Styles Group
Underwater Acoustics

ASSESSMENT OF UNDERWATER NOISE EFFECTS

PROPOSED OFFSHORE (<25M DEPTH) SAND EXTRACTION: MANGAWHAI-PAKIRI COAST

PREPARED FOR

McCallum Brothers Limited

DATE

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Styles Group
Underwater Acoustics

Assessment prepared by Styles Group for McCallum Brothers Limited.

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Executive Summary

Styles Group has been engaged by McCallum Brothers Ltd (MBL) to undertake the underwater noise effects modelling associated with the resource consent application to extract sand offshore (up to 25m depth) along the Mangawhai-Pakiri coast. This report describes the modelling of the underwater dredging noise effects that been undertaken in order for the Cawthron Institute to complete their assessment of the potential for dredging noise to adversely affect marine mammals. Thus, it is important to note that the effects concluded to potentially occur from the consent renewal are not discussed in this report but rather contained entirely within the report from Cawthron. An assessment on airborne noise effects is provided in a separate report.

The 2019-built *William Fraser* incorporates the latest industry advances that make the new vessel much more efficient than older trail suction hopper dredgers, such as MBL's previous vessel, the 1968 *Coastal Carrier*. There is also extensive isolation of the engine mounts, and hull insulation that reduce the on-board noise emissions from the vessel into the water column. Therefore, the main noise sources associated with the activity will be the draghead making contact with the seafloor, the water jetting and the movement of the sand slurry up the pipe to the hopper. We have therefore based our assessment on the loudest operational stage (active dredging).

Section F2.18.2 Objective of the Auckland Unitary Plan (Operative in Part) pertains to the health and well-being of marine fauna, and this assessment considered the noise effects on invertebrates, fish and marine mammals. However, given the physical properties of the dredging noise in this case and differing hearing mechanisms between the different phylums or classes, the potential noise effects on invertebrates and fishes are not expected to be greater than those predicted for marine mammals. Therefore, the effects modelling was undertaken specifically for marine mammals, with effects radii on fishes and invertebrates being inside those of the marine mammals.

Nine marine mammal species have been identified within the area, five of which (the more common species) were focused on. Those five were common dolphins, bottlenose dolphins, killer whales, Bryde's whales and NZ fur seals. Of those species, three functional hearing groups have been identified: low-frequency (LF) cetaceans, mid-frequency (MF) cetaceans and Otariid pinnipeds (OW).

In order to assess potential noise effects on those species, two data needs were identified: (1) understanding the existing soundscape; and (2) understanding the source levels and true propagation coefficients of the *William Fraser* inside the current consent area. These investigations were completed between March and November 2019, with two passive acoustic monitoring arrays being deployed inside the southern consent area off Pakiri, and a single measurement array (containing 6 SoundTrap recorders) used to investigate the noise levels of the *William Fraser* and propagation losses (used to adjust the acoustic models herein). Those data revealed a typical soundscape for an open coastal area (with sounds from fish and marine mammals, snapping shrimp, vessels, dredging and weather (wind and waves), generating daily sound pressure levels between 96 and 111 dB re 1 μPa) and

dredging noise levels below those from larger TSHDs previously assessed in New Zealand waters (average source level of the *William Fraser* approximately 168 dB re 1 µPa @ 1m).

Predicted noise emissions from the TSHD *William Fraser* were evaluated in terms of critical distances for which injury (PTS, where hearing sensitivities do not return to normal following noise exposure), temporary threshold shifts (TTS, whereby hearing sensitivities do return to pre-exposure thresholds after a period of time following noise exposure), risk of behavioural effects (as a percentage over range), and auditory masking (whereby noise interferes with a biologically-important signal that marine mammals rely on). The key methods used to assess those effects were the safe distance method, dose-response calculations, listening space reductions (*LSR*s) and generalised sonar equations.

Injury (PTS) from the sand extraction activities using the TSHD *William Fraser* is not expected to occur at any stage of the dredging within the consented area, for any species. Temporary threshold shifts are also not expected to occur for any species beyond 1m from the proposed TSHDs. These findings are based on the source levels and subsequent exposure levels being below the 2018 NMFS thresholds for PTS and TTS beyond 1m.

Audibility of the dredging noise from the *William Fraser* is calculated to be within 5.8km, beyond which, acoustic disturbance is theoretically not possible. Based on the measured ambient sound levels and published hearing thresholds for the species listed above, there is a risk of auditory masking and behavioural effects occurring at a limited range from the *William Fraser*. There is also a risk of auditory masking for fish; however they are substantially smaller than for the marine mammals. The risk for moderate behavioural responses (defined as those moderate or extensive changes in swimming speeds, direction and/or diving behaviours, cessation of vocalisations for a moderate or extended period, and/or avoidance of the area) was less extensive than low behavioural responses (defined as minor changes in respiration rates, swimming speeds and direction). For example, the 50% probability of a moderate behavioural response in the delphinids was just 4m compared to 51m for a low response. Those ranges extend to 54m and 87m, respectively, for a 25% probability of risk.

The degree of auditory masking (and spatial extent) was highest for fur seals (exceeding 76% reduction in the available listening space (i.e. the volume of ocean surrounding an animal within which a biologically-important sound can be detected) within 15m of the TSHD, followed by bottlenose/common dolphins (maximum of 72% LSR), killer whales (70% LSR) then Bryde's whales (maximum of 68% LSR). The spatial extent of any masking (i.e. greater than 1% LSR) was highest for fur seals, followed by killer whales, bottlenose/common dolphins and then Bryde's whales.

1.0 Introduction

1.1 Background

Styles Group has been engaged by McCallum Brothers Ltd (MBL) to undertake the underwater noise effects modelling associated with the resource consent application to extract sand offshore (up to 25m depth) along the Mangawhai-Pakiri coast. The current proposal that this assessment pertains to relates to a new consent application to undertake sand extraction in a new area (referred to herein as the new application area).

The new application area is located between the current inshore coastal permit held by MBL and the offshore permit held by Kaipara Ltd (Figure 1) and is between the 15m and 25m depth contours.

A full description of the activity is provided in the Description of Activity[1].

There is a growing body of evidence to suggest that loud anthropogenic noise, of similar intensity and bandwidths to dredging noise may have adverse effects on a range of marine taxa, including marine mammals (Williams et al. 2016). Noise emissions from TSHDs have in the past been characterised as a potential source of acoustic disturbance in marine mammal species (Todd et al. 2014). During a typical production cycle, sources of noise vary between dredgers, as well of course between dredger types (CEDA 2011, WODA 2013). Typical noise sources from operating TSHDs include noise from the propellers, inboard pumps, engines, the draghead and the underwater pipe used to transport the aggregate from the draghead to the hopper (WODA 2013).

1.2 Potential Noise Receivers

Objective F2.18.2 of the Auckland Unitary Plan requires that:

> 1) *Underwater noise from identified activities is managed to maintain the health and well-being of marine fauna and users of the coastal environment.*

Policy F2.18.3 of the Auckland Unitary Plan seeks to:

> 3) *Enable the generation of underwater noise where that noise is associated with the following activities:*
>
> a) *the operational requirements of vessels;*
>
> b) *construction or operation of marine and port activities, marine and port facilities, marina activities, marine and port accessory structures and services, maritime passenger facilities and dredging, that do not*

[1] Description of Activity document from Jacobs dated December 2019.

involve underwater blasting, impact and vibratory piling, or marine seismic surveys; and

c) *sonar not including marine seismic surveys.*

While marine dredging for mineral extraction is not one of the activities identified in Policy F2.18.3, the activity is Discretionary (Activity A28 in Table F2.19.4), and therefore an assessment of the proposal against the objectives and policies is still relevant.

There is a growing body of evidence to suggest that loud anthropogenic noise, of similar intensity and bandwidths to dredging noise may have adverse effects on a range of marine taxa, including marine mammals (Williams et al. 2016). Noise emissions from TSHDs have in the past been characterised as a potential source of acoustic disturbance in marine mammal species (Todd et al. 2014). During a typical production cycle, sources of noise vary between dredgers, as well of course between dredger types (CEDA 2011, WODA 2013). Typical noise sources from operating TSHDs include noise from the propellers, inboard pumps, engines, the draghead and the underwater pipe used to transport the aggregate from the draghead to the hopper (WODA 2013).

In this case, the identified potential noise receivers are those marine fauna expected to be present in the area at some time during the dredging operations – namely marine mammals, fish and invertebrates. While the use of sound underwater by marine mammals is widely known, the sensitivity of fish to underwater noise has also been well documented. Like marine mammals, fish use underwater sounds to sense their environment, as well as coordinate certain behaviours such as reproduction or territorial defence (Hawkins 1986; Rountree et al. 2006; Pine et al. 2017b).

Notwithstanding, however, the potential noise effects from the application are not expected to be greater for invertebrate and fish species, than for marine mammals[2]. This is because the hearing biology of fish and invertebrates differ substantially than for marine mammals, particularly in terms of how they detect sound. Below 5 kHz, many fishes show a good degree of hearing sensitivity (Ladich & Fay 2013) by detecting either the particle motion and/or pressure component of the sound. Fish with swim bladders are able to detect both pressure and particle motion (**REF**) and therefore have the capacity to detect noise over greater distances than fish without swim bladders (as fish without swim bladders are not sensitive to sound pressures). A study investigating the audibility of ship noise in the Hauraki Gulf found fish and crustaceans are theoretically capable of detecting ship noise (of a similar power spectrum to the *William Fraser*) within a maximum of 2.97 km based on conservative detection thresholds of 6 dB (Pine et al. 2016). A more recent study involving fish within the Hauraki Gulf calculates a maximum baseline communication range[3] of 43.5m (Putland et al.

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[2] This is because fish and invertebrates show a high degree of sensitivity to particle motion which propagate over shorter distances compared to the pressure wave (i.e. the pressure component that marine mammals are more sensitive to), and no direct evidence of tissue damage in invertebrates and fish exists over the same distances as for marine mammals.

[3] Communication range is the radius around an animal within which acoustic communication is possible. This differs from audibility ranges, which is the range within which mere detection of a sound above ambient noise is possible.

2017), though the detection threshold was set at a substantially higher level compared to Pine et al. 2016. The noise emissions from the *William Fraser* is therefore highly unlikely to be detected by fish within the Goat Island marine reserve, situated approximated 3.2 km from the southern extent of the proposed consent area (based on the increased sound pressure levels associated with productive reef systems and the measured spectrum of the *William Fraser*). Both Pine et al. (2016) and Putland et al. (2017), among others, do show that audibility and communication ranges in marine mammals are substantially further than for fish. Therefore, the underwater acoustic modelling undertaken for this report has been performed specifically for marine mammals.

A total of nine marine mammal species have been identified as species potentially affected, due to their presence in the area (whether frequent, seasonal, or infrequent), by the Cawthron Institute. However, of those, four species are considered to be more likely affected based on their presence in the area being most common. Those are common dolphins (*Delphinus delphis*), bottlenose dolphins (*Tursiops truncatus*), killer whales (*Orcinus orca*) and Bryde's whales (*Balaenoptera edeni*). Underwater noise effects modelling have therefore been undertaken on each of those species. With respect to their bio-acoustics, these species also share similarities with the other species less frequent in the area, and therefore focusing on the most common species can be considered as an appropriate proxy for those (including those data-deficient ones). We have also assessed potential noise effects on New Zealand fur seals (*Arctocephalus forsteri*), as they do not share similar hearing physiologies or effects criteria to the four cetacean species listed above.

1.3 Scope of this Assessment

The scope of this assessment was to model the underwater dredging noise to assess the potential extent for which dredging noise may induce hearing threshold shifts (both permanent and temporary), behavioural responses and auditory masking in marine mammals. We did consider fishes as well, however the effects on fishes were less sensitive to the most pervasive component of dredging noise and were, therefore, not worse than those expected for the marine mammals. Noise effects on fish have therefore not been mapped. This acoustic report is targeted at aiding the assessment of effects on marine mammals (provided in Clement & Johnston 2019) by providing the critical distances for which the potential onset of noise effects may occur for the species of concern. As such, discussion of the results from the acoustic modelling and impact zones in the context of literature and the relevant objectives and policies of the Auckland Unitary Plan, as well as recommended monitoring and mitigation, is not included herein. Instead, they are provided as part of the assessment of effects on marine mammals (Deanna & Johnston 2019).

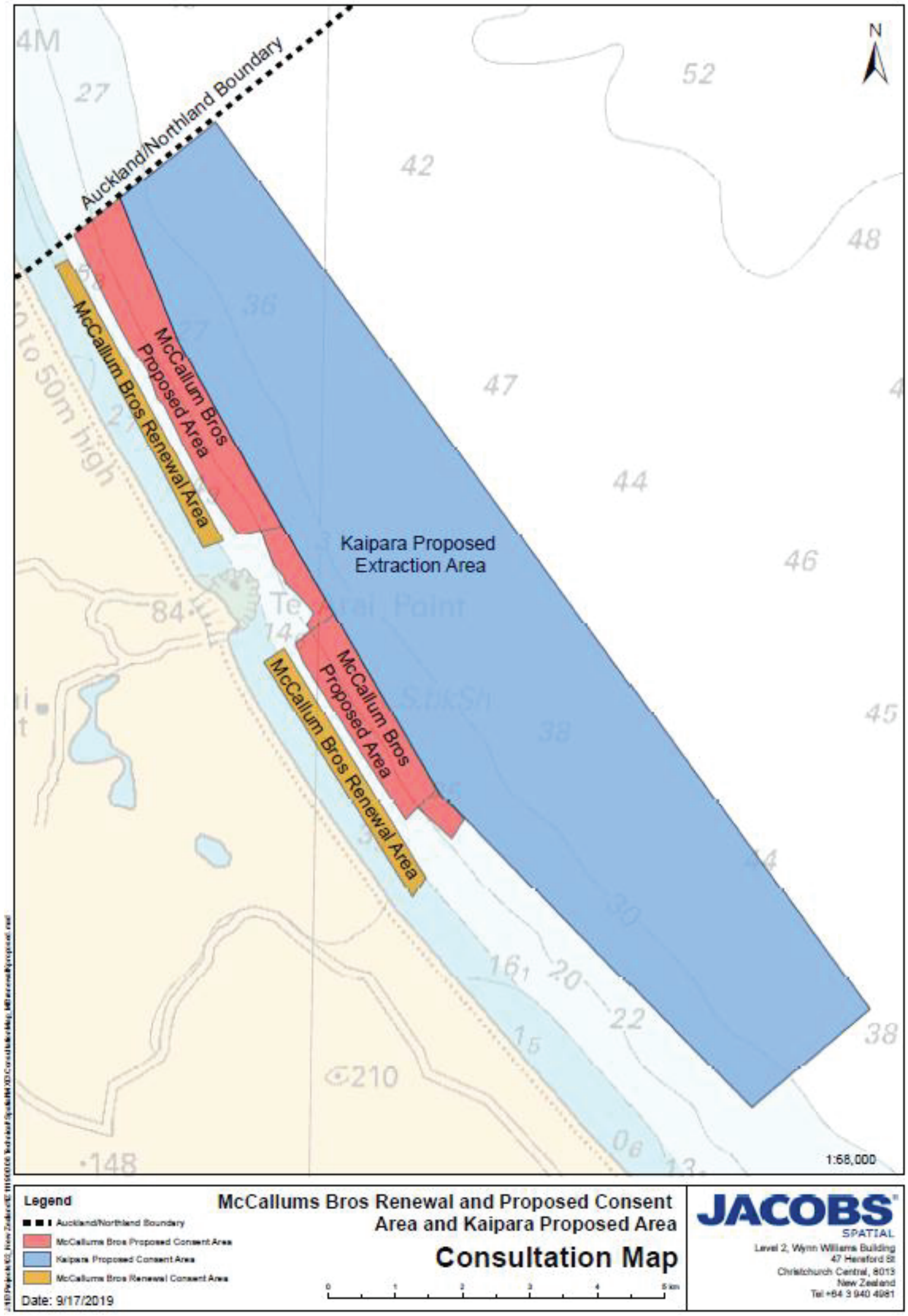

Figure 1 McCallum Bros Ltd Pakiri Sand Extraction new consent application area (taken directly from the Description of Activity document from MBL dated 19 December 2019.

2.0 The Existing Underwater Soundscape

Marine mammals (as well as fish and marine invertebrates) depend on underwater sound for critical life processes. These processes include, but are not limited to, keeping group members together while navigating turbid coastal waters, communication between family members, locating prey during feeding, mediating mating behaviours, and avoiding predation (Montgomery et al., 2006; Popper et al., 2001; Radford et al., 2007; Richardson and Thomson, 1995; Slabbekoorn et al., 2010; Stanley et al., 2010). Their ability to communicate and sense their environment using sound is therefore linked to the ambient sound environment; whereby the biologically-important signal must be audible over the background sound level within some critical bandwidth. Coastal activities, including pile driving, dredging, shipping, drilling, etc, can cause ambient sound levels over wide frequency bandwidths to rise – to the point where biologically-important signals for marine mammals can be masked, leading to increased stress and sub-lethal behavioural responses (Southall et al. 2007; Nowacek et al. 2007). Underwater noise pollution can therefore degrade marine mammal habitats within sites where offshore (up to 25m depth) activities take place.

Notwithstanding, the extent of which possible effects may occur is not always homogenous across sites or regions but vary according to the physical environment. Generally, noise effects can only occur if the invading noise source is audible (audibility being a function of both the ambient sound levels and hearing thresholds of the listener). Therefore, in order to properly assess the spatial extent of possible acoustic disturbances, the ambient soundscape must be fully considered and incorporated into the effects modelling (in the context of the species' hearing thresholds).

2.1 Methodology

2.1.1 Study Site and Recorders

To characterise the ambient soundscape within the area, we deployed four SoundTrap 300HF recorders (two arrays, providing sampling redundancy) off the northern end of Pakiri Beach (Figure 2). That area was chosen due to dredging operations occurring nearer Mangawhai. The sampling rates were at 96 kHz, while the click detectors operated at the full sampling rate of 576 kHz. The arrays were deployed along the 30m depth contour between 19 March and 25 April 2019, and then again between 9 May and 10 June 2019.

The hydrophone component of the SoundTrap recorders was calibrated by the manufacturer and field-calibration checks before and after deployments were undertaken using a calibrated piston phone (GRASS Type 42AA, SPL 114 dB re 20 µPa, nominal frequency 250 Hz), a calibrated (using a Brüel & Kjaer Type 4231 Sound Calibrator) sound level meter (Brüel & Kjaer 2250 Type 1 SLM with a Brüel & Kjaer ½ inch Condenser Microphone Type 4189) and specialist acoustic software. Electronic calibration of the recorder component was done at the start of every recording event by comparing a set of automated tones of known frequency and voltage amplitude to the full-scale response level provided by the manufacturer for the appropriate gain setting, and verified using the piston phone.

Figure 2: Google Earth image showing the locations of the two hydrophone arrays used to characterise the existing soundscape.

The two arrays were identical for sampling redundancy reasons.

