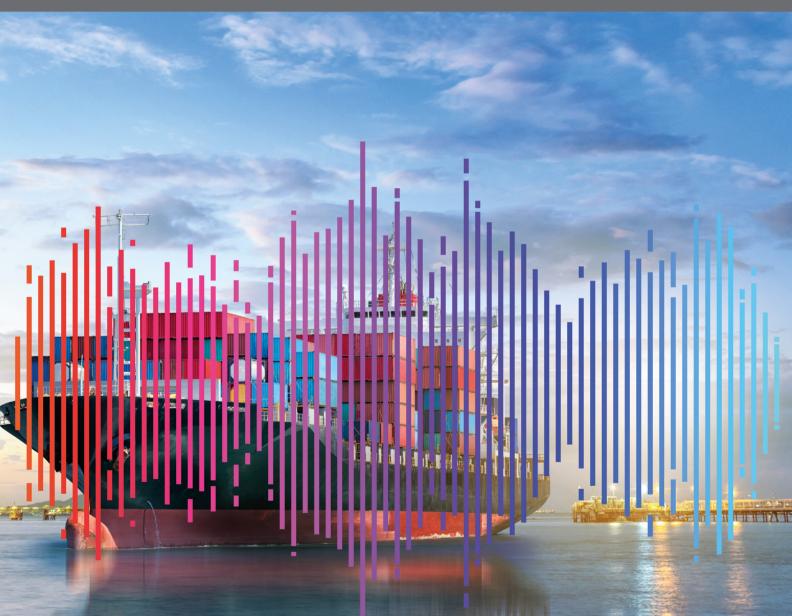
ACOUSTICAL IMPACT OF SHIPS AND HARBOURS: AIRBORNE AND UNDERWATER N&V POLLUTION

EDITED BY: Davide Borelli and Tomaso Gaggero

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ACOUSTICAL IMPACT OF SHIPS AND HARBOURS: AIRBORNE AND UNDERWATER N&V POLLUTION

Topic Editors:

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Noise and vibrations generated by ships affect a wide range of receivers: crew and passengers inside the vessel, inhabitants of the coastal areas and marine fauna outside it. Recent studies suggest that a large percentage of people living in urban areas close to harbors and a number of marine species, at different evolutionary levels (in particular mammals and cephalopods), suffer from ship N&V emissions in air and in water. The present degree of knowledge of the phenomena involved in the noise emissions inside and outside ships is quite different, as a result also of the time elapsed since the negative effects were realized and therefore studied. The development of the normative framework in the various areas reflects these differences, but there are expectations for improvements on all fronts that need to be supported by the scientific community presenting the latest research results in this particular field of acoustics.

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Editorial: Acoustical Impact of Ships and Harbors: Airborne and Underwater N&V Pollution

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Keywords: ship noise, underwater radiated noise, maritime acoustics, animal bioacoustics, sound propagation

Editorial on the Research Topic

Acoustical Impact of Ships and Harbors: Airborne and Underwater N&V Pollution

Aim of this Frontiers research topic is to analyse the different aspects of the impact of noise emitted by human activities and ships in particular. As ships have the peculiarity of operating at the interface between two fluids (air and water), noise generation takes place both in air and underwater, involving two different families of sources, propagation paths, and receivers. As regards airborne noise, sources are represented by the funnels, air intakes, and discharges and in general all the openings that put in communication the inside of the ship with the surrounding environment. The receivers are the inhabitants of port areas or channels with intense ship traffic. For what concerns ships underwater radiated noise, main noise sources are the propellers and the engines. While noise coming from the engines features a series of energy transformations, as vibrations are transmitted to the hull that radiates noise into water, the propeller is a much more efficient noise source which generates noise directly inside the water, especially when cavitation phenomena occurs. The widespread shipping traffic is responsible for a diffused broadband increase in the noise levels, while other noise sources such as air guns and military sonars generate very high level impulsive sounds. Receivers affected by underwater noise are potentially all the species living in the oceans, but attention is mainly focused on the consequences of noise on marine mammals. The effects on those species can range from temporary to permanent hearing losses or even death for high power noise sources to behavioral changes and communication problems for broadband diffused sources like shipping.

The normative framework development and the scientific studies in both fields of ship noise emissions (airborne and waterborne) featured a strong increase in the last decade. For what concerns the airborne noise, albeit ships, as noise sources, present characteristics which are similar to other typical transport systems (such as road vehicles, trains, etc.) when moving, and can be treated as an industrial plant if in a stationary situation, at the moment no instruments nor standards to specifically characterize, assess, and control this kind of noise are available. On the other hand, the human perception of noise and noise exposure consequences have been deeply studied. On the contrary, standards for the measurements of underwater noise from ships are already available, and some voluntary class notations to certify the low noise emission for ships have been issued by most of the classification societies taking advantage of the experience gained in the naval field. A lack of knowledge is in fact present as regards the impact that noise has on marine mammals. To this aim, the focus of the research is the assessment of the noise footprint of human activities both numerically, by means of models, and experimentally, by means of infield measurements. To reach this goal, a deeper knowledge of all the elements of the noise chain (source, transmission path, and receiver) is necessary.

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Borelli D and Gaggero T (2018) Editorial: Acoustical Impact of Ships and Harbors: Airborne and Underwater N&V Pollution. Front. Mar. Sci. 5:83. doi: 10.3389/fmars.2018.00083 Borelli and Gaggero Ship and Harbors Acoustical Impact

As regards the ship characterization as an underwater noise source, Karasalo et al. presented a study to estimate the noise source spectra of ships based on long term measurements in the Baltic sea. Data from over 2000 close-by passages, recorded during 3 months were used. A procedure for ship source spectra estimation was presented based on: sound recordings by a single hydrophone placed close to a shipping line; Automatic Identification System data to localize ships and gain information of their operative conditions and a model to estimate sound propagation. The acquired data were compared with source models available in literature, finding a good agreement between models and measurements for frequencies higher than 200 Hz. Such kind of study is particularly important as very few data regarding commercial vessels underwater noise emissions are available in literature and it is extremely difficult and expensive to carry out ad-hoc measurements.

Concerning sound transmission at sea, van der Schaar et al. presented a study on noise propagation in the Arctic. The study took advantage of seismic surveys carried out by Statoil is summer 2013. Two different recorders were installed in the Greenland Sea, allowing the estimation of propagation losses acting on sound emitted by the air guns. The seismic surveys were carried out at distances ranging from 50 to 300 km, and around 10,000 shots were detected and analyzed. Results showed that it is difficult to find a unique "log(R)" transmission loss law. Studying anthropogenic sound propagation in the Arctic is particularly important because anthropogenic actives are rapidly increasing in an uncontaminated environment which is more vulnerable. Moreover, the presence of ice influences sound transmission allowing the tuning of mathematical models.

As regards the effects on cetacenas, Parsons reviewed the problem of military sonars and their impact on mass strandings. The study underlines that there is a high level of uncertainity in this particular issue of marine science, and that there is a need for precaution due to several factors, e.g., the difficulty of finding and seeing strandings eve if they occur or the fact that most cetaceans sink upon death, making injury, or mortality at sea

caused by noise unlikely to be observed. The suggestion is that all navies should implement best practices, effective monitoring, and mitigation measures, as well as the governments need to develop criteria for assessing and investigate atypical mass strandings.

Again, concerning cetaceans, Marley et al. analyzed the underwater soundscape of bottlenose dolphins habitat within the Swan-Canning River system in Western Australia. In this highly urbanized estuary in Perth, acoustical data were recorded and analyzed across 8 years. Among the multiple sound sources, the two most prevalent ones were vessels traffic and snapping shrimps. The analysis was carried out taking into account both spatial and temporal variations, and showed that vessels noise was the most disruptive sound, since its peculiar spectral and temporal characteristics tend to overlap and likely mask dolphin whistles, thus influencing their behavior.

This Frontier research topic represented an excellent opportunity for researchers to publish original works dealing with the impact of anthropogenic noise on the marine fauna. The published papers covered all the main aspects of the problem presenting studies regarding the assessment of noise effects by air guns and sonars on cetaceans, ship characterization as a source of underwater noise and noise propagation in an extreme environment such as the Arctic region.

AUTHOR CONTRIBUTIONS

All authors listed have made a substantial, direct and intellectual contribution to the work, and approved it for publication.

Conflict of Interest Statement: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Estimates of Source Spectra of Ships from Long Term Recordings in the Baltic Sea

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Estimates of the noise source spectra of ships based on long term measurements in the Baltic sea are presented. The measurement data were obtained by a hydrophone deployed near a major shipping lane south of the island Öland. Data from over 2,000 close-by passages were recorded during a 3 month period from October to December 2014. For each passage, ship-to-hydrophone transmission loss (TL) spectra were computed by sound propagation modeling using

- 1. bathymetry data from the Baltic Sea Bathymetry Database (BSBD),
 - 2. sound speed profiles from the HIROMB oceanographic model,
 - 3. seabed parameters obtained by acoustic inversion of data from a calibrated source,
 - 4. AIS data providing information on each ship's position.

These TL spectra were then subtracted from the received noise spectra to estimate the free field source level (SL) spectra for each passage. The SL were compared to predictions by some existing models of noise emission from ships. Input parameters to the models, including e.g., ship length, width, speed, displacement, and engine mass, were obtained from AlS (Automatic Identification System) data and the STEAM database of the Finnish Metereological Institute (FMI).

Keywords: ship noise, underwater radiated noise, URN, Automatic Identification System, AIS, propagation modeling, Baltic sea

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1. INTRODUCTION

As ship traffic is increasing in the Baltic Sea, noise pollution and its impact on underwater fauna is becoming a concern. For example, the behavior and breeding patterns of fish and sea mammals have been found to be negatively affected by anthropogenic underwater radiated noise (URN) (Rolland et al., 2012). This has raised interest in gaining improved quantitative insight into underwater noise caused by ship traffic.

As a basis for gathering information on URN, measurements on ships accompanied by models describing the URN as a function of ship parameters are frequently used. Examples of this are Hatch et al. (2008), where measurements on ships off the coast of Massachusetts were combined with crude transmission loss (TL) estimates to establish the relative contribution of the URN

Estimates of Source Spectra of Ships

from large vessels to the total ocean noise, and Wales and Heitmayer (2002) establishing an ensemble average spectrum based on recordings of 272 ships between 1986 and 1992 in the Mediterranean Sea and the Eastern Atlantic Ocean. Recently, Simard et al. (2016) estimated source levels of 191 cargo ships and tankers passing the St. Lawrence Seaway during a 16 week period in 2012. More elaborate procedures for estimating URN as prescribed by the ANSI S12.64 (ANSI, 2009) standard have also been used. The standard requires cooperation by the measured ship and puts restrictions on the measurement range in terms of water depth. This effectively prohibits the procedure from being used for gathering statistics on large numbers of ships, in contrast to the above mentioned references. Application of the standard thus usually concerns single ship measurements (Arveson and Vendittis, 2000; De Robertis et al., 2013). Several models describing URN have been proposed (Breeding et al., 1996; Wittekind, 2014; Audoly and Rizzuto, 2015; Brooker and Humphrey, 2015), using combinations of ship parameters to derive frequency dependent equivalent (omni-directional) point source representations of the noise radiating ship.

In this paper, a procedure of gathering data of noise emissions from ships trafficking the Baltic sea is outlined. The data are extracted by a single hydrophone recording continuously from October to December 2014, capturing the noise from more than 2,000 ship passages. In order to estimate equivalent point sources representing the noise emitted at CPA (Closest Point of Approach) of each individual ship passage, the environmental influence on the recorded signal was eliminated by modeling the transmission loss from the ship to the hydrophone by a wave number integration code (Karasalo, 1994), taking into account influences of the layered seabed and the temporally and spatially varying sound speed in the water volume. Reliable sound speed profiles were obtained from the High Resolution Operational Model for the Baltic Sea (HIROMB) (SMHI, 2016). For the seabed, less high resolution data are available and estimates of the seabed structure and parameters were determined by a dedicated transmission loss measurement and geo-acoustic inversion. The approach is similar to that by Simard et al. (2016), but employs a more elaborate procedure for estimating the seabed parameters motivated by the relative shallowness of the observation site (\sim 40 m). Furthermore, a wider range of ship types are included in the analysis, covering both passenger ferries and tugboats.

The resulting noise source library produced contains 1/3-octave source levels for each ship passage, along with ship identifiers in terms of IMO (International Maritime Organization) and MMSI (Maritime Mobile Service Identity) numbers. These ship identifiers were subsequently used to extract ship parameters (displacement, engine mass, number of operating engines, cavitation inception speed etc.) from the STEAM (Jalkanen et al., 2012) database of the Finnish Meteorological Institute (FMI), then used as input to available noise source models. Comparisons of these model predictions with the experimentally observed noise source spectra are presented and discussed.

The purpose of this study is to investigate a cost-effective procedure for assessing the single monopole source model of

ship noise, by using a single hydrophone deployed near a shipping lane and the passing ships as sources of opportunity. The procedure enables recording URN data from large numbers of ships of different types using simple instrumentation only, and thus provides a useful complement to more advanced measurement procedures at dedicated measurement ranges.

2. EXPERIMENTAL SITE

The experimental site is located south of the island Öland, where a hydrophone was deployed at N 56° 0.212′, E 16° 17.413′, continuously recording acoustic data during the period October-December 2014. The location was chosen in the vicinity of a major shipping lane, having a few thousand ship passages within a kilometer during the trial period. The hydrophone was attached to an anchor via a line, hovering \sim 3 m above the seabed. Bathymetry data for the area were retrieved from the Baltic Sea Bathymetry Database (Baltic Sea Hydrographic Commission, 2016) (Figure 1) while sound speed profiles, updated every 6 h throughout the period were obtained from the High Resolution Operational Model for the Baltic Sea (HIROMB) (SMHI, 2016). Furthermore, some data regarding the bottom sediment types were obtained from The Geological Survey of Sweden (SGU) (2016). These data indicate that the seabed at the experimental site consists mainly of silt and/or clay (Figure 2). However, such data are not unambiguously translated into acoustic parameters needed for sound propagation modeling. Further, these data only give information on the top sediment layer, thus neglecting the often important effects of underlying sediment layers or bedrock. A more detailed survey of the acoustic bottom parameters was therefore performed as described in the following section.

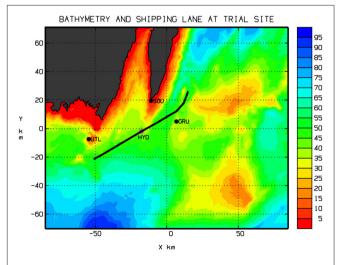


FIGURE 1 | Bathymetry (meters) at the experimental site, with the hydrophone position marked as HYD. The shipping lane is indicated by the trajectory of the closest passage on October 2, 2014. The black dots marked UTL, SOU, GRU show the positions of the lighthouses Utlängan, Ölands Södra Udde, and Ölands Södra Grund, respectively.

3. TRANSMISSION LOSS TRIAL

In order to determine geoacoustical parameters capturing the sound propagation effects at the experimental site, a transmission loss measurement was performed. A loudspeaker emitting 30 s continuous wave pulses at 100, 150, 250, 350, 450, and 550 Hz was towed at distances 90–2,215 m from the bottommounted hydrophone (**Figure 3**). The signal from a hydrophone hanging from the towing boat together with data from the bottom-mounted hydrophone were then used to determine the transmission loss between the two hydrophones as

$$TL = 10 \log_{10} \left(\frac{p_1^2}{p_2^2} \right) \tag{1}$$

where p_1 and p_2 is the pressure at the towed and the bottom mounted hydrophone, respectively. The bottom mounted Wildlife SM2M measurement system was calibrated in a standing wave tube resulting in sensitivity curves shown in **Figure 4**.

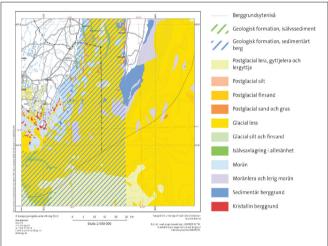


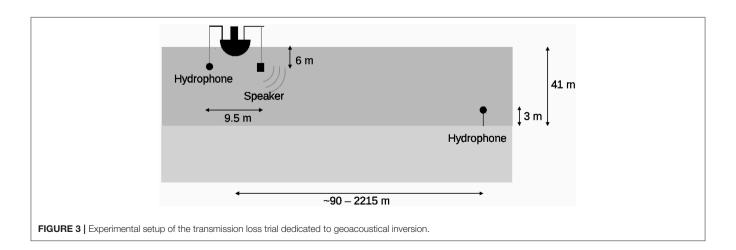
FIGURE 2 | Sediment types at the experimental site [(The Geological Survey of Sweden (SGU), 2016)].

For frequencies below 100 Hz and above 800 Hz, a constant extrapolation of the sensitivity is assumed. It should be noted that previous measurements using this equipment have indicated that below 100 Hz, the sensitivity may in fact be much lower, and hence the low frequency results should be taken with some caution, as discussed in Section 5.

The parameters of a range-independent seabed composed of a sediment layer above a bedrock halfspace were estimated from the observed TL data by geo-acoustic inversion using the differential evolution method (Snellen and Simmons, 2008), with the XFEM code (Karasalo, 1994) for range-independent layered media as forward model. The assumption of range-independence is motivated by the weak bathymetry variations observed in **Figure 6**, where the depth ranges from 41.6 to 43.9 m in a 2.5 × 2.5 km square centered at the hydrophone. The bounds of the parameter search regions and the obtained estimates are listed in columns 2–4 of **Table 1**. The choice of the search regions was guided by the map of sediment types shown in **Figure 2** combined with data on typical acoustic parameters for sediment and rock materials (Ainslie, 2010, Table 4.18), (Bourbié et al., 1987, Table 5.2)

It should be noted that the purpose of the inversion is to find a simplified seabed model for which the predicted transmission losses are good approximations to the experimentally observed. The seabed model is then useful for reliable modeling of the bottom interactions at transmission loss prediction, however its parameters and structure do not necessarily correspond to those of the actual physical seabed. This argument is illustrated by **Table 2** and **Figure 5** below. Column 4 of **Table 2** shows the seabed parameters obtained by acoustic inversion but with a different initialization of the random number generator used by the differential evolution algorithm. Both the sediment thickness and the material parameters of the individual layers are seen to be significantly different from those in column 4 of **Table 1**.

Figure 5 compares the transmission losses $TL_1(r)$ and $TL_2(r)$ as function of source range r in the 63, 127, 254, and 640 Hz 1/3 octave bands, using soundspeed data for 2014-10-18 combined with the seabed parameters in, respectively, **Table 1** $[TL_1(r)$, black] and **Table 2** $[TL_2(r)$, red]. The source depth is 5 m.



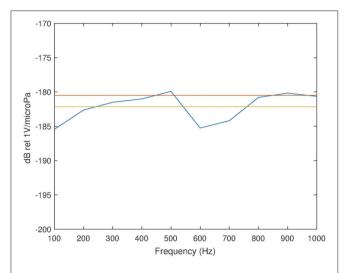


FIGURE 4 | Sensitivity of the receiver chain as function of frequency: Calibrated in standing wave tube (blue), frequency average (yellow), and factory value (red).

The differences $|TL_1(r) - TL_2(r)|$, averaged over range r, are shown in the upper right hand corners of the four frames. The average differences of ≈ 1 dB or less indicate the uncertainty induced by unknown seabed parameters on the transmission losses used for source level estimation in Section 4 below. Similar results, not shown here, were obtained for a selection of dates in October–December 2014.

4. ESTIMATION OF THE SOURCE LEVELS

Estimates of the noise source spectra were computed for all ship passages of the hydrophone at range 1,000 m or less in the trial period October 2–December 29. The numbers of such passages and individual ships were 2,088 and 943, respectively. The noise source level was estimated in 21 1/3-octave bands with center frequencies $f_k = 10 \times 2^{(k-1)/3}$ Hz, ranging from $f_1 = 10$ Hz to $f_{21} = 1016$ Hz.

The estimate SL_k of the noise source level (dB) in frequency band k was obtained as

$$SL_k = RL_k + TL_k. (2)$$

 RL_k and TL_k are, respectively, estimates (dB) of the noise level at the hydrophone and the transmission loss from the source to the hydrophone when the ship is at its closest point of approach (CPA), i.e., when the range from the ship to the hydrophone is minimal. The computation of these estimates is described in Sections 4.1 and 4.2 below.

4.1. Estimation of the Noise Level at the Hydrophone

Let r_{hyd} denote the position of the hydrophone, r(t) the position of the ship as function of time t, and R(t) the range from the ship

TABLE 1 | Parameters of two-layer seabed model.

