APPENDIX 14-A

Rationale for Inclusion or Exclusion of Other Certain and Reasonably Foreseeable Projects and Activities in the Cumulative Effects Assessment of Marine Mammals

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Appendix 14-A Rationale for Inclusion / Exclusion of Other Certain and Reasonably Foreseeable Projects and Activities in the Cumulative Effects Assessment of Marine Mammals

The assessment includes consideration of the potential for an interaction between a potential Project-related residual effect on marine mammals and the effects of other certain and reasonably foreseeable projects and activities on this VC. The rationale for inclusion or exclusion of each certain and reasonably foreseeable project and activity identified in **Table 8-8 Project and Activity Inclusion List**, from the cumulative effects assessment, is presented in **Table 14-A1**.

Table 14-A1 Rationale for Inclusion or Exclusion of Other Certain and Reasonably Foreseeable Projects in the Cumulative Effects Assessment of Marine Mammals

Other Certain and Reasonably Foreseeable Project/Activity	Included (I)/Excluded (E)	Rationale for Inclusion/Exclusion[1]
Projects		
BURNCO Aggregate Project, Gibsons, B.C.	E	No potential for cumulative interaction due to distant location from Roberts Bank, and number of vessels is anticipated to be small compared to overall traffic (**Table 8-8**); therefore, any cumulative change is not anticipated to be measurable.
Centerm Terminal Expansion, Vancouver, B.C.	E	Potential for cumulative interaction with RBT2 as the project will contribute to underwater noise from vessel traffic within the Strait of Georgia; however, the number of vessels is anticipated to be small compared to overall traffic (**Table 8-8**), and any cumulative change is not anticipated to be measurable.
Fraser Surrey Docks Direct Coal Transfer Facility, Surrey, B.C.	I	Potential for cumulative interaction with marine mammals as underwater noise is anticipated from barge and vessel traffic.
Gateway Pacific Terminal at Cherry Point and associated BNSF Railway Company Rail Facilities Project, Blaine, Washington	I	Potential for cumulative interaction with marine mammals as underwater noise is anticipated from barge and bulk carrier traffic.
Gateway Program - North Fraser Perimeter Road Project, Coquitlam, B.C.	E	Not relevant to this VC assessment due to land-based nature of project.
George Massey Tunnel Replacement Project, Richmond and Delta, B.C.	E	No potential for cumulative interaction due to Project location upstream on Fraser River.

Other Certain and Reasonably Foreseeable Project/Activity	Included (I)/Excluded (E)	Rationale for Inclusion/Exclusion[1]
Kinder Morgan Pipeline Expansion Project, Strathcona County, Alberta to Burnaby, B.C.	I	Potential for cumulative interaction with marine mammals as the project is anticipated to contribute to underwater noise from tanker traffic within the Strait of Georgia.
Lehigh Hanson Aggregate Facility, Richmond, B.C.	E	No potential for cumulative interaction with marine mammals as project is expected to have a negligible contribution to future underwater noise levels.
Lions Gate Wastewater Treatment Plant Project, District of North Vancouver, B.C.	E	No potential for cumulative interaction with marine mammals due to land-based project; noise generated from discharges to Burrard Inlet anticipated to be negligible.
North Shore Trade Area Project – Western Lower Level Route Extension, West Vancouver, B.C.	E	Not relevant to VC assessment due to land-based nature of project.
Pattullo Bridge Replacement Project, New Westminster and Surrey, B.C.	E	Not relevant to VC assessment due to land-based nature of project.
Southlands Development, Delta, B.C.	E	Not relevant to VC assessment due to land-based nature of project.
Vancouver Airport Fuel Delivery Project, Richmond, B.C.	I	Potential for cumulative interaction with marine mammals as the project will contribute to underwater noise from tanker and barge traffic within the Strait of Georgia.
Woodfibre LNG Project, Squamish, B.C.	E	Potential for cumulative interaction with RBT2 as the project will contribute to underwater noise from vessel traffic within the Strait of Georgia; however, the number of vessels is anticipated to be small compared to overall traffic (**Table 8-8**), and any cumulative change is not anticipated to be measurable.
Activities		
Incremental Road Traffic Associated with RBT2	E	Not relevant to this VC assessment due to land-based nature of activity.
Incremental Train Traffic Associated with RBT2	E	Not relevant to this VC assessment due to land-based nature of activity.
Incremental Marine Vessel Traffic Associated with RBT2	I	Potential for cumulative interaction with marine mammals as the activity will contribute to underwater noise from container vessel traffic within the Strait of Georgia.