2.1.2 Overview of Analysis Procedure

The sound pressure levels (*SPL*s), daily in-band sound energy contributions (as a percentage), power spectral densities (*PSD*s), and third octave band levels (*TOL*s) were calculated, along with the statistical variation. The *PSD* data were plotted as long-term spectral averages (*LTSA*). *LTSA* plots are useful in allowing large time series data to be viewed and analysed in a more efficient way. In total, 3,318 sound files were generated and viewing all those files as individual spectrograms would be near impossible. *LTSA* plots provide the means to compress the data and view all recordings as a whole, revealing an overview of the spectrum for the monitoring period. The .WAV files were uploaded to PAMScan's file directory and restructured into 1-min time-bins. Acoustic analyses were then undertaken on each minute and the 1-min time averaging was then applied (generating the time-averaged data for the *LTSA*). The statistical analyses of the *PSD*s (percentile levels (1st, 5th, 25th, 50th, 75th, 90th and 99th percentiles) and spectral probability densities) and *TOL*s (boxplots) were then performed and plotted. Bandpass filters for four bandwidths (10 – 100 Hz, 100 – 1000 Hz, 1 – 10 kHz and 10 – 32 kHz) were then applied to the raw waveform for each 1 minute of data. The bandpass filtered 1-min L_{eqs} were then calculated for each time-bin, generating a single mean value for each time bin and frequency bandwidth. The *SPL* data were then batch processed by individual days (each 1-min bin was time stamped, determined by the file name of each recording) and averaged over each day (so a single mean value for each day was obtained). The in-band energy contributions of each frequency band were calculated at the same time as the daily L_{eqs}, expect an additional algorithm was used that took the daily acoustic energy in each frequency band, compared it to the total energy in the whole bandwidth, then multiplied it by 100 to generate a percentage. The results were plotted for the whole monitoring period, showing the change in each day.

2.2 Results: The Existing Soundscape

All figures in this section are provided in Appendix B, and referenced as such.

The relative broadband (10 Hz – 48 kHz) daily *SPL*s varied by approximately 15 dB re 1 µPa (between 96 dB re 1 µPa (on 25 March) and 111 dB re 1 µPa (on 14 April) over the survey period and are provided in Figure 18. The hourly broadband *SPL*s are provided in Figure 19, and do show dawn and dusk choruses (typical of nearshore habitats (Pine 2013)). Broken down into smaller bandwidths, the bands between 100 Hz and 10 kHz showed similar trends to the broadband levels. Unlike busy harbours, the *SPL*s below 100 Hz were generally lower than the 1 – 10 kHz band, reflecting the fewer vessels in the study area.

The relative in-band daily *SPL* contributions over the survey period are also provided Figure 20. The relative in-band daily *SPL* contributions were generally higher for the lower frequency band 100 – 1000 Hz, followed by the mid-frequency band 1 – 10 kHz. This did differ on occasion, with inclement weather (as particularly evident in the *LTSA* data, Figure 21) increasing levels below 100 Hz (Figure 20).

The *LTSA* plot is provided in Figure 21. The *LTSA* plot reveals similar trends to the in-band contributions, but at a much finer frequency resolution (every 1 Hz). Figure 22 and 23 respectively show the statistical plots for the *PSD* and *TOL* data from the whole survey period (between March and June 2019). These plots are useful for showing the variation in the decibel levels for each frequency or frequency band, as well as the corresponding spectral probabilities. This descriptive statistical analysis revealed considerable variation in the frequency-dependent sound levels. Generally, the ambient noise floor (represented by the 1^{st} and 5^{th} percentiles in the *PSD* data, Figure 22) shows a shallow slope with no distinct rises between 500 and 2000 Hz – which is typical of a sandy bottom habitat with limited vessel traffic (Radford et al. 2010). The higher percentiles (the 95^{th} and 99^{th} percentiles and represented the more transient events) do reflect the presence of vessels (also seen in the *LTSA* data) but have limited influence on the averaged and median sound levels.

Overall, the ambient soundscape is typical of a sandy beach habitat that is not inside or near a busy harbour.

3.0 Marine Mammal Detections

3.1 Data analysis

Automated detectors and classifiers for marine mammal vocalisations were run through all acoustic data from the northern array. The detectors were focused on dolphins (species unidentifiable beyond a broadband click species, i.e. bottlenose and common dolphins) and whales (baleen whales, assumed to be Bryde's whales based on their resident status within the Hauraki Gulf and time of the year during which this study took place).

Dolphins were detected in the acoustic dataset based on their echolocation clicks and whistles using PAMGuard[4] (Figure 3, Putland 2017). Detection data from the on-board click detector in the SoundTrap 300HF units were also processed in PAMGuard, as well as the .WAV snippets. Snapping shrimp were a prevalent source of false positives in the detection data and so positive detections were only logged when a train source was identified and manually verified by assessing the waveform, spectrum, Wigner plot, *PSD* and in some cases, playing back the edited audio file itself. Once a true positive was confirmed, the start time, end time and duration of that detection event were logged, as well as the minimum inter-click interval (to determine the presence of foraging buzzes, and thus foraging activity). A single detection event was defined as the time between the first and last confirmed vocalisation (either echolocation clicks or whistles) after no vocalisations were detected for more than 30min followings the last detection (Pine et al. 2017).

Whale vocalisations were detected using a custom-written detector similar to that described by Hendricks et al. (2018), but modified for the Mangawhai – Pakiri region. The detector first runs through an adaptive entropy band detector, then runs an additional algorithm based on the spectrogram itself to confirm the presence and location of a whale's call within the recording. The detector worked by breaking the signal into 10-second windows and performing the processing in each window with 50% overlap. Using an optimised-sized Hann window, the entropy in each window was calculated and compared those to a dynamic threshold based on the background entropy after being scaled for the variance per unit of time. When the entropy dropped below the set threshold, the program indexed the number of successful triggers and flagged the detection when the number of successive triggers reached the required minimum (Hendricks et al. 2018). The start and end times were then extracted and a 120 second spectrogram around the detection was generated and saved as a .PNG image for reference. Those spectrograms were then binarised based on adaptive thresholds inside a predetermined window to identify the call's contours for cross-correlation with known calls. If the contours of the detection met the required criteria, the .PNG was moved to a new directory for quick manual verification.

[4] PAMGuard is an open source software designed specifically for bioacoustic analyses of passive acoustic data and the detection and classification of marine mammal vocalisations. It is the most commonly used software for passive acoustic monitoring of marine mammals worldwide. See www.pamguard.org

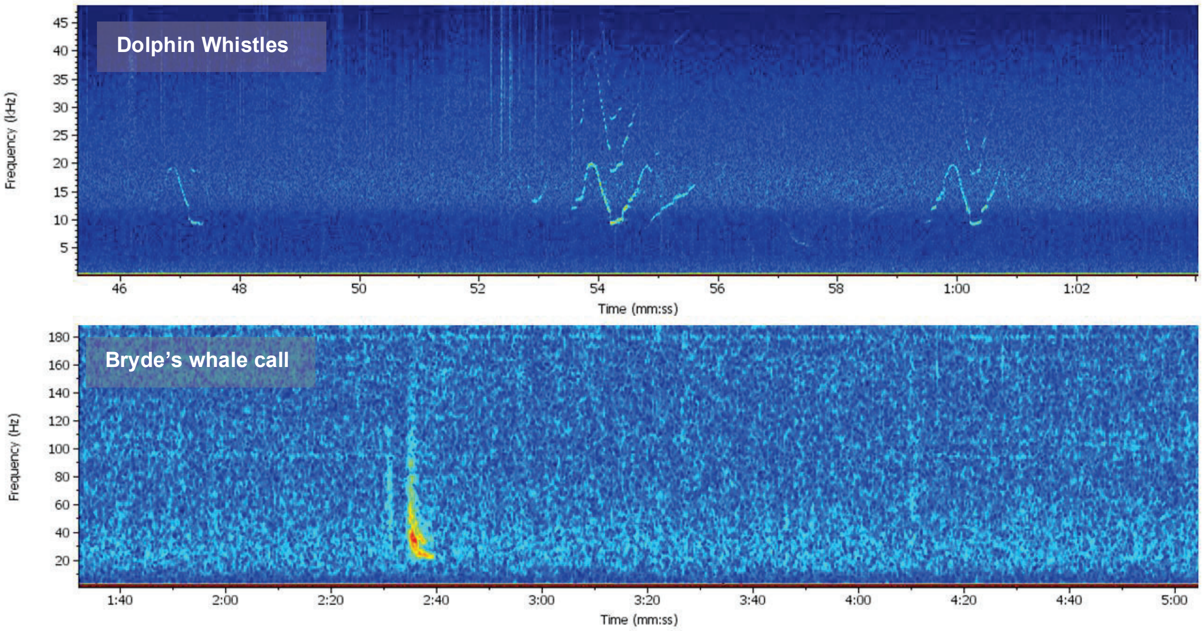

Figure 3: Spectrogram showing examples of detected dolphin whistles (lower end of echolocation clicks also visible between 52 and 54 seconds) (Top Panel); and a single Bryde's whale call (Bottom Panel).

3.2 Results: Marine Mammal Detections

A total of 64 detection events (comprising of thousands of echolocation clicks spanning 22hr 24min) of dolphins (either bottlenose or common dolphins) were confirmed, while 477 Bryde's whale calls were detected over the 69 recording days (Figure 4). Of the 64 dolphin detection events, 36% of them contained feeding buzzes. A summary of dolphin detections are provided in Table 1 and the activity plots (including foraging activity) are provided Figures' 5 through 7.

Table 1: Durations of the dolphin detection events

Median (h:mm:ss)	Average (h:mm:ss)	Min (h:mm:ss)	Max (h:mm:ss)
0:12:00	0:20:22	0:03:00	1:36:00

Bryde's whale vocalisations were detected at least once during 25 days out of the 69 day deployment (i.e. 36% of all days monitored contained at least 1 whale call).

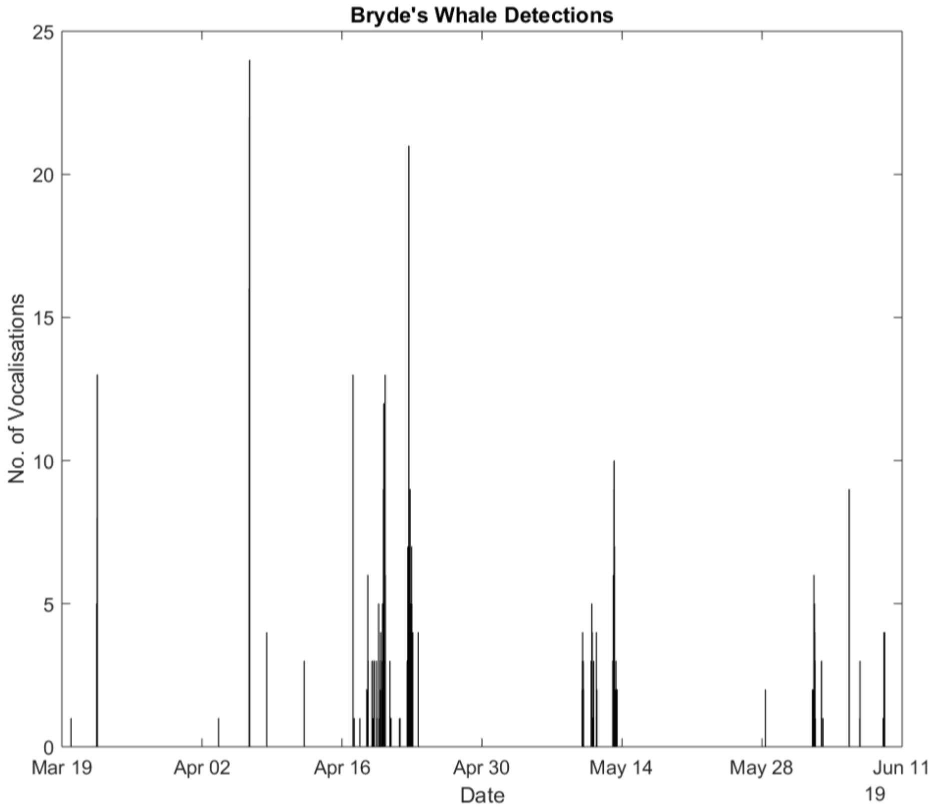

Figure 4: Acoustic detections of Bryde's whales during the monitoring period.

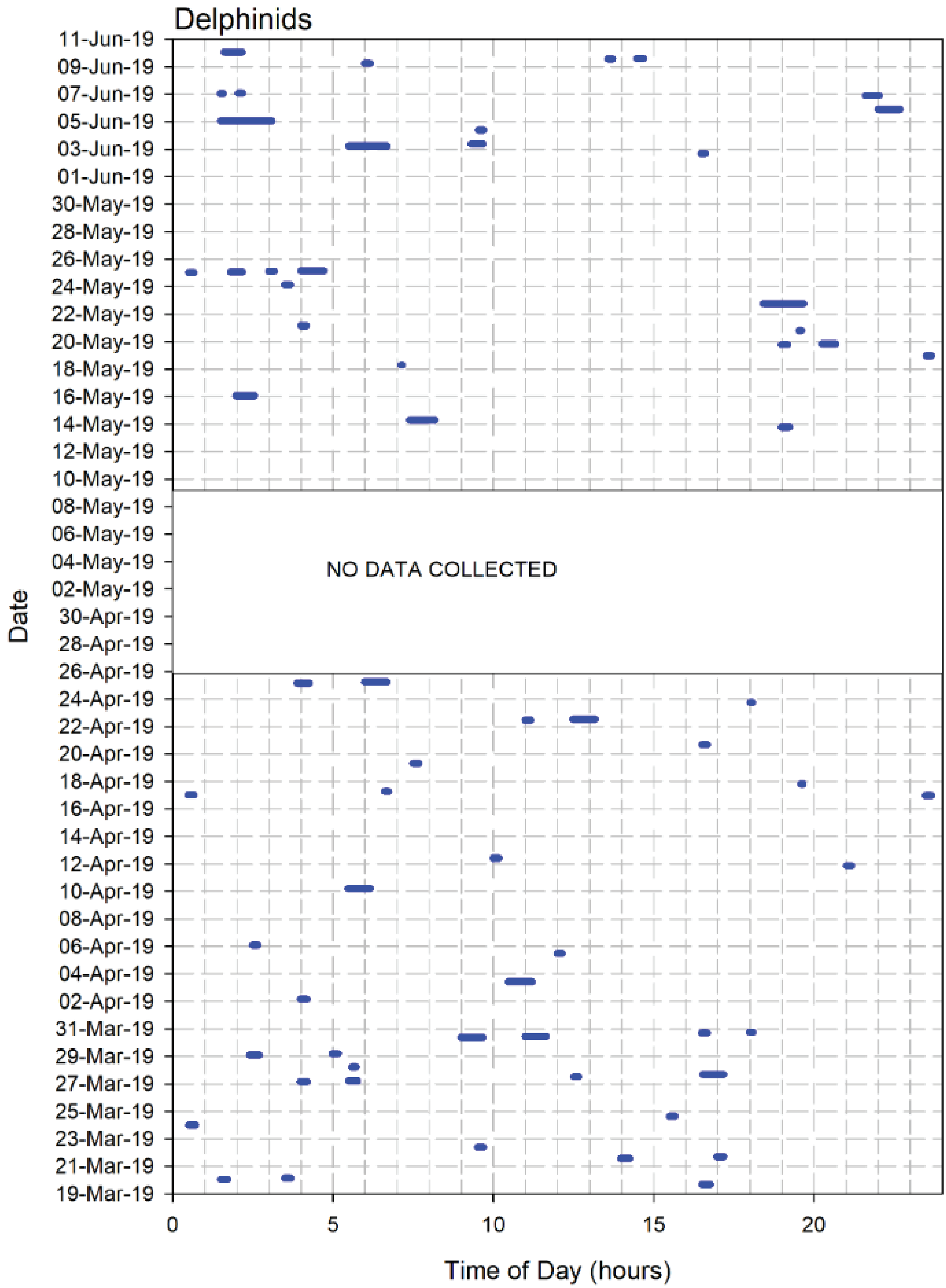

Figure 5: Actograms showing bottlenose/common dolphins during the monitoring period.

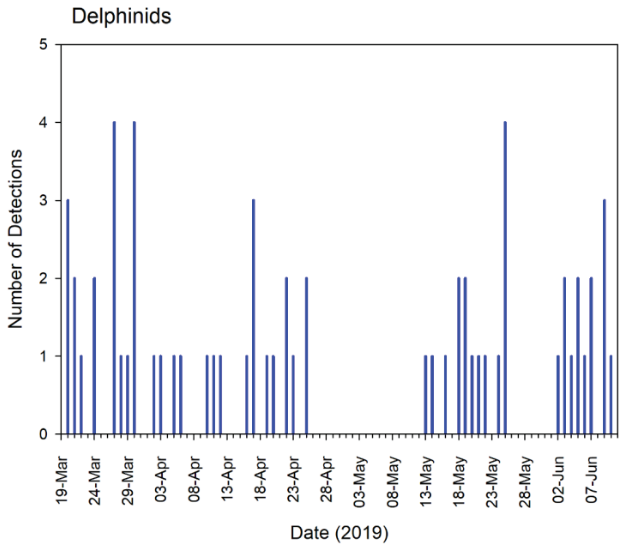

Figure 6: Number of acoustic detections of bottlenose/common dolphins per day over the monitoring period.

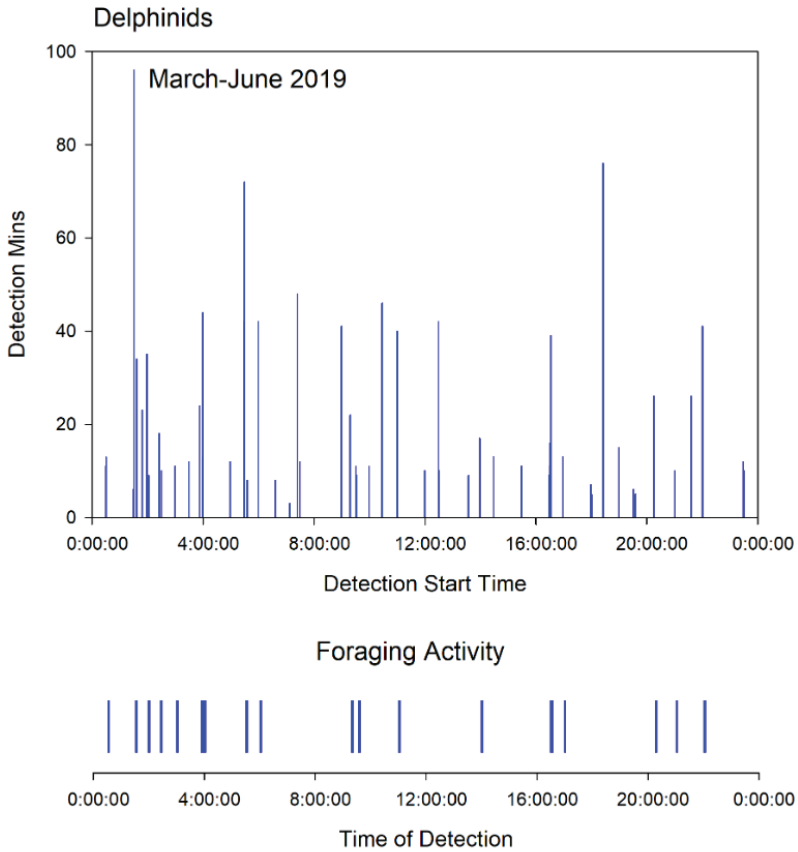

Figure 7: Plots showing detection durations of bottlenose/common dolphins (presented as detection minutes) (top panel) and occurrence of feeding buzzes (bottom panel) over the day.

These data have not been time averaged, but are the actual durations overlaid for the monitoring period

4.0 Noise Modelling: Methodology

The overall approach to the underwater noise modelling undertaken in this project is given **Figure 8**. Each component is described below.