Parameter	Lower bound	Upper bound	Estimate
Sediment thickness, m	3	7	3.6
Sediment density, kg/m ³	1,000	2,500	1,216
Sediment soundspeed, m/s	1,450	2,000	1,936
Sediment absorption, dB/ λ	0.01	2.5	0.03
Bedrock density, kg/m ³	2,500	3,100	2,550
Bedrock soundspeed, m/s	3,000	6,000	4,307
Bedrock absorption, dB/ λ	0.0	2.0	0.1

TABLE 2 | Parameters of alternative two-layer seabed model.

Parameter	Lower bound	Upper bound	Estimate
Sediment thickness, m	3	7	4.6
Sediment density, kg/m ³	1,000	2,500	2,317
Sediment soundspeed, m/s	1,450	2,000	1,476
Sediment absorption, dB/λ	0.01	2.5	0.04
Bedrock density, kg/m ³	2,500	3,100	2,926
Bedrock soundspeed, m/s	3,000	6,000	5,584
Bedrock absorption, dB/λ	0.0	2.0	0.07

to the hydrophone

$$R(t) = |r(t) - r_{\text{hvd}}|. \tag{3}$$

The function r(t) was defined as the piece-wise linear interpolant to AIS position data. Denote the minimum of R(t) by $R_{\text{cpa}} = R(t_{\text{cpa}})$.

Then the estimates of the noise levels N_k , (k = 1, ..., 21) at the hydrophone excited by the ship from its CPA were computed as follows:

- 1. A time-interval with length $T_{\rm tot} = 240$ s was selected, centered at $t_{\rm cpa}$, and subdivided into M = 60 consequtive subintervals T_j , (j = 1, ..., M) with equal lengths $T_{\rm tot}/M = 4$ s.
- 2. Denote by $s_j(t)$ the signal received by the hydrophone in subinterval T_i , j = 1, ..., M and by

$$W_j = \int_{T_i} s_j(t)^2 dt \tag{4}$$

the energy of $s_i(t)$.

3. The subinterval j_{cpa} for which W_j is maximal was found and the short-time Fourier spectra $\hat{s}_{j_{\text{cpa}}}(f)$ of $s_{j_{\text{cpa}}}(t)$ were computed by FFT. Then the noise levels RL_k at the hydrophone excited by the ship from its CPA were estimated by

$$RL_k = 10 \log_{10} \left\{ \int_{f_k^-}^{f_k^+} |\hat{s}_{j_{\text{cpa}}}(f)|^2 df \right\} \qquad k = 1, ..., 21$$
 (5)

where $f_k^- = 2^{-1/6} f_k$ and $f_k^+ = 2^{1/6} f_k$ are the bounds of the 1/3-octave band with center frequency $f_k = 10 \times 2^{(k-1)/3}$ Hz.

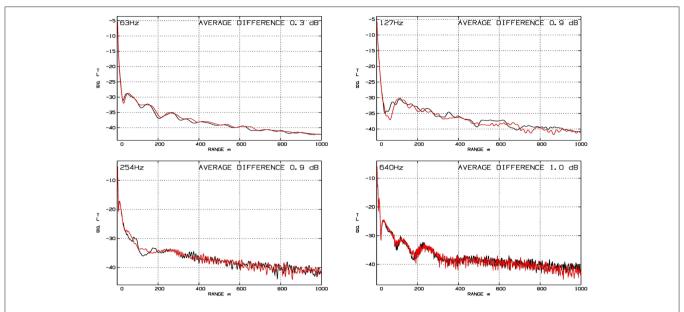


FIGURE 5 | TL to the hydrophone as function of source range using sound speed data of 2014-10-18 combined with seabed parameters in, respectively, **Table 1** (black) and **Table 2** (red). Source depth 5 m.

To summarize, the received signal segment corresponding to sound emitted from the CPA of the ship was identified as the 4 s time-window in which the received sound energy (4) is maximal. The simpler alternative of using the timepoint $t_{\rm cpa}$ explicitly proved to be unreliable due to inaccurate time-synchronization between the AIS and the hydrophone data. Note that Equation (2) with RL_k defined by Equation (5) holds only when the received noise is dominated by that from the ship, a condition which was reasonably well-satisfied for ship passages within the selected maximal range of 1 km.

4.2. Estimation of Transmission Loss

The transmission losses TL_k , (k=1,...,21) from the CPA to the receiver hydrophone were estimated by sound propagation modeling. The following simplifying assumptions on the underwater medium were used:

- 1. The geometry and the medium parameters are rangeindependent, with water-depth equal to that at the hydrophone.
- 2. Variations of the sound speed profile within 1 day are negligible.

Assumption 1 was considered reasonable since (i) The variations of the water depth are only ca 2 m within the maximal range (1 km) to the CPAs used for the estimates as shown in **Figure 6** and (ii) Data on the sound speed profile were available at a single spatial location only. Similarly, assumption 2 was found reasonable by inspection of the sound speed profile data. **Figure 7** shows the sound speed profile at the measurement site every 6 h throughout the measurement period.

Under these assumptions the soundfield was computed with a full-field method for range-independent layered media (Ivansson and Karasalo, 1992; Karasalo, 1994; Karasalo and deWinter,

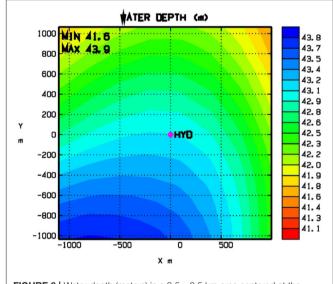


FIGURE 6 Water depth (meters) in a 2.5×2.5 km area centered at the hydrophone marked as HYD.

2006), based on adaptive high-order wavenumber integration and solution of the depth-separated wave equation by exact finite elements. The method is accurate at all ranges to the CPA, including in particular CPAs in the immediate nearfield of the hydrophone. Further, the modeled transmission loss is independent of the direction to the CPA, so that the estimates of the TLs from all CPAs on a given day and a given frequency were obtained by a single run of the propagation model to obtain the TL on a dense range grid followed by computation of the TLs from the individual CPAs by interpolation in range.

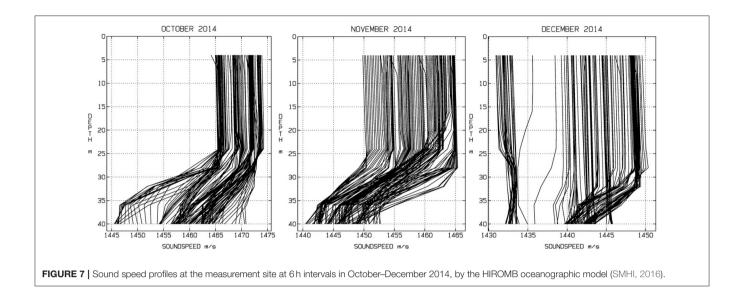


TABLE 3 Number of passages and average speed, length, and displacement per ship category.

Ship type	No. of passages	Avg. speed (kn)	Avg. length (m)	Avg. displ. (tons)
Tanker	244	12	118	15,769
Dredging	1	5	48	1,933
Fishing	7	9	39	1,411
Law	2	7	70	4,112
Other	4	11	104	9,141
Passenger	61	21	191	23,118
Pleasure	1	0	33	317
Towing	3	6	27	481
Tug	30	7	28	503
Military	3	-	-	-
Total	2,088	10	110	1,187

The transmission loss TL_k in 1/3 octave band nr k was estimated by

$$TL_k = 10 \log_{10} \left\{ \frac{1}{N} \sum_{j=1}^{N} 10^{TL(f_j)/10} \right\} \qquad k = 1, ..., 21$$
 (6)

where N=7, TL(f) is the TL (dB) at frequency f and f_j are frequencies covering 1/3 octave band nr k with $\log f_j$, (j=1,...,N) equidistant.

5. RESULTS

In **Table 3**, the number passages based on ship category is shown, together with statistics on speed, length, and displacement. Estimated median source spectra for the four categories with the most ship passages are given in **Figure 9**. One cargo ship

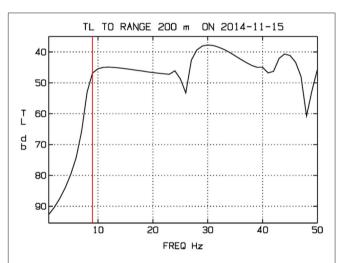
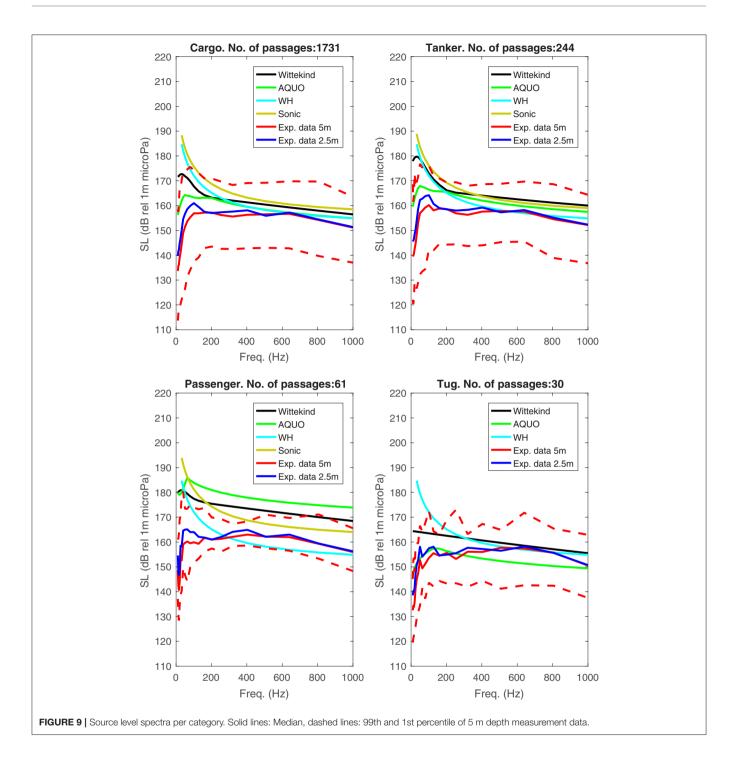


FIGURE 8 | Model-predicted TL to the hydrophone from a ship at range 200 m, using sound speed data of 2014-11-15. The red line at indicates the cutoff frequency of the shallow-water medium.

was excluded because of incomplete data. For each ship passage, predictions of the source levels using four models are given:

- 1. The Wittekind model (Wittekind, 2014), requiring seven input parameters: cruise speed, displacement, cavitation inception speed, block coefficient, engine mass, number of engines in use, and an engine mount parameter.
- 2. The AQUO model (Audoly and Rizzuto, 2015), estimating the source levels based on ship category, cruise speed, and ship length.
- 3. The Wales-Heitmayer (WH) model (Wales and Heitmayer, 2002), providing an estimate based on statistics obtained from measurements on 272 ships. The WH model is independent of ship parameters, and hence gives a baseline spectrum which is identical for all ships.



4. The SONIC model (Brooker and Humphrey, 2015), giving a correction to the WH model based on cruise speed and a ship category specific reference speed. This model is not applicable to Tug.

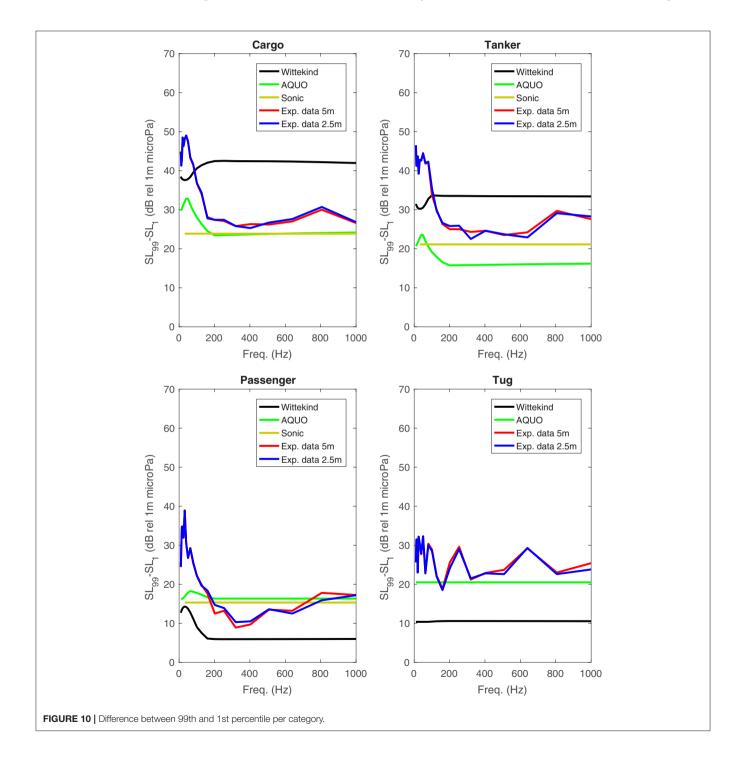
Input parameters for the AQUO and the Wittekind models were obtained from AIS and the STEAM database. For more details, the reader is referred to the Appendix (Supplementary Material).

The accuracy of the source level predictions varies; for three of the ship categories, Cargo, Tanker, and Tug, the medians agree with the experimental data within 10 dB for frequencies above 200 Hz. For Cargo and Tanker, the AQUO model gives slightly better agreement with measurement data than Wittekind for most frequency bands. The Wittekind model clearly overestimates the source level for Passenger ferries. Meanwhile, the WH baseline spectrum agrees fairly well for this category. For frequencies below 200 Hz, the model-measurement

agreements are generally poorer. Two possible causes for this are (i) deterioration of the calibration of the hydrophone at low frequencies and (ii) degradation of the signal to noise ratio caused by decrease of the transfer function amplitude with decreasing frequency. The second of these effects is investigated in **Figure 8** showing the transmission loss on 2014-11-15 as function of frequency from a ship at range 200 m to the hydrophone. By the figure the cutoff frequency of the shallow-water medium is $\sim\!\!9$ Hz. Thus the center frequencies of all the considered

1/3-octave bands are above cutoff, hence the model-measurement discrepancies at low frequencies are more likely caused by poor hydrophone calibration than by low S/N.

Furthermore, Scrimger and Heitmeyer (1991) noted when comparing the source spectra from passenger ships, cargo ships, and tankers, that the differences between these were not significant. Similar observations can be made here by noting that the median levels of the experimental data for the these categories differ by no more than around 7 dB. This is to be compared to



Estimates of Source Spectra of Ships

the \sim 20 dB predicted by the AQUO model and the Wittekind models' \sim 13 dB.

The source strengths are estimated from the experimental data assuming two different source depths, 2.5 and 5 m, to investigate the effect of this model parameter on the transmission loss. While the assumed source depth is not included explicitly in the source strength models, it is seen in **Figure 9** to influence the predicted source strength by up to \sim 5 dB.

Statistical variability measures in terms of the difference between the 99th and 1st percentile are shown in **Figure 10**. For Cargo and Tanker the variability of the measured source strengths is in the range of 20–30 dB above \sim 200 Hz while increasing to >40 dB at around 100 Hz, agreeing with what is observed by Simard et al. (2016). The observed Passenger ships notably show a smaller variability, not exceeding 20 dB for frequencies >200 Hz.

In Figures 11, 12 the source level estimates from two individual ships (the Ro-Ro Passenger ship Finnlady and the Ro-Ro Cargo ship Finnsky) passing near the hydrophone

on 21 and 16 separate occasions, respectively, are depicted. For Finnlady, the median values of the Wittekind model matches the experimental data poorly, as previously observed for passenger ships. Moreover, the predicted variations are rather large, showing that the noise radiated from a single ship can vary as much as 20 dB. For the Finnsky ship, the variations are substantially smaller (~10 dB) and of the same magnitude as predicted by the AQUO and Sonic models. In **Figures 13**, **14** frequency averaged source levels, $\sum_{k=1}^{21} SL_k/21$, are plotted for each ship passage along with data on ship speed, hydrophone-CPA distance and wind speed. For neither of the ships, a clear correlation between these parameters and source level is discernible. Hence, the cause of the large source level variations for the Finnlady ship is as of now unclear. For a more rigorous investigation of the observed variations, more detailed data on the actual operating conditions for each ship passage would be needed, including e.g., data on engine power input, propeller cavitation and load carried by the ship.

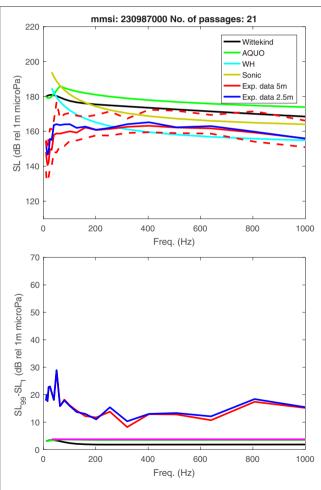


FIGURE 11 | Top: Source level spectra of the Passenger/Ro-Ro Cargo ship Finnlady (IMO 9336268, MMSI 230987000) estimated from 21 close passages. Solid lines: Median, dashed lines: 99th and 1st percentile. Bottom: Difference between 99th and 1st percentile.

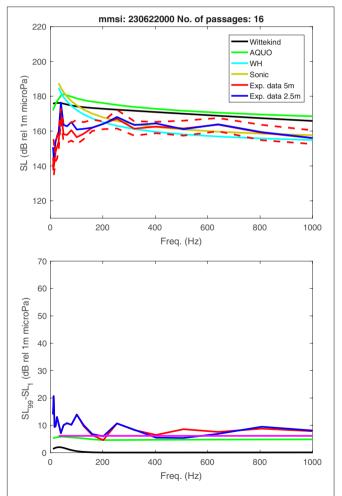


FIGURE 12 | Top: Source level spectra of the Passenger/Ro-Ro Cargo ship Finnsky (IMO 9468906, MMSI 230622000) estimated from 21 close passages. Solid lines: Median, dashed lines: 99th and 1st percentile. **Bottom**: Difference between 99th and 1st percentile.

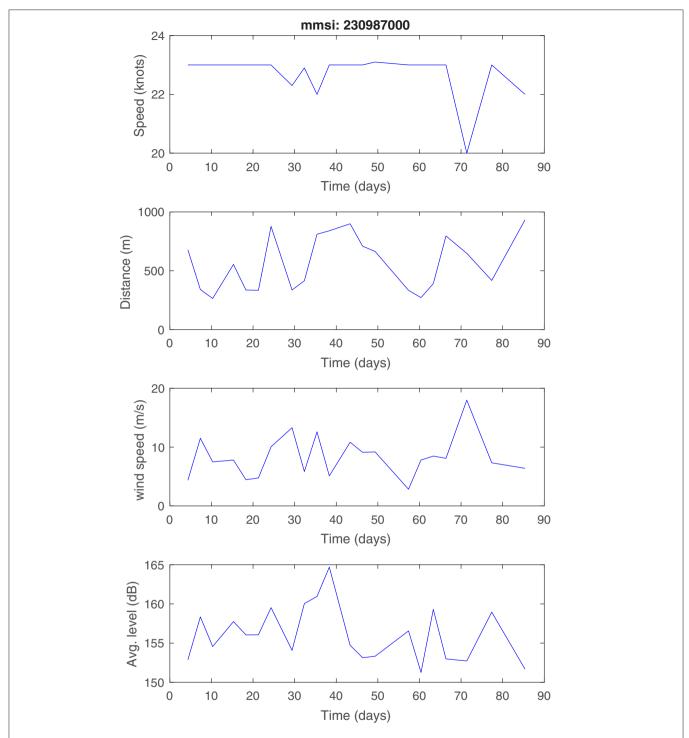


FIGURE 13 | Source level averaged over frequency for each passage of the Passenger/Ro-Ro Cargo ship Finnlady (IMO 9336268, MMSI 230987000) along with data on ship speed, hydrophone-CPA distance and wind speed. Day zero is taken as October 1, 2014.

