. The highest (or peak) mean value for the relative proportion of individuals from each population and the amount of foraging time that they would be exposed to a seismic survey of a given radius of disturbance (From figures 7-9).

	5 km		25 km		50 km		100 km	
	Mean Peak Proportion	Mean Peak Time	Mean Peak Proportion	Mean Peak Time	Mean Peak Proportion	Mean Peak Time	Mean Peak Proportion	Mean Peak Time
Humpback Bering Sea	0.27	5.52	0.71	80.06	0.93	179.9	0.99	304.02
Humpback wAP	0.02	0.59	0.59	7.46	0.08	26.06	0.16	67.30
Blue whale CA Current	0.02	0.62	0.62	6.71	0.10	17.59	0.19	34.05

Table 2. Mean proportion of foraging time that individuals in each region would be exposed to a seismic survey with the different simulated radii of disturbance.

	5 km %	25 km %	50 km %	100 km %
Humpback whale Bering Sea	86	100	100	100
Humpback whale wAP	3	4	4	6
Blue whale California current	7	11	14	19

4. DISCUSSION

Surprisingly, there was a considerable difference in the likelihood of exposure between humpback whales foraging in the Bering Sea versus those foraging in the western Antarctic Peninsula. It took a large disturbance radius (*i.e.* 100 km) before only 6% of the population of humpback whales in Antarctica or 19% of the blue whale population off of California were exposed. In contrast, a high proportion of the Bering Sea humpback whales were exposed with a 5 km disturbance radius. Indeed, the likelihood of exposure for blue whale falls within the range observed for the two populations of humpback whales. It is important to consider whether these tracks are truly representative of the different populations. This would require tracks from more individuals over the entire season as time of year can affect foraging patterns (Curtice *et al.*, 2015). However, the different sensitivities are intriguing and may be associated with differences in foraging patterns observed between the three locations. Humpback whales foraging in the Bering Sea had spatially localized foraging regions compared to humpback whales in the western Antarctic Peninsula that are likely related to differences in their prey. In the Bering Sea, humpback whales feed on fish that form highly aggregated prey patches, while in the Antarctic they feed on krill. While krill patches in Antarctica are also dense, the foraging patch dynamics are likely to be quite different. It is interesting that humpback whales feeding on krill are more similar to blue whales also feeding on krill than they are to humpback whales feeding on fish. While tentative these simple examples show the importance of understanding the ecology of a population or species when considering the likelihood of exposure to disturbance. This can also be thought of as the “context” of the animal’s environment that has been identified as an important confounding variable in behavioral response studies (Friedlaender *et al.*, 2016; Harris *et al.*, 2016; Isojunno *et al.*, 2016).

The above results are consistent with a similar analysis carried out on northern elephant seals and California sea lions (Costa *et al.*, 2016). In this case the widely foraging northern elephant seals had a low probability of exposure, whereas almost all of the California sea lions would be exposed to the 100 km disturbance radius. In this study humpback whales in the Bering Sea are comparable to California sea lions with small overlapping foraging areas, while humpback whales in the western Antarctic Peninsula and blue whales are more like northern elephant seals that feed over a large geographic range. In the case of northern elephant seals, when the spatial

domain of the disturbance was small relative to the entire foraging range the lost foraging opportunities were minor, having an almost undetectable effect on the seals' reproductive output (Costa *et al.*, 2016).

Bioenergetics models need reliable information on how the animals spends their time. In our study we used ARS derived from tracking data as an index of foraging behavior. As mentioned earlier, while this approach has been widely used there are few examples where the relationship between ARS and successful foraging has been validated (Robinson *et al.*, 2012; Whitlock *et al.*, 2015). Furthermore, the quality of the tracking data can be quite variable depending on the design and intent of the original study. For example, satellite tags are often duty cycled (reduced transmission frequency) to increase the life of the tag and the duration of the track (Mate *et al.*, 2007). Duty cycling reduces the frequency of observations effecting the quality of the behavioral data obtained. Further, state space models are particularly sensitive to variations in time series and duty cycling can result in serious biases (Lonergan *et al.*, 2009; Breed *et al.*, 2011). Output of state space models may also overestimate foraging behavior as behavior may vary as animals are known to partition foraging by day/night. Further, due to errors in ARGOS locations the likelihood of 'resting' behavior falling under ARS and not transiting is quite high (Gurarie *et al.*, 2009). For example, blue whales do not feed at night (Friedlaender *et al.*, 2015) in the California current while humpback whales in the western Antarctic Peninsula do (Friedlaender *et al.*, 2013).

While we are not aware of similar analysis having been carried out with other marine species, the approach described here can be applied to any animal whose movements can be tracked. New tag technologies and attachment methods are making it possible to collect high quality tracking data for extended periods of time from a variety of marine vertebrates (Andrews *et al.*, 2008; Block *et al.*, 2011; Hammerschlag *et al.*, 2011; Maxwell *et al.*, 2013; Schorr *et al.*, 2014; Hart *et al.*, 2015; Block *et al.*, 2016). Additionally, the incorporation of dive data along with tracking data provides a more robust data set to develop time-activity budgets that allow considerable insight into the behavior and energetics of these animals in general and their potential response to disturbance in particular (Mate, 2012; Del Raye *et al.*, 2013; Naito *et al.*, 2013; Whitlock *et al.*, 2013; Jorgensen *et al.*, 2015; Whitlock *et al.*, 2015).

5. CONCLUSION

Our goal was to provide an overview of the various life history parameters needed to be considered when assessing the likelihood of marine animals being exposed to acoustic disturbance. All of the life history patterns described here are found among marine vertebrates, from species like the sea otter, *Enhydra lutris*, which has an extensive species range but small individual home ranges, to species like white sharks *Carcharodon carcharias*, Bluefin tuna *Thunnus orientalis*, leatherback sea turtles *Dermochelys coriacea*, elephant seals *Mirounga spp.*, and baleen whales, which have large species ranges, with individuals transiting over much of that range. We provided three case studies of migratory species to show how the approach described here can be used to develop a framework for estimating the likelihood that marine vertebrate populations would be exposed to disturbance. Such information will be enable the development of general guidelines for risk assessment. Case studies could be carried out across representative species to develop a more general risk assessment guideline, along the lines of those developed for the hearing sensitivity of the different types of marine mammals (Southall *et al.*, 2007).

ACKNOWLEDGMENTS

This paper was developed in association with the Office of Naval Research supported PCAD/PCOD and was supported by Office of Naval Research grant N00014-08-1-1195, the E&P Sound and Marine Life Joint Industry Project of the International Association of Oil and Gas Producers.

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