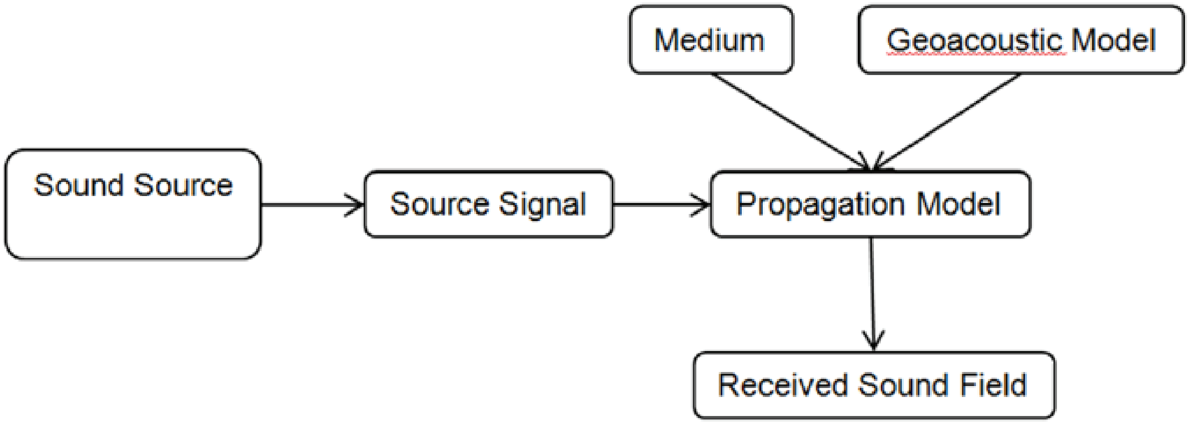

Figure 8: Flow diagram showing the overall approach to the acoustic modelling.

Each box represents a key component to the model environment described below.

4.1 The modelled area

The modelling was undertaken within and around the proposed area shown in Figure 1, defined by the extraction area coordinates in Appendix B. The modelling was not repeated in the southern consented area because the environment and input data are the same.

4.2 Sound source: the TSHD vessel

The recently commissioned purpose-built trailing suction dredge, the *William Fraser* shown in Figure 9 will be used to undertake the sand extraction.

Figure 9: The TSHD *William Fraser*.

Table 2: Summary table of the operational characteristics of the William Fraser.

Vessel details	*William Fraser*
Length	68 m
Beam	16 m
Deck Size	43m x 10 m
Hopper Capacity	900 m^3
Loaded Draft	4.2 m
Vessel extraction speed	1.5 – 2 knots
Draghead Width	1.5 m
Sand pump capacity	400 mm
Sand screen size	2.5 mm
Width of extraction trench	Average 1.5 m
Depth of extraction trench	Average 65 mm
Extraction trench shape	Trapezoidal
Volume required to be extracted from seabed to fill hopper	1560 m^3
Length of extraction track needed to fill hopper	Up to 15 km
Time to fill hopper	4 – 5 hrs
Number of trips in 30 consecutive days for 15,000 m^3 limit	17 trips

4.3 Sound Source Characterisation: The *William Fraser*

Underwater noise measurements of the *William Fraser* were undertaken on the 28th November 2019, during fine weather conditions (variable 10 knot breeze, sea state zero and no swell). A measurement array was deployed that consisted of six SoundTrap 202STD recorders (Ocean Instruments Ltd, Auckland, New Zealand) (Figure 10). The hydrophones were calibrated using the same method described in Section 2.1.1 *Study Sites and Recorders* and operated continuously.

The array was deployed the morning of the 28th of November, and each hydrophone was bottom-mounted along the 30m (the inner hydrophones, ST 1, 2, 3, and 4) and 35m (ST 5, and 6) contours. The hydrophones were set at 3 m above the seafloor, with a subsurface float (2 L volume) set a further 2 m above the hydrophone. This was done to ensure the subsurface float was far enough away so to not contaminate the measurements. The differing depths between the inner and outer hydrophones are not expected to cause any differences in the noise levels recorded in this case. The rationale for the outer hydrophones was to simultaneously record the noise emissions of the *William Fraser* at two distances that were in-line of each other to further investigate the empirical frequency dependent propagation loss. The inner hydrophones (ST 1,2,3,4) were placed between 200 and 300m apart, while the outer ones were placed 400m away to the east from ST 2 and 3 (the middle of the inner 'line') (Figure 10). This shape of the array effectively allowed for four replicates as the TSHD passed the array (whilst actively dredging, i.e. draghead down with pump and generator operating), for multiple bearings. The vessel operated as normal, with no issues reported. Once it passed, the vessel continued north for approximately 1.4 km after passing the last hydrophone of the array (ST1), before turning around and passing the array again, southbound. The TSHD followed the 30m contour, as per the offshore consent owned by Kaipara Ltd but operated by MBL.

Figure 10: Google Earth image showing the GPS track of the TSHD *William Fraser* in relation to the measurement (hydrophone) array (ST1 through 6) on 28 November 2019 in fine weather conditions.

The vessel was tracked using a Garmin Map62 GPS unit, logging the vessels' position in relation to the array every few seconds (with an error of 3m). The same GPS unit was used to mark the GPS positions of each of the hydrophones, and those were used to calculate the horizontal distances between the vessel and hydrophones for every 10 seconds (since the *SPL* data was averaged over a 10 second period).

During the measurements, the research vessel left the area but remained 10 km away. The times when other vessels were visible anywhere were recorded and checked against the hydrophone data to ensure no contamination. In addition, bespoke vessel detectors were used to ensure no vessel noise was confounding the results. If there was any contamination (i.e. another vessel was detectable on the hydrophone (using both power spectra and detection of modulation of noise methods), those data were excluded from the analysis (Figure 11).

4.3.1 Data analysis

Time-series of the recorded power spectral densities (*PSD*s) were calculated and plotted to examine the quality of the data from all six hydrophones. The received third octave band levels (*TOL*s) were also calculated and plotted, providing the frequency-dependent sound pressure levels that were used to represent the critical bandwidths of cetaceans in the effects modelling (Erbe et al. 2016; Pine et al. 2018).

The *PSD*s and *TOL*s were calculated using a 1-sec Hamming window and 50% overlap with 10-sec averaging. The broadband (10Hz – 48 kHz) *SPL*s, as 1-sec and 10-sec averages, were calculated for each horizontal distance between the TSHD *William Fraser*'s GPS position and the respective hydrophone position. This analysis was performed using the Haversine formula, after the source and receivers latitude and longitude coordinates' were time-synced.

It is important to note that the Haversine formula assumes the earth to be a perfect sphere, however the distances between the *William Fraser* and all hydrophones were inside 3.1km, the margin of error from assuming a perfect sphere is trivial. For each distance, the 10 second *SPL*s (both broadband *SPL*s and *TOL*s) were plotted (and can be viewed as an animation through time), showing the fine-scale variations in the received sound pressures over distance as the TSHD passed the array. These data were also used to compare the propagation modelling and improve its performance using an empirical *PL* coefficient, similar to that undertaken by Pine et al. (2014).

For the purposes of the underwater noise modelling, the received sound pressures measured at the *William Fraser*'s closest point of approach (CPA) to each hydrophone (Figure 11) while actively dredging were back-calculated to a reference distance of 1 metre. This was done using the published sound propagation formulas by Pine et al. (2014) but modifying the spreading coefficient based on the empirical data collected in this study (see Figure 33 in Appendix F). The overall source spectra used in the effects modelling was the averaged spectra over the 4 closest hydrophones (thus 8 replicates – 4 hydrophones, two passes of the TSHD each).

Only the source spectra of the *William Fraser* was used in the effects modelling due to the decommissioning of the *Coastal Carrier* in October 2019, however both spectra are provided in Figure 12 for comparison.

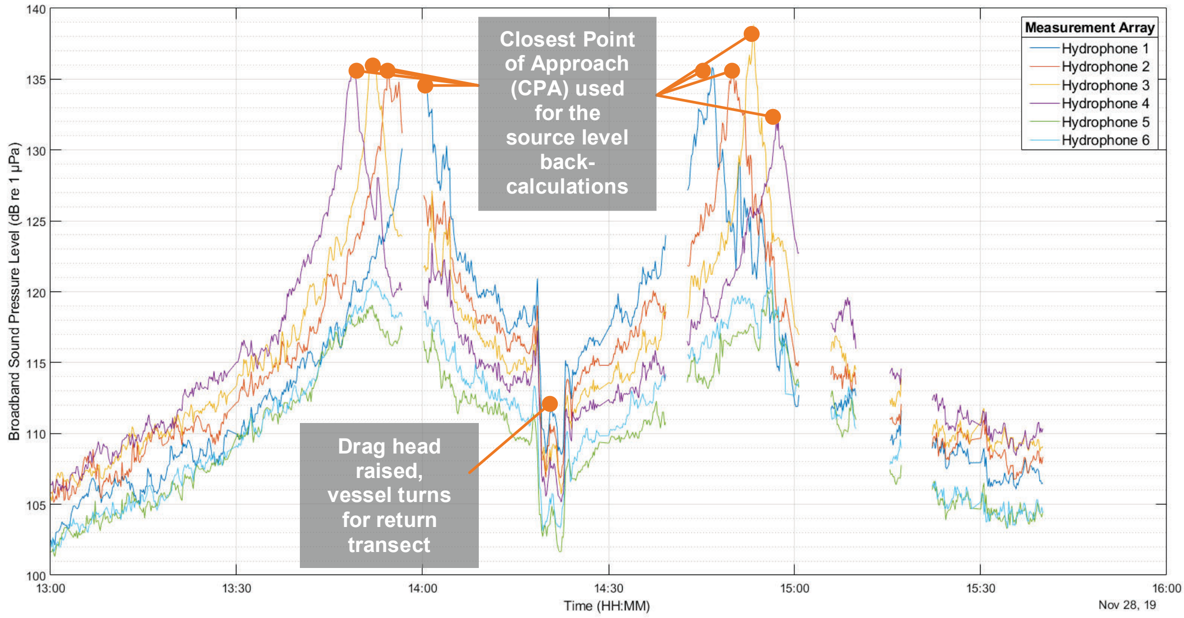

Figure 11: Measured *SPLs* from the inner hydrophones (ST 1, 2, 3, 4) as the *William Fraser* moves through the northern consent area, actively dredging, passing the measurement array.

Data containing contaminating vessel noise (extraneous) were removed from the analysis

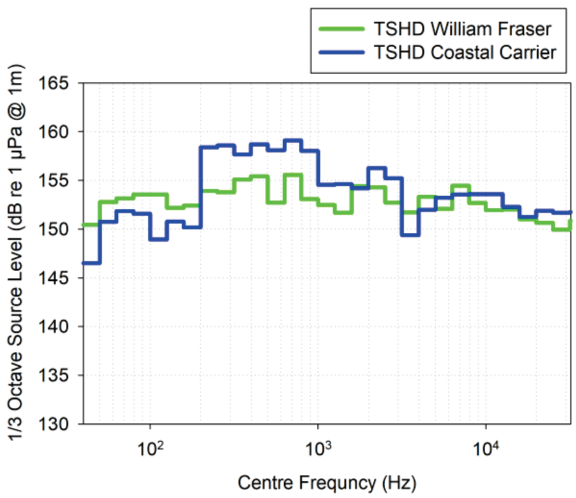

Figure 12: Averaged 1/3 octave source spectrum for the two TSHD vessels used by MBL.

Note the *Coastal Carrier* was decommissioned October 2019 and will not be used in the new consent. The *Coastal Carrier*'s spectrum is provided for comparative purposes only and was measured on the 28th June 2019 using the same methods described in Section 4.3 *Sound Source Characterisation*. The underwater noise modelling and effects results for the *Coastal Carrier* are provided in Appendix F.

4.4 Bathymetry

Sound propagation in shallow water typically follows a normal mode whereby a sound wave of a particular frequency moves sinusoidally through an acoustic waveguide (i.e. the water column of seafloor) (Jensen et al. 2011). However, sound propagation in shallow water is highly influenced by boundary effects and the extent of those effects is related to water depth, as well as the seafloor and surface roughness. Bathymetry data for the acoustic modelling within this project were those obtained by MBL and the National Institute of Water and Atmosphere (NIWA 2016). The NIWA dataset was region-wide, while the bathymetry provide by MBL was of the consent area only. The two datasets were overlaid and the MBL bathymetry was relied upon in the consent area based on being the more recent dataset (since dredging that has occurred over the past few years and the depths within the consenting area may differ to the NIWA dataset). The bathymetry from NIWA was obtained using multibeam and single beam sounding lines spaced 50-120m apart. The bathymetry from MBL was obtained using side-scan sonar and multibeam sounding lines spaced an average of 30m apart (that decreased as depth reduced the further inshore the bathymetric survey was undertaken). This is the most comprehensive digitised bathymetry data available for the Mangawhai – Pakiri region and was uploaded directly to the acoustic model platform (Figure 13).

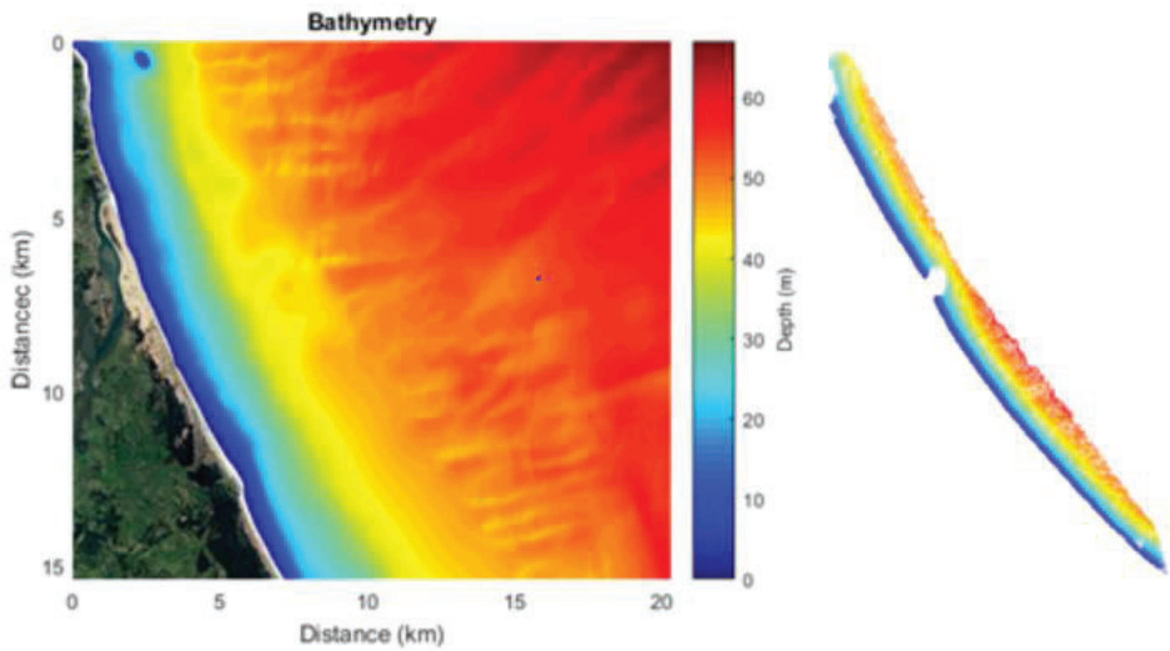

Figure 13: Bathymetry rasters used in the acoustic modelling.

The left raster is from the NIWA dataset, while the right raster is the MBL dataset from 2019. The two were used together to cover the region. Note the colour bar on the left raster does not correspond to the MBL raster.

4.5 Sea-floor Composition

The composition of the seafloor and sediments has a direct influence on the sound propagation as part of the ocean acoustic medium. Sediment type and seafloor roughness also influences the boundary effects through sound absorption, sound speed changes and reflections. These factors mean that the sound field at any given location from the sound source can be highly variable due to changes in the seafloor composition and geo-acoustic properties. The seafloor composition within the existing consent area is well understood, with a range of comprehensive studies having been undertaken. Information on the seafloor composition within the modelled area has been obtained by MBL at many locations from Mangawhai to south Pakiri during 2011 and 2017 (Figure 14). Those data were uploaded to the model platform and the sediments outside the consent area were based on hydrographic charts.

Figure 14: Google Earth image showing the locations of sediment samples from 2011 and 2017 used in the acoustic modelling.

The geo-acoustic properties for different sediment types that were used in the modelling are provided in Table 2.

Table 2: Geo-acoustic properties for various sediment types within the project area.

Sediment Type	Density (kg/m^3)	Compressional wave velocity (m/s)	Absorption (dB / lambda)
Sand-silt-clay	1600	1560	0.20
Sand-silt	1700	1605	1.0
Silty sand	1800	1650	1.1
Very fine sand	1900	1680	1
Fine sand	1950	1725	0.8
Coarse sand	2000	1800	0.9
Gravel	2000	1800	0.6

4.6 Sound Speed Profiles

The speed of sound underwater is dependent on the temperature, density (salinity) and depth. Within the Hauraki Gulf, sea surface temperatures vary between summer and winter months, with significant mixing down to 40m depth occurring during the winter months (Zeldis 2013). The absence of this mixing during the summer months gives way to the development of a thermocline (a layer in which there is a steep temperature gradient) and halocline (a layer in which there is a steep salinity gradient) which causes a change in sound speeds with depth. While long term sea surface data from various locations around the Hauraki Gulf (including near the project area) is available, long-term (i.e. spanning seasons) thermocline and halocline data for the Mangawhai – Pakiri area is not available[5]. Information on the thermocline and halocline depths for the acoustic modelling has therefore been from a proxy location within the outer Hauraki Gulf of similar depths and hydrology to Pakiri (Zeldis 2013). Based on these similarities, we expect the thermocline and halocline differences between Pakiri and the proxy location to be negligible in terms of sound speed effects and we therefore consider the assumption to be appropriate. The use of environmental data from nearby proxy locations has occurred in many published scientific investigations, both abroad and within New Zealand, when specific data in the study's location is unavailable. Please refer to Appendix D for the plots showing the thermoclines and haloclines that were used to calculate the sound speed profiles (provided in Figure 15).

[5] CTD cast data is available from December 2019 and these were compared with the proxy data with good agreement.

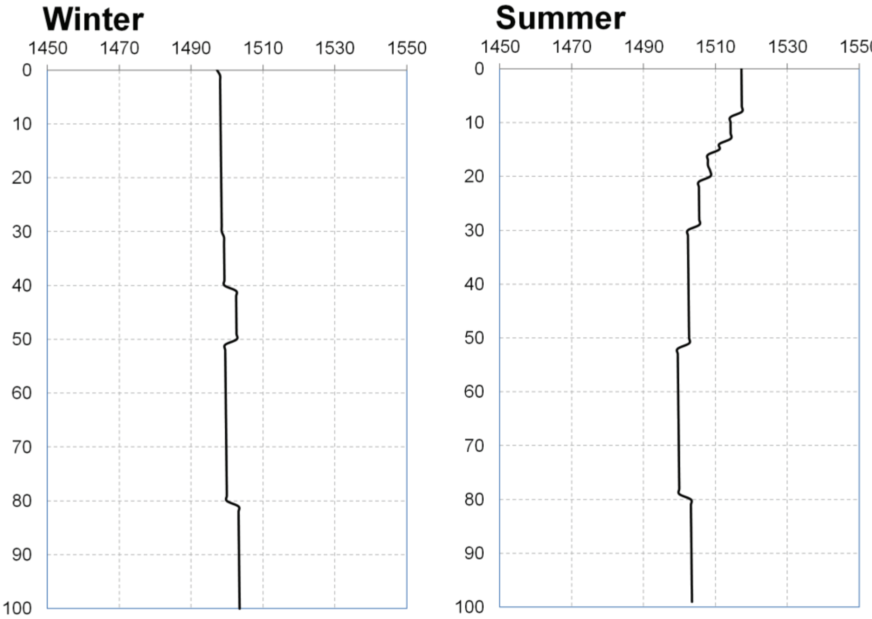

Figure 15: Sound speed profiles calculated for the project area.