6. SUMMARY AND CONCLUSIONS

A procedure was presented for estimating the underwater radiated noise (URN) source spectra of individual ships using (i) sound recordings by a single hydrophone positioned near a major shipping lane in the Baltic Sea, (ii) AIS data on ship traffic in the area and (iii) sound propagation modeling to estimate the transmission loss from the closest

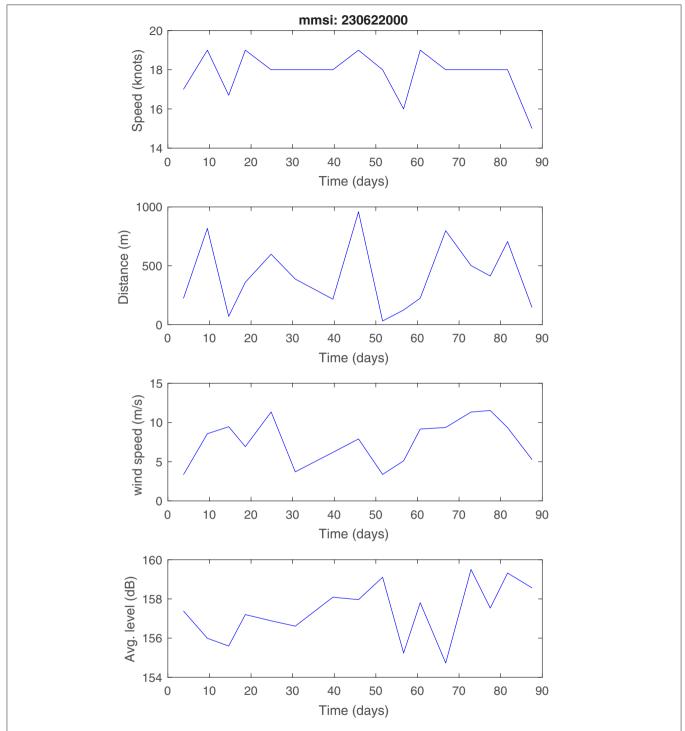


FIGURE 14 | Source level averaged over frequency for each passage of the Passenger/Ro-Ro Cargo ship Finnsky (IMO 9468906, MMSI 230622000) along with data on ship speed, hydrophone-CPA distance and wind speed. Day zero is taken as October 1, 2014.

point of approach of the ships to the hydrophone. Acoustic seabed parameters were estimated by geo-acoustic inversion of data from a transmission loss trial and sound speed profiles were obtained from the HIROMB oceanographic model.

The procedure was applied to estimate the source strength of over 900 individual ships from more than 2,000 close passages of the hydrophone during a 3 month period in 2014. Comparisons to source strength models found in the literature show, that for Tankers and Cargo ships, which are the most common ships

in the Baltic Sea, these can provide reasonably good estimates of the median source levels above $\sim\!\!200$ Hz. For the two other categories and for multiple passages by the same ship, larger discrepancies are present. The poor model-prediction agreement observed for low frequencies is likely due to the lack of reliable hydrophone calibration data. In future studies, it is desirable to extend the lower frequency limit of the methodology in order to cover the lowest indicator frequency (63 Hz) of the Marine Strategy Framework Directive adopted by the European Union.

The estimation procedure and the collected data can, in light of the discrepancies observed, be used to enhance the reliability of existing source strength models. Considering that the awareness of the adverse effects of URN on underwater fauna is increasing, such models along with statistics as presented here can be used to more accurately direct URN mitigation efforts.

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AUTHOR CONTRIBUTIONS

IK and MÖ provided estimates of the seabed parameters by acoustic inversion of the transmission loss trial data and of the source spectra from the long-term recordings. PS carried out the transmission loss trial. JJ and LJ provided the Wittekind noise source results based on FMI STEAM model, and participated together with ML and RB in manuscript writing and result analysis.

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Conflict of Interest Statement: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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APPENDIX

Acquisition of Ship Parameters

For the Wittekind noise source model, several parameters are required which are not part of commercially available databases of ship specifications. The engine mass data were obtained from the Marine Engines catalog (NEXUS MEDIA, 2005) and augmented with data from engine manufacturers. Based on these data, linear relationships between engine power and engine mass was determined and used for cases where no data could be obtained from other sources. How to determine the number of operating engines and Block coefficient ($C_{\rm b}$) is described by Jalkanen et al. (2012), where power prediction formulas are employed to determine the number of operating engines. Hull form parameters were estimated according to Watson (1998).

Engine mounting parameter for resiliently and rigidly mounted engines were assigned according to Rowen (2003). Cavitation inception speed was estimated in terms of block coefficient and vessel design speed, $V_{\rm d}$, as

$$V_{\text{CIS}} = \min\{\max[(1.42 - 1.2C_b)V_d, 9], 14\}.$$
 (A1)

Cavitation inception speed can significantly affect the predicted noise source levels, but this parameter is very difficult to predict based on available information of the propulsors of the world fleet. Undoubtedly, prediction of $V_{\rm CIS}$ using expression (A1) may introduce a significant source of uncertainty to noise source modeling and further work is needed to describe cavitation inception speed as accurately as possible.





Arctic Anthropogenic Sound Contributions from Seismic Surveys during Summer 2013

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Statoil deployed three acoustic recorders from fall 2013 to 2014 in the Arctic region as part of a broad scientific campaign. One recorder was installed in the Barentsz Sea southeast of Spitsbergen. Two other recorders were installed in the Greenland Sea north-east of Greenland. All recorders were operating at a duty cycle of 2 min on and 30 min off, sampling at 39,062 Hz and recording in 24 bits. The Greenland recorders both captured air gun surveys performed during the summer months of 2013, allowing to estimate the transmission loss in the Arctic over long ranges. This paper presents "log(R)" transmission loss curves for these scenarios that can help assessing the acoustic shipping impact for future expeditions.

Keywords: acoustics, arctic, airgun, noise measurement, propagation loss

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1. INTRODUCTION

With the opening of shipping routes and improved economic availability of the arctic the anthropogenic activities have been increasing over the last few years in that area (Stephenson et al., 2011). At the same time, sound pollution has become an important issue where there is concern not only about how sound may affect marine mammals (Southall et al., 2007) but also concerning its affect on fish (Casper et al., 2013), cephalopods (André et al., 2011), and other organisms (Solé et al., 2016). These affects can lead in extreme cases to direct harm of an animal or more often to masking of acoustic signals reducing communication of forraging ranges (Jensen et al., 2009). In the European Union, this concern about sound pollution has resulted in a special section in the Marine Strategy Framework Directive where sound levels have to be monitored and high intensity sounds have to be cataloged (European Parliament and the Council of the European Union, 2008). It is likely that similar requirements will become the norm for operations outside of EU coastal waters such as the arctic zone which has a rich marine mammal diversity. Actual measurement of sound contributions from activities is not always possible or practical due to costs or the difficult artic environment. In many cases source sound levels related to these activities may be available, but the sound propagation ranges, and the sphere of influence, is often decided using modeling techniques (Sigray et al., 2016).

This report presents opportunistic data of sound measurements made by two recordings deployed by Statoil during the 2013–2014 season that recorded seismic surveys using air guns performed in the area at distances up to 300 km away. The surveys were performed during the months August, September, and October of 2013 (a duration of about 50 days). Availability of the survey ship position then permitted to compare received sound levels with source distance and to estimate the parameters of the most basic sound propagation loss model: $C \log_{10}(R)$. Where C is

often set to 20 for spherical loss, to 10 for cylindrical loss, or somewhere in between to account for sound channels with partically reflecting surfaces. In this report, *C* is estimated for long range propagation loss using the received levels from the air gun used in the seismic survey. These results may be compared with for example the shallow water loss curves for the Barentsz Sea provided in Jensen et al. (2000) that show very high low frequency losses (over 100 dB at 50 Hz at 10 km range) or empirical data in deep water in DiNapoli and Mellen (1986) (around 82 dB at 50 Hz at 100 km range). Additionally this may serve as input or control for low frequency arctic modeling (Gavrilov and Mikhalevsky, 2006; Alexander et al., 2013).

2. MATERIALS AND METHODS

2.1. Recording Equipment

The Greenland I recorder was deployed during the Oden Arctic Technology Research Cruise 2013 on August 23 at 78° 30′N and 10° 0′E (**Figure 1A**) and recovered the next year on September 17. The location was about a 120 km away from the continental slope (a zone where sperm whale activity could be expected). The Greenland II recorder was deployed during the Oden Arctic Technology Research Cruise 2013 on August 22 at 76′ 30′N and 14° 20′E and recovered the next year on September 17. The depth at both deployment locations was around 200 m. The recorders were attached to a line suspended a few tens of meters above the sea floor using a subsurface float. The lines were recovered with the use of an acoustic release.

In both recorders, the data was recorded with a duty cycle of 2 min on and 30 min off sampling at 39,062 Hz in 24 bits. The sampling frequency was chosen to allow a longer deployment time than 1 year in case its recovery would be delayed due to weather conditions. The hydrophone (AGUAtech Low-Power Scientific Measurement Hydrophone) sensitivity was $-160~{\rm dB}$

re 1 V/ μ Pa; the data was quantized between ± 2.5 V. The hydrophones were connected to channel B on the two recorders which had gain correction parameters of -0.732 dB (Greenland I) and -0.576 dB (Greenland II), respectively.

2.2. Sound Speed Profiles

Sound speed profile data was obtained from the NOAA-NODC World Ocean Database 2013 to have some idea of the propagation properties of the environment. Pressure and salinity information was entered into the UNESCO equation to compute a sound speed profile. Three profiles are shown in **Figure 1B**. The red line was obtained from cruise #4832 (cast #12258746) recorded at 78.832 latitude and –9.998 longitude on September 10, 2003. In this case, pressure was not recorded and it was estimated from the depth. The green line was obtained from cruise #9719 (cast #3288290) recorded at 76.958 latitude and –14.203 longitude on September 9, 1984. The blue line was obtained from cruise #10547 (cast #9922885) recorded at 77.573 latitude and –12.3 longitude on September 3, 2000.

All three profiles were made around the same time of year as the seismic survey was performed. The recorders were installed below a possible acoustic channel at a depth of around 100 m. In any case, the airguns were towed well-above the sound channel, limiting the amount of acoustic energy that would have been trapped by it.

Based on the recordings, the frequency band considered to be most interesting was the third octave band centered on 40 Hz. Higher frequencies were not always as clearly apparent at long distances and below 20 Hz the airgun energy started to reduce. Absorption of this frequency in sea water is below 0.01 dB/km (Mellen et al., 1987; Ainslie and McColm, 1998) and was ignored in this analysis in light of the presence of greater sources of error and propagation distances under 300 km.

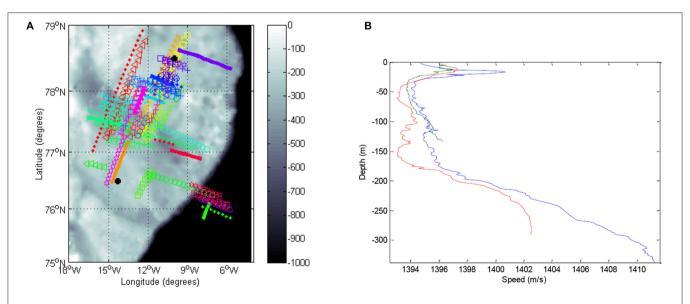
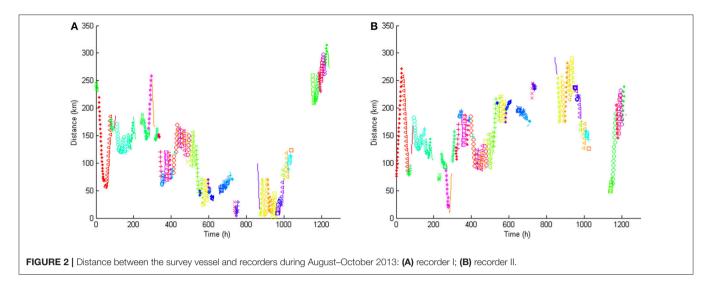


FIGURE 1 | Overview of the deployment area: (A) survey vessel and recorder positions together with bathymetric information (the color bar shows the depth in meters); (B) Sound speed profiles at the two recorders (red: Greenland I, green: Greenland II) and a location in between the two recorders (blue).



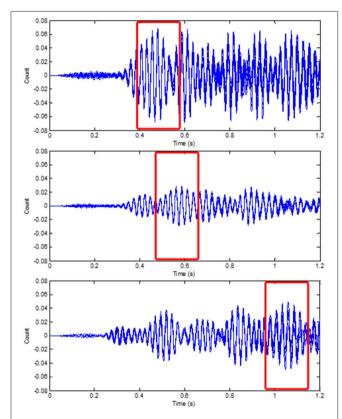


FIGURE 3 | Three air gun shot sequences from the Greenland II recorder. All images show the pulses detected in a single run file (2 min) synchronized using cross correlation with the first pulse in the recording. The top two ones were fired from a distance of 20 km, the bottom one from 50 km.

2.3. Airgun Shot Detection

Airgun shots were detected automatically using a basic magnitude threshold detector requiring peaks to be at least twice over the background noise level. The background noise level was estimated before each detected shot by taking a

0.5 s sample 5 s before the detected peak. The configured duty cycle only provided 2 min of continuous data at a time. For detected shots to be included in the analysis at least 5 shots had to be detected in the recording and not more than 13; the latter would indicate a recording with a large amount of impulsive noise from other sources. Additionally, detections were eliminated if they did not follow a pattern of about 11s intervals. Shots with received peak levels over 160 dB re 1 μ Pa² were excluded as they were likely affected by saturation. A shot was defined as starting 0.1s before the detected peak and ending 0.1s after it. Considering the large number of available shots that were detectable well-above the background noise, no efforts were made to fine-tune the detector to detect weaker impulses. However, as explained further below, the shots were not always equally well-defined due to bottom and surface interactions. In total, 10,076 shots detected at Greenland I and 11,391 shots detected at Greenland II were used.

2.4. Seismic Survey Data

Positional data was made available through a datasheet provided by Statoil containing the position and time of the air gun shots. All gun shot recordings were made in the months August–October 2013. The survey tracks are shown with a different color/shape combination for each run in Figure 1A. The positional data was not entirely consistent with occasional mixing of shots made under the same operating name and time, but at a very different position. These positions were much less frequent than the regular gun shots and were filtered away using a median distance estimate taken over ship positions from a time interval around the recorder timestamp.

A broadband source level estimate of the array is published in the GUNDALF array modeling suite report (Goppen, 2011) and gives as 252 dB re 1 μ Pa at 1 m for the zero to peak level (RMS pressure 229 dB re 1 μ Pa at 1 m). These values are understood to be used for long range modeling and are not correct very close to the array. The source level in a frequency band around 40 Hz was not known.

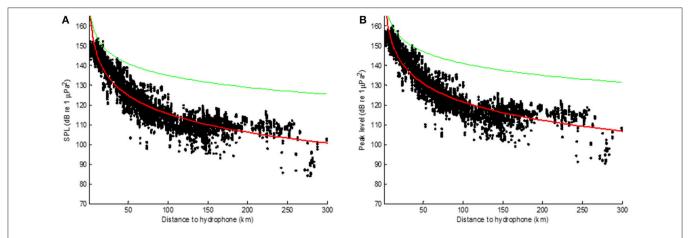


FIGURE 4 | Received level in the third octave band centered on 40 Hz (A) and received peak level (B) at the Greenland I recorder. The green line represents spherical loss; the red line is a logarithmic fit with optimal parameters –31 log₁₀(R) (with residual RMS = 5.0 dB) for both graphs.

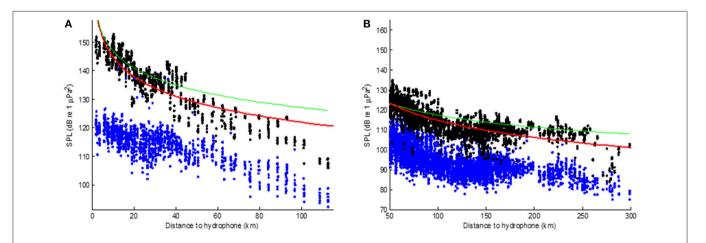


FIGURE 5 | Logarithmic fit on the sound levels of the third octave band centered on 40 Hz for measurements up to 150 km away using data from hours 860 to 990 (A) and measurements between 50 and 300 km away (B) from the Greenland I recorder. Level measurements are in black; background noise measurements are in blue. The green line represents spherical loss; the red line is a logarithmic fit. The optimal fit up to 50 km range (A) has parameters –16 log₁₀(R) (residual RMS = 4.2 dB) and beyond 50 km (B) –29 log₁₀(R) (residual RMS = 4.6 dB).

The distance between the survey vessel and each recorder is shown in **Figure 2**. The color and shape of the different runs follows the same scheme as in **Figure 1A**.

2.5. Time Synchronization

It was assumed that the ship timestamps were synchronized through GPS; the recorder times were configured before deployment, but they may have some unknown drift. To synchronize the clocks it was initially planned to find the start of an airgun run after a long pause that was recorded. There are more than enough pauses in the airgun deployment, but the duty cycle of the recorders made it more difficult to find such an event. Unless the clocks are very much out of synchronization, it seems that shooting started before the shots were registered in the shot datasheet, and also continued for some time after. This made it impossible to find a reliable moment for synchronization. In order to find the ship range for each detected shot on the

recorder all ship ranges in a 5 min time interval before and after the shot (10 min in total) were collected and the final range was evaluated through a median. The error with respect to the distance of the ship is considered to be small, minimizing at the same time the effect of spurious airgun activations discussed above.

2.6. Shot Measurement

For the comparison of measurements, it is important to understand what the automated algorithm is measuring. **Figure 3** shows airgun signals passed through a third octave band filter centered on 40 Hz from three different survey vessel locations and registered at the Greenland II recorder. Each image shows the superposition of all the signals detected inside a single run file of 2 min. The top two images were recorded with the vessel at roughly 20 km distance. The bottom image with the vessel at 50 km distance.

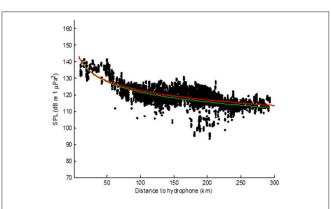


FIGURE 6 | Received level in the third octave band centered on 40 Hz at the Greenland II recorder. The green line represents spherical loss; the red line is a logarithmic fit with optimal parameters $-19 \log_{10}(R)$ with residual RMS =4.2 dR

There is an obvious difference in recorded levels between the two 20 km recordings, but first the detection algorithm is evaluated. The detector looks for the peak level and then takes a window of 0.1 s on both sides to compute the SPL. The airgun signals in the images actually consist of multiple pulses (about three or more) of different intensity; the strongest pulse is not always at the same position. This means that the algorithm may not always take the SPL measurement over the same part of the recorded signal, as shown with the rectangles in the images. The total window length of 0.2 s was selected to cover a complete single pulse and the level of the strongest pulse present in the signal is what is used to evaluate transmission loss, in addition to changes in the peak level itself. Considering the sampling rate of the recording (many times higher than necessary for the band being measured) the peak level itself should be fairly accurately estimated.

There is a fairly large difference in received levels at 20 km distance in **Figure 3**. The bathymetry profiles between the ship position and Greenland II recorder was inspected and were found to be similar: a gradual increase in elevation of 40–50 m. Ship shielding at seismic frequencies is not expected to play any role here. There was a difference in the array orientation but the airgun array is assumed to have omnidirectional radiation patterns. The low received levels in **Figure 3** were recorded just when a pause was made. It is not known if reduced levels were used as an Acoustic Deterrent Device which would have some effect on the transmission loss estimation; however, if enough shots were made at constant level these outliers can be ignored by the modeling process.

3. RESULTS

3.1. Greenland I

First the received sound pressure level in the third octave band centered on 40 Hz (SPL) and the received peak level (PL) in the same band are considered at the Greenland I recorder. **Figure 4** shows the SPL and PL against the survey vessel distance up to a distance of 300 km. The red line is a logarithmic fit on the

measurements; the green line represents spherical transmission loss. The root-mean-square of the residuals of each fit is reported as the "residual RMS." The transmission loss follows a pattern much larger than spherical. In arctic waters it is not unusual to find high losses at some distance from the source due to dissipation of reflective rays.

The logarithmic fit in Figure 4 does not follow the curve of the data very well and a single "log(R)" rule does not seem sufficient. To look at the loss close to the source a smaller selection of data was taken based on the distance graphic in Figure 2. At the end of September and beginning of October (860-990 h) the survey vessel came closest to Greenland I. Matching the track colors and shapes in Figure 1A these tracks appear to go almost straight over the recorder, providing very similar conditions for the measurements. The received levels of that time period are displayed in Figure 5 (black dots) together with background noise levels (blue dots) showing that all the shots are well-above background noise levels. The transmission loss was now close to spherical, but still quite high. From around 40 km distance the "near distance" model starts to fail and another fit is required. Figure 5B shows the transmission loss for distances from 50 km together with background noise level estimates. The logarithmic fit follows the data fairly well. The received levels have a spread of around 10 dB at each distance. Using a combination of the two models a prediction could be made below 5 dB error (based on the residual RMS-values).