APPENDIX 14-B

Southern Resident Killer Whale Noise Exposure and Acoustic Masking Technical Report

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PROPOSED ROBERTS BANK TERMINAL 2
TECHNICAL REPORT

Marine Mammals

Southern Resident Killer Whale Underwater Noise Exposure and Acoustic Masking Study

Prepared for:
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December 2014

SMRU
CANADA

Technical Report / Technical Data Report Disclaimer

The Canadian Environmental Assessment Agency determined the scope of the proposed Roberts Bank Terminal 2 Project (RBT2 or the Project) and the scope of the assessment in the Final Environmental Impact Statement Guidelines (EISG) issued January 7, 2014. The scope of the Project includes the project components and physical activities to be considered in the environmental assessment. The scope of the assessment includes the factors to be considered and the scope of those factors. The Environmental Impact Statement (EIS) has been prepared in accordance with the scope of the Project and the scope of the assessment specified in the EISG. For each component of the natural or human environment considered in the EIS, the geographic scope of the assessment depends on the extent of potential effects.

At the time supporting technical studies were initiated in 2011, with the objective of ensuring adequate information would be available to inform the environmental assessment of the Project, neither the scope of the Project nor the scope of the assessment had been determined.

Therefore, the scope of supporting studies may include physical activities that are not included in the scope of the Project as determined by the Agency. Similarly, the scope of supporting studies may also include spatial areas that are not expected to be affected by the Project.

This out-of-scope information is included in the Technical Report (TR)/Technical Data Report (TDR) for each study, but may not be considered in the assessment of potential effects of the Project unless relevant for understanding the context of those effects or to assessing potential cumulative effects.

EXECUTIVE SUMMARY

The Roberts Bank Terminal 2 Project (RBT2 or Project) is a proposed new three-berth marine terminal at Roberts Bank in Delta, B.C. that could provide 2.4 million TEUs (twenty-foot equivalent units) of additional container capacity annually. The Project is part of Port Metro Vancouver's Container Capacity Improvement Program, a long-term strategy to deliver projects to meet anticipated growth in demand for container capacity to 2030.

The Project is located in federally designated critical habitat for the endangered southern resident killer whale (SRKW, *Orcinus orca*), therefore, it is important to understand the potential effects of underwater noise from the Project on SRKW. The overall objective of the SRKW Underwater Noise Exposure and Acoustic Masking Study was to determine when underwater noise produced both from Project-related activities and/or regional vessel traffic could result in behavioural responses and/or echolocation masking of SRKW.

Four scenarios were investigated:

- S1 - Existing commercial vessel traffic (2012);
- S2 - Future commercial vessel traffic with no new projects except RBT2, and future incremental vessel traffic associated with RBT2 (2030)[1];
- S3 - Future commercial vessel traffic due to certain and foreseeable projects without RBT2, or incremental vessel traffic associated with RBT2 (2030); and
- S4 - Future commercial vessel traffic with RBT2, incremental shipping traffic associated with RBT2, and future vessel traffic due to certain and foreseeable projects (2030).

For each of the scenarios, this study estimated:

- The spatial extent of exposure of SRKW to underwater noise from commercial vessel traffic;
- The resulting number of low-severity and moderate-severity behavioural responses; and
- The degree of additional acoustic masking that might occur in cases where no behavioural responses were predicted.

Noise models included transiting of commercial vessels within and outside of PMV jurisdiction, and berthing within PMV jurisdiction. Construction noise was not included in this modelling study (see Regional Commercial Vessel Traffic Underwater Noise Modelling Study (JASCO 2014)).

[1] Expected conditions between 2012 and 2030 include no new projects but increases in vessel traffic at Westshore and Deltaport terminals i.e., Deltaport Terminal Road and Railway Improvement Project (DTRRIP).