Depths (y-axis) are in metres below the surface and sound speeds (x-axis) are metres per second.

4.7 Propagation Modelling

The underwater noise modelling was simply defined as:

$$SPL_{freq}\,(R) = SL_{freq} - PL_{freq}\,(R)$$

where SPL_{freq} at distance R was the predicted sound pressure level for some frequency bandwidth, SL_{freq} was the source level at that frequency band and PL_{freq} was the propagation loss over distance R for that frequency band. The propagation loss (*PL*) was determined using a combination of range dependent parabolic equation (*PE*) and ray trace (*RT*) models in dBSea, for frequencies below and above 1.4 kHz, respectively, for 180 radials over a 10m grid with 0.5 m depth resolution. Since the ray trace model is based on Snell's Law, it is applicable if a signal's wavelength is much less than the layer in which it is propagating. Therefore, ray tracing was only applied for frequencies above 1.4 kHz (based on the halocline depth during the summer (due to the density change either side of the halocline) and the depth during the winter model, as no halocline is present. The *PL* for three frequencies within each 1/3 octave band between centre frequencies 50 Hz and 32 kHz were calculated and then averaged within each 1/3 octave bandwidth to represent the *PL* for a specific band. Third octave bands were chosen for the modelling as they are often used to represent the critical bandwidths of marine mammal hearing[6] (Erbe et al. 2016; Pine et al. 2018).

[6] This is done when the true critical bandwidths are unknown for the species of concern.

The validity of the acoustic model was tested by comparing the modelled *PL* coefficients for each 1/3 octave bandwidth with the empirical data collected from the *William Fraser* and *Coastal Carrier*. There was good agreement between the modelled and the empirical data, and the results are provided in Appendix G.

4.8 Effects Modelling for Marine Mammals

The overall aim of the acoustic modelling is to provide the acoustic footprint of the dredging works in order to inform an assessment of the potential impacts on marine mammals[7]. It is important to note that these impacts are discussed in the assessment of effects on marine mammals (please refer to Clement & Johnston 2019) and not in this report. We have provided advice to the Cawthron Institute regarding underwater noise effects and, for conciseness, those impacts and recommendations are not discussed herein. However, the methods used to reach those noise effects conclusions and recommendations are provided herein, as well as an overview of the key results.

The potential noise effects, and severity, depend on the distance between the source and receiver, with injury (permanent threshold shifts, PTS) occurring close to the source, followed by temporary threshold shift (TTS), behavioural responses and auditory masking. PTS effects are not expected for the consent renewable because the source levels of the TSHD were too low.

The receiver's locations were assumed to occur within the top 15m of the water column, and thus the modelled SPLs below 15m depth were excluded from the noise effects modelling. This is based the dive profiles and behaviour of Bryde's whales within the Hauraki Gulf (the same population as this assessment is investigating), where the data showed the whales spending most their time within 13m of the surface (Constantine et al. 2015). We have assumed the receiver depths for the dolphin species, based on DTAG data of other delphinid species (Silva et al. 2016).

4.8.1 Temporary Threshold Shifts

When a receiver is exposed to high noise levels over an extended period of time, the cells within the inner ear begin to fatigue and become less sensitive. Therefore, a change in the receiver's hearing threshold occurs, and the degree at which those thresholds change is referred to as a threshold shift. If hearing returns to normal after a certain time post-exposure, the threshold shift is temporary (termed TTS), but if not, then it is referred to as PTS. The amount of threshold shift depends on the duration of noise, rise times, duty cycles, sound pressure levels within the receiver's critical bandwidths' (i.e. the spectral composition of the noise) and, of course, the overall energy.

Some exposure guidelines for hearing effects (TTS/PTS) prescribe a cumulative sound exposure level threshold, which relates to the amount of time (for example a 24-hour period,

[7] Noise effects on fish and invertebrates are well inside the zones for marine mammals, and therefore not explicitly stated herein.

as is the case with NMFS (2018)) that the noise source is present (after M-weighting the noise):

$$SEL_{cum} = SPL + 10\ log_{10}(duration\ of\ exposure\ (seconds))$$

where SEL_{cum} is the cumulative sound exposure level, assuming a constant received *SPL* with no temporal variability over space and time (NMFS 2018). However, if one was to assume a stationary (or very slow moving) receiver (i.e. a marine mammal) and a moving source at a constant speed and direction (typical of TSHDs actively dredging), then the actual exposure will vary over space and time (i.e. the rate at which exposure increases will be the greatest when the receiver is closest to the TSHD and decrease with increasing range as either the TSHD or receiver moves away). Thus, if the problem is addressed from the perspective of the marine mammal, then the simple SEL_{cum} equation above does not reflect reality particularly well. In this case, the approach for assessing potential noise impacts on marine life is the *safe distance*, R_0, method[8]. This concept is also provided within NOAA's technical guidance stating that it *"allows one to determine the distance the receiver would have to remain in order to not exceed some predetermined exposure threshold".* The safe distance method accounts for the source velocity, spectrum, duty cycle, and is independent to the exposure duration (i.e. suitable for moving sources, whether continuous or impulsive). The safe distance method was calculated using the same equation from Sivle et al. (2014), but expressed in a simpler manner:

$$R_0 = \frac{\pi}{E_0 v} SD$$

where *S* is the source factor, *D* is the duty cycle, *v* is the transit speed and E_0 is the exposure threshold[9] (NMFS 2018). A key assumption to this method is that the sound source is simple – i.e. the source moves at a constant speed and in a constant direction (such as that of a TSHD actively dredging). Since the exposure thresholds for TTS used herein as those from the NMFS (2018) guidance (Table 3), the source levels of the *William Fraser* were M-weighted. This was done for every 1 Hz, and then recombined to generate the broadband source level used for the TTS zone calculations. Previous research undertaken by Cawthron Institute has identified nine different species of marine mammals within or near the current project area (Clements & Johnston 2019). Among those nine species, four functional hearing groups were identified: low-frequency and mid-frequency cetaceans and otarrid pinnipeds.

Table 3: NMFS (2018) auditory threshold criteria for a non-impulsive noise source for the functional hearing groups relevant to this consent application.

Functional Hearing Group	Functional Hearing Range		Non-Impulsive Noise	
	f_1 (kHz)	f_2 (kHz)	TTS Threshold (SEL*, weighted)	PTS Threshold (SEL*, weighted)

[8] First described by Sivle et al. (2014) and described in the 2018 Revisions to the Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing (Version 2) from NOAA, April 2018.

[9] As a pressure value of the NMFS 2018 thresholds.

Functional Hearing Group	Functional Hearing Range		Non-Impulsive Noise	
LF	0.2	19	179	199
MF	8.8	110	178	198
OW	0.94	25	199	219

* Cumulative sound exposure level.

4.8.2 Behavioural Responses

There is a substantial amount of literature on the behavioural effects of noise on marine mammals – either direct evidence-based studies, opportunistic studies, or observations – that have been summarised in several reviews (for example Richardson et al. 1995; Hildebrand 2005; NRC 2005; MMC 2007; Nowacek et al. 2007; Weilgart 2007; NAS 2017). Behavioural effects are highly varied and may include changes in swimming behaviours (directions and speeds), diving behaviours (durations, depths, surface intervals), time spent on the surface, respiration rates, fleeing the noise source and changes to vocalisations. Predicting the zones within which behavioural effects may be seen is the most difficult noise effect to quantify due their dependency on the context, species and location (see Ellison et al. 2012; Gomez et al. 2016 for reviews on the issue of context dependency on marine mammal behaviour).

There is no widely-accepted regulatory guidance on behavioural effects currently in existence – it is still a research problem. The only preliminary guidance in existence for behavioural responses is a single unweighted decibel value of 120 dB_{rms} re 1 μPa (from the National Oceanic and Atmospheric Administration (NOAA)), but it has not had a wide-spread uptake (Gomez et al. 2016). One of the issues of using a single noise threshold for behavioural responses is that the data currently available are not very comparable (Nowacek et al. 2007; Southall et al. 2007; Eillison et al 2012; Gomez et al 2016) with limited relationships between the severity of the behavioural response and the received level of underwater noise (Gomez et al. 2016).

Some underwater noise assessments in New Zealand still consider the 120 dB_{rms} re 1 μPa contour, stating the reason being it's the only measure of behavioural effects on marine mammals. However, because of the uncertainty in assessing the risk of behavioural effects within and between species (based on the highly contextual nature of behavioural effects), the application of a simplistic noise threshold value for behaviours should be avoided (Faulker et al. 2018). Recent scientific research assesses behavioural zones based on the probability of occurrence using dose-response curves specific for the species of interest (Joy et al. 2019). Dose-response curves show the relationship between the probabilities of a behavioural effect occurring at a given level of noise exposure (Joy et al. 2019). The dose-response formulas have been used by the US Navy (US Navy 2008, 2012) and the scientific community for a number of years – primarily for impulsive signals. However, a recent scientific investigation from the Sea Mammal Research Unit (SMRU), JASCO Applied Sciences and the Ports of Vancouver in British Columbia, Canada, has been published that provides a specific dose-response function and thresholds for southern resident killer whales and a continuous noise source (Joy et al. 2019). The thresholds used make use of the most up-to-date data for killer whales and behavioural effects (specifically those effects classed as low[10] or moderate[11] (respectively a Southall severity score of 2-3 and 4-6 (Southall et al. 2007)). Briefly explained, the researchers took empirical studies on killer whales and noise (42 studies in total) and correlated the estimated received noise levels with the behavioural response type (i.e. the Southall severity scores from Southall et al. 2007) to get a regression curve (linear relationship). From there, the researchers calculated two received levels that corresponded to the 50% probability of either a low or moderate behavioural response occurring and used those to generate the dose-response curve for killer whales. The dose-response curve in this assessment was calculated using:

$$R = \frac{1 - \left(\frac{L-B}{K}\right)^{-A}}{1 - \left(\frac{L-B}{K}\right)^{-2A}}$$

where *R* was the risk from 0 to 1 (i.e. the probability of an effect occurring) at the noise level *L*, *B* was the basement received level (*RL*) at which the risk of an effect occurring is so low it does not warrant calculating, *K* was the *RL* increment above *B* at which there is 50% risk and *A* was a transition sharpness parameter (Joy et al. 2019). The *RL* at which there was a 50% risk of an effect was set at 129.5 (for a low response (Southall severity 2.5)) and 137.2 dB re 1 µPa (for a moderate response (Southall severity 5)) (Joy et al. 2019).

Since this method was based on more accurate data (and a killer whale, which is a species that can occur in the Mangawhai – Pakiri area, with hearing biology similar to other delphinids), we applied the same method and assumptions to our data. However, we altered the basement received level, *B*, to be the averaged 1-min *SPL* of ambient noise over our 69 day monitoring period. This provided a conservative baseline level specifically related to Pakiri that is more useful and scientifically based than the unweighted threshold level of 120 dB_{rms} re 1 µPa for all marine mammals.

For Bryde's whales, however, the *RL* at which there was a 50% risk of a low behavioural response was set a 120 dB re 1 µPa (since that level was the lowest at which bowhead whales, another mystecete species and one of the only whales with estimated levels of exposure (from continuous noise) linked to a certain behavioural response (Southall et al 2007). This is conservative. No assessment for moderate behavioural effects for Bryde's whales was done because we do not know what such a threshold would look like and is therefore too speculative to be meaningful. The same basement levels and transition sharpness values were applied for Bryde's whales as for the odontocetes (represented by the killer whales and bottlenose dolphins).

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[10] Low behavioural responses are defined as minor changes in respiration rates, swimming speeds and direction (Joy et al. 2019).

[11] Moderate behavioural responses are defined as moderate to extensive changes in swimming speeds, direction and/or diving behaviours, moderate or prolonged cessation of vocalisations, and/or avoidance (Joy et al. 2019).

4.8.3 Auditory Masking

Several species of marine mammals (and fish) are known to have hearing ranges that overlap with low-frequency anthropogenic noise – such as vessels (Pine et al. 2016; Putland et al. 2018) or machinery such as renewable energy devices (Pine et al. 2019). For example, bottlenose dolphins (*Tursiops truncates*) and common dolphins (*Delphinus delphis*) have shown hearing sensitivities to signals as low as 100 Hz, while killer whales (*Orcinus orca*) show sensitivity down to 500 Hz (Hall & Johnson 1972; Popov & Klishin 1998; Szymanski et al. 1999). Therefore, auditory masking - the interference of a biologically important signal (such as vocalisations from conspecifics or predator/prey etc) by an unimportant noise that prevents the listener from properly perceiving the signal (Erbe 2008) – is expected to occur (Pine et al. 2019). Therefore, dredging noise (along with other anthropogenic noise sources commonly seen in coastal waters), has the potential to interfere with an animal's ability to perceive their natural acoustic environment (Erbe et al. 2016; Popov & Klishin 1998). The inclusion of auditory masking in underwater noise effects assessments is best practice because behavioural effects generally occur at moderate levels of masking and thus understanding the spatial limits of masking is important (Pine et al. 2019).

We assessed auditory masking for marine mammals by quantifying the reduction in a species listening space. An animal's listening space is the immediate area (volume of ocean) surrounding it within which it can detect and perceive a biologically important signal. The listening space method was used instead of sonar equations in this case because the call structures of the species of interest at the source are poorly understood, while the listening space method is more sensitive to changes in the existing sound environment (Pine et al. 2018). Those changes could be better modelled by the computer using the empirical data outlined in Section 2.0 and 3.0 above. As the anthropogenic noise source (in this case, the *William Fraser*) approaches an animal (or vice versa), the animal's listening space will decrease to a new, smaller listening space. The difference between the original and the smaller listening space under masking conditions is termed the listening space reduction (*LSR*).

The method for calculating the *LSR* is fully described by Pine et al. (2018) who define the *LSR* as:

$$LSR = 100\left(1 - 10^{-2\frac{\Delta}{N}}\right)$$

where N is the frequency-dependent *PL* slope coefficient and Δ is the difference between the perceived base ambient noise level NL_1 and TSHD noise level NL_2 at a given distance (NL_2 was the modelled sound pressure levels, as described in Section 4.6 *Propagation Modelling*). The ambient noise levels were those described in Section 2.0 *The Existing Underwater Soundscape*. It is important to note that NL_1, being the perceived base ambient noise level, is the maximum of the listener's hearing threshold (audiogram value) and the ambient level inside a critical band, approximated herein by 1/3 octave bands (Erbe et al. 2016; Pine et al. 2018). Audiogram values for bottlenose dolphins and killer whales (reconstructed from Nedwell et al. 2004) were used to estimate hearing thresholds in each critical band. There are no audiograms available for the New Zealand fur seal or Bryde's whale. Consequently, a Northern fur seal (*Callorhinus ursinus*) audiogram (Nedwell et al. 2004) and modelled audiogram for the fin whale (Cranford & Krysl 2015) were used herein. The use of modelled audiograms do require special care when used, as they are based on the structure of the skull and no true hearing data is available (as AEP experiments, or behavioural audiograms, are not able to be done on mytecetes). However, their use in scientific studies is done when no other data exists – as is the case with this assessment.

The *PL* slope coefficient was calculated by curve fitting the empirical *PL*s of each 1/3 octave band between 50 Hz and 32 kHz (using the same data collected as described in Section 4.2.2 *Data Analysis*) over a distance that represented the listener's maximum listening range under natural sound conditions (the ground-truthed *PL* model was also used for those distances beyond a few kilometres). This was done using a simplified sonar equation without signal gain (to increase conservativeness):

$$SE = SL - PL - NL_1 - DT$$

where signal excess (*SE*) is set to zero to indicate detection onset, NL_1 was the 5th percentile ambient noise level and *DT* was the detection threshold (conservatively set at 10 dB for common dolphins (Clark et al. 2009; Kastelein et al. 2013; Putland et al. 2017; Pine et al. 2018; Pine et al 2019). This was done because the *PL* slope can have some range-dependence.

The empirical source levels, ambient levels and audiograms are provided in Figure 16.

The *LSR* was then calculated for each 1/3 octave band at each depth step – resulting in an *LSR* map for each band. Those maps were then overlaid on top of each other (forming a 3D matrix) and averaged across layers to provide an overall 2D *LSR* map for the project area (Pine et al. 2018).

It is important to note the three important assumptions applied to the auditory masking model: (1) the listener exhibits omnidirectional hearing; (2) the sound propagation field is omnidirectional; and (3) no masking release mechanisms occurred. The exclusion of masking release is an important assumption as it means the results are likely to be conservative (i.e. has the potential to overstate true masking). Marine fauna have evolved in a naturally noisy environment, with many natural sources (such as waves and conspecific or heterospecific vocalisations etc) active as effective maskers (Radford et al. 2014). It therefore stands to reason that they have evolved to counteract naturally occurring maskers, ensuring their vocalisations can be detected by a listener over ambient noise levels. Anti-masking strategies by the sender are predominately altering the call's characteristics, such as increasing call amplitude (Lombard effects), changing the spectral characteristics of the call (such as lowering or raising the fundamental or peak frequencies) to reduce spectral overlap, or altering the temporal dynamics of the call, such as increasing call rates or repetition (Radford et al. 2014; Erbe et al. 2016). There may also be repeating information at multiple frequencies within a call's harmonics (such as in some fish calls, graded structures in dolphin vocalisations and whale calls). In addition, masking release at the listener may occur when the call and masking noise are coming from different direction (termed spatial release from masking) or when the masking noise is amplitude modulated over a bandwidth much wider than the critical band of the listener (termed comodulation masking release) (Erbe et al. 2016). All these masking release mechanisms have been documented in marine mammals and fish, and thus the importance of this particular assumption.

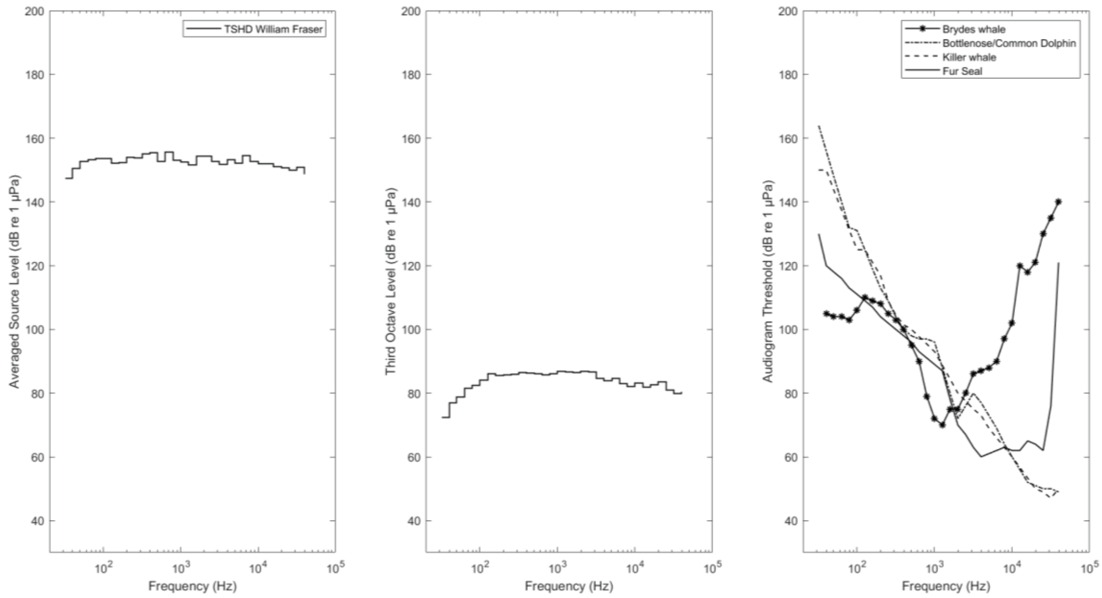

Figure 16: 1/3 Octave source level for the *William Fraser* (left panel), median 1/3 octave ambient sound levels measured between March and June 2019 (middle panel) and species audiograms (right panel) reproduced from Nedwell et al. 2004.