The background noise levels plotted in **Figure 5** seem to be coupled to the distance to the survey vessel in a very similar way as the shots. The operational noise from the survey propagates well at least up to 300 km.

3.2. Greenland II

The data from the Greenland II recorder will be presented in a similar fashion as those from Greenland I. The peak level measurements followed the 40 Hz levels closely and are not provided. In **Figure 6**, the 40 Hz SPL measurements are shown with a logarithmic fit (red) and spherical transmission loss (green). The transmission loss behaves much more spherical than at Greenland I, with slightly less than spherical loss at large distances

At close range the model does not fit the data very well. On September 3 and 4 the survey vessel made two very similar tracks nearby the recorder. These correspond to the orange and purple tracks close to the Recorder II in **Figures 1A**, **2** (260–285 h). The variation in received levels was large, which was discussed above in relationship with **Figure 3**. Fitting a logarithmic model only on the levels received at a distance of 50 km or more, as was done in **Figure 5** for Greenland I, resulted in parameters –18 $\log_{10}(R)$ with residual RMS = 4.2 dB. As with the Greenland I recorder, the background noise levels follow a similar pattern as the airgun shots related to the survey vessel distance (not shown in plot), indicating that the survey itself is the dominant contributor during that time.

Interestingly, in e.g., **Figure 6**, which shows the received peak levels as in **Figure 4B**, there is a clear drop in both airgun shot levels and background noise levels of around 20 dB. This could be due to the bathymetry. From that position, there was an

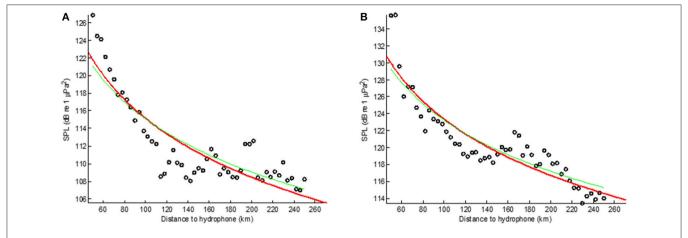


FIGURE 7 | Logarithmic fit on the sound levels of the third octave band centered on 40 Hz for measurements from 50 km away and binned at 4 km intervals; (A) Greenland I and (B) Greenland II. The green line represents spherical loss, the red line an optimal logarithmic fit, in both cases –22 log₁₀(R); the residual RMS was 2.3 dB (A) and 2.0 dB (B). These curves are based on summer data from August to October 2013.

underwater ridge at about 136 km from the ship, 50 km from Recorder II with its peak around 25 m above the recorder depth, possibly blocking a fair amount of sound. A similar effect was not as apparent in the data from Recorder I as most of the survey tracks were made on the north side of it.

4. CONCLUSION

From the data presented above, it appears difficult to define a "log(R)" when the source is close to the recorder. **Figures 5**, **6** give very different estimates of the transmission loss with the former showing a loss much larger than spherical and the latter a loss somewhat smaller. However, there was less data available at close distances which made it more difficult to average out level fluctuations due to operational or environmental circumstances.

A large amount of data was available for both recorders for larger distances, but at any particular distance the spread of the received levels was in the order of 10 dB. To clean up the data

the levels were binned at 4 km intervals from 50 up to 250 km. The results are shown in **Figure 7**, where the transmission loss at both recorders is reasonably modeled by spherical transmission loss. There are bumps in the curves where the propagation path may have been optimal or partially blocked, but as an order of magnitude estimation a model using $-22 \log_{10}(R)$ seems to be a good approximation for this area during summer months.

AUTHOR CONTRIBUTIONS

My, MA, and SD contributed to the data analysis of all recordings. AH, JW, and MA contributed to the deployment and acquisition of the data. All authors contributed to the draft and revisions of this article.

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Impacts of Navy Sonar on Whales and Dolphins: Now beyond a Smoking Gun?

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The risks military sonar poses to cetaceans received international attention with a highly-publicized mass stranding of Cuvier's beaked whales (Ziphius cavirostris), Blainville's beaked whales (Mesoplodon densirostris), and northern minke whales (Balaenoptera acutorostrata) in the Bahamas in 2000. This was the first time that the US Government determined a stranding to be the result of mid-frequency active sonar use. Subsequently attention has been drawn to other mass strandings coincident with naval exercises, including events preceding the 2000 mass stranding. The list of species for which mass strandings have been linked to naval exercises has also increased to include other beaked whales, dwarf and pygmy sperm whales (Kogia spp.), pilot whales (Globicephala spp.), several dolphin species (Stenella sp. and Delphinus delphis), and harbor porpoises (Phocoena phocoena). In particular, there have been several mass strandings in the northern Indian Ocean coincident with naval exercises - including one of the largest (200-250 dolphins)-which have received little attention. Changes in beaked whale behavior, including evasive maneuvering, have been recorded at received levels below <100 dB re 1 μPa (rms) and mass stranding may occur at received levels potentially as low as 150-170 dB re 1 µPa. There is strong scientific evidence to suggest that a wide range of whale, dolphin and porpoise species can also be impacted by sound produced during military activities, with significant effects occurring at received levels lower than previously predicted. Although there are many stranding events that have occurred coincident with the presence of naval vessels or exercises, it is important to emphasize that even the absence of strandings in a region does not equate to an absence of deaths, i.e., absence of evidence does not mean evidence of absence. Strandings may be undetected, or be unlikely to be observed because of a lack of search effort or due to coastal topography or characteristics. There may also be "hidden" impacts of sonar and exercises not readily observable (e.g., stress responses). Due to the level of uncertainty related to this issue, ongoing baseline monitoring for cetaceans in exercise areas is important and managers should take a precautionary approach to mitigating

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impacts and protecting species.

INTRODUCTION

The risks sonar poses to cetaceans received international attention with a highly-publicized mass stranding of Cuvier's beaked whales (Ziphius cavirostris), Blainville's beaked whales (Mesoplodon densirostris), and northern minke whales (Balaenoptera acutorostrata), in the Bahamas, in 2000 (Balcomb and Claridge, 2001). This was the first time that the US Government determined a stranding to be the result of midfrequency active sonar use¹ (Anonymous, 2001), although the link between naval exercises and beaked whale strandings had first been documented in the 1970s (Van Bree and Kristensen, 1974). Following the Bahamas strandings, concerns started to be expressed about the threats posed to cetacean populations by active sonar and scientists started to point to evidence of more sonar-related strandings in various parts of the world (Parsons et al., 2008a; Dolman et al., 2011) This concern led to several court cases in the US and legal injunctions against military exercises using sonar (Zirbel et al., 2011a); sonarrelated resolutions from international treaty organizations; and statements of concern by professional organizations (see Parsons et al., 2008a; Dolman et al., 2011; Simmonds et al., 2014).

Although there was mounting scientific evidence that sonar could cause impacts on cetaceans, the issue of military sonar was—as Parsons et al. (2008a) put it—a "smoking gun" in relation to its possible link to cetacean strandings, injuries and mortalities. The largely precautionary approach, by legal bodies and organizations, to protect cetaceans from a possible and on-going threat, appears to be supported by the general public. A survey restricted to the Washington DC area found that 51% of respondents believed that naval sonar impacted marine mammals and, moreover, three-quarters (75.2%) thought that the Navy should not be exempt from environmental regulations during peacetime. They also believed that "sonar use should be moderated if it impacts cetaceans" (75.8%; p. 49) and there was bipartisan support for such protection (Zirbel et al., 2011b).

This paper provides an update on the latest scientific data on the effects of sonar on cetaceans, showing that the impacts of military sonar on a variety of cetacean species are now more than a "smoking gun," that all navies need to fully assess the likely true extent of these impacts, and immediately implement best practice, including effective monitoring and mitigation measures.

BEAKED WHALE STRANDINGS

Beaked whale mass strandings are relatively unusual events and draw attention when they occur (see Parsons et al., 2008a; for a previous summary).

An analysis of "atypical" mass strandings² of beaked whales found enough evidence for a statistically significant correlation between 12 of these events (out of 126 beaked whale mass standings since 1950) and naval exercises in the Caribbean and Mediterranean (D'Amico et al., 2009; Filadelfo et al., 2009a). A further 27 beaked whale mass stranding events occurred either adjacent to naval facilities or at the same time as nearby naval vessels could have been using active sonar (D'Amico et al., 2009; Filadelfo et al., 2009a). It should be noted that due to a lack of availability of data on naval sonar use, it is entirely possible more of these beaked whale mass strandings may have been linked to naval sonar use, or exercises.

Subsequently, in 2014, five Cuvier's beaked whales stranded on the coast of Crete during the *Noble Dina 2014* joint exercise with the Israeli, Greek and US Navies (Dolman, 2014). In early 2015, during the hunt for a Russian submarine off the coast of western Scotland and Ireland, a further eight Cuvier's beaked whales ("atypically") stranded on the coast of Ireland (Sibylline, 2015). Also in 2015, three beaked whales stranded simultaneously but in different locations along the southern coast of Guam. It was confirmed that a joint US-Japanese naval exercise, incorporating sonar use and anti-submarine activities, was being conducted in nearby waters when the strandings occurred (23–27 March 2015; Kuam News, 2015).

There have also been mass strandings where anthropogenic underwater noise has been suspected to be a factor; however, there was not enough information to make a link. For example, in 2008 there was a high level of cetacean strandings reported (56 animals over a 7-month period)—including Cuvier's and Sowerby's beaked whales (*Mesoplodon bidens*) and long-finned pilot whales (*Globicephala melas*)—off the coast of Ireland and Scotland (Dolman et al., 2010). However, a full investigation was not conducted due to the carcasses' advanced state of decomposition. In the winter of 2014-15 (Amos, 2015; Siggins, 2015), there was a recurrent increase in Cuvier's beaked whale strandings in this region (n = 15), which appeared to have occurred at the same time as a high level of anti-submarine activity, although active sonar use in this region was denied by the Royal Navy (Farmer, 2015).

THE RECEIVED LEVELS OF SONAR AND BEAKED WHALES IMPACTS

In 2007, a US government-convened panel published guidelines for the level of noise at which injury occurs to cetaceans (Southall et al., 2007). They considered impulsive sound at levels of 230 dB re:1 μ Pa peak pressure was an uppermost "safe" exposure limit for marine mammals, including beaked whales. The 2007 limit has since been adopted by many noise producers and managers as an absolute level at which injury impacts to cetaceans occur [(for example, a European Union advisory group used these criteria

 $^{^{1}}$ Mid-frequency active sonar has a frequency range of 1kHz-10kHz. One of the systems most frequently used and/or associated with stranding events is the AN/SQS 53C system (3.5kHz with most energy in the 2.5kHz-4.5Hz range) with a source level of 235 dB rms re 1 μ Pa @ 1m. Low frequency active sonar has a frequency range of 100-500Hz and is utilized at approximately the same source level as mid-frequency sonar.

 $^{^2\}mathrm{These}$ "atypical" mass strandings are when multiple animals come ashore, but the strandings may occur over sizeable geographic area over a short time frame (Frantzis, 1998). D'Amico et al. (2009) use a definition of two or more animals stranding within a six day period over a 40 nautical mile (74km) stretch of coastline.

for their advice on harmful sound levels in EU waters (Tasker et al., 2010; Genesis, 2011)]. The approach used by Southall et al. (2007) has been criticized on methodological and statistical grounds, such as inconsistency of weighting functions and problems with pseudo replication that downplay the sensitivity of animals to sound (Tougaard et al., 2015; Wright, 2015). For example, the proposed levels were developed using limited available evidence, where levels at which temporary (TTS) and permanent threshold shift (PTS) and other responses occur in a small number of captive cetaceans from a limited number of species [i.e., common bottlenose dolphins (*Tursiops truncatus*) and beluga whales (Delphinapterus leucas)], particularly animals kept in the US Navy's marine mammal research facilities. Critics have noted that trained, captive cetaceans, often in noisy facilities and exposed to high sound level experiments many times, may not respond in the same way as naïve, wild animals (Parsons et al., 2008a; Wright et al., 2009). Scientists studying US captive cetacean responses to sound have also highlighted that using these animals to directly predict the behavior of wild animals can lead to biased and/or inaccurate predictions. Such studies are "likely not directly transferrable to conspecifics in the wild. The dolphins have years of experience under stimulus control, which is a necessary condition for the performance of trained behaviors, and they live within an environment with significant boating activity. These factors likely impact the threshold of responsiveness to sound exposure, potentially in the direction of habituation or increased tolerance to noise" (Houser et al., 2013, p. 130). In fact, the original panel that published the 230 dB re:1 µPa peak pressure safe level noted that cetacean strandings, and thus injury and probable death, could occur at much lower levels due to behavioral changes occurring at much lower sound levels than their criterion (Southall et al., 2007). NOAA Fisheries has since introduced updated guidance (National Marine Fisheries Service, 2016) which, for example, notes that the 230 dB re:1 µPa peak pressure is an impulsive (one off) exposure level that could cause PTS in species, such as beaked whales (224 dB re:1 µPa peak pressure for TTS). The updated guidance notes that a cumulative sound exposure level (over 24 h) of 185 dB re 1 µPa2s could likewise cause PTS or 170 dB re 1 μPa²s for TTS (National Marine Fisheries Service, 2016). However, this guidance is written in an extremely technical format and is far from accessible to non-specialists.

That behavioral impacts occur at lower sources levels than noted above, is an important caveat is frequently overlooked by noise managers when developing mitigation measures to active sonar. Following the 2000 Bahamas mass stranding it was estimated that these whales were exposed to sound levels no higher than "160–170 dB rel μPa @ 1 m for 10–30 s" (p. 286 in Hildebrand, 2005a); or even 150–160 dB re 1 μPa for 50–150 s (Hildebrand, 2005b), a level clearly much lower than the (now widely-used) noise impact guideline level of 230 dB re: 1 μPa (Southall et al., 2007), which would result in a much larger impact radius around the active sonar source.

Subsequent at-sea studies investigated the specific responses of tagged Blainville's beaked whales to military sonar (n = 6). Tyack et al. (2011) found that one animal stopped feeding above 138 SPL dB (or a cumulative sound exposure level (SEL) of 142

dB re 1 μ Pa²-s), while a second experiencing a received level of 146 re 1 μ Pa swam 10s of km away (an average of 54 \pm 10 km) from the center of the testing range and remained out of the area for 2–3 days (McCarthy et al., 2011; Tyack et al., 2011). The calls of beaked whales in the area also decreased during sonar exposure and did not recover to pre-exposure levels for up to 108 h after exposure, although calls were produced even at estimated exposure levels of 157 dB re 1 μ Pa (rms) (McCarthy et al., 2011).

In a more recent study on tagged Cuvier's beaked whales (n=2), the animals began to respond at received levels of 89 dB re 1 μPa (rms) by ceasing to beat their tail flukes (DeRuiter et al., 2013). One animal stopped echolocating, ceased foraging, and swam rapidly away from the source at a received level of 98 dB re 1 μ Pa (rms). The avoidance response lasted for 1.6 h. The other whale demonstrated similar responses, and displayed an abnormal diving pattern for 7.6 h after exposure to sonar. One of the whales was incidentally exposed to sonar levels similar to those that produced a response (78-106 dB re 1 µPa rms) from a naval vessel that was using sonar 118 km away, according to the ships' log (DeRuiter et al., 2013). The researchers stated that "current US management practices assume that significant behavioral disruption almost never occurs at exposure levels this low" (DeRuiter et al., 2013). In fact, significant impacts to beaked whales could occur at levels lower, and from sound sources at greater distances from animals, than previously thought, arguably making current US mitigation guidelines for mid-frequency active sonar ineffective at preventing wide-scale impacts to whales.

Miller et al. (2015) determined that Northern bottlenose whales (Hyperoodon ampullatus) showed a "high sensitivity ... to acoustic disturbance, with consequent risk from marine industrialization and naval activity" (p. 1). At a received sound pressure level (SPL) of 98 dB re 1 µPa, a tagged whale turned to approach the sound source, but at a received SPL of 107 dB re 1 µPa, the whale began moving in an unusually straight course and then made a near 180° turn away from the source, and performed the longest and deepest dive (94 min, 2339 m) recorded for this species (Miller et al., 2015). Animal movement parameters differed significantly from baseline for more than 7 h until the tag fell off 33-36 km away (Miller et al., 2015). No clicks were emitted during the response period, indicating cessation of normal echolocationbased foraging. A sharp decline in both acoustic and visual detections of conspecifics after exposure suggests other whales in the area responded similarly (Miller et al., 2015). Sivle et al. (2015) also noted avoidance behavior by bottlenose whales to a 1-2 kHz sonar signal, starting at a sound pressure level of 130 dB re 1 μPa. They noted "severe" responses to the sonar exposure (as ranked by experts grading the responses), including cessation of feeding and long-term avoidance (Sivle et al., 2015).

Reponses to (simulated) sonar signals (3.5–4 kHz) were also noted for Baird's beaked whale (*Berardius bairdii*) by Stimpert et al. (2014). The researchers noted that "within 3 min of exposure onset, the tagged whale increased swim speed and body movement, and continued to show unusual dive behavior for

each of its next three dives," with reactions by the whale occurring at a received level of approximately 127 dB re 1 μ Pa.

A number of studies suggest population-level impacts in beaked whales from repeated exposures to naval activities (Dolman and Jasny, 2015). A Blainville's beaked whale population on the Navy's AUTEC naval range, in The Bahamas, had lower abundance and recruitment success (calf to female ratio) than another off-range Bahamas population, based on a 15-year field study (Claridge, 2013). Further, adult females showed high residency at the navy range, putting them at risk, especially when pregnant and lactating (Claridge, 2013). In California, naval activities were proposed as one of two plausible hypotheses, along with ecosystem change, to explain a precipitous decline in beaked whale populations in the California Current ecosystem (Moore and Barlow, 2013).

The studies above document behavioral changes in beaked whales at relatively low levels of mid-frequency sonar exposure that can be expected to occur at distances many hundreds of miles from the sonar source. It should be noted, however, that the degree of responses by animals, and the received level of sound at which these responses occur, might be affected by the context in which the sound is received. For example, a mother and calf might be more "skittish" than a solitary male; an animal that urgently needs to feed may show less of a behavioral change than one that is relatively well-fed; a young animal that is more vulnerable to predation might react more quickly to an intense noise than a larger adult; a habituated animal might respond at higher received levels than a naive animal; or a chronically stressed animal might responded differently to a nonstressed animal (see section Absence of Evidence Does Not Mean Evidence of Absence-the Need For Precaution Below; Beale and Monaghan, 2004; Beale, 2007; Wright et al., 2007; Guerra et al., 2014; Forney et al., 2017).

Even if the changes in beaked whale behavior resulting from sonar use do not lead to stranding events, they could still lead to sub-lethal impacts and significantly impact the health of individuals, and potentially populations, by affecting biologically important behaviors, such as preventing normal feeding or separating family members. The degree to which this happens is currently an important question for cetacean conservation, in all species (e.g., Parsons et al., 2015). For example, even minor reductions in feeding behavior as the result of human disturbance were estimated to have dramatic effects on the energy budget of cetaceans, which could translate into substantive negative impacts on cetacean fitness and health (Christiansen et al., 2013).

To quantify this energetic impact, Williams et al. (2017) tried to estimate the energetic cost of beaked whales evading sonar. Using the energetic costs of bottlenose dolphin fluke strokes (3.31 \pm 0.20 J kg $^-$ 1 stroke $^-$ 1), the cost of high speed evasion responses in cetaceans, including observed escape responses of beaked whales to naval sonar (increased fluking rates and longer bursts of powered swimming), was estimated. Williams et al. (2017) reported a theoretical 30.5% increase in beaked whale metabolic rate, with an elevated rate being maintained for more than 90 min after the exposure to noise. Even increasing the amplitude of vocalizations—so that calls may be heard in a noisy environment—may have an energetic cost (Holt et al., 2015).

However, the impact of these energetic costs on cetacean health, both short- and long-term, needs to be evaluated.

There are several modeling efforts underway to estimate the health and population-level impacts of behavioral disturbances upon cetacean populations, with beaked whales being a particular cause for concern. The most notable are the PCOD and PCAD models (see King et al., 2015 and Harwood et al., 2016 for details). One particularly enlightening study, on gray whales (Eschrichtius robustus), predicted that an energy loss of 4% because of disturbance events during the year of pregnancy would result in reproductive failure (Villegas-Amtmann, et al., 2015). Moreover, a 30–35% energy immediately before pregnancy would mean that a female would lack sufficient energy to become pregnant (Villegas-Amtmann, et al., 2015). Death would occur at a 40-42% energy loss (Villegas-Amtmann, et al., 2015). This equates to a loss of only 10 days of feeding opportunities due to disturbance theoretically leading to an unsuccessful pregnancy or loss of a whale calf (Villegas-Amtmann, et al., 2015).