To achieve the Study objectives, underwater noise from commercial vessel traffic was examined in two spatial areas i.e., 'regional' and 'focused'. The regional model area encompassed the largest area that an acoustic propagation model could be technically conducted where there was adequate SRKW relative density data to provide good predictive results (see Marine Mammal Habitat Use Studies - Southern Resident Killer Whale Network Sighting Synthesis (Hemmera 2014)). Due to seasonal differences in sound propagation and SRKW relative density in summer and winter, the regional model was conducted during both seasons. Estimated average broadband underwater noise in January and July, and relative SRKW density estimates for summer and winter were spatially overlaid and normalised to generate maps of noise exposure levels.

For all four scenarios, Haro Strait, Boundary Pass, and Active Pass had the highest noise exposure. Calculations of change in noise exposure from existing underwater conditions (S1) to underwater noise produced from existing regional vessel activities and incremental shipping traffic associated with the Project (S2) showed that the increase was small, but concentrated in Haro Strait, Boundary Pass, and the approaches to RBT2. Changes from S1 to S3 and S4 were also small and concentrated in the same areas and the southern approaches to Rosario Strait. Less noise exposure was calculated for winter than summer, which was driven by low SRKW density in winter. Underwater noise from commercial vessel traffic did not differ greatly from S1 to S2, S3, or S4 indicating that underwater noise from commercial vessel traffic was high in existing conditions. Mean broadband [10 Hz to 63 kHz] noise levels across the regional study area for S1 were 117.54 and 122.14 dB re 1 μPa for summer and winter, respectively. These mean levels increased by less than 0.3 dB under the future scenarios (JASCO 2014).

A finer-scale "focused" simulation model was developed to estimate the number of low-severity and moderate-severity behavioural responses by SRKW, and also any additional amount of acoustic masking under the four scenarios. High-severity behavioural responses were not predicted to occur as a result of underwater noise produced by the Project or regional commercial vessel traffic. Because of the added complexity of the model and the additional computational requirements for the noise inputs, the spatial extent of the focused model area was smaller than the regional model, but still incorporated most of high-use SRKW critical habitat in Canada and the U.S. Model inputs included broadband underwater noise estimates during a 24-hour period, SRKW relative density, SRKW monthly probability of occurrence, and behavioral disturbance dose-response curves to estimate the number of low-severity and moderate-severity behavioural responses. Estimates of 50 kHz power spectral density (PSD) levels were used with a click masking model to estimate, only when a behavioural response was not predicted, the number of additional minutes a whale might instead experience masking in a year. The focused model was run for the 365 days of a modelled year and repeated 1,000 times to generate appropriate estimate variability. Inputs were varied as appropriate based on time of year.

From scenario S1 to S2, there was an increase of 74 low-severity (5.0% increase) and 26 moderate-severity (4.2% increase) behavioural responses per year per SRKW individual in the focused model area. The variability in behavioural response estimates within scenarios was larger than the difference between scenarios. Masking increased less than 1% of the year for all scenarios; however, masking was only calculated when low-severity and moderate-severity behavioural responses were not occurring. No high-severity behavioural responses were predicted.

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AIS – Automatic Identification System

BCCSN – B.C. Cetacean Sightings Network

CCIP – Container Capacity Improvement Program

CI – Confidence Interval

DFO – Fisheries and Oceans Canada

LSA – Local Study Area

NOAA – National Oceanic and Atmospheric Administration

PCoD – Population Consequences of Disturbance

RBT2 – Roberts Bank Terminal 2

PSD – Power Spectral Density

SARA – Species at Risk Act

SRKW – Southern Resident Killer Whale

SL – Source Level

SPL – Sound Pressure Level

TEU – Twenty-foot Equivalent Unit

VTOSS – Vessel Traffic Operational Support System

1.0 INTRODUCTION

This section provides an overview of the Southern Resident Killer Whale (*Orcinus orca*; SRKW) Underwater Noise Exposure and Acoustic Masking Study, including Project background information, rationale for the study components, and major objectives.