4.8.4 Audibility Ranges

In order for any noise effects to occur, the noise has to first be audible to a receiver. It is important to note, however, that just by detecting a noise source does not equate to an effect occurring. Notwithstanding, the limits of audibility do provide us a maximum area within which the risk of any effect occurring is theoretically greater than 1 %. By calculating the limits of audibility for each of the species of concern, it allows regulatory bodies to better understand the acoustic footprint of the proposed dredging for particular species or groups.

Audibility limits were calculated based on the hearing sensitivities of killer whales, common dolphins, bottlenose dolphins and New Zealand fur seals (approximated based on the Northern fur seal, being the closest phylogenetic relative to the NZ fur seal for which audiogram studies have been undertaken), in the context of the ambient soundscape. A conservative approach was taken – detection thresholds, auditory gain functions and directivity of hearing sensitivities have been left out of the calculations because they are unknown for the species of concern. Masking release mechanisms have also been left out for the same reason. The key assumption, therefore, is that detectability of the dredging noise is omnidirectional[12] and directly relates to the difference between the ambient sound level, the dredging noise and hearing thresholds at each critical band.

5.0 Noise Modelling: Results

5.1 Seasonal Variation in the Acoustic Propagation

The propagation of sound levels did show variation between summer and winter months, with the warmer surface temperatures and shallow thermocline increasing the propagation of dredging noise (Figure 17).

Spring and autumn months are expected to be within these two seasonal extremes, and thus not specifically investigated herein. The noise effects modelling were therefore performed during summer conditions, being the more conservative of the two seasons.

5.2 Noise Effects

5.2.1 Temporary Threshold Shifts

During full dredging conditions (pump on, draghead down and loading the hopper), the TSHD *William Fraser* is not expected to induce a TTS beyond 1m distance due to the M-weighted source levels being below the required sound exposure level thresholds[13].

5.2.2 Behavioural Effects

Behavioural effects can be expected to occur within limited ranges from the TSHD (Table 1). The specific distances at which there is a 75% chance for some behavioural effect to occur (whether low or moderate in severity) is approximately 15m for dolphin species and 124m for Bryde's whales. A 50% probability of some behavioural effect occurring is restricted within approximately 51m and 241m for dolphins and Bryde's whales, respectively. Based on the baseline noise conditions that we measured between March and June 2019, the risk of behavioural effects occurring beyond approximately 411m and 1234m (respectively for dolphins and Bryde's whales) are not expected.

[12] Also assumed in peer reviewed scientific publications, such as Pine et al. 2016; Pine et al. 2018; Pine et al. 2019; Putland et al. 2017; Stanley et al. 2018).

[13] Those thresholds being the NMFS (2018) thresholds for LF- MF- and OW functional hearing groups.

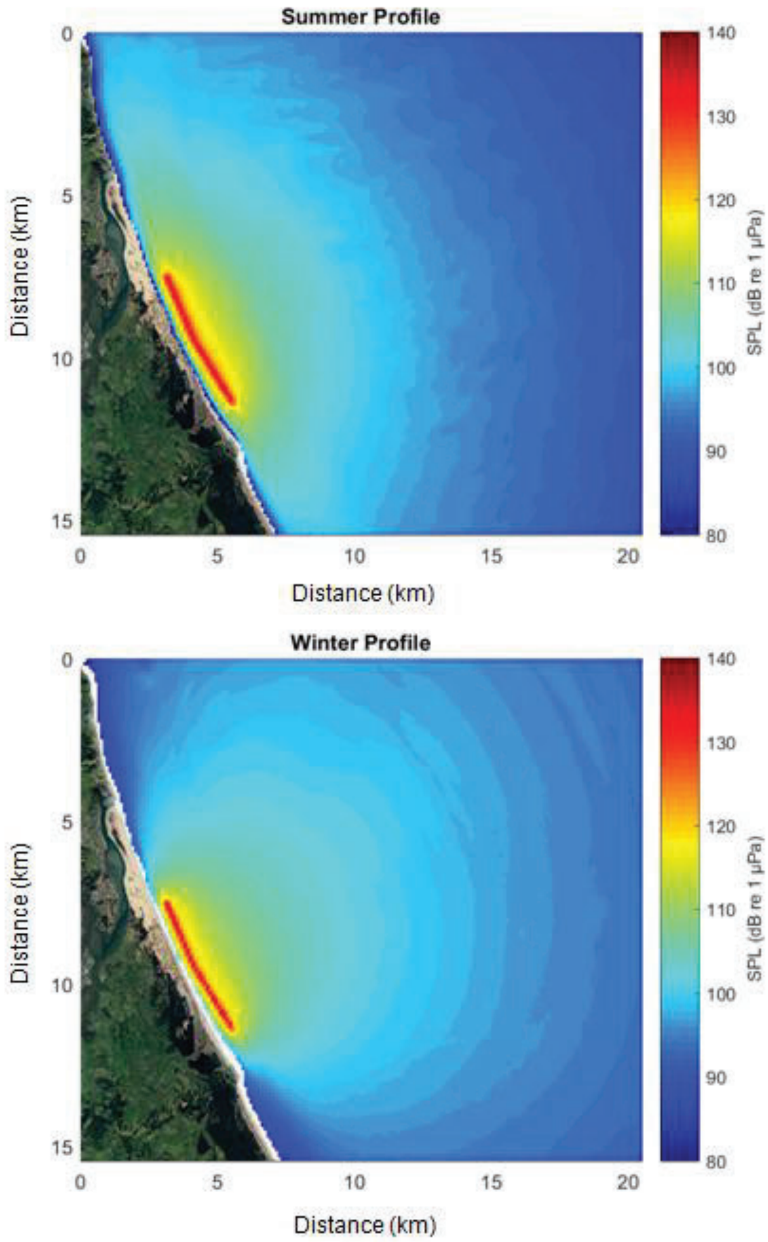

Figure 17: Comparative sound fields (broadband (10Hz – 48 kHz) un-weighted sound pressure levels) for a single measurement sample of the TSHD during summer and winter sound speed profiles and surface roughness.

All noise effects modelling were based on the more conservative summer profile.

Table 4: Distances at which 75, 50, 25 and 0% risk of low and moderate behavioural responses for each of the species of interest.

Species	Behavioural Response	Risk isopeth (m)			
		75%	50%	25%	0%
Killer Whale Bottlenose Dolphin Common Dolphin	Low (minor changes in respiration rates, swimming speeds/direction)	15	51	87	411
	Moderate (moderate to extensive changes in swimming speeds/direction and/or diving behaviours, moderate or prolonged cessation of vocalisations, and/or avoidance)	NA	4	54	308
Bryde's Whale	Low (minor changes in respiration rates, swimming speeds/direction)	124	241	314	1234

5.2.3 Auditory Masking

Auditory masking effects, and the extent of such, varied between species, and are provided in Table 2. Higher masking impacts, in terms of *LSR*, were seen for fur seals with averaged *LSR*s exceeding 70% within 100m, respectively. The spatial extent of any masking effects also occurred over longer distances for seals compared to cetacean listeners.

Table 5: Distances at which 75, 50, 25 and 0% listening space reduction (*LSR*) occurs for each of the species of interest.

Species	Distance from the TSHD			
	75% LSR	50% LSR	25% LSR	0% LSR
Killer whale	NA	348	2777	5637
Bottlenose/Common dolphin	NA	318	2757	5630
Bryde's whale	NA	92	1230	4013
Fur seal	17	466	2857	5657

5.2.4 Audibility Limits

The limits of audibility for each of the species of interest are provided in Table 6. The audibility limit for Bryde's whales is expected to be smallest, followed by bottlenose/common dolphins, fur seals and then killer whales. This is expected, based on the hearing thresholds of each species and the *William Fraser*'s spectrum. It is important to note, however, is that the audibility limits (like the *LSR* metric) is the averaged audibility limit over several critical bands for each species[14].

Table 6: Distances within which audibility is possible.

Species	Audibility radius (m)
Killer whale	5784
Bottlenose/Common dolphin	5702
Bryde's whale	4107
Fur seal	5760

6.0 Summary

Styles Group has been engaged by MBL to undertake underwater noise effects modelling of the sand extraction operations associated with the sand extraction operations in the proposed consent area. Noise effects on fishes were considered and concluded to be lesser than those for marine mammals and therefore not outlined in the results.

Changes to marine mammals' hearing thresholds (PTS/TTS), risk of behavioural effects, auditory masking and general audibility were assessed. Audibility of the dredging noise occurred over the greatest distances for all species (approximately 4.1km for Bryde's whales, 5.8km for fur seals, 5.7km for bottlenose/common dolphins and 5.8km for killer whales), followed by auditory masking effects (starting within 4.01 – 5.66km) and behavioural effects (starting within 411m and 1.23km). No risk of hearing temporary threshold shifts beyond 1m from the TSHD was found, based on the empirical data collected from the actively dredging *William Fraser*. No risk of PTS for all species was identified.

7.0 Literature Cited

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[14] Similar to the *LSR* method as outlined in Pine et al. 2018 and 2019.

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Appendix A Glossary of Terms

Acoustic waveguide	A medium or structure that guides sound waves by restricting the wave movement in one of more dimensions, resulting in the efficient transmission of the sound wave.
Ambient sound	Ambient sound is the total of all noise within a given environment, comprising a composite of sounds from sources near and far.
Biologically important signal	An acoustic signal that, once detected and perceived, provides the receiving animal some information that is important to its survival and/or reproductive output.
Critical band	The frequency band of sound, contained within a broadband noise spectrum, that contains the energy equal to that of a pure tone centred in the critical band and just audible in the presence of broadband noise (Erbe et al. 2016).
dB (decibel)	The basic measurement unit of sound. The logarithmic unit used to describe the ratio between the measured sound pressure level and a reference level of 1 micropascals (0 dB) (or 20 micropascals for airborne sound).
Detector	A detector is a computer program that automatically detects the presence or absence of a particular signal that the algorithm is trained to detect.
Halocline	A strong change in salinity in a body of water with depth, where the salinity is markedly different above and below the layer in which the salinity change occurs.
$L_{eq(t)}$ (dB)	The equivalent sound pressure level with the same energy content as the measured varying acoustic signal over a sample period (t). The preferred metric for sound levels that vary over time because it takes into account the total sound energy over the time period of interest.
Power spectral density (PSD)	The dB level of the power spectrum, presented every 1 Hz.
Sub-lethal	Sub-lethal effects are biological (including ecological), physiological or behavioural effects on individuals that survive exposure to the invasive noise.
Sound pressure level (SPL)	The logarithmic unit used to describe the ratio between the measured sound pressure level and a reference level of 1 micropascals (0 dB) (or 20 micropascals for airborne sound). Unless stated otherwise, the SPL refers to the root-mean-square (rms) sound pressure.
Soundscape	Similar to ambient sound, the acoustic soundscape is the sum of multiple sound sources arriving at a receiver (whether animal or hydrophone).
SoundTrap (ST)	An autonomous underwater acoustic logger used in marine science research from Ocean Instruments New Zealand.
Sound exposure level	The dB level of the time integral of the squared pressure over the duration of the sound event, expressed as dB re 1 $\mu Pa^2 \cdot s$.
Source level	The sound pressure level transmitted by a point-like source that would be measured at 1 metre distance, and expressed as dB re 1 µPa @ 1m.
Temporary Threshold shift	An increase in the threshold of hearing (i.e. the minimum sound intensity required for the receiver to detect a signal) at a specific frequency that returns to its pre-exposure level over time.

Thermocline	A sudden change in temperature in a body of water with depth, where the temperature is markedly different above and below the layer in which this temperature change occurs.

Styles Group
Underwater Acoustics

Appendix B Pakiri Proposed Consent Area Co-ordinates

	Application Area			
Number on Map	NZTM2000 Projection		NZGD2000 Projection	
	Northing	Easting	Latitude	Longitude
9	6000390	1747380	174 38 16.29835 E	36 07 48.00831 S
10	6000000	1747510	174 38 21.76062 E	36 08 00.59008 S
11	5999890	1747620	174 38 26.23431 E	36 08 04.09860 S
12	6000860	1747810	174 38 33.17741 E	36 07 32.52440 S
13	5997970	1748730	174 39 11.93578 E	36 09 05.77878 S
14	5998030	1749060	174 39 25.09593 E	36 09 03.64970 S
15	5997490	1749260	174 39 33.46569 E	36 09 21.05808 S
16	5997350	1749390	174 39 38.76220 E	36 09 25.52797 S
17	5997240	1749430	174 39 40.43775 E	36 09 29.07448 S
18	5996820	1749840	174 39 57.12873 E	36 09 42.47252 S
19	5996630	1749730	174 39 52.85817 E	36 09 48.69787 S
20	5996530	1749800	174 39 55.72747 E	36 09 51.90319 S
21	5996330	1749580	174 39 47.06244 E	36 09 58.51412 S
22	5996080	1749680	174 39 51.23501 E	36 10 06.56921 S
23	5993740	1751200	174 40 53.67506 E	36 11 21.63460 S
24	5993910	1751400	174 41 01.56136 E	36 11 16.00686 S
25	5993460	1751890	174 41 21.48531 E	36 11 30.32951 S
26	5993750	1752080	174 41 28.88732 E	36 11 20.81386 S
27	5994090	1751720	174 41 14.24282 E	36 11 09.98688 S
28	5996780	1750150	174 40 09.55881 E	36 09 43.59746 S
29	5998100	1749380	174 39 37.84899 E	36 09 01.20135 S
30	6002960	1746960	174 37 57.77445 E	36 06 24.85782 S
31	6002420	1746270	174 37 30.54839 E	36 06 42.75257 S

Appendix C Soundscape Characterisation Results

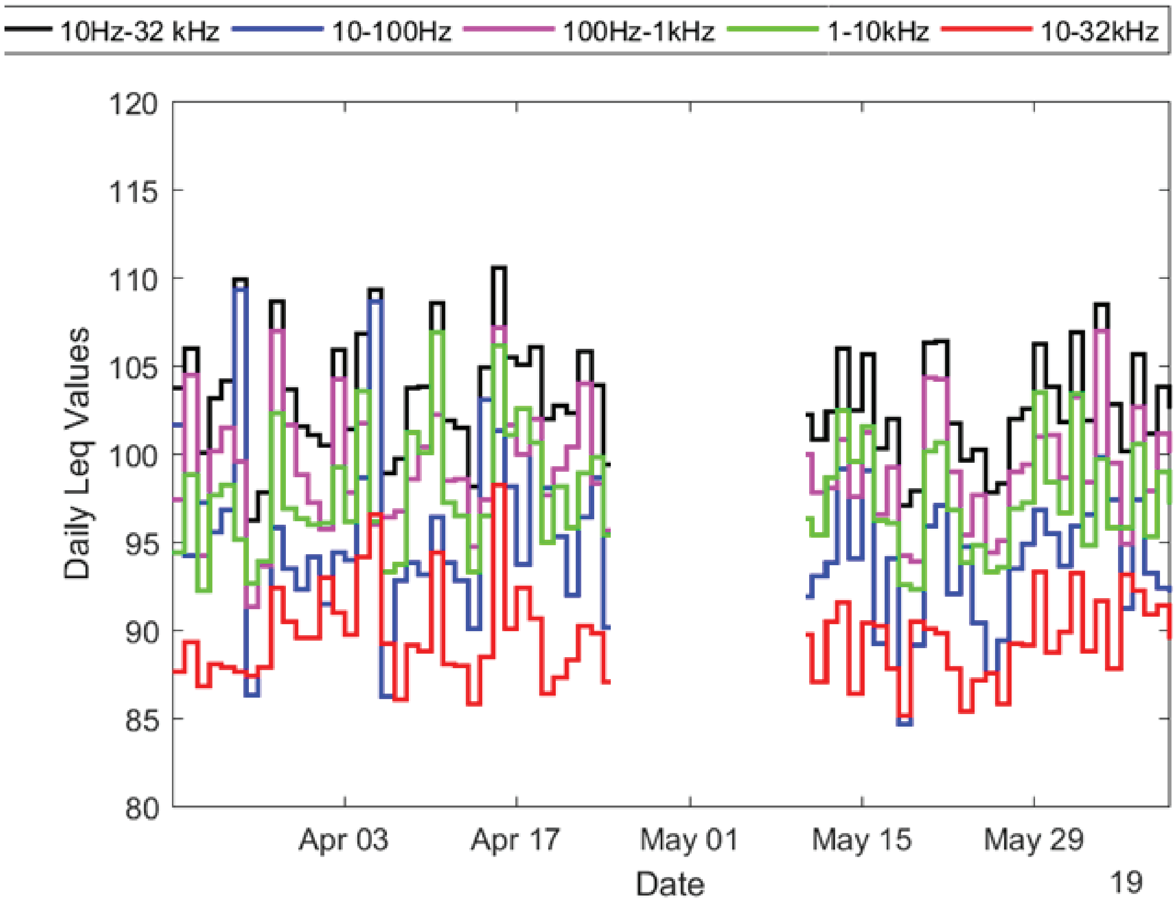


Figure 18 Daily sound pressure levels (presented as daily Leq values, dB re 1 µPa) measured within the southern consent area off northern Pakiri Beach between May and June 2019.