OTHER CETACEAN SPECIES AFFECTED BY ACTIVE SONAR

A young male beluga whale was exposed to mid-frequency sound frequencies [19-27 kHz;140-160 dB (no reference level given)] and exhibited significantly increased heart rate, with the rate increasing with the intensity of the sound level (Lyamin et al., 2011). Heart rate increased no matter how many times the whale was exposed to the sound and the animal showed no signs of habituation. The respiration rate of the animal also increased significantly at the beginning of exposures. Such "severe tachycardia" is the heart's reaction to a stressor. This started at very low noise levels (i.e., 140 dB), suggesting a relatively severe physiological stress response to anthropogenic noise exposure in this whale. One would expect similar, substantive, yet not readily observable and effectively "hidden" stress responses to occur in other cetacean species with similar physiologies (such as beaked whales). Although shortterm (acute) stress responses are essential for the survival of animals, allowing them to undergo "fight or flight" responses, continued (chronic) activation of substantive stress responses can be physiologically detrimental to animals (Wright et al., 2011).

Tagged blue whales (*Balaenoptera musculus*) in the Southern California Bight displayed behavioral responses to experimental mid-frequency active sonar. Although the sound levels produced in the experiments were orders of magnitude below most military systems, the blue whales responded by stopping feeding, increasing swimming speed and traveling away from the sound source, with displacement occurring at a received level of 140 dB re 1 µPa, with other responses, such as cessation of feeding, occurring at lower source levels (Goldbogen et al., 2013). Baleen whales thus alter biologically important activities in the presence of sonar sounds. Moreover, the researchers expressed their concerns that "frequent exposures to midfrequency anthropogenic sounds may pose significant risks to the recovery rates of endangered blue whales" because they ceased feeding and were displaced (p. 6 in Goldbogen et al., 2013).

Northern minke whales (*Balaenoptera acutorostrata*) had been noted previously (Parsons et al., 2008a) to strand during military sonar-related beaked whale mass stranding events (e.g., in 2000 in the Bahamas and in 2005 in North Carolina; Anonymous, 2001; Balcomb and Claridge, 2001; Hohn et al., 2006). Moreover, it has been noted that during naval exercises in Scotland, minke whale sighting rates significantly decreased (Parsons et al., 2000). It was subsequently proposed that minke whales hear well within the range of mid-frequency active sonars, and thus they are likely to be at risk from them over wide ranges, although this species is often overlooked in exercise planning (Tubelli et al., 2012). Subsequently, Sivle et al. (2015) found that minke whales exhibited "high speed avoidance" (p. 469) when exposed to $1-2 \, \text{kHz}$ sonar signals, with avoidance starting at sound pressure levels of 130 and 146 dB re 1 μ Pa,

Minke and blue whales may not be the only baleen whales that are vulnerable. Between 1982 and 2007, of 180 gray whale (Eschrichtius robustus) standings that occurred in California, 22% coincided in time and location with military exercises (Filadelfo et al., 2009b). Although the monthly pattern of whale strandings in relation to military exercises was statistically insignificant, nonetheless a substantial proportion of gray whale strandings did occur coincident with naval exercise periods and the situation warrants precautionary management and further investigation into whether this species may also be vulnerable to military noise (Filadelfo et al., 2009b). Indeed, a study noted that migrating gray whales moved around a stationary sound source emitting low frequency active sonar sounds (0.1–0.5 kHz), based on land-based observations (Buck and Tyack, 2000; Croll et al., 2001; Tyack, 2009), with avoidance occurring at a received level of approximately 140 dB re 1 µPa (Buck and Tyack, 2000). Minor movement to avoid a loud sound source may not seem like a major impact at first glance, but as mentioned above, Villegas-Amtmann, et al. (2015) estimated that just 10 days of lost foraging opportunities due to disturbance could lead to an unsuccessful pregnancy/loss of a calf in gray whales (Villegas-Amtmann, et al., 2015).

Humpback whales (Megaptera novaeangliae) changed their singing behavior, lengthening their songs-with some ceasing altogether—when exposed to low frequency active sonar (Miller et al., 2000). As humpback whale song plays a significant role in their mating behavior (Parsons et al., 2008b), this may have biological significance. Sivle et al. (2015) noted humpback whales responded to 1-2 kHz active sonar, although the responses were less severe, at received levels higher than did minke and bottlenose whales. However, Sivle et al. (2016) found that the first exposure of 12 humpback whales to military low-frequency sonar (1.3-2.0 kHz with SPLs at the source up to 160-180 dB re 1 µPa) led to a statistically significant, 68% reduction in lunge feeding rates. Moreover, during a second exposure, the feeding rate was 66% below normal, pre-exposure levels. Such a significant reduction in feeding might have an impact on the energy budget of these whales.

The following species, other than beaked whales, have stranded coincident with naval exercises: dwarf sperm whales (*Kogia sima*); pygmy sperm whales (*K. breviceps*); short-finned pilot whales (*Globicephala macrorhynchus*); long-finned pilot

whales (G. melas); pygmy killer whales (Feresa attenuata); and several dolphin species (Stenella attenuata and S. coeruleoalba) (Kaufman, 2004, 2005; Department of the Environment and Heritage, 2005; Hohn et al., 2006; Wang and Yang, 2006; Parsons et al., 2008a). Some of these standings occurred even though naval vessels were 90 nautical miles away from the stranding area (Kaufman, 2005)—a distance which is now known to be within the range that sonar exercises could potentially cause cetacean behavioral changes (DeRuiter et al., 2013). It should be noted that the strandings of long-finned pilot whales were usually associated with high frequency sonar (50-200 kHz) usage, as opposed to mid-frequency active sonar (the latter is often considered to be the sound source of most concern) (Department of the Environment and Heritage, 2005). Another species that could be added (although not stranding as such), is the melon-headed whale (Peponocephala electra). This species has entered unusually shallow waters in response to sonar exposure—a so-called "milling event" (Southall et al., 2006).

Even sperm whales (*Physeter microcephalus*) have been documented responding to sonar. Isojunno et al. (2016) and Curé et al. (2016) reported avoidance behavior, interruption of foraging and/or resting behavior, and an increase in social sound production in response to 1–2 kHz active sonar. Sperm whales stopped foraging at cumulative received sound exposure levels (SEL) of 135–145 dB re 1 μ Pa (Curé et al., 2016). They also displayed avoidance and social call changes in response to 6–7 kHz sonar, although the responses were less pronounced (Curé et al., 2016; Isojunno et al., 2016).

In recent years, more dolphin species have been found during mass stranding events coincident with naval exercises. In June 2008, a mass stranding of common dolphins (Delphinus delphis) was associated with a naval exercise in Falmouth Bay, UK and at least 26 of these animals died. The researchers who evaluated the standing event determined "naval activity to be the most probable cause of the Falmouth Bay [mass stranding event]" (Jepson et al., 2013). One of the largest dolphin stranding events to date, however, occurred 6-7 March 2009 on Gaddani Beach on the Balochistan coast of Pakistan, 50 km northwest of Karachi, when a mass stranding of 200-250 pan-tropical spotted dolphins (Stenella attenuata) occurred on the second day of a multinational naval exercise, AMAN 09 (5-14 March 2009, involving 20+ warships from the US, UK, France and Australia) (Kiani et al., 2011). This event was the largest (atypical) mass stranding recorded of this species by an order of magnitude. It seems highly likely that this unusual mass mortality was also caused by naval exercises.

A common dolphin mass stranding (*Delphinis capensis*; n=11) occurred on the Iranian coast on 22 January 2011 (Mohsenian et al., 2014). Although this paper's authors stated that they had been told that no Iranian naval activity had occurred prior to the mass stranding (Mohsenian et al., 2014), a large multi-national naval exercise involving the Indian, French and US navies in the Arabian Sea had commenced on 11 January 2011 (Anonymous, 2011). These mass strandings of dolphins in the northern part of the Indian Ocean have received little to no attention by government agencies in Europe and the US.

Other delphinids may also be vulnerable to active sonar. For example, killer whales (*Orcinus orca*) exposed to mid-frequency active sonar in Norway responded at received levels much lower than currently addressed by US Navy mitigation measures (Miller et al., 2014). In fact, Harris et al. (2015) found that killer whales were more likely to respond to sonar at lower received levels than sperm whales or long-finned pilot whales.

Recent research has also highlighted the susceptibility of porpoises to naval activities. In one incident, 85 harbor porpoises (*Phocoena phocoena*) stranded along approximately 100 km of Danish coastline from 7 to 15 April 2005 (Wright et al., 2013). Bycatch was established as the cause of death for most of the individuals, and military vessels from various countries were confirmed in the area from 7 April, *en route* to the largest naval exercise in Danish waters to date (Wright et al., 2013). Although sonar usage could not be confirmed, it is likely that ships were testing sonar equipment prior to the main exercise. Thus, naval activity cannot be ruled out as a possible contributing factor (Wright et al., 2013).

In fact, recent acoustic exposure experiments suggest that harbor and finless porpoises (*Neophocaena phocaenoides*) may be more sensitive to anthropogenic sound than previously thought. Previous predictions had extrapolated their sensitivity to sound based on results from common bottlenose dolphins (*Tursiops truncatus*), but experimental results show this appears to have underestimated the sound levels at which impacts (behavioral and TTS) to harbor porpoises might occur (Tougaard et al., 2015). For porpoises, Tougaard et al. (2015) found that impacts strongly depend on the frequency of the sound, with avoidance reactions occurring just 40–50 dB above the hearing threshold for a particular frequency, with TTS occurring at about 100 dB above the hearing threshold.

There is a substantive and growing body of corroborating evidence to suggest that a wide range of whale, dolphin and porpoise species can be impacted by sound produced during military activities. The risk active sonar poses is not limited to beaked whales only. In fact, there may be more individuals of non-beaked whale species that have stranded coincident with military exercises than beaked whales. In addition, the level of sound at which impacts can occur is generally lower than previously believed. Thus, there is an urgent need for nations to require more strategic and widespread active sonar management. A more concerted effort to monitor for cetacean strandings, including delphinids, and to plan mitigation measures for all naval exercises—especially in the Indian Ocean—is warranted.

ABSENCE OF EVIDENCE DOES NOT MEAN EVIDENCE OF ABSENCE-THE NEED FOR PRECAUTION

Although there are many stranding events that have occurred coincident with the presence of naval vessels or exercises, it is important to emphasize that even when strandings do not occur coincident with naval exercises, this does not mean there have been no deaths or other negative impacts.

It is highly likely that injury or mortality at sea caused by noise will not be observed, as explained previously (Fernández et al., 2005; Parsons et al., 2008a). In some locations, even if animals do strand, it is unlikely carcasses will be observed or recovered, either because they wash away before they are seen or the location is too remote for them to be observed at all.

To illustrate this: there have been 11 cetacean mass stranding events in the Hawaiian Islands in a 22-year period, of which six have coincided with military exercises (Faerber and Baird, 2010). However, despite the occurrence of beaked whales in these waters, none of these mass strandings have involved beaked whales. Through 2006, only nine single beaked whale strandings were recorded on Hawaii's coasts (Faerber and Baird, 2010). Due to this paucity of records of beaked whale strandings, the US Navy has stated that there are no impacts on vulnerable beaked whales in this location from military activities (Faerber and Baird, 2010). However, an analysis of topography and coastal characteristics indicates that a variety of factors—a lack of beaked whale habitat close to shore, a prevalence of steep cliffs, lower human densities on the coast-decreases the likelihood of strandings occurring and/or being detected in Hawaii compared to elsewhere (e.g., Canary Islands) (Faerber and Baird, 2010). Faerber and Baird (2010, p. 610) stated that "it is inappropriate to conclude there has been no impact on beaked whales from anthropogenic activity in the Hawaiian Islands." This conclusion could be extrapolated to any location where coastal features make strandings unlikely, or unobservable, or locations where there is a lack of public awareness about the need to report stranded cetaceans so necropsies can be done, or a lack of search effort for cetacean carcasses, at sea or beached, during active sonar exercises. Moreover, most cetaceans sink upon death (Allison et al., 1991), which means discovery of any cetacean killed during exercises in deep waters is unlikely. Indeed, most of the world's coastlines can be considered regions of low reporting for cetacean mortalities.

Decomposition is also relevant, as time is a critical factor to collecting pathology evidence. For example, Morell et al. (2015) has developed a novel technique that requires carefully examining the microscopic hair cells inside the ear of the whale and appears able to pinpoint damage as well as the frequency of the damage, which is critical for identifying the sound source. Ears need to be removed within just a few hours of death to be analyzed.

Other impacts include biologically important behavioral changes, over scales that far exceed current management measures, which are difficult to accurately predict or to take into account. Moreover, absence of behavioral changes, such as moving from feeding habitat, is not necessarily an indicator of no impact. For example, in Australia two sites occupied by dolphins were investigated—an area where dolphin-watching occurred and an area undisturbed by dolphin-watching 17 km away. At the site where there was no dolphin-watching, dolphin behavior changed more significantly than at the site where dolphin-watching (and therefore noise disturbance) occurred (Bejder et al., 2006). This would normally lead to the conclusion that boat traffic had little impact on animals at the site where dolphin-watching occurred, i.e., they were habituated, but the study also looked at changes in

dolphin abundance at the two sites over 10 years. The researchers found that in the area where dolphin-watching occurred, there was a significant decline in dolphin numbers (14%) linked to an increase in dolphin-watching activity (Bejder et al., 2006). The researchers concluded that the most sensitive animals moved away from the area, but this effect would have been hidden without more detailed examination. As a result of this study, the Australian government implemented restrictions on dolphin-watching boats (to one), and thus reduced disturbance in the area.

Therefore, even if there are cetaceans still visible in an area while military exercises are being, or have been, conducted, managers should not conclude that there has been no effect on cetaceans in the area. The animals being observed could possibly be less sensitive animals that have remained in an area, and more sensitive animals (such as pregnant females) may have been displaced. Even with detailed observations on the movement of individual animals in the population, one cannot say categorically there has been not been a significant impact of a sound-producing activity.

Moreover, a lack of visible behavioral response by an animal might be an indication that an animal is extremely stressed already. A stressed, starving or sick animal may not display any observable response if they do not have the energy or capability to react behaviorally; for example, if the disturbance location is the only viable feeding area, the animal may not leave (Beale and Monaghan, 2004; Beale, 2007; Wright et al., 2007; Forney et al., 2017). In short, absence of a behavioral response to noise does not necessarily translate to absence of a significant or life-threatening impact. Sonar management should account for this and a "precautionary" approach should be taken, with efforts undertaken to minimize noise exposure even though there might not be immediately obvious impacts upon cetacean behavior.

Finally, there are "hidden" responses by animals that may not be readily visible. As noted above, animals may undergo a substantial stress response at relatively low levels of noise exposure, and chronic stress could well lead to major physiological and health impacts (Wright et al., 2009). The level and impacts of stress in populations of animals that face chronic sound exposure (such as those within sonar testing ranges or in regular military exercise areas) need to be studied urgently. There are many non-invasive methods of studying stress hormone levels in cetaceans that are now viable (Hunt et al., 2013) and this could be done relatively easily on potentially impacted populations.

UNCERTAINTY IN MARINE SCIENCE

In Australia, over 10 years of data were required to determine that there was a disturbance impact on a dolphin population (Bejder et al., 2006). In the Bahamas, it took 15 years to gather enough data to note a decline in beaked whale abundance on a military testing range (Claridge, 2013).

A lack of longitudinal data and studies gathering baseline data before the onset of sound-producing events are common problems with cetacean research. In addition, there are logistical difficulties in collecting data and observing the behavior of animals that may spend significant amounts of time underwater. This is particularly true for deep-diving beaked whales, where the likelihood of detecting a whale at the surface in normal conditions may only be one in a hundred, according to Barlow and Gisiner (2006). All militaries need to commit to long-term surveillance monitoring, as well as impact monitoring.

For numerous reasons, collecting data in the marine environment is logistically much more difficult, and more expensive, than in the terrestrial environment (Norse and Crowder, 2005). For example, for 60 years no one noticed the extinction of a limpet species (*Lottia alveus*), even though the area it inhabited was studded with marine laboratories and stations (Carlton et al., 1991). Perrin's beaked whale (*Mesoplodon perrini*; Dalebout, 2002) was only discovered recently, and confirmed sightings of a living individual have yet to be made in the wild, despite the species inhabiting the waters off California, one of the most surveyed regions in the US and the world, with probably one of the greatest densities of marine mammal biologists in the country.

Because of the high degree of variability and uncertainty in cetacean data, the ability to detect trends is very limited (Gerrodette, 1987; Taylor and Gerrodette, 1993; Taylor et al., 2007), even for well-studied species and populations. It can take a decade or more to detect a decline in the best studied dolphin populations (Wilson et al., 1999; Thompson et al., 2000). Scientific uncertainty is a major problem for assessing cetacean conservation status (Parsons et al., 2015). However, lack of data and effort for beaked whales, coupled with difficulties in studying them, makes discerning their conservation status particularly difficult (Parsons, 2016). The percentage of precipitous declines that would not be detected was 90% for beaked whales (where a precipitous decline is a 50% decrease in abundance in 15 years, at which point a stock could be legally classified as "depleted" under the U.S. Marine Mammal Protection Act) (Taylor et al., 2007). Even where declines in marine mammal populations have been identified, the ultimate cause of declines can sometimes be difficult to determine due to a wide range of subtle contributing factors (Merrick et al., 1987; Alverson, 1992; Marmontel et al., 1996). The difficulty with monitoring the effects of anthropogenic impacts on cetaceans and the huge level of uncertainty involved have been noted as key issues that need to be addressed via scientific research, in order to better conserve, manage and protect cetaceans (Agardy et al., 2007; Dolman, 2007; Dolman and Jasny, 2015; Parsons et al., 2015; Parsons, 2016).

The importance of not delaying conservation action when a concern exists, but scientific data and analysis have not incontrovertibly established the threat exists, i.e., "the precautionary principle," has been enshrined in a number of international laws (Hey, 1991), including the 1992 Convention on Biological Diversity (Principle 15 of the so-called "Rio Summit"). Because of this level of uncertainty and difficulty in establishing beyond a reasonable doubt trends and threats in cetacean populations, it has been argued that in order to effectively conserve and manage populations one must be precautionary, as otherwise catastrophic declines in cetacean populations could occur before science catches up with the

problem (Mayer and Simmonds, 1996; Parsons et al., 2010, 2015; Parsons, 2016). It may be a long time before technology and methods are easily available to answer the many still unanswered questions about the exact nature and degree of the impacts of sound on cetaceans, especially when we know that many of the mitigation measures in place for protecting cetaceans against the impacts of sound are untested "best guesses" or, indeed, are known to be ineffectual (Parsons et al., 2009). Therefore, it is essential that as precautionary and conservative an approach to management is taken as possible with respect to the effects of military sonar on cetaceans. Although there is now a better idea of the scale and range of species that are affected, and the means by which strandings might occur and possibly the levels of sound that are most harmful, there are still many unknowns. Management of cetaceans needs to be precautionary because of these large number of unknowns, and at present this is mostly not the case. As Simmonds et al. (2014) and Erbe et al. (forthcoming) note, the science about the impacts of underwater noise on marine mammals is advancing, but management is lagging behind.

Many militaries have committed to investigate and mitigate their activities to protect marine mammals. However, there is an additional need for militaries to commit to conducting adequate baseline monitoring in areas where exercises routinely occur, to understand and to plan better to avoid deaths and, more importantly, to avoid behavioral impacts at appropriate ranges, and to mitigate accordingly. There is also a need for governments to develop criteria for assessing—and to commit to independently and thoroughly investigate—all atypical mass strandings in future.