1.1 PROJECT BACKGROUND

The Roberts Bank Terminal 2 Project (RBT2 or Project) is a proposed new three-berth marine terminal at Roberts Bank in Delta, B.C. that could provide 2.4 million TEUs (twenty-foot equivalent units) of additional container capacity annually. The Project is part of Port Metro Vancouver's Container Capacity Improvement Program, a long-term strategy to deliver projects to meet anticipated growth in demand for container capacity to 2030.

Port Metro Vancouver has retained Hemmera to undertake environmental studies related to the Project. This technical report has been prepared by SMRU Canada Ltd. on behalf of Hemmera and describes the results of the SRKW Underwater Noise Exposure and Acoustic Masking Study.

1.2 SRKW NOISE EXPOSURE OVERVIEW

This technical report describes (1) findings from a regional model that estimates the spatial overlap between the relative density of SRKW and underwater noise from commercial vessel traffic, and (2) a focused model that estimates the number of low-severity and moderate-severity behavioural responses plus any additional amount of time that acoustic masking of SRKW echolocation clicks occur due to underwater noise from commercial vessel traffic. High-severity behavioural responses were not predicted to occur as a result of underwater noise produced by the Project or regional commercial vessel traffic. Study components, major objectives, and a brief overview are provided in **Table 1**.

Table 1 SRKW Noise Exposure Study Components and Major Objectives

Component	Major Objective	Brief Overview
1) Regional SRKW Noise Exposure	• Model the spatial overlap between SRKW relative density and underwater noise from commercial vessel traffic.	• Identify spatial overlap of SRKW and underwater noise from commercial vessel traffic during summer and winter periods under existing conditions and three future development scenarios.
2) Focused SRKW Noise Exposure	• Estimate behavioural responses of SRKW and/or masking of SRKW signals using SRKW relative density and underwater noise from commercial vessel traffic.	• Calculate the number of low-severity and moderate-severity behavioural responses of SRKW to underwater noise produced from commercial vessel traffic under existing conditions and three future development scenarios in the focused model area and in the local study area (LSA). • Estimate any additional amount of time that masking of SRKW echolocation clicks by underwater noise from commercial vessel traffic may be occurring under existing conditions and three future development scenarios in the focused model area and in the local study area (LSA).

Anthropogenic ocean noise has increased over the last few decades, primarily as a result of commercial shipping traffic (Hildebrand 2009).The Strait of Georgia is a highly used area with maritime traffic comprised of domestic and international vessels. The overall size of vessels is expected to increase in B.C. waters, particularly those in the largest size class (i.e., Post-Panamax) (Ministry of Transportation and Ministry of Small Business and Economic Development 2005).

In many parts of the world, shipping noise is the dominant source of underwater noise at frequencies that are fundamental to the way that marine mammals behave (Wright 2008). Marine mammals depend on sound for a wide range of activities, including communication, foraging, and navigation (Tyack 2008), therefore, anticipated increase of shipping traffic in the Strait of Georgia and nearby areas is of concern for marine mammal populations. The SRKW, which is listed as an Endangered species under Schedule 1 of the *Species at Risk Act* (*SARA*) (DFO 2011) uses waters off the southern coast of B.C. and within the proposed Project area, which are federally designated critical habitat (DFO 2011). SRKWs are at risk because of a small population size of 78 individuals (as of January 15th 2015) (Center for Whale Research 2015), low reproductive rate, and a variety of anthropogenic threats (e.g., decreased availability and quality of prey, environmental contamination, and physical and acoustic disturbance) (DFO 2011).

The effects of underwater noise exposure on the long-term population viability of SRKWs are poorly understood. The Recovery Strategy for the Northern and Southern Resident Killer Whales in Canada (DFO 2011) has identified this as a data gap and listed acoustic disturbance as a threat to SRKW. Underwater noise produced during the construction and operation of RBT2, and potential cumulative effects from regional commercial vessel traffic, has the potential to behaviourally disturb individuals and mask SRKW echolocation clicks and social calls necessary for foraging and communication.