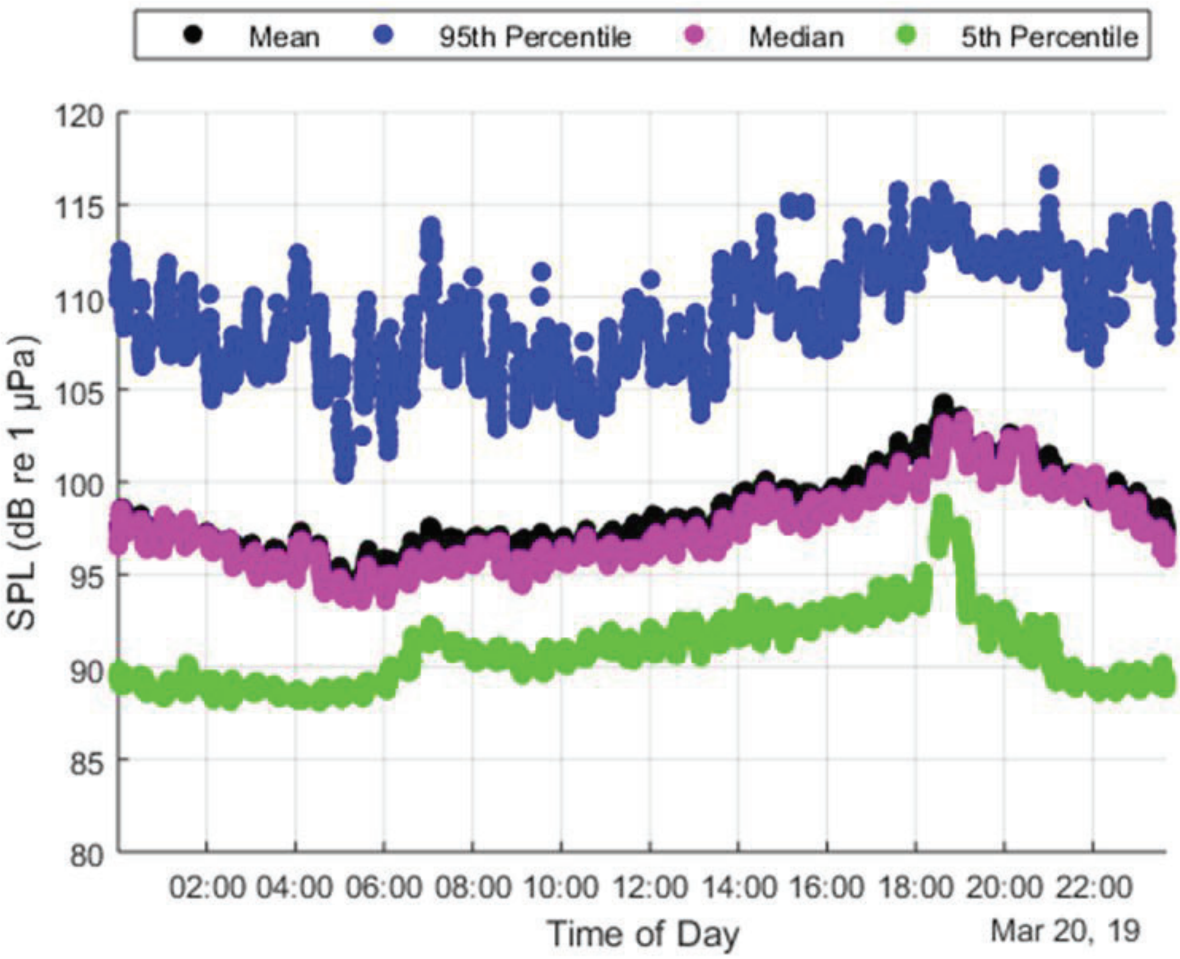


Figure 19 Descriptive statistics of the sound pressures levels over each 24 hour period between March and June 2019. Each data point represents the 1-min *Leq*.

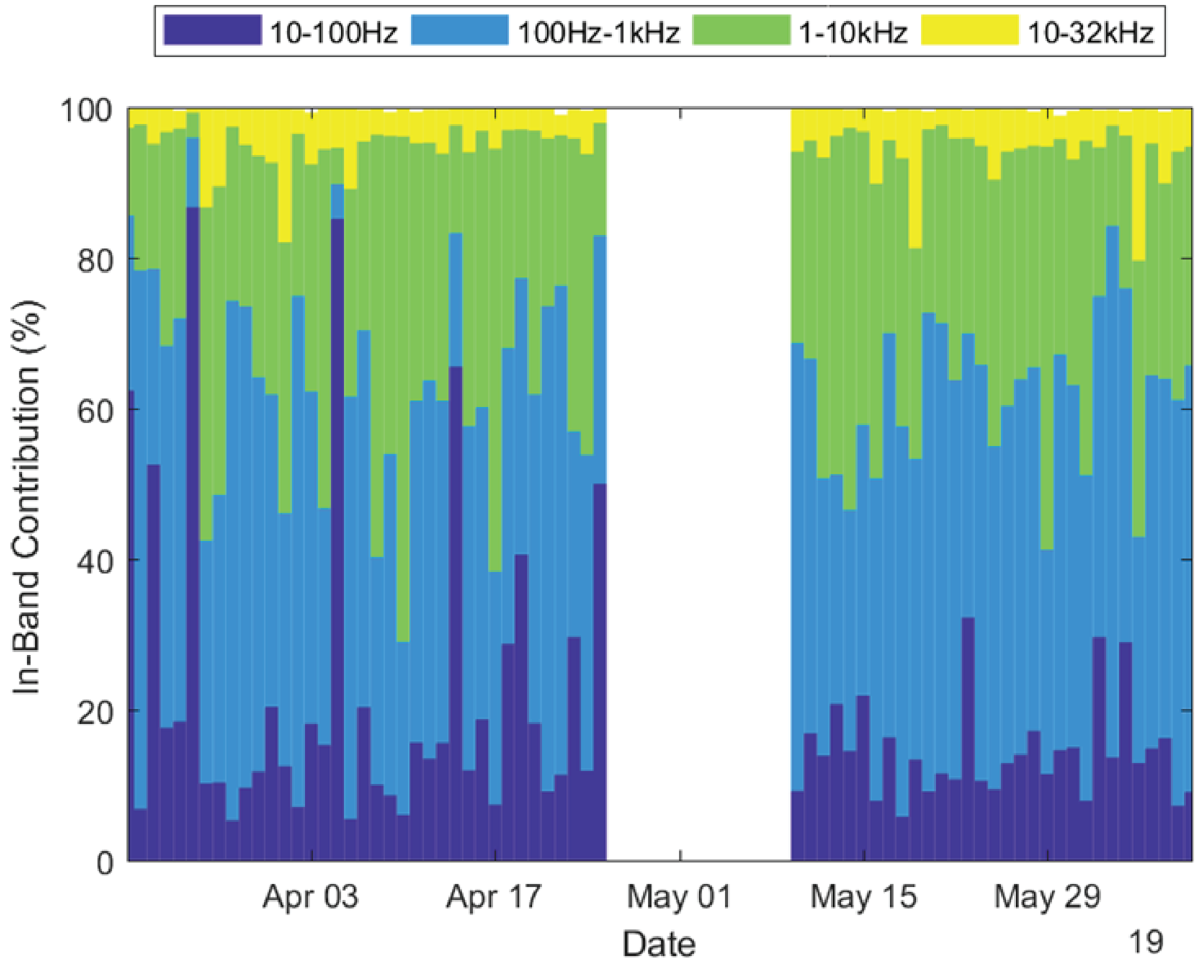


Figure 20 Daily in-band contributions (%) per day.

The in-band contribution shows the percentage contribution of each frequency band to the total broadband energy for each 24 hour period.

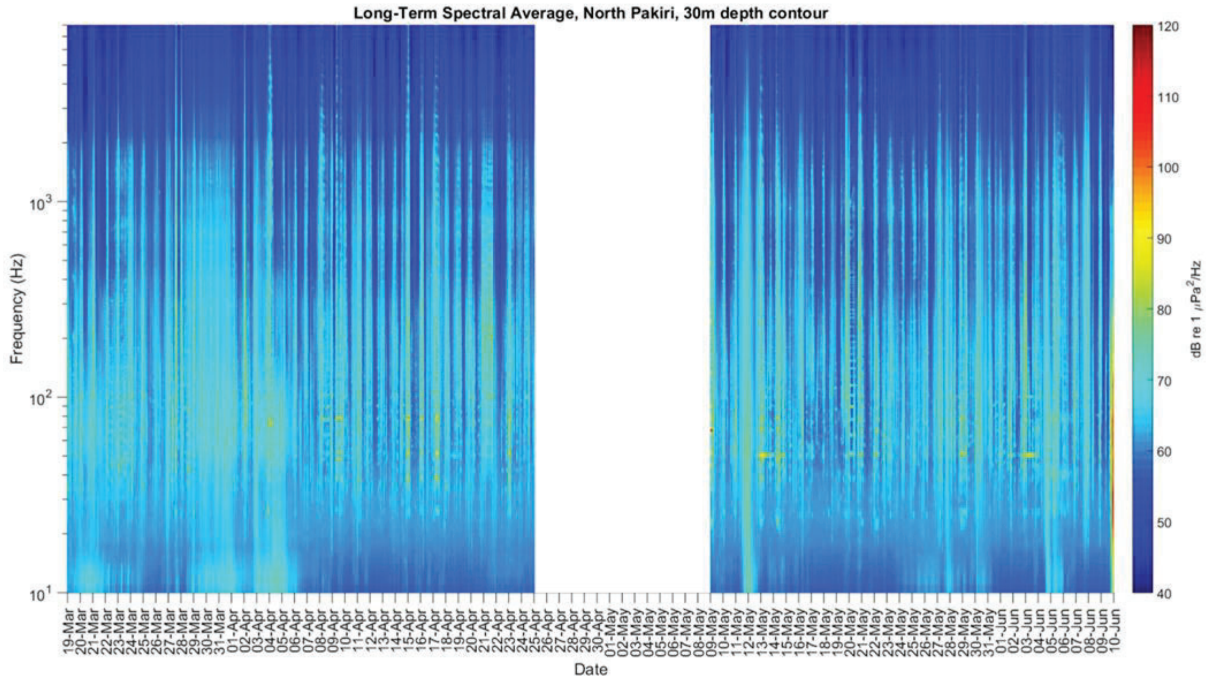


Figure 21 Long term spectral average (LTSA) of the power spectral density (*PSD*, dB re 1 μPa^2/Hz) over the monitoring period. No data were collected between the 25 April and 09 May.

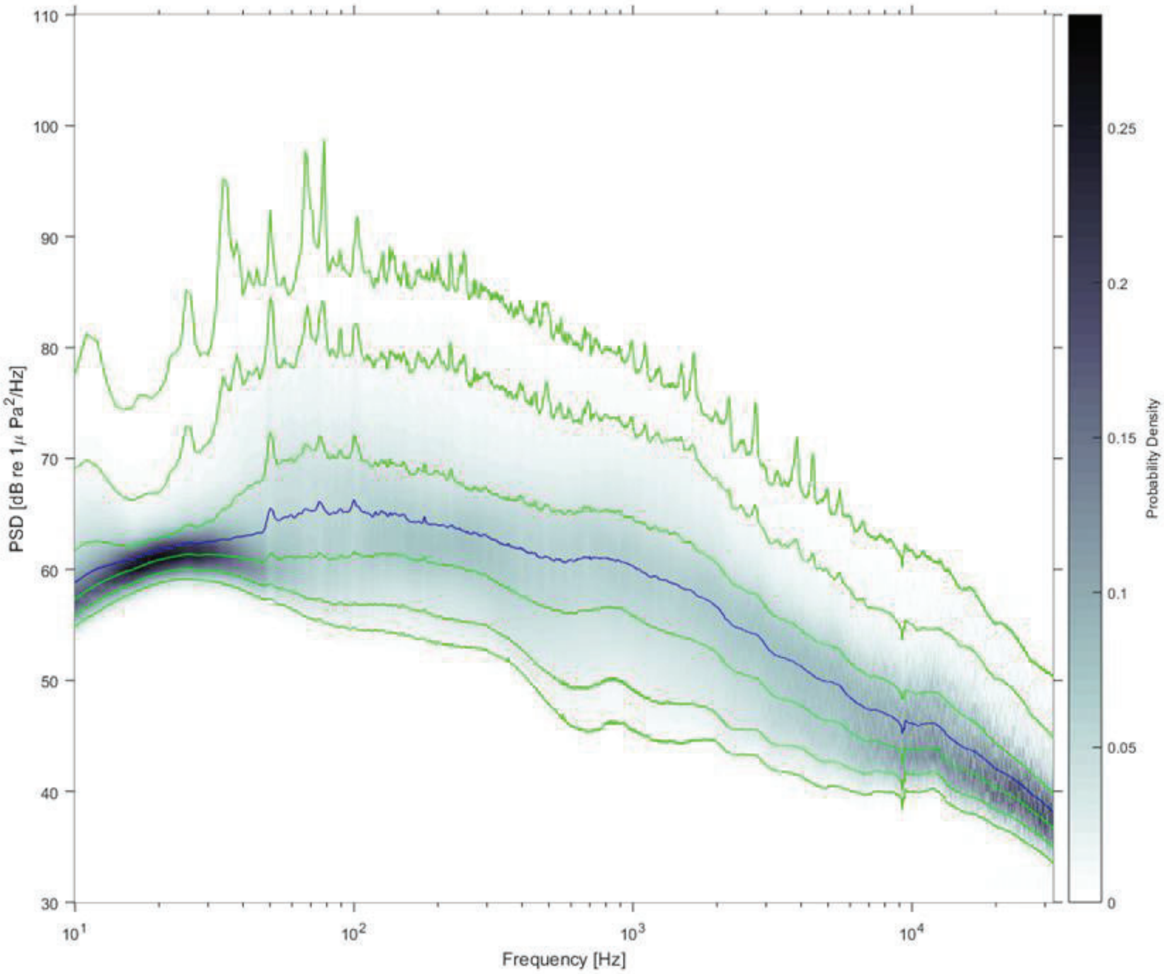


Figure 22 Plots showing the variation in power spectral densities (*PSD*, dB re 1 µPa²/Hz) and the spectral probability density for the monitoring period.

The blue line represented the median (50th percentile) levels, while the green lines, starting at the top line, represent the 99th, 95th, 75th, 25th, 5th and 1st percentile levels. The colour bar presents the spectral probability density from 10 Hz to 32 kHz. These plots were generated using the LTSA data shown in Figure 21.

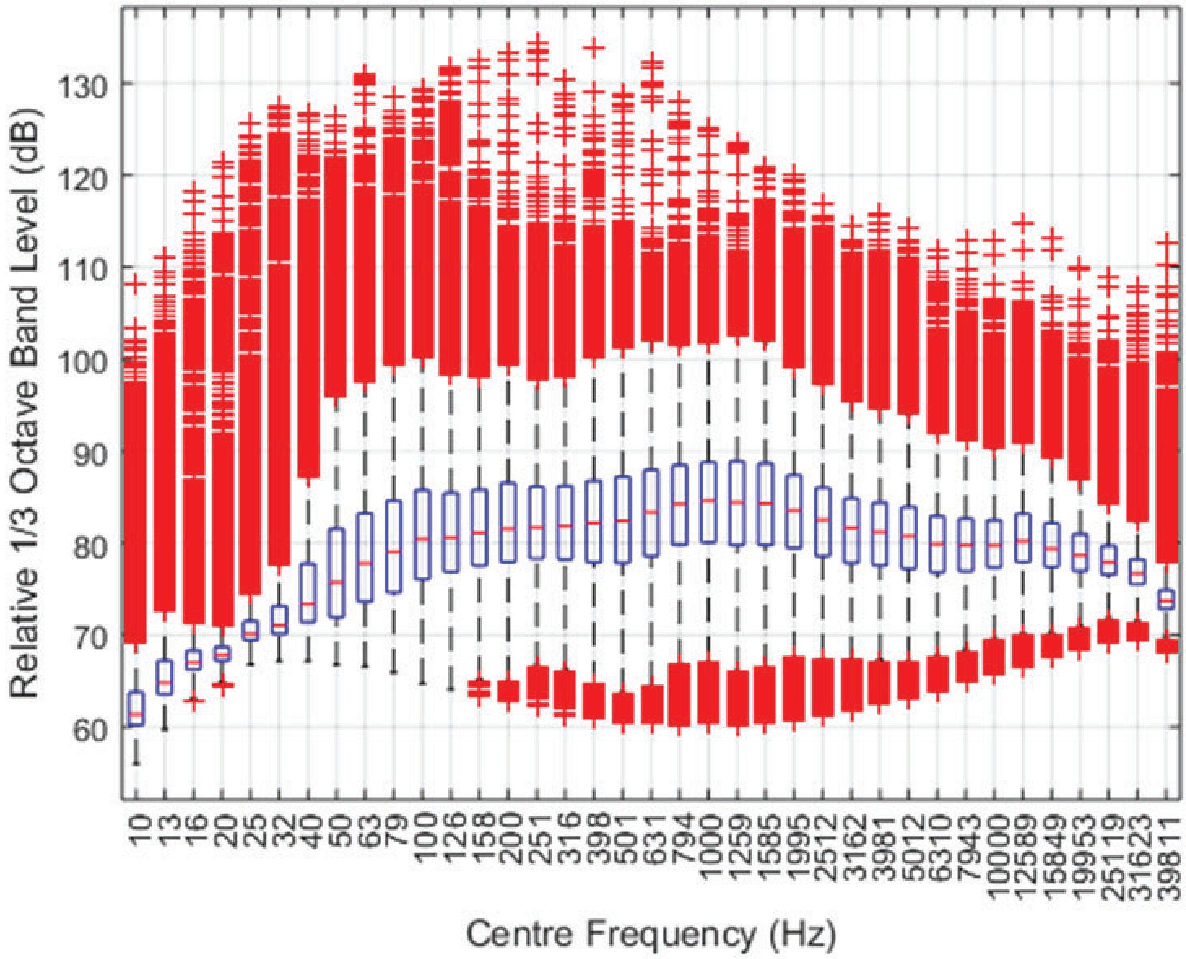


Figure 23 Boxplot showing the variation in the 1/3 octave band levels (dB re 1 µPa) for the monitoring period.

The plot was generated from the 1-min averages in each 1/3 octave band for each day of the monitoring period.

Appendix D Temperature and Salinity Data

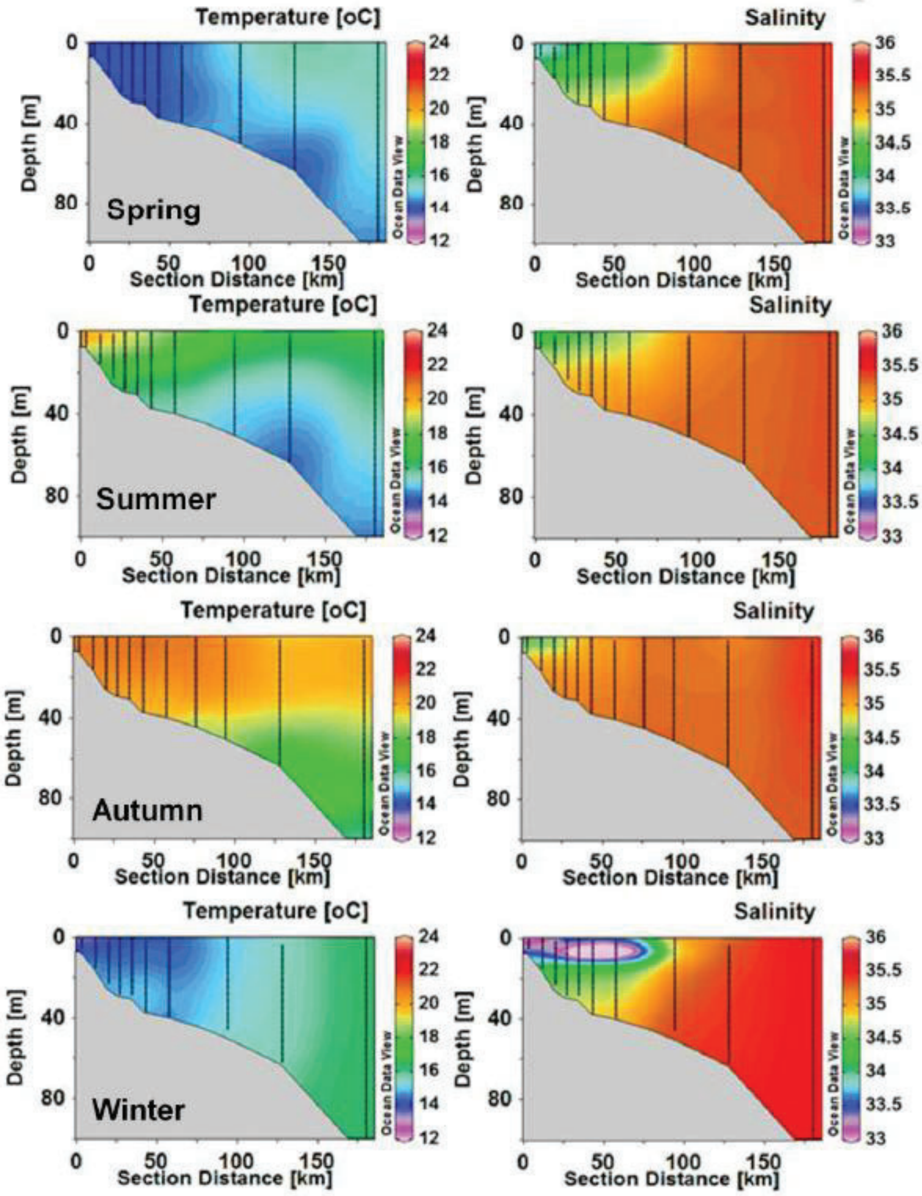


Figure 24: Thermo- and halocline depths used to calculate the sound speed profiles.