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Spatial and Temporal Variation in the Acoustic Habitat of Bottlenose Dolphins (*Tursiops aduncus*) within a Highly Urbanized Estuary

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There is growing awareness of underwater noise in a variety of marine habitats, and how such noise may adversely affect marine species. This is of particular concern for acoustically-specialized species, such as dolphins. In order to ascertain the potential impacts of anthropogenic noise on these animals, baseline information is required for defining the soundscape of dolphin habitats. The Swan-Canning River system in Western Australia flows through the city of Perth, and experiences numerous anthropogenic activities. Despite this, the river system is home to a community of Indo-Pacific bottlenose dolphins (Tursiops aduncus). To provide a baseline soundscape description of dolphin habitat, over 11,600 h of acoustic data were analyzed from five sites within the Swan River (from Fremantle Inner Harbor to 20 km upstream) across an 8-year period. Multiple sound sources were recorded at these sites, including: snapping shrimp; fishes; dolphins; pile-driving; bridge and road traffic; and vessel traffic. The two most prevalent sound sources, vessel traffic and snapping shrimp, likely have very different effects on dolphin communication with the former expected to be more disruptive. Sites were characteristic in their prominent sound sources, showing clear among-site variations, with some sites being "noisier" than others based on broadband noise levels, octave-band noise levels, and power spectrum density percentiles. Perth Waters had the highest broadband noise (10-11 kHz; median 113 dB re 1 μPa rms), whilst Heirisson Island was quietest (median 100 dB re 1 μPa rms). Generalized estimating equations identified variation in broadband noise levels within sites at a fine temporal scale, although sites differed in the significance of temporal variables. At Mosman Bay, a long-term dataset spanning eight years highlighted inter-annual variation in broadband noise levels, but no overall upwards or downwards trend over time. Acoustic habitats of the Swan River displayed significant variations at a variety of temporal and spatial scales throughout areas frequented by the local dolphin community. Such variations should be quantified when assessing dolphin acoustic habitat as they may provide significant clues to dolphin behavior.

Keywords: underwater soundscape, spatio-temporal variation, anthropogenic noise, Indo-Pacific bottlenose dolphins

INTRODUCTION

Marine habitats are characterized by a unique combination of topographic structures, environmental conditions, and species compositions. These features contribute, either directly or indirectly, to the acoustic conditions of a particular environment as abiotic (e.g., wind, waves, precipitation, ice break-up, earthquakes) and biotic (e.g., crustaceans, fishes, marine mammals) sound sources. Habitats with human activities also have the added contribution of anthropogenic sound sources. As a result, the "soundscape" of any particular habitat varies in space and time depending on the prevalence of the sound sources within it (Krause, 2008; Pijanowski et al., 2011).

The distribution and occurrence of marine species is often related to physical features, such as depth, seafloor slope, or proximity to shore (Forney, 2000; Cañadas et al., 2002; Elwen and Best, 2004; Elwen et al., 2006). In other cases, species occurrence may be linked with more transient environmental variables, such as sea surface temperature, salinity, or primary productivity (Forney, 2000; Azzellino et al., 2008; Mannocci et al., 2014). Given the acute attenuation of light in water, many marine organisms rely on acoustics to investigate their environment (Nybakken and Bertness, 2005). As a result, introduced anthropogenic underwater noise has been increasingly recognized to act as a chronic, environmental stressor, which can affect both individual animals and ecosystem linkages (Weilgart, 2007; Hatch and Fristrup, 2009; Erbe, 2010; Boyd et al., 2011; Wright et al., 2011; Erbe et al., 2014; Finneran, 2015; Williams et al., 2015b). Thus, for acousticallyspecialized fauna, species occurrence may also be influenced by the soundscape of a marine habitat.

Of the acoustically-specialized marine fauna, cetaceans show some of the most elaborate and extreme adaptations for auditory perception and sound production underwater (Tyack and Miller, 2002). Using sound allows these animals to overcome the challenges of limited vision to fulfill a series of vital processes, such as orientation, communication, and foraging. However, these auditory adaptations also make cetaceans especially susceptible to the impacts of anthropogenic noise. The potential effects of underwater noise on cetaceans are widely recognized, ranging from minimal short-term effects to severe long-term effects (Richardson et al., 1995; Nowacek et al., 2007; Southall et al., 2007; Tyack, 2008). At low levels often corresponding with long ranges from the source, anthropogenic noise may be merely detectable by marine mammals. At higher levels, noise may interfere with animal communication and acoustic signal detection, or cause displacement, behavioral disturbance or induce stress. In extreme cases, acoustic exposure might even lead to hearing loss or physical injury (Erbe, 2012).

Coastal areas are among those marine habitats most at risk from human activities (McIntyre, 1999; Moore, 1999). As a result, coastal species—such as bottlenose dolphins (*Tursiops* sp.)—are among those marine fauna most vulnerable to anthropogenic threats (Thompson et al., 2000; DeMaster et al., 2001). In coastal habitats, the most ubiquitous source of anthropogenic underwater noise is vessel traffic, which has resulted in numerous dolphin behavioral response studies. Results have found evidence

of physical and acoustical changes to dolphin behavior, such as alterations to inter-breath intervals, inter-animal distances, movement patterns, activity states, whistle duration or rates, and frequency shifts in whistle characteristics, among others (Hastie et al., 2003; Buckstaff, 2004; Bejder et al., 2006; Lusseau, 2006; Nowacek et al., 2007; Weilgart, 2007; Ellison et al., 2012; Steckenreuter et al., 2012; New et al., 2013; Pirotta et al., 2015; Heiler et al., 2016). Significant changes to foraging success or energy demands (from altered movement, behavior or vocal production patterns) could also affect individual health, reproductive rates, or even long-term population survival. This is of particular concern for small dolphin communities, which tend to exhibit naturally low reproductive rates (Wilson et al., 1999; Ross, 2006). Therefore, knowledge regarding the response of dolphins to vessel traffic is of relevance to managers regulating activities in coastal areas.

However, in order to ascertain the potential impacts of anthropogenic noise on cetacean distribution, population dynamics, and behavior, there is first a requirement for baseline information defining the soundscape of cetacean habitats. Such baseline studies involve describing the habitat in terms of prominent sound sources, levels of acoustic energy in particular frequency bands, and patterns of ambient and anthropogenic noise (e.g., Parks et al., 2009; Erbe et al., 2014; Rice et al., 2014; Guan et al., 2015). Once identified, the acoustic characteristics of critical cetacean habitats can be further examined to determine the potential impact of man-made noise. Such studies can go on to inform management decisions regarding human-use of these areas, and determine whether conservation efforts are best directed toward "fixing" noisy habitats or preserving the remaining quiet areas (Erbe et al., 2014; Williams et al., 2014, 2015a). Given that the underwater soundscape contains sounds driven by weather conditions, environmental variables, and the presence of both marine fauna and human activities, it follows that an acoustic habitat will not be static in its composition. Consequently, whilst generalizations may be made about the acoustic characteristics of some underwater environments, many marine habitats will also display spatial and temporal variability in their acoustic components (Parks et al., 2009; Radford et al., 2010; McWilliam and Hawkins, 2013; Rice et al., 2014; Erbe et al., 2015; Guan et al., 2015; Marley et al., 2016a). Thus, to understand the role acoustic characteristics may play in driving the habitat use of cetaceans, there is a need to quantify the marine soundscape over large areas and across long periods of

The Swan River is an estuarine river system flowing through the Western Australian state capital of Perth. It is joined in its middle reaches by the Canning River, and together these rivers form an extensive system with a combined shoreline of approximately 300 km length. The Swan River estuary has a mean depth of 6 m and covers a surface area of approximately 31 km² (Robson et al., 2008). The system is composed of three distinct regions: an entrance channel at the river mouth; several shallow basins in the middle reaches of the river; and the riverine upper reaches. Despite transiting through a major metropolitan area (>1.4 million people), the Swan-Canning River is home to a small resident community of approximately 18

adult bottlenose dolphins (*T. aduncus*), plus juveniles and calves (Chabanne et al., 2012; SRT, 2015). The dolphins show daily use of this river system and high site fidelity (Chabanne et al., 2012). Research investigating the spatial and temporal patterns of dolphin occurrence within the rivers has shown that animals are distributed heterogeneously, with certain areas experiencing higher numbers of dolphin sightings than others (Moiler, 2008; Beidatsch, 2012; Marley et al., 2016b). In particular, the Fremantle Inner Harbor has been identified as a seasonal "hotspot" strongly linked with dolphin foraging behavior (Moiler, 2008). Other hotspots of dolphin sightings include Freshwater Bay, Melville Waters, Matilda Bay and Canning Bridge, located within the shallow basins region (Moiler, 2008; Beidatsch, 2012). Yet the dolphins are also sighted throughout the rest of the river system, with their range extending to the upper reaches of both the Swan and Canning Rivers (Beidatsch, 2012; SRT, 2015).

Like many urban estuaries, this river system experiences a range of environmental stressors. For example, in the past the Swan River has suffered from toxic algal blooms, nutrient enrichment, anoxia, pollution, introduced and invasive species, coastal flooding, and habitat modification (Rate et al., 2000; Robson and Hamilton, 2003; Morgan et al., 2004; Gosbell and Clemens, 2006; Eliot, 2012; Smale and Childs, 2012; Adolf et al., 2015; Hourston et al., 2015). These stressors were highlighted by the deaths of six dolphins within the river in 2009, which was hypothesized to be the result of a lowered immune system from multiple pressures, such as contaminant exposure and human activities (Holyoake et al., 2010). Parts of the Swan River have been shown to receive high levels of human activities. Visual monitoring at Perth Waters and the Fremantle Inner Harbor has revealed high levels of vessel traffic engaged in a range of activities (Marley et al., 2016b). Acoustic monitoring at The Narrows—a site mid-way along the Swan River in the Perth Waters area revealed that vessel noise was present in approximately 52% of hourly underwater recordings across a six-week period (Marley et al., 2016a). Similarly, the Fremantle Inner Harbor has been found to contain various sources of anthropogenic noise, including: vessel traffic, train and vehicle traffic passing over a nearby bridge; machinery noise; and wharf construction (Salgado Kent et al., 2012). As underwater noise levels and characteristics increasingly become considered as an indicator of habitat quality, there is a need to characterize the soundscape of the Swan-Canning river system with regard to its bottlenose dolphin population and anthropogenic activities. While past studies have highlighted the variation in soundscape at specific locations (Salgado Kent et al., 2012; Marley et al., 2016a), they have not described how these change in time and over a broader spatial range.

This study aims to examine spatial and temporal variability in the soundscape of the Swan River. Acoustic data collected from five locations along the river across 8 years were used to: (1) identify and compare prominent sound sources defining each site, (2) compare the spatial variability in soundscapes at four locations in the Swan River, (3) identify significant temporal scales (hourly, daily, monthly) of variability within the four sites, (4) describe long-term variability in the soundscape at peak

vessel traffic periods (using one exemplary site), and (5) relate prominent sound sources and their spatio-temporal variability to dolphin communication. In particular, the prevalence of vessel noise within the river system was used to determine whether some sites are "noisier" than others and thus have a potential to affect how dolphins use these habitats.

METHODS

The Swan-Canning estuary is located along the Western Australian coast. Five locations within the estuary over a distance of 20 km were selected for collecting acoustic data (Figure 1). From west to east, these locations were: Fremantle Inner Harbor (in the lower reaches of the river); Mosman Bay (middle reaches); Matilda Bay (middle reaches); Perth Waters (middle reaches); and Heirisson Island (upper reaches). These five study sites comprise a mixture of dolphin sighting hotspots and areas of human activity along the lower, middle, and upper reaches of the Swan River.

The Fremantle Inner Harbor is part of the state's biggest general cargo port and Australia's fourth largest container port (http://www.fremantleports.com.au), experiencing high levels of vessel traffic from commercial and recreational sources (Marley et al., 2016b). However, it has also been identified as a dolphin sighting hotspot, with animals reportedly spending several hours foraging within the Inner Harbor, regardless of vessel densities (Moiler, 2008; Marley et al., 2016b).

Mosman Bay is up-river of the narrow entrance channel at the river mouth. A long tidal sandbar stretches across from the opposite bank, funneling water flow, vessel traffic, and animals as they move down from the wide, shallow basins of the middle river reaches into the narrow, cliff-lined lower reaches. Three water ski areas line the periphery of this area with several boat pens located at the northern side, and the main Swan River ferry route passing through the middle of the bay. Dolphins transit through this site, with opportunistic foraging occurring around the boat pens. Mosman Bay has been identified as a spawning site for mulloway (*Argyrosomus japonicus*; Farmer et al., 2005). Consequently, this is the site of a long-term fish acoustic monitoring study for the species, which exhibits characteristic spawning-related vocalizations of high source level (Parsons et al., 2013).

Matilda Bay is a dolphin sighting hotspot, which is primarily used for foraging (Moiler, 2008). This small bay has a boat ramp and series of boat pens located on the north-eastern shore, and is adjacent to the main ferry route utilizing the Swan River. The southern river shore opposite Matilda Bay is used as a personal watercraft freestyle area.

Perth Waters comprises a wide (ca. 1.5 km), shallow basin within the middle reaches of the Swan River. It has not been identified as a dolphin hotspot; however, animals traveling between the middle and upper reaches of the Swan River must transit through this area (Marley et al., 2016b). This site contains the Barrack Street ferry terminal, and is also used by recreational boaters and crab fishermen. Additionally, at the time of this study, Elizabeth Quay (http://www.mra.wa.gov. au/projects-and-places/elizabeth-quay) was under development, involving construction and dredging activities.

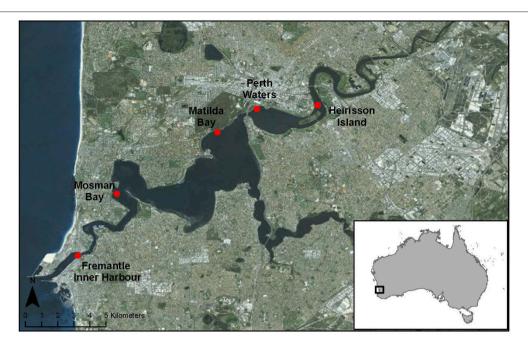


FIGURE 1 | Map of the Swan River with the five acoustic monitoring sites indicated.

Finally, Heirisson Island marks the beginning of the riverine upper reaches of the Swan River, characterized by a narrowing of the river as it winds through the Perth Hills. Although not a dolphin hotspot, animals are regularly sighted in the upper reaches of the Swan River. Heirisson Island experiences vessel traffic from both recreational boats and tourism ferries, and is also adjacent to a seasonally-used powerboat racecourse.

Data Collection

A total of 13 underwater acoustic recorders were used to collect soundscape data. Underwater acoustic recorders were one of two types. Low-frequency underwater sound recorders were custombuilt by Curtin University's Centre for Marine Science and Technology (CMST) and equipped with external hydrophones, entering the housing via a bulkhead connector to an impedance matching pre-amplifier with 20 dB gain. Digitized recordings (16 bit) were written on a flash card and, when full, to a hard disk in the logger. High-frequency recorders were assembled at CMST, using the same pre-amplifier as in the low-frequency recorders, and a programmable 16-bit data acquisition board made by Wildlife Acoustics Inc. Digitized recordings were written to four 128 GB SD cards. Both recorder types were calibrated by applying white noise of known power spectral density. High-pass filters (8 Hz cut-off) were employed to filter out high levels of lowfrequency noise, enhancing the dynamic range of the recorder at the frequencies of interest.

The acoustic recorders were placed on the seabed during all deployments. The recorder was connected to a weighted ground line leading to a main weight; as there was no surface line, the authors grappled for the recorder during recovery. The exception to this was the Fremantle Inner Harbor recorder,

which was deployed from a small jetty and tied off by two surface lines. Recorders were deployed on the riverbed for several weeks (Table 1). This allowed for temporal variations in the acoustic environment to be documented over hours, days and weeks, thus giving a representative insight into the acoustic conditions of each deployment over the temporal scales at which soundscape variations are likely to occur (Parsons et al., 2016). In some cases, multiple deployments within the same site were achievable, allowing longer-term measurements of underwater noise (Figure 1; Table 1). The specific recording dates, locations, settings and duty cycles used for each deployment are summarized in Table 1, along with hydrophone sensitivities. Mosman Bay deployments typically recorded for 5 of every 15 min at a sampling frequency of 6 kHz; however, two deployments were at 4 and 5 kHz. The Heirisson Island deployment was part of a separate study targeting high-frequency vessel noise, and so was set to record 40 of every 43 min at a sampling frequency of 96 kHz. The remaining deployments all recorded 10 of every 15 min, with sampling frequencies of either 22 kHz (Matilda Bay, Perth Waters first and second deployments) or 96 kHz (Perth Waters third deployment and Fremantle Inner Harbor). This variation in sampling frequency was a result of whether a low- or highfrequency acoustic recorder was used, which was dependent upon equipment availability.

The study was carried out in accordance with the recommendations of the National Health and Medical Research Council Australia code for the care and use of animals for scientific purposes 8th Edition (2013). The protocol was approved by the Curtin University Animal Ethics Committee (Approval Number AEC-2013-28).

TABLE 1 | Summary of acoustic data collection within the Swan River.

ID	Location	Start date	End date	Days total	Sampling frequency (kHz)	Total gain (dB)	Duty cycle	Hydrophone sensitivity (dB re 1 μPa/V)
MB	Matilda bay	27/11/2013	19/01/2014	53	22	40	10 of every 15 min	-194.0
PW1	Perth waters	11/01/2014	05/03/2014	53	22	40	10 of every 15 min	-197.7
PW2	Perth waters	05/06/2014	28/07/2014	53	22	40	10 of every 15 min	-197.7
HI	Heirisson island	13/10/2014	27/10/2014	14	96	44	40 of every 43 min	-202.8
PW3	Perth waters	17/02/2015	09/04/2015	51	96	44	10 of every 15 min	-202.8
FIH	Fremantle inner harbor	30/04/2015	14/06/2015	45	96	44	10 of every 15 min	-202.8
Mos07	Mosman bay	01/01/2007	01/02/2007	31	4	40	5 of every 15 min	-197.7
Mos09	Mosman bay	01/01/2009	01/02/2009	31	6	40	5 of every 15 min	-196.8
Mos10	Mosman bay	01/01/2010	01/02/2010	31	6	40	5 of every 15 min	-196.8
Mos11	Mosman bay	01/01/2011	01/02/2011	31	5	40	5 of every 15 min	-196.0
Mos12	Mosman bay	01/01/2012	01/02/2012	31	6	40	5 of every 15 min	-197.0
Mos13	Mosman bay	01/01/2013	01/02/2013	31	6	40	5 of every 15 min	-197.5
Mos15	Mosman bay	01/01/2015	01/02/2015	31	6	40	5 of every 15 min	-197.7

Acoustic Analyses

Data were first reviewed in Matlab (Version R2013a, The MathWorks Inc.) using the toolbox CHORUS (Gavrilov and Parsons, 2014). This allowed prominent sound sources for each deployment to be identified. Protocols for further processing of acoustic data broadly followed the methodology of Marley et al. (2016a) and were applied to data collected from each deployment. Recordings were analyzed in Matlab, and were first Fourier transformed in 1 s windows, producing a time series of power spectral density (PSD). Cable noise, where it existed, was identified as brief broadband spikes and the corresponding 1s windows were discarded from further analysis. Due to the range of sampling frequencies employed, data were down-sampled to correspond with the lowest sampling frequency of 22 kHz. The exception to this were recorders used in Mosman Bay, which had an original sampling frequency of 4-6 kHz (here, down-sampled to 4 kHz), as these were part of a separate study targeting fish calls. Hence, these data were not used in the spatial comparison, and instead were utilized for a long-term temporal overview of soundscape changes in the Swan River.

For all datasets, the PSD was averaged into 10 s windows and the first 10 s PSD average of each minute was plotted in a weekly spectrogram (Mon–Sun). These spectrograms allowed initial visual inspection and comparison of the data. The first 10 s PSD average of each minute was used to compute PSD percentile plots. To reduce computational effort, the PSD was further averaged into a series of adjacent frequency bands, each 10 Hz wide. The *n*th percentile of each plot gives the level that was exceeded n% of the time, with the 50th percentile representing the median. Thus, these plots illustrate the statistical variability of underwater sound for each deployment across the study period, allowing visual comparison of acoustic power vs. frequency, both within and between sites.

The 1 Hz PSDs of underwater sound were converted to linear units, averaged over 10 min of every acoustic recording and integrated into adjacent octave band levels (OBLs). This resulted in time series of noise levels in each octave band, with one

sample corresponding with each acoustic recording, allowing comparison of the noise levels in each OBL across both sites and years. Dolphin whistles in the Fremantle Inner Harbor have been reported to range between 1.1 and 18.4 kHz, with a minimum frequency of 1.1 to 9.0 kHz (Ward et al., 2016). Given this frequency range, it is possible to consider which sound sources may overlap with dolphin communication frequencies and identify which river sites may pose concern given noise levels in their upper OBLs.

Spatial and Temporal Variation

For each recording, broadband noise levels (NL_BB) were calculated as the root-mean-square sound pressure level over the duration of each file. These data were used to compare spatial and temporal variations in NL_BB, both between and within sites. Spatial comparisons were made across all sites except for Mosman Bay, as this dataset was down-sampled at a lower frequency than the other sites. Spatial variation was examined by conducting a Kruskal-Wallis test on the Fremantle, Matilda Bay, Perth Waters, and Heirisson Island datasets. Wilcoxon-Mann-Whitney Tests identified the source of any differences, and the power of these tests was assessed through *post hoc* tests in G*Power (Vr 3.1.9.2).