These figures have been taken directly from Zeldis (2013) and were measured from within the Hauraki Gulf off Coromandel.

Appendix E Underwater Noise Effects Modelling Results: The *William Fraser*

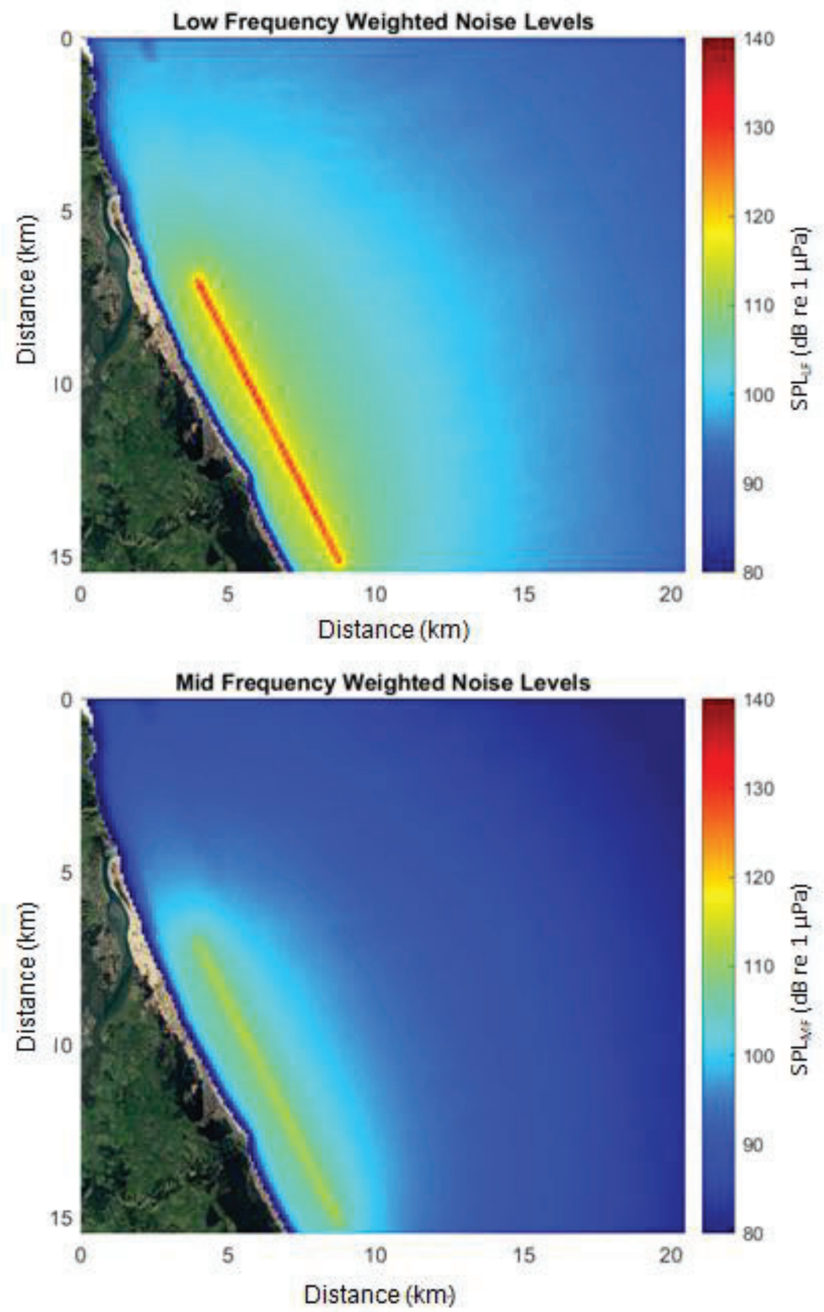


Figure 25: Modelled broadband (10Hz – 38 kHz) sound pressure levels for low-frequency and mid-frequency functional hearing groups for the TSHD *William Fraser* under full dredging conditions (pump on, draghead down, vessel underway and hopper being loaded).

Note the plots above show an entire dredge transect over multiple hours and is not the acoustic footprint or effect at a single point in time (as provided in Section 5.0).

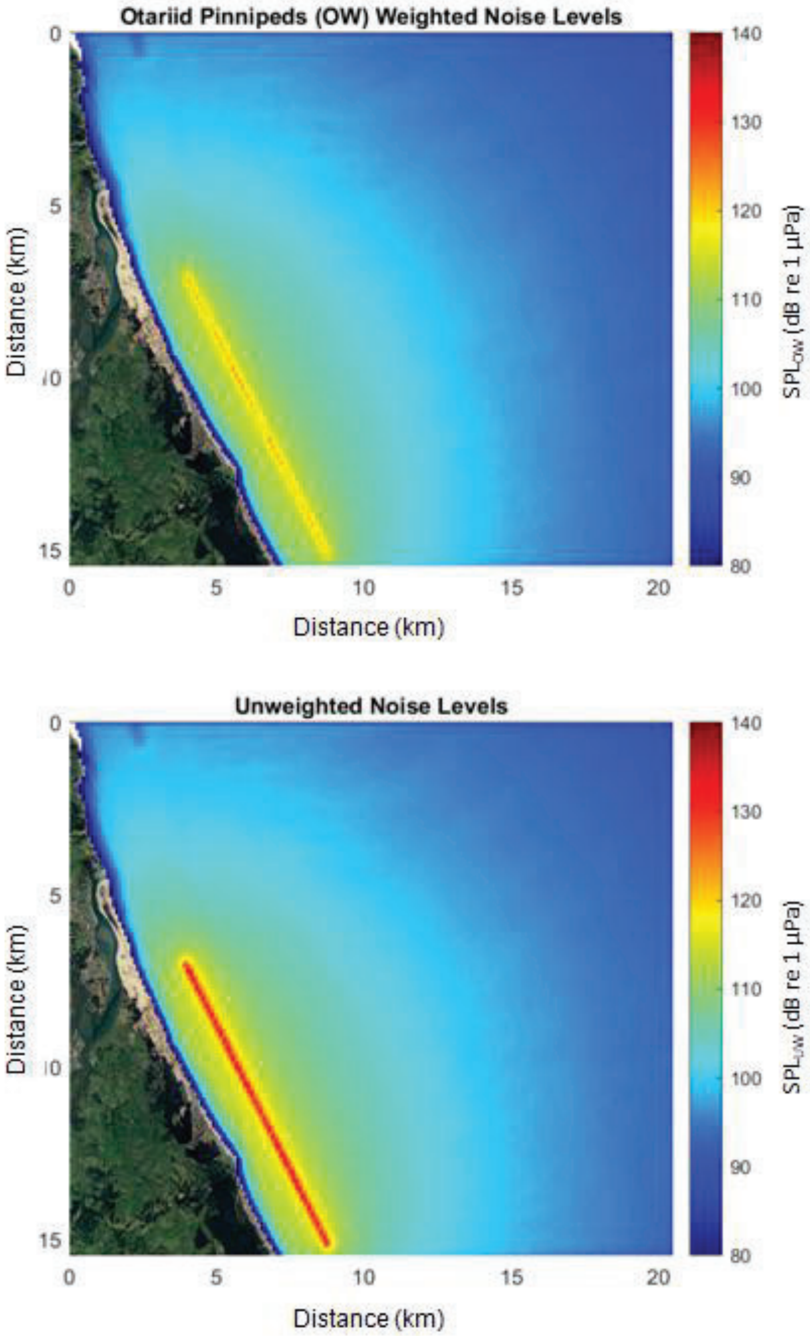


Figure 26: Modelled broadband (10Hz – 38 kHz) sound pressure levels for otariid pinniped functional hearing group (top panel) and unweighted levels (bottom panel) for the TSHD *William Fraser* under full dredging conditions (pump on, draghead down, vessel underway and hopper being loaded).

Note the plots above show an entire dredge transect over multiple hours and is not the acoustic footprint or effect at a single point in time (as provided in Section 5.0).

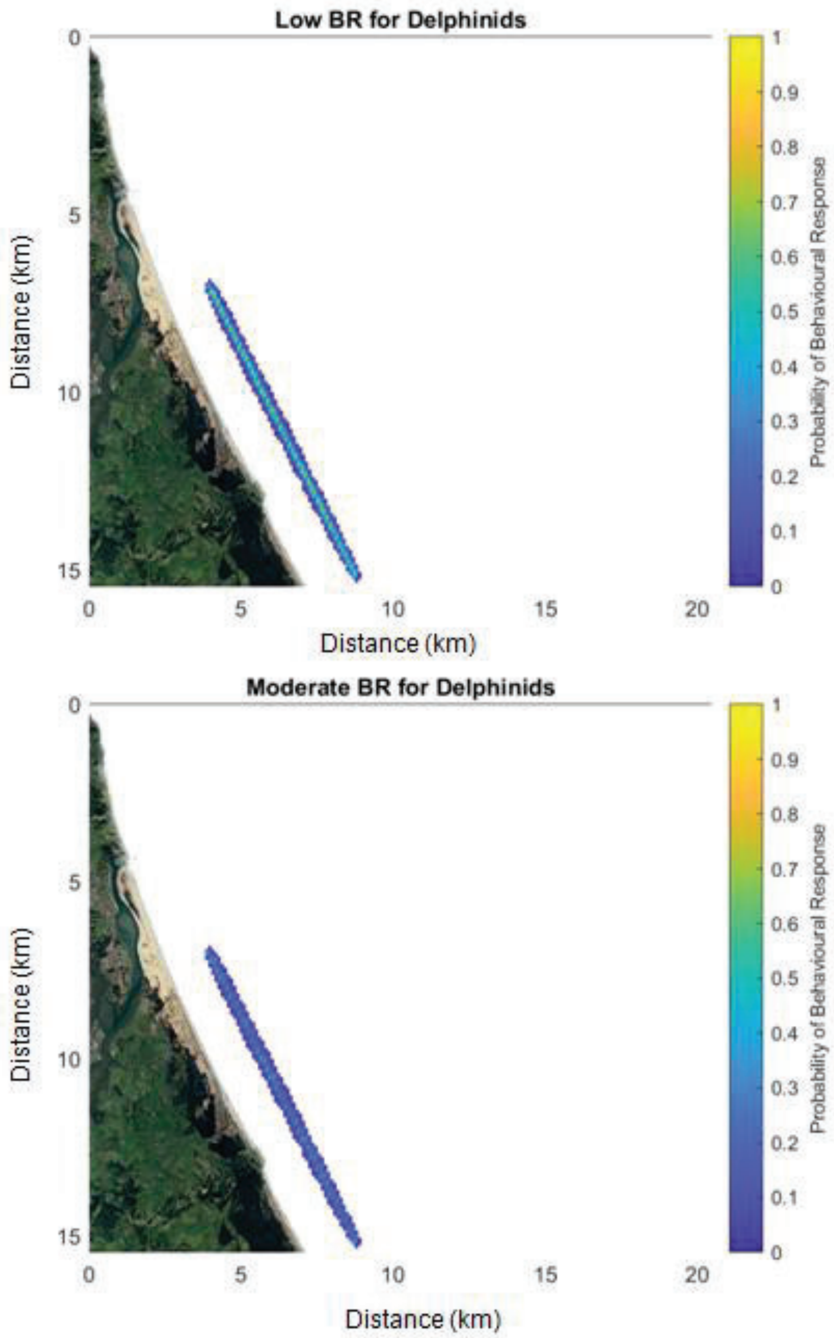


Figure 27: : Low and moderate behavioural response risk (top and bottom panels, respectively) for dolphin species (either bottlenose or common dolphins) for the TSHD *William Fraser* under full dredging (pump on, draghead down, vessel underway and hopper being loaded).

Note the plots above show an entire dredge transect over multiple hours and is not the acoustic footprint or effect at a single point in time (as provided in Section 5.0).

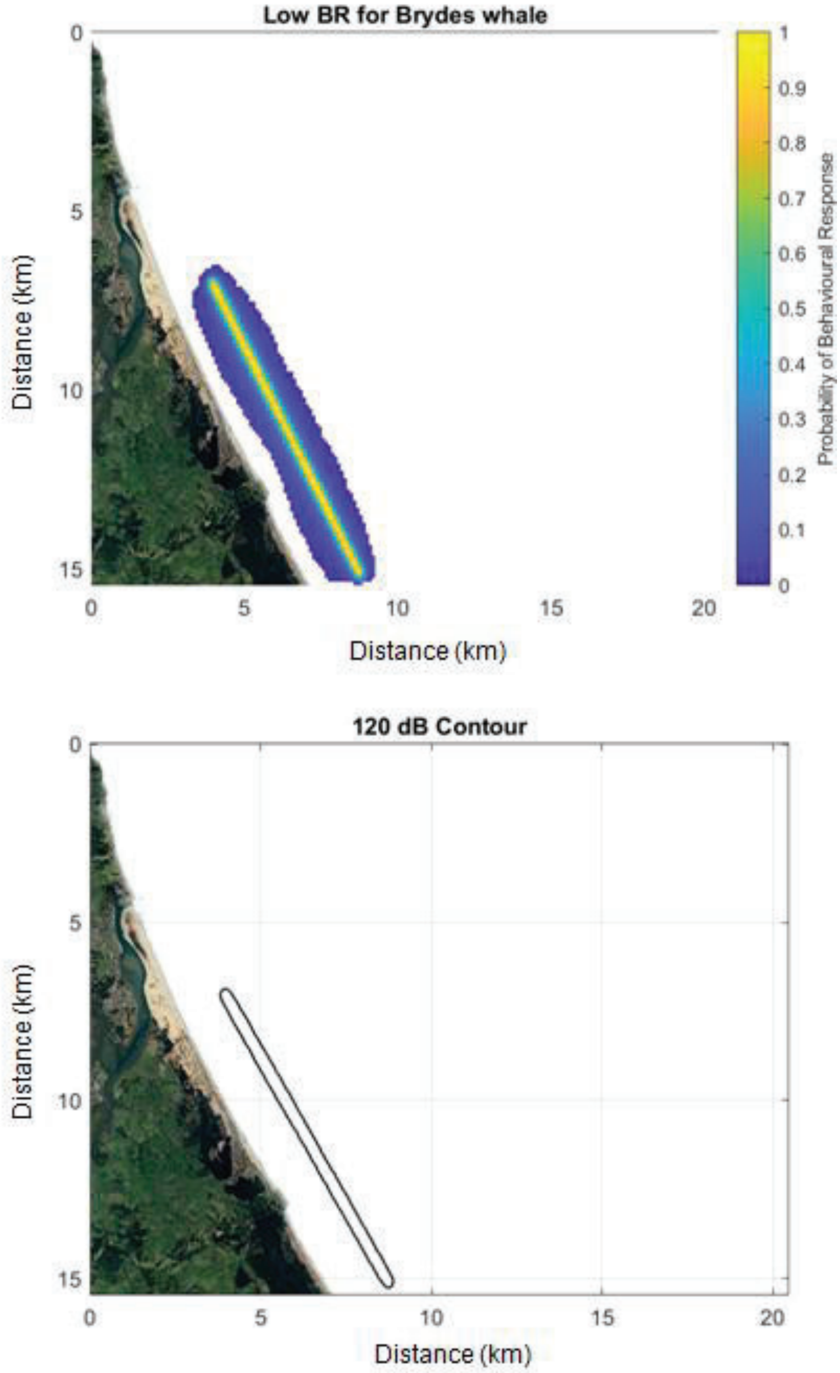


Figure 28: Low behavioural response risk for Bryde's whales (top panel) and the 120 dB re 1 µPa contour for comparison (bottom panel) for the TSHD *William Fraser* under full dredging (pump on, draghead down, vessel underway and hopper being loaded).

The dose-response formulas used to generate the probability of risk in Bryde's whales was based on a 50% risk occurring at that 120 dB re 1 µPa threshold as a conservative measure.

Note the plots above show an entire dredge transect over multiple hours and is not the acoustic footprint or effect at a single point in time (as provided in Section 5.0).

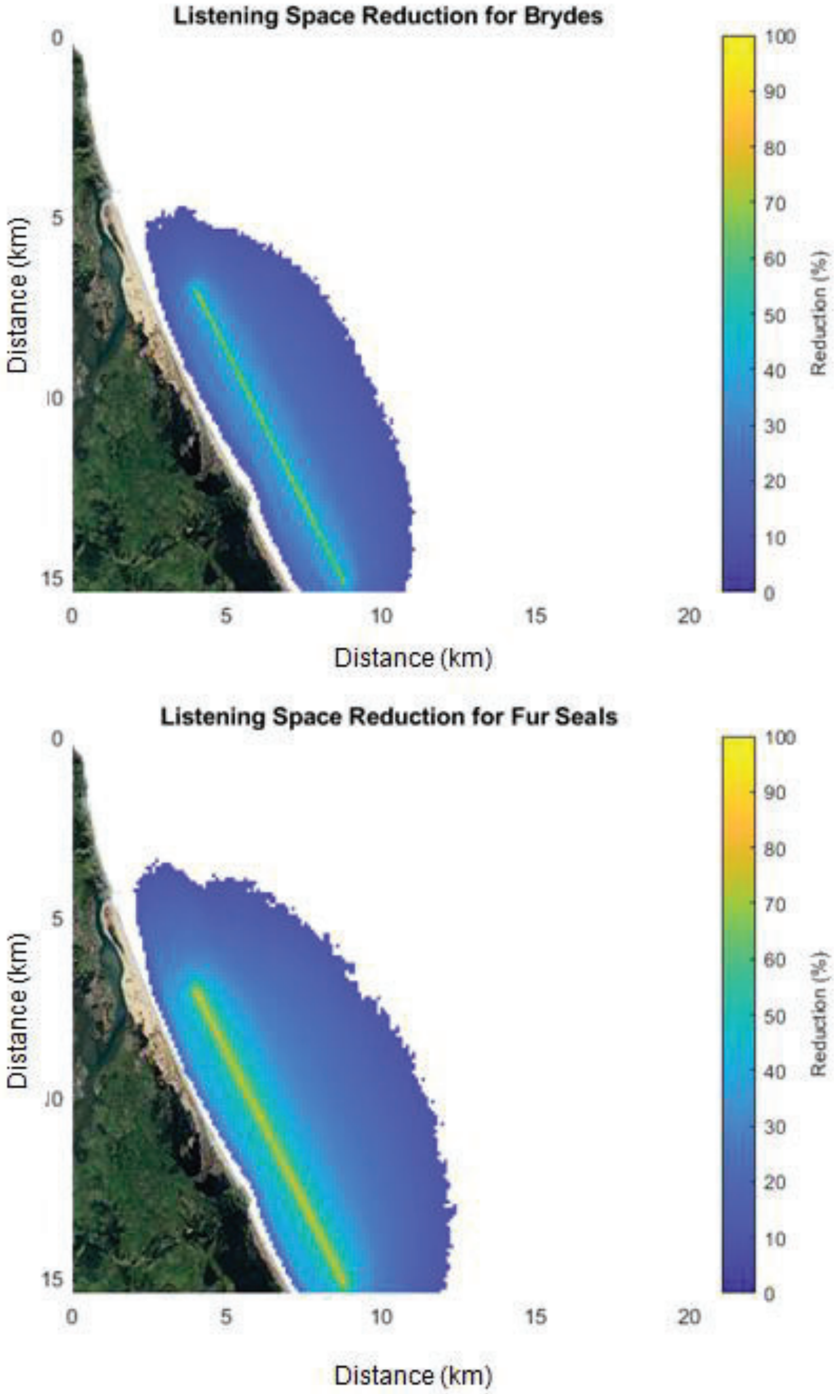


Figure 29: Plots showing the spatial extent of listening space reductions (*LSR*) for Bryde's whales and fur seals for the TSHD *William Fraser* under full dredging (pump on, draghead down, vessel underway and hopper being loaded).