To examine short-term temporal variation within sites, Generalized Estimating Equations (GEEs) were applied to the Fremantle, Matilda Bay, Perth Waters, and Heirisson Island datasets. For the purposes of these analyses, data were downsampled to select only one recording per hour. Temporal variation in NL_BB was examined for hour of day ("Hour"), day type (weekday or weekend; "DayType"), and month of the year ("Month"). GEEs were deemed suitable for these analyses as they account for temporal autocorrelation whilst identifying temporal variation, thus allowing the use of repeated measures data (Zuur et al., 2009; Photopoulou et al., 2011; Bailey et al., 2013). Modeling followed the methods of Marley et al. (2016a), with DayType and Month included as factors. However, as time of day forms part of a cycle, the variable Hour (h) was converted

to a cyclical covariate using sine and cosine vectors, termed H_s and H_c respectively (Zuur et al., 2009; Bailey et al., 2013):

$$H_s = \sin\left(\frac{2\pi \times h}{24}\right)$$
$$H_c = \cos\left(\frac{2\pi \times h}{24}\right)$$

This allowed hours at the start and end of the day to be considered close to each other (e.g., 23:00 and 01:00 h). A similar approach has been undertaken by other studies to include circular variables as model terms (Griffin and Griffin, 2003; Bailey et al., 2009, 2010, 2013; Pirotta et al., 2013; De Boer et al., 2014; Marley et al., 2016a). This approach was not applied to Month due to datasets generally being limited to only a few months.

The GEE model used a gamma error distribution with a log-link function. Gamma distributions are appropriate for continuous response variables which have positive values (Zuur et al., 2009). Variance Inflation Factors (VIFs; Zuur et al., 2010) were calculated, but revealed no collinear variables in the model. However, a Runs Test indicated that there was an issue with correlation in the model residuals (p < 0.001); therefore, a blocking structure was selected to model this correlation.

GEEs account for temporal autocorrelation via within-cluster correlations to increase the estimation efficiency, thus allowing maximum use of sequential or repeated measures data (Zuur et al., 2009; Bailey et al., 2013). To select clusters for the model blocks (ID), the autocorrelation of the model residuals by ID was plotted to check for a decline in correlation over time. During each separate "Date" (a sequential value beginning on Day 1 of sampling and ending on the final day of sampling), the correlation of observations made hourly declined to approximately zero within a 24-h period. Thus, separate days were treated as independent, and so "Date" was selected to define clusters of data points within which residuals were allowed to be autocorrelated. Given that the data were serially correlated and that GEEs are robust in providing consistent estimates of mean parameters even when the correlation structure is mis-specified, an AR-1 correlation structure was selected as the most logical option for the model.

Selection of the best model was assessed via a quasi-likelihood criterion (QIC; Pan, 2001) and model fit was assessed by plotting observed vs. fitted values and fitted values vs. scaled Pearson's residuals. Once the final model was selected, repeated Wald's tests were used to assess the significance of each temporal variable, and partial residual plots of significant terms were created.

Long-term temporal variation of high vessel traffic periods was assessed for Mosman Bay. This site included 7 years of data collected between 2006 and 2015. Although several months of data were recorded each year, only January was retained because it was consistently captured each year and also represents the austral summer, when high levels of anthropogenic activities were expected to occur. In January, daily mulloway choruses were recorded in the late evening. To explore sources associated with human "rush hour" as opposed to peak mulloway chorusing,

only acoustic data from the morning (06:00–12:00 h) were used (Marley et al., 2016a,b). A Kruskal-Wallis test was applied to this multi-year dataset to identify variations between years; the source of differences were then identified by Wilcoxon-Mann-Whitney Tests. Statistical power was again assessed using G*Power (Vr 3.1.9.2).

All statistical analyses were conducted in R (R Core Team, 2015) with the aid of the *geepack* (Yan, 2002; Yan and Fine, 2004; Højsgaard et al., 2006), *MESS* (Ekstrom, 2014), *MRSea* (Scott-Hayward et al., 2014), and *stats* (R Core Team, 2015) packages.

RESULTS

A total of over 11,600 h of acoustic data were collected during 14 deployments at five sites within the Swan River. Of these, approximately 6,450 h from seven deployments at four sites were analyzed for spatial and short-term temporal comparisons, whilst 5,200 h from seven annual deployments at Mosman Bay were analyzed to assess long-term temporal variation.

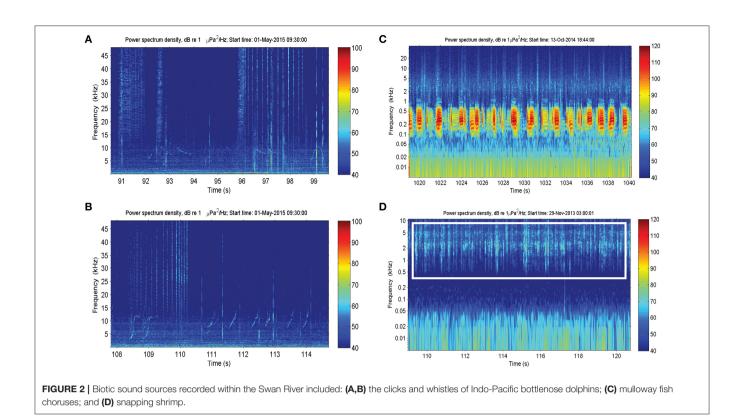
Prominent Sound Sources

Prominent sounds recorded in this study came from biotic and anthropogenic sources. These included: snapping shrimp; fish choruses; dolphin clicks and whistles; impulse pile-driving; trains and/or vehicles passing over nearby bridges; and vessel traffic.

Dolphin sounds were most abundant in the Fremantle Inner Harbor, where both whistles and echolocation clicks were frequently recorded (Figures 2A,B). While sounds likely produced by fish occurred at all locations, fish choruses were only observed at the Heirisson Island site (Figure 2C), although they are known to occur in other areas of the river such as Blackwall Reach and Mosman Bay (Parsons et al., 2013). Snapping shrimp were observed in all locations to varying degrees (e.g., Figure 2D).

There were also a number of additional anthropogenic sounds. Pile-driving was heard at Heirisson Island, due to adjacent shore-based construction works (Figure 3A). The sound of pile-driving recorded in water consists of series of sharp pulses every few seconds (see Erbe, 2009, for pile driving recorded in equally shallow water). High-frequency "blips" thought to originate from vessel echo-sounders were observed in Fremantle Inner Harbor (Figure 3B). In Matilda Bay, series of very low-frequency pulses were observed (Figure 3C). Similar pulses have been previously reported at the neighboring Narrows Bridge site (Marley et al., 2016a), where they were hypothesized to be the result of train or vehicle traffic crossing the bridge. The Matilda Bay deployment site was adjacent to a busy main road, which may be the source of this sound.

One of the most striking features was the variability in sounds produced by vessel traffic (Figure 4). In the Fremantle Inner Harbor, there was near-continuous background noise from transiting vessels and idling engines, in addition to sounds from near-passing vessels (Figure 4A). In other areas, vessel sounds included steady tones (Figures 4B,C), series of engine revs increasing with engine rotations per minute (Figure 4D;



Erbe et al., 2016a), undulating tones with many harmonics from jet skis (**Figure 4E**; Erbe, 2013), and bands across the low and high frequencies (**Figures 4B,F**, respectively). Considering all the variations observed, vessel noise has the potential to range from 5 to over 20 kHz. The highest frequency sounds were observed in the presence of small powerboats engaged in high-speed races near Heirisson Island (**Figure 4F**).

Spatial Variation

Significant variation in NL_BB occurred among the four Swan River sites (Fremantle Inner Harbor, Matilda Bay, Perth Waters, and Heirisson Island; Kruskal-Wallis test $X^2=4,252.6$, df=3, p<0.001; **Figure 5**). All sites were significantly different from each other (Wilcoxon-Mann-Whitney Tests all had p<0.001). The effect size of site ranged from 0.12 to 1.15, achieving a power of 0.77–1.00. Median NL_BB's for the four Swan River sites were: Fremantle Inner Harbor 106 dB re 1 μ Pa; Matilda Bay 107 dB re 1 μ Pa; Perth Waters 113 dB re 1 μ Pa; and Heirisson Island 100 dB re 1 μ Pa.

The soundscapes of the four Swan River sites were further compared by investigating PSD percentile plots (**Figure 6**). The most obvious feature of the Fremantle Inner Harbor dataset was the presence of vessel traffic at 0.05–1 kHz. As a result of this sound source, the Fremantle site was only as quiet as other Swan River sites < 5% of the time. In addition to noise from vessel traffic, there was near-continuous background anthropogenic noise from Port operations, such as machinery and engine noise. Despite this site being located closest to the ocean and containing numerous structures for settlement, noise from snapping shrimp was not often detected. Shrimp clicks

were detected sporadically, and did not dominate the weekly spectrograms or PSD percentile plots (unlike at other sites, such as Matilda Bay or Perth Waters). Dolphin clicks and whistles were frequently present in manually reviewed acoustic files; however, these transient events did not cause any obvious spikes in PSD percentile plots.

The Matilda Bay acoustic measurements resulted in higher PSD levels in the lower frequencies (**Figure 6**), which corresponded with observed trends in the weekly spectrograms. Matilda Bay had the strongest prevalence of snapping shrimp of all sites. Numerous vessel transits were visible in the weekly spectrograms, particularly during the daytime; yet these did not form the same strong bands of vessel noise observed in the Fremantle Inner Harbor. Matilda Bay exhibited some of the quietest recorded ambient noise levels in the 100–1,000 Hz band.

Three deployments occurred at Perth Waters. The first and second deployments were similar in terms of overall noise levels in the lower frequencies, whilst in the higher frequencies the second and third deployments showed greater similarity (Figure 6). Spectrograms from this site also showed daily patterns of low-frequency noise bands were present, which were particularly prominent in the second deployment. Snapping shrimp noise was a strong feature at this site and was slightly louder than in Matilda Bay (Figure 6).

One deployment occurred at Heirisson Island, the most prominent feature of which was the presence of a fish chorus from 50 to 500 Hz (**Figure 6**). Snapping shrimp were minimally observed at this site, which is located in the upper, riverine reaches of the Swan River. At the higher frequencies (>1 kHz),

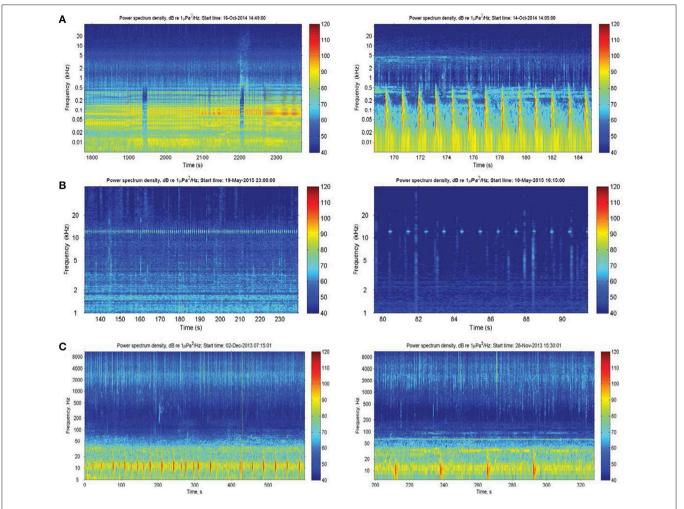


FIGURE 3 | Anthropogenic sound sources within the Swan River included: (A) pile-driving; (B) vessel echosounders; and (C) unknown series of low-frequency impulses thought to originate from vehicle traffic.

there is evidence of high-frequency vessel traffic approximately 1% of the time. This corresponds to high-speed powerboats, which occasionally race in this area during the austral summer months (**Figure 4F**). Powerboats also contributed to the frequency band of fish chorusing (**Figures 2C**, **4F**) but occurred temporally out of sync, with boats recorded during the day and fish at night.

When individual OBLs are considered, it is evident that some levels vary between sites more substantially than others (Figure 7). Levels at Matilda Bay and Perth Waters were higher than other sites in the OBL centered at 20 Hz, reflecting the presence of unidentified low-frequency anthropogenic sounds (Figure 3C). All sites were similar at the 40 Hz OBL. Fremantle Inner Harbor was highest at mid-range OBLs centered at 80, 160, 320, and 640 Hz, reflecting the high level of vessel traffic at this site. Levels at Heirisson Island also had a wide range across these OBLs, due to the presence of a fish chorus. Levels at Matilda Bay were highest in the OBLs centered at 1,280, 2,560, and 5,120 Hz, followed by the Perth Waters deployments. This energy reflected the prevalence of snapping shrimp at these sites.

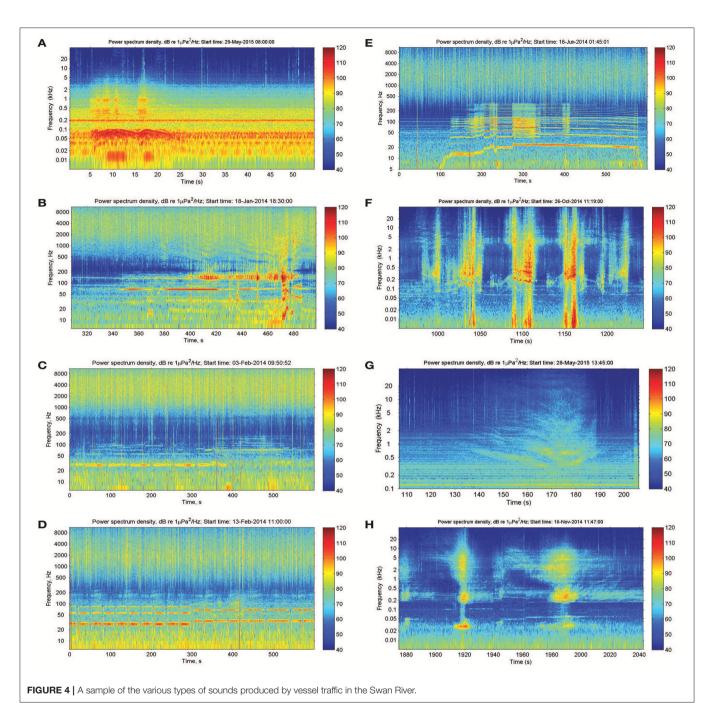
Short-Term Temporal Variation

There were significant temporal variations within each site based on GEE results (**Table 2**; **Figure 8**). NL_BB at Fremantle Inner Harbor varied by Hour ($X^2 = 55.848$, p < 0.001), DayType ($X^2 = 5.212$, p = 0.022), and Month ($X^2 = 8.301$, p < 0.001). Noise levels were typically higher during the day at this site, peaking at approximately 09:00 h. Weekday noise levels were higher than those of the weekend, and May was noisier than June.

The Matilda Bay GEE retained Hour ($X^2 = 59.07$, p < 0.001) and Month ($X^2 = 515.05$, p < 0.001) as significant variables. At this site, noise levels gradually increased throughout the day before peaking at 20:00 h. Noise levels increased over the austral summer months (November to January).

Perth Waters retained Hour ($X^2 = 11.64$, p < 0.001) and Month ($X^2 = 780.37$, p < 0.001) as significant variables. Noise levels sharply increased between 08:00 and 10:00 h then peaked between 19:00 and 21:00 h before decreasing overnight. April was the quietest month.

Heirisson Island only retained Hour ($X^2 = 223.739$, p < 0.001) as a significant variable. It showed a gradual increase in



noise levels throughout the day, then a sharp peak between 18:00 and 22:00 h; this period coincided with the evening fish chorus identified in the weekly spectrograms.

Long-Term Temporal Variation

In Mosman Bay, NL_BB measured in January differed among the seven years of measurement (Kruskal-Wallis $X^2 = 102.75$, df = 6, p < 0.001; **Figure 9**). NL_BB in 2010 was greatest, whilst NL_BB in 2013 and 2015 were most similar. The effect size of year ranged from 0.01 to 0.49, achieving a power of 0.06–1.00.

The Mosman Bay PSD percentile plots and OBLs show that the soundscape at this site was very similar over the years (**Figures 10, 11**). Most noise occurred in the 70–300 Hz frequency band. Closer examination in the weekly spectrograms suggested the noise was produced by vessel traffic. About 5% of the time, noise in this band was above 100 dB re 1 $\mu Pa^2/Hz$ for all years considered. This noise only dropped to below 70 dB re 1 $\mu Pa^2/Hz$ less than 5% of the time. There was no trend of decreasing or increasing noise levels at any frequency over the 7-year period of acoustic monitoring in Mosman Bay.

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Relevance to Dolphin Communication

Dolphin whistles could be expected to overlap with OBLs centered at 1,280, 2,560, and 5,120 Hz (Figure 7). High mean

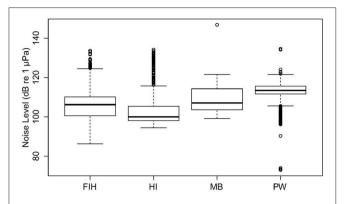


FIGURE 5 | Overall broadband noise levels (NL_BB) of four Swan River sites: Fremantle Inner Habour (FIH), Heirisson Island (HI), Matilda Bay (MB) and Perth Waters (PW).

noise levels at these OBLs were present at Matilda Bay and Perth Waters due to the presence of snapping shrimp. In comparison, Heirisson Island and the Fremantle Inner Harbor had lower mean values. However, the high variability of levels at the Fremantle Inner Harbor resulted in levels occasionally surpassing those of Matilda Bay and Perth Waters. Due to the relatively low levels of snapping shrimp noise in Fremantle Inner Harbor when compared to other study sites, these "noisiest" periods are likely attributable to high vessel traffic.

When considered as individual sound sources, snapping shrimp and vessel traffic produce noise across a wide frequency band that can overlap with dolphin whistles (Figures 2D, 4, respectively). However, the spectro-temporal structures of these sounds differ considerably. Whilst colonies of snapping shrimp produce frequent impulsive broadband clicks, each lasting a few milliseconds, vessels produce continuous broadband noise from propeller cavitation and long-lasting tonal sounds due to engine and propeller rotations (e.g., Erbe et al., 2016a). Propeller cavitation is a stochastic process, and the resulting power spectrum has characteristics of pink noise. Shrimp snaps, on the other hand, show a higher degree of comodulation across

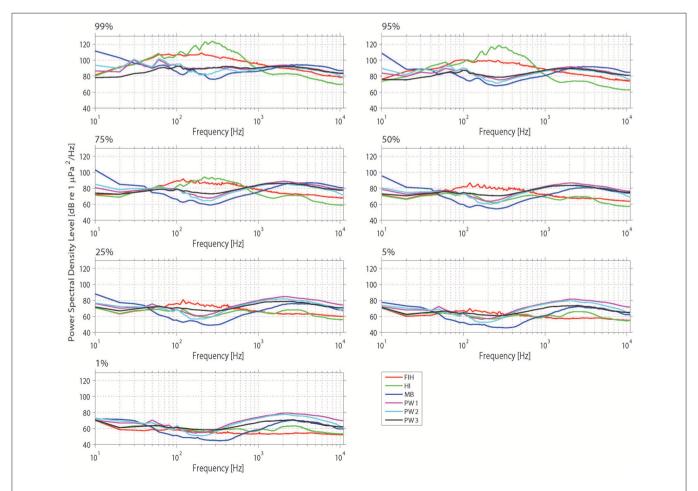


FIGURE 6 | Power spectrum density (PSD) percentiles (averaged into 10 Hz bands) from the four sites considered in the spatial analysis. The nth percentile gives the level that was exceeded n% of the time.