Note the plots above show an entire dredge transect over multiple hours and is not the acoustic footprint or effect at a single point in time (as provided in Section 5.0).

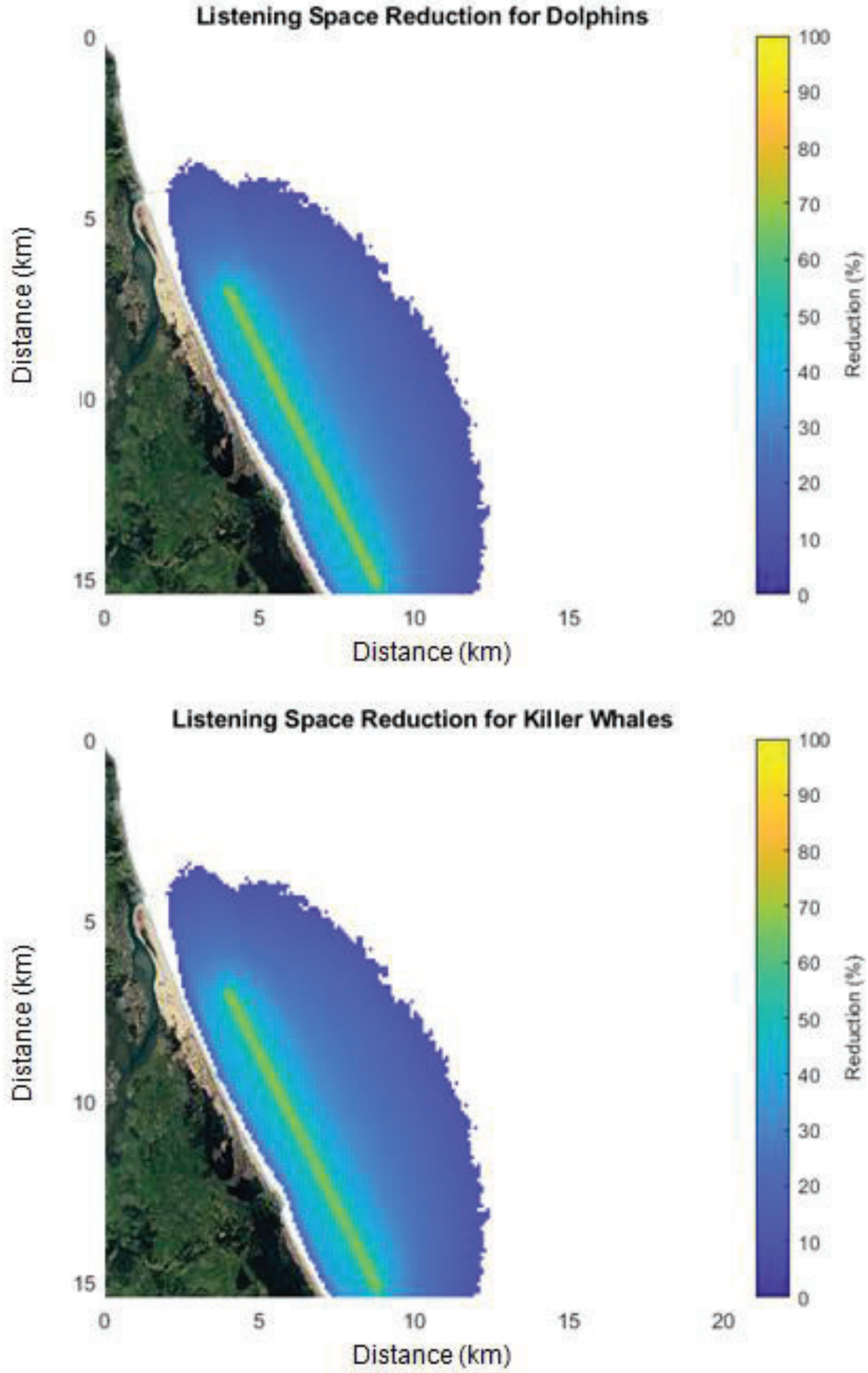


Figure 30: Plots showing the spatial extent of listening space reductions (*LSR*) for dolphins (bottlenose/common dolphins) and killer whales for the TSHD *William Fraser* under full dredging (pump on, draghead down, vessel underway and hopper being loaded).

Note the plots above show an entire dredge transect over multiple hours and is not the acoustic footprint or effect at a single point in time (as provided in Section 5.0).

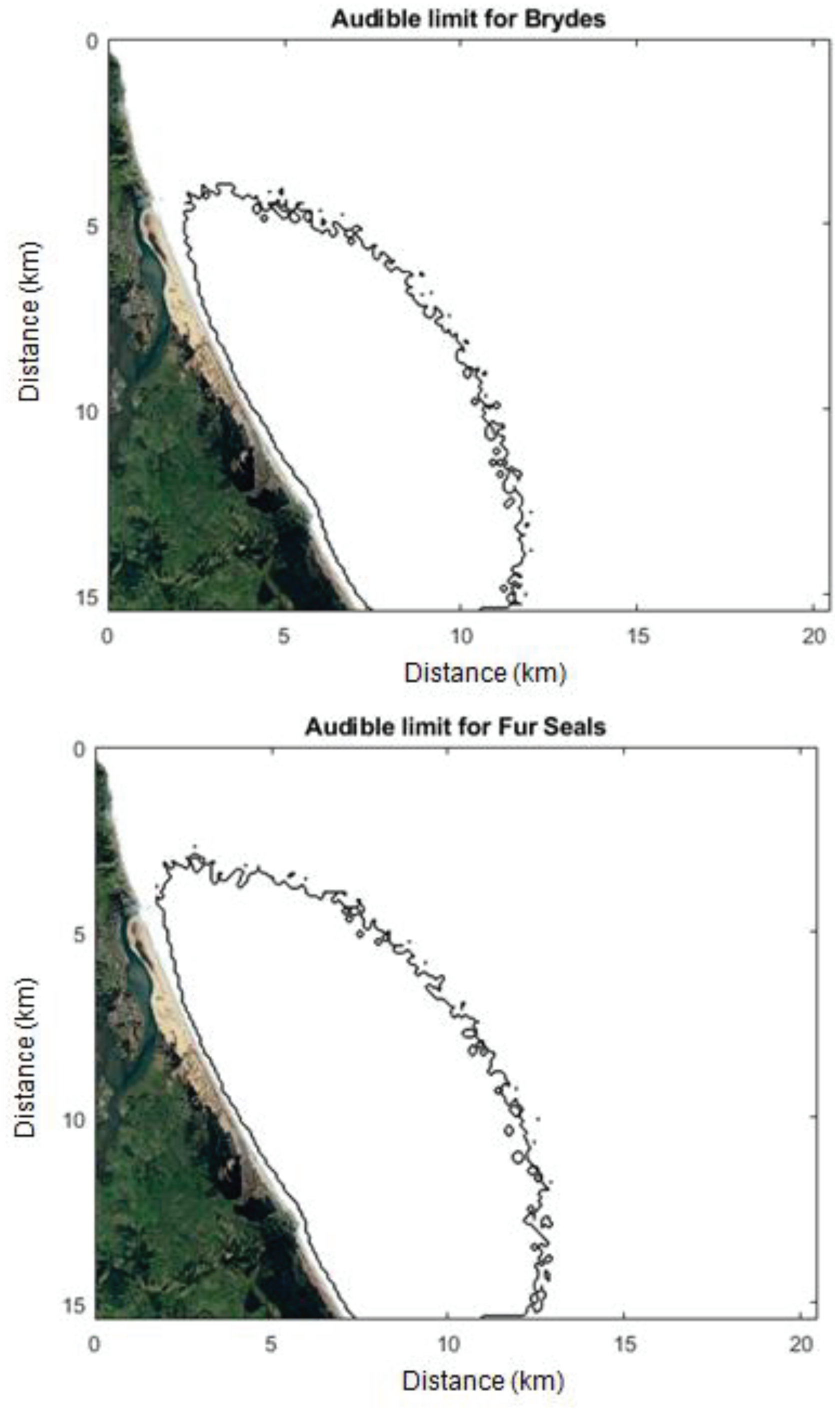
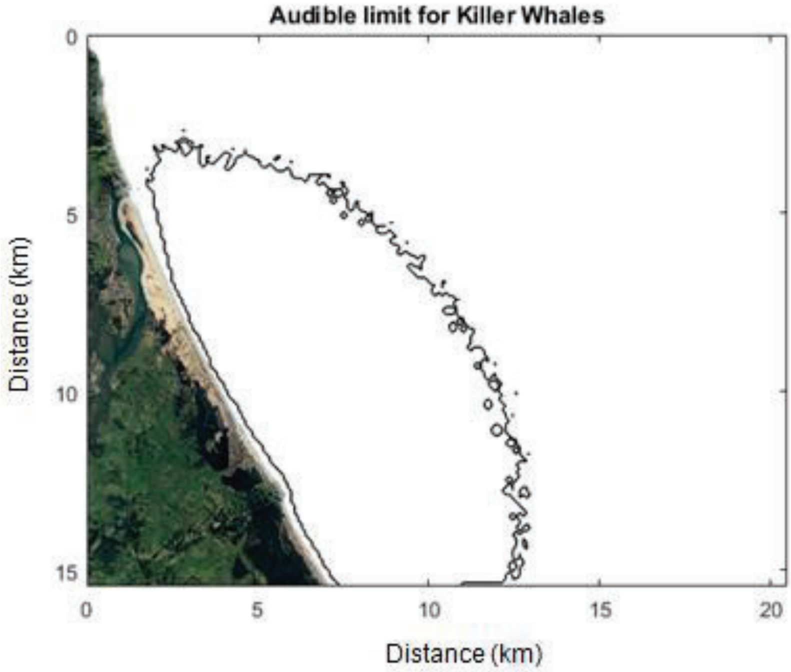


Figure 31: Plots showing the audibility limits for Byrde's whales and fur seals for the TSHD *William Fraser* under full dredging (pump on, draghead down, vessel underway and hopper being loaded).

Note the plots above show an entire dredge transect over multiple hours and is not the acoustic footprint or effect at a single point in time (as provided in Section 5.0).



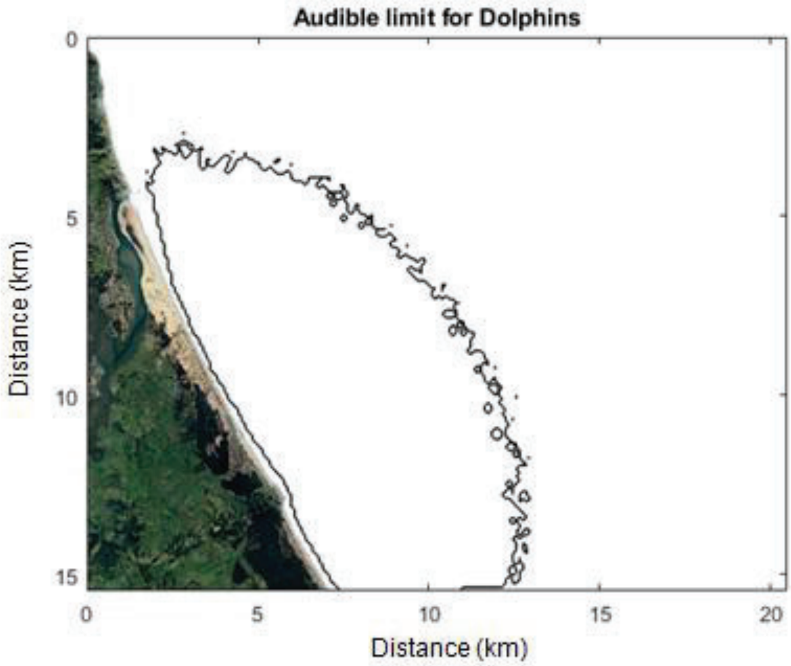


Figure 32: Plots showing the audibility limits for killer whales and dolphins (bottlenose/common dolphins) for the TSHD *William Fraser* under full dredging (pump on, draghead down, vessel underway and hopper being loaded).

Note the plots above show an entire dredge transect over multiple hours and is not the acoustic footprint or effect at a single point in time (as provided in Section 5.0).

Appendix F Model Ground-Truthing

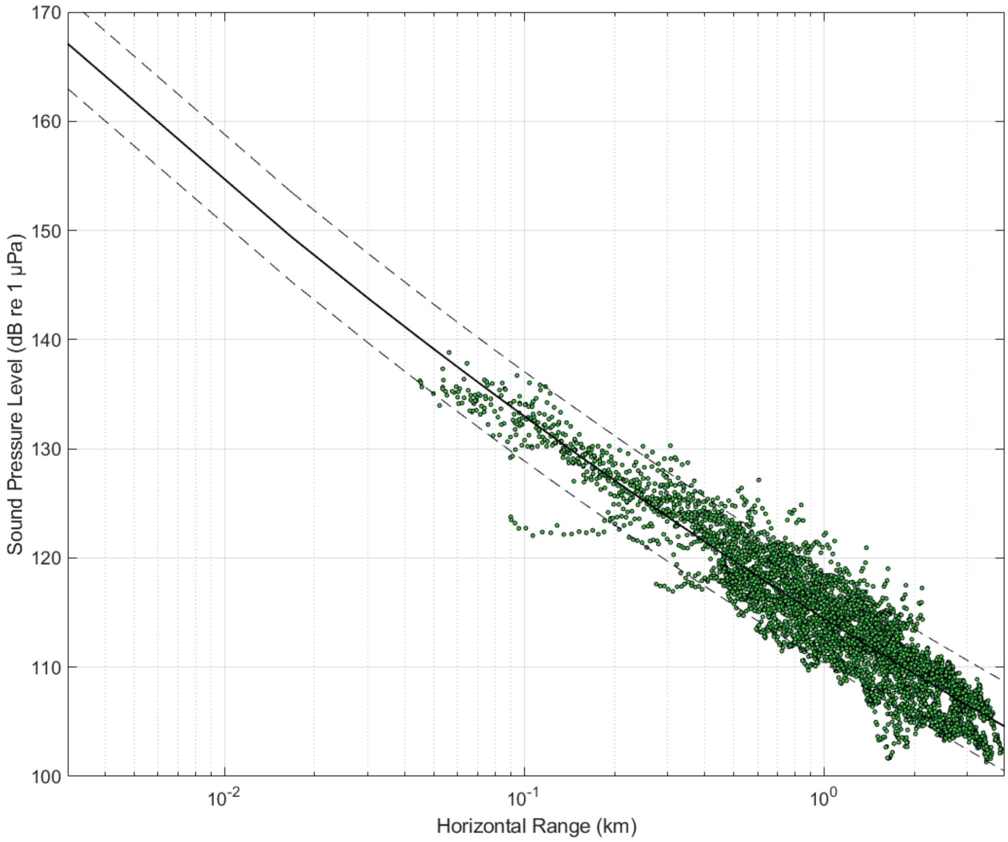


Figure 33: Curve-fitted data (and 90% confidence interval) from each of the six hydrophones in the measurement array. Each data point is the 10-second averaged *SPL* (10Hz – 48kHz) as the TSHD *William Fraser* operated in the northern consent area on 28 November 2019.

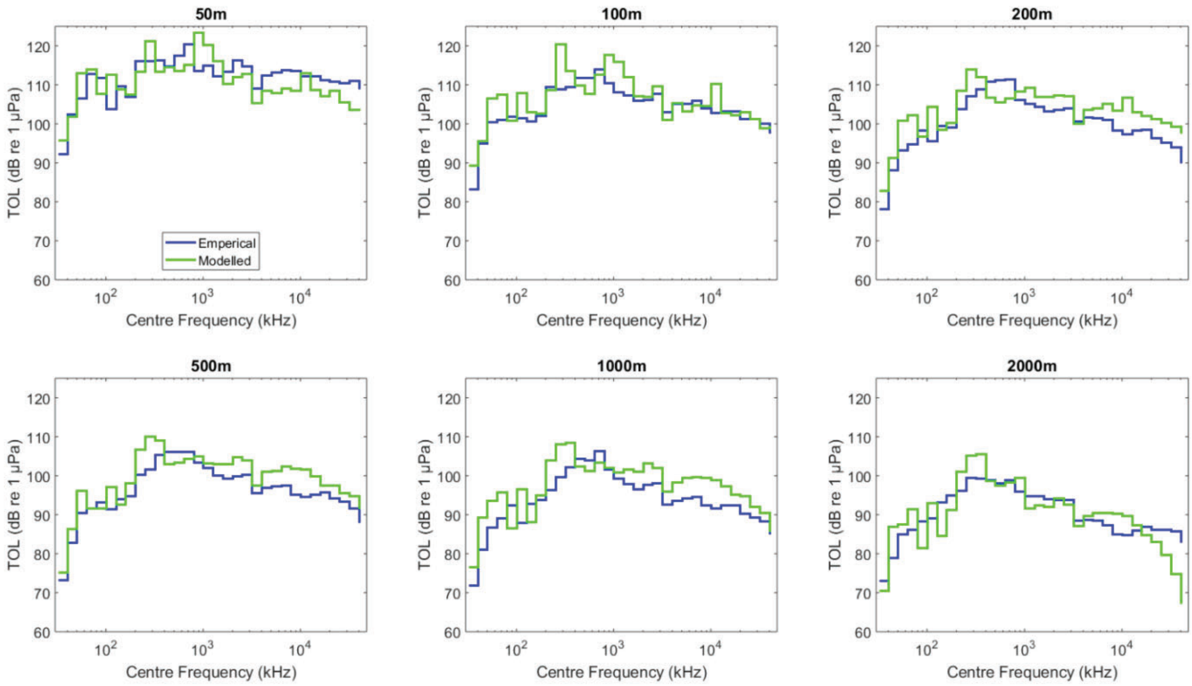


Figure 34: Modelled 1/3 octave sound pressures at various ranges compared to the empirical data.

The modelled levels incorporate the area-specific coefficient determined from the curve-fitted data for each three frequencies within a 1/3 octave band. This was based on the *Coastal Carrier* because the outputs are related to the frequency-dependent propagation loss only, which is independent to the type of dredger (given the modelled area is unchanged).