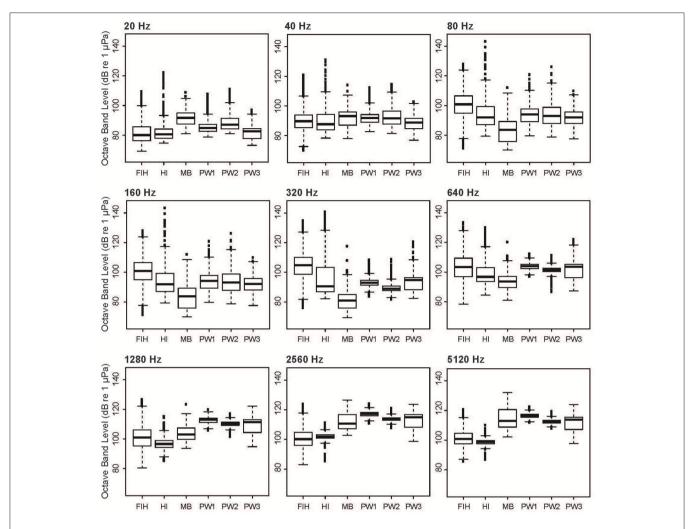


FIGURE 7 | Variation in selected octave-band levels between the four sites considered in the spatial analysis: Fremantle Inner Harbor (FIH); Heirisson Island (HI); Matilda Bay (MB); and Perth Waters (PW1-3).

frequency (Branstetter et al., 2013). It is the different temporal structures and comodulation degrees that will likely reduce the risk of acoustic masking by snapping shrimp over that of vessels (Erbe, 2008; Erbe et al., 2016b). In other words, high levels of vessel traffic in Fremantle Inner Harbor are more likely to mask dolphin whistles than high levels of snapping shrimp in Matilda Bay.

DISCUSSION

This study describes the acoustic habitat in the core range of the Swan River dolphin community, at varying spatial and temporal scales. Overall, there were two predominant sound sources which occurred at multiple sites: snapping shrimp and vessel traffic. From the acoustic perspective of the resident dolphin community, both of these sound sources overlap with the frequency range of dolphin whistles used for communication. However, whilst snapping shrimp sounds are brief, impulsive, repetitive, and their spectrum comodulated across multiple frequencies, the propeller cavitation noise

produced by vessel traffic is temporally continuous and spectrally not comodulated. Thus, the latter is more likely to interfere with dolphin whistles, particularly where multiple vessels are simultaneously contributing to the soundscape. Additionally, the number of sound sources identified and their changeability throughout the river system clearly illustrates the spatially variable acoustic environment experienced by this community of bottlenose dolphins. High within-site temporal variability observed over small and large temporal scales (hours to years) adds another layer of complexity to this acoustic environment.

In this study, the most ubiquitous sound source was vessel noise, which was present at all sites to some degree. The Swan River is known to be a site of high vessel traffic, used by vessels of numerous types engaged in a range of activities (Marley et al., 2016b). However, despite their prevalence, vessel sounds were not consistent. In fact, the extreme variation in vessel sounds was in marked contrast to the much lower variability in characteristics of prominent biotic sounds, such as shrimp snaps and fish calls, which were of comparatively predictable duration

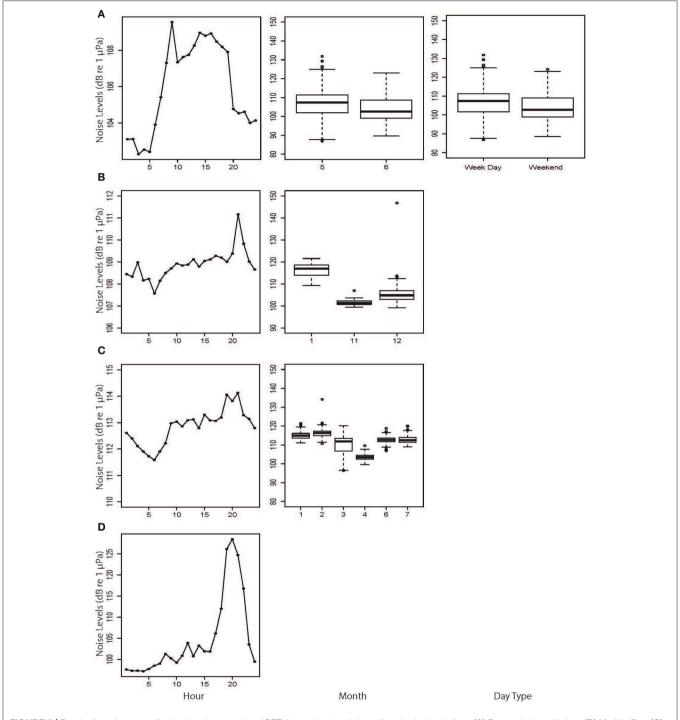


FIGURE 8 | Results from the generalized estimating equations (GEEs) based on hourly broadband noise levels from (A) Fremantle Inner Harbor, (B) Matilda Bay, (C) Perth Waters, and (D) Heirisson Island. Only significant explanatory variables are shown.

and frequency. The high variability in vessel acoustic features is a result of differences in vessel type, speed and behavior; the physical characteristics of the environment; and varying distances from the receiver (see Erbe, 2013; Erbe et al., 2016a for variability of underwater noise from jetskis and small boats with

outboard motors, which are the most common type of vessel in the Swan River). For example, some vessels produced bursts of low-frequency "revs" with relatively few harmonics, whilst others produced mid-frequency tonal sounds with several harmonics for a few minutes, and many dominated the entire frequency

TABLE 2 Summary of generalized estimating equation (GEE) models investigating temporal patterns within Swan River sites at the scale of Month, DayType (Weekday or Weekend), and Hour (a cyclical variable represented by *Hs* and *Hc*).

Site	Parameter	Coefficient estimate	Standard error	Wald	P	
FREMANT	LE INNER HARBOR (FIH)					
	Intercept	4.68328	0.00797	3.46e+05	<2e-16	**:
	Month6	-0.03107	0.01013	9.40e+00	0.022	**
	Weekend	-0.03078	0.01350	5.20e+00	0.0226	*
	H_S	-0.00659	0.00366	3.24e+00	0.0718	
	H _C	-0.02787	0.00373	5.59e+01	7.8e-14	***
MATILDA	BAY (MB)					
	Intercept	4.753883	0.005365	7.85e+05	<2e-16	***
	Month 11	-0.135897	0.005881	5.34e+02	<2e-16	***
	Month 12	-0.100433	0.006875	2.13e+02	<2e-16	***
	H_S	-0.005619	0.000736	5.83e+01	2.3e-14	***
	H _C	0.000249	0.000732	1.20e-01	0.73	
PERTH W	ATERS (PW)					
	Intercept	4.748788	0.002937	2.61e+06	<2e-16	***
	Month 2	0.007966	0.003615	4.85e+00	0.02758	*
	Month 3	-0.044825	0.007522	3.55e+01	2.5e-09	***
	Month 4	-0.109140	0.005000	4.76e+02	<2e-16	***
	Month 6	-0.028517	0.003224	7.83e+01	<2e-16	***
	Month 7	-0.028411	0.003263	7.58e+01	<2e-16	***
	Hs	-0.006599	0.000464	2.02e+02	<2e-16	***
	H _c	-0.002113	0.000619	1.16e+01	0.00065	***
HEIRISSO	N ISLAND (HI)					
	Intercept	4.64622	0.00506	8.42e+05	<2e-16	***
	H_S	-0.08831	0.00590	2.24e+02	<2e-16	***
	H _C	-0.00496	0.00628	6.30e-01	0.43	

Significance level: $\leq 0.001^{***}$; $\leq 0.01^{**}$; $\leq 0.05^{*}$.

band for the whole recording period of 10 min. Low-frequency sounds (centered at 35 Hz) have previously been recorded from passenger ferries operating in Perth Waters (Marley et al., 2016a). Some of these ferries also venture to other parts of the river, traveling past Heirisson Island, Perth Waters, Matilda Bay and the Fremantle Inner Harbor. Vessels are often observed milling in some parts of the river (e.g., Perth Waters), where they circle an area at low speeds for a prolonged period of time, and the engine may be stopped and started several times as the vessel moves between particular spots (Marley et al., 2016b). Such vessel behavior, which often coincides with fishing or crabbing activities, likely contributes to the low-frequency acoustic environment of the river system. The highest frequency vessel sounds were observed in the presence of small powerboats engaged in high-speed races near Heirisson Island. Such races can involve several competing powerboats at any one time, with the most powerful boats claimed to "regularly achieve over 170 kph" (http://www.wasbc.com.au). This race site is situated at the start of the upper riverine reaches of the Swan River system, where the river narrows to only 450 m wide. Given the high-frequency noise produced by these vessels and the narrow nature of the river in this area, there could be potential for displacement of dolphins whose communication whistles could be masked.

The wide array of sound characteristics—even from the same source type-contributed to the spatial and temporal variability of the acoustic environment experienced by this dolphin community. Each of the sites considered had its own characteristic combination of contributing sound sources. The minimum distance between any two of these sites is approximately 2.5 km, highlighting the site-specific nature of acoustic habitats within the same system. This agrees with findings of previous studies. Soundscape studies in New Zealand (Radford et al., 2010), Pacific Panama (Kennedy et al., 2010), the U.S. east coast (Rice et al., 2014) and Taiwan (Guan et al., 2015) have also noted site-specific sound fields at locations several kilometers apart, generally as the result of biotic or anthropogenic activities. Additionally, there was considerable temporal variation within sites. The Fremantle Inner Harbor was noisiest between the hours of 08:00-20:00 h, particularly on weekdays. The presence of both recreational vessel traffic and port activities at these times combine to increase average noise levels. Matilda Bay and Perth waters also displayed increased noise levels during the day, likely from vessel traffic. Noise levels at Heirisson Island only slightly increased during the day; instead, the 'noisiest' period occurred between the hours of 18:00-22:00 h as a result of the evening fish chorus. In the Swan River, the "noisiness" of each site also varied according to the frequency

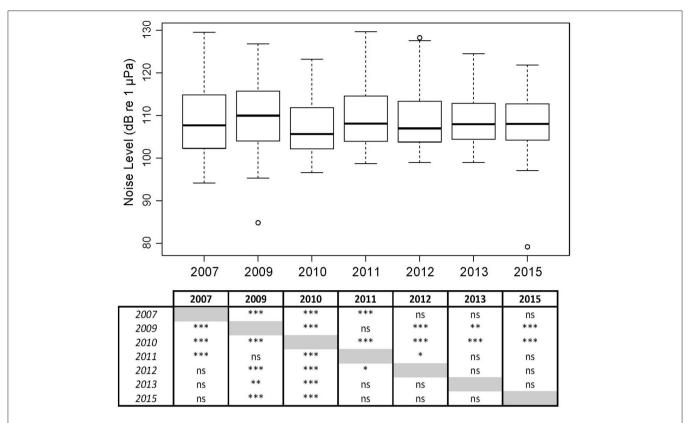


FIGURE 9 | Overall broadband noise levels (NL_BB) at Mosman Bay over the month of January across a 9-year period, including a table of significant differences as determined by Wilcoxon-Mann-Whitney Tests: ≤ 0.001 ***; ≤ 0.01 **; ≤ 0.05 *.

band considered; all sites were uniformly quieter in the lowest frequency OBLs, but at the mid- and upper-frequency OBLs there were considerable differences between sites. Overall, snapping shrimp sounds were prevalent at Matilda Bay and Perth Waters, whilst Heirisson Island displayed a strong fish chorus, and the Fremantle Inner Harbor was dominated by sounds from vessel traffic and port activities. In the OBLs centered on frequencies also utilized by dolphins, noise from vessel traffic and snapping shrimp caused the greatest variation.

How dolphins may respond to vessel traffic in the Swan River is still under investigation. A previous study on dolphin occupancy in response to vessel traffic found that despite similarities in vessel densities, dolphins showed differential use of two monitored sites within the river (Marley et al., 2016b). Fewer dolphin sightings were recorded at Perth Waters when vessel densities were high, whereas vessel traffic appeared to have no relationship with dolphin occupancy in the Fremantle Inner Harbor (Marley et al., 2016b). The acoustic data presented here from Perth Waters 1, Perth Waters 3 and the Fremantle Inner Harbor overlap temporally with the visual observations presented in Marley et al. (2016b). It can be clearly seen that dolphins are not experiencing the same acoustic environment at these two separate sites; yet dolphins remained present at the anthropogenically busiest, noisiest one. Future research investigating whether certain vessel characteristics (physical, behavioral or acoustical) elicit responses in Swan River dolphins beyond changes in animal occupancy would provide insight into finer scale responses. These responses could be physical behavioral changes such as alterations to swim speed, activity state, movement patterns, or acoustical behavioral changes such as variations in whistle frequency, repetition, or duration.

To determine the level at which dolphin communications are being masked at anthropogenically noisy sites (such as the Fremantle Inner Harbor), future work on source levels and transmission of whistles in the Swan River is required. In addition, data on how the structure of Swan River dolphin whistles may change in different contextual scenarios should be measured. The acoustic characteristics of whistles appear to vary between dolphin populations, in terms of frequency content, bandwidth, duration, extrema, steps and inflection points (Ding et al., 1995; May-Collado and Wartzok, 2008; Hawkins, 2010; Ward et al., 2016). These differences may be due to distance (e.g., separate vocal evolution, low exchange rates of individuals) or as a result of context (e.g., group composition, behavior). Variations in whistle characteristics could also reflect different environments in terms of physical or environmental characteristics, such as water depth, sediment type, salinity and/or temperature. Ambient noise is increasingly becoming considered an indicator of environmental quality, which could also influence features and use of dolphin whistles (Buckstaff,

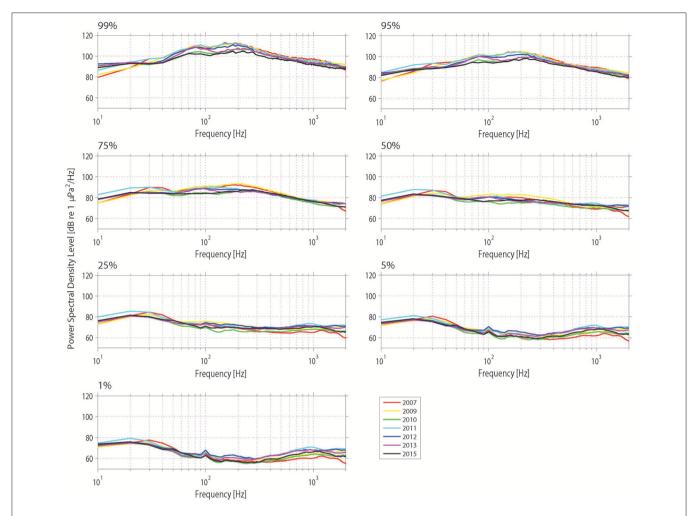
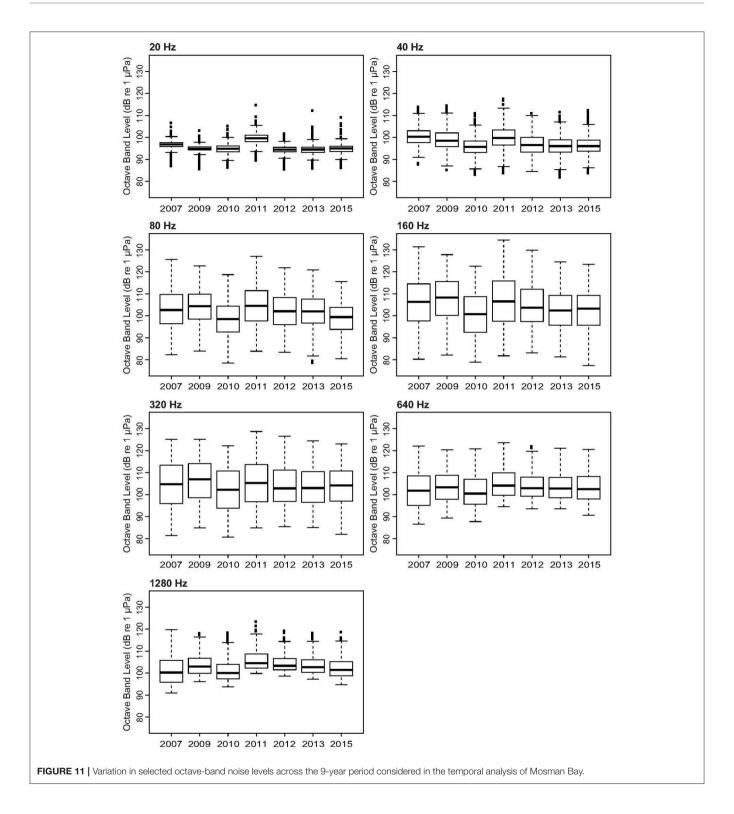


FIGURE 10 Power spectrum density (PSD) percentiles (averaged into 10 Hz bands) at Mosman Bay over the month of January across the 9-year period considered in the temporal analysis. The *n*th percentile gives the level that was exceeded n% of the time.

2004; Morisaka et al., 2005; Guerra et al., 2014; May-Collado and Quiñones-Lebrón, 2014; Heiler et al., 2016). Furthermore, different dolphin populations also appear to vary in the source levels of the whistles they produce (Jensen et al., 2012). To date, the only published analysis of the Swan River dolphins' repertoire describes the characteristics of whistles recorded in the Fremantle Inner Harbor and does not include source levels or contextual analysis (Ward et al., 2016). Additionally, little is known about the source levels of different sound types in the Swan River, particularly from anthropogenic activities. Anthropogenic noise has the potential to degrade habitat through a loss of "acoustic space." In areas which experience high levels of anthropogenic noise, habitat fragmentation may even occur if animals are unable or unwilling to transit through noisy areas in order to reach necessary habitat (Rice et al., 2014). Thus, it would be beneficial to document the structure and source levels of dolphin whistles and human activities at multiple sites throughout the Swan River to see if differences exist in "noisy" vs. "quiet" habitats.

The lack of any long-term increase or decrease in noise levels at Mosman Bay suggests a degree of temporal stability within this

site. There is growing concern regarding the increasing level of underwater noise in many coastal areas as a result of expanding anthropogenic activities. Mosman Bay is the site of a longterm acoustic monitoring study due to a prominent fish chorus associated with seasonally-breeding mulloway. To focus on potential increases in anthropogenic noise and avoid including the evening fish chorus in analyses, only data from 06:00 to 12:00 h were assessed. This period was expected to overlap with the morning vessel "rush hour" reported in other studies at different points in the river (Marley et al., 2016a,b), during a month of increased recreational time due to the austral summer holidays. Although years were not all the same, their average noise levels were all within 3 dB, displaying no overall upwards or downwards trend in yearly noise levels within Mosman Bay. The relatively low effect size associated with the majority of yearly comparisons suggests that, although significant, differences are not considerable. Despite considerable urban growth within Perth over this time period (average annual rate of population grown 2.7% between 2007 and 2015; ABS, 2016), noise levels at Mosman Bay did not increase. Thus, despite some inter-year



variability, there appears to be a general long-term stability in the soundscape of this site.

Such acoustic stability in localized soundscapes could be beneficial for long-lived animals such as dolphins, as this may indicate the possibility of predictability in a variable environment. If dolphins are able to use acoustics to aid predictions of when and where different sound sources occur in the river system, this may influence their habitat use. For example, fish calls may indicate prey availability and signal an "attractive" area, whereas vessel or pile-driving noise may signal

an "unattractive" area where animals are at risk of disturbance or harm. This information could be particularly important for the decision-making of mother-calf pairs, who may have a greater preference for quieter areas for vital nursing and resting activities. Previous studies within the Swan River have found differential site use by dolphins in response to site-specific environmental and anthropogenic variables (Moiler, 2008; Beidatsch, 2012; Marley et al., 2016b). It could therefore be particularly beneficial to ascertain the relationship between the acoustic characteristics of the environment and dolphin occurrence or behavior.

In conclusion, the Swan River is a highly variable acoustic environment, experiencing a large range of different sound sources. The most ubiquitous noise in the Swan River came from vessel traffic, which was persistent at all sites considered, followed by snapping shrimp which had site-specific prevalence. Although these two sound sources are both strong acoustic components of the Swan River soundscape over similar frequency ranges, their impact on dolphin communication is likely to contrast due to structural differences, with vessel noise suggested to be the more detrimental. The prominence of these sound sources varied spatially, resulting in characteristic soundscapes at different sites within the same river system. Some sites are therefore "noisier" than others, with the Fremantle Inner Harbor the noisiest from an anthropogenic perspective. However, there was variation in "noisiness" within sites, with different sites showing temporal variation in broadband noise levels at the scale of hours, days, months or even (to some degree) years. This spatial and temporal variation illustrates the acoustic complexity of the Swan River soundscape. How dolphins effectively navigate this spatially and temporally complex environment, and at what stage anthropogenic noise becomes too much to maintain a healthy dolphin community, has yet to be determined. Thus,

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when considering dolphin acoustic habitat, it is beneficial to consider the context of soundscape contributors—their frequency structure, duration, variability within source type, and spatio-temporal prevalence.

AUTHOR CONTRIBUTIONS

Conceived and designed study: SM, CS, CE, and MP. Collected and analyzed the data: SM, with contributions to data collection from MP. Advised on analyses: CE, CS, and IP. Wrote the manuscript: SM, with contributions to drafting, critical review, and editorial input from CE, CS, MP, and IP.

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