An understanding of SRKW habitat use and the underwater soundscape is necessary to understand potential effects of underwater noise produced from RBT2 activities and commercial vessel traffic on SRKWs. The SRKW Underwater Noise Exposure and Acoustic Masking Study builds upon the SRKW Network Sighting Synthesis Study (Hemmera 2014) by incorporating summer and winter SRKW relative density maps with underwater noise from commercial vessel traffic outlined in the Regional Commercial Vessel Traffic Underwater Noise Modelling Study (JASCO 2014) into a predictive exposure model. In the regional model, month-long underwater noise data were used to determine areas where SRKW and underwater noise from commercial vessel traffic currently overlap, and how these areas are expected to change with the addition of RBT2/incremental vessel traffic associated with RBT2 and/or other projects in the study area.

For the focused model, SRKW relative densities during summer and winter were used during two representative 24-hour periods of underwater noise to identify areas of SRKW-underwater noise overlap, and to estimate the exposure to masking and behavioural response of SRKW under existing and three future development scenarios. Due to the complexities of modelling noise over large areas (JASCO 2014) and the spatial limits of the SRKW Network Sighting Synthesis Study (Hemmera 2014) (**Appendix A: Figure A 1** and **Figure A 2**) - different spatial extents were used for the regional and focused studies. The focused model area was applied on a smaller spatial scale, but still encompassed areas of critical habitat highly used by SRKW.

2.0 REVIEW OF AVAILABLE LITERATURE AND DATA

Human activities introduce a wide range of sounds into the ocean that have the potential to affect marine mammal populations. Human-generated underwater sound sources include recreational and commercial vessels, seismic exploration, construction, fishing activities, military and commercial sonar, acoustic deterrent devices, and renewable energy sources (e.g., wind turbines, tidal turbines, and wave farms). Sounds from vessels, particularly large ships, are the most common source of underwater noise at frequencies less than 300 Hz in the world's oceans (Hildebrand 2009, Heise and Alidina 2012).

Large container ships, with their powerful engines and slow turning propellers, typically produce high source levels (SLs) at low frequencies (Richardson et al. 1995). Although most shipping noise is low in frequency, modern ships are known to generate 1/3-octave SLs over 150 dB rms re 1 μPa at 1 m at 30 kHz (Arveson and Venditis 2000), which overlap with vocalisations of many odontocete (i.e., toothed whale) species (Aguilar Soto et al. 2006). Broadband SL of ships can exceed 188 dB re 1 μPa at 1 m (McKenna et al. 2012). Ship noise increases with ship size, power, load, and speed (Richardson et al. 1995).

An increase in underwater noise has the potential to affect marine mammals through behavioural changes, range displacement, communication interference, decreased foraging efficiency, hearing damage, and physiological stress (Tyack 2008). Numerous studies have focussed on the behavioural, acoustic, and physiological responses of marine mammals to noise (Nowacek et al. 2007). Examples include changes in dive patterns, increased vocalisation rates and SLs, heightened stress levels, and hearing loss (Mooney et al. 2009, Di Iorio and Clark 2010, Holt et al. 2011, Tyack et al. 2011, Rolland et al. 2012)). Many factors can influence responses of individual marine mammals to underwater noise, including the sound characteristics, the physical and behavioural state of the animal, the population demographics, and the ecological context in which the animal encounters the sound (Ellison et al. 2011).

By building on other studies conducted to inform the RBT2 EA, this study (1) estimates the spatial overlap of SRKW relative density and underwater noise from commercial vessel traffic, and (2) quantifies changes in the number of behavioural responses and/or echolocation masking occurrences from underwater noise between existing conditions and three future development scenarios.

3.0 REGIONAL NOISE EXPOSURE STUDY

This section presents the methods and results for the regional study component of the SRKW Underwater Noise Exposure and Acoustic Masking Study.

3.1 METHODS

The spatial and temporal scope, and model inputs of the regional study component are presented below.

3.1.1 Overview and Scenarios

The SRKW Underwater Noise Exposure Study determined the overlap of SRKW relative density and current underwater noise from commercial vessel traffic (S1), and compared this existing scenario to predicted commercial vessel traffic noise during three future development scenarios (i.e., S2, S3, and S4; **Table 2**)

Table 2 Scenarios Considered in this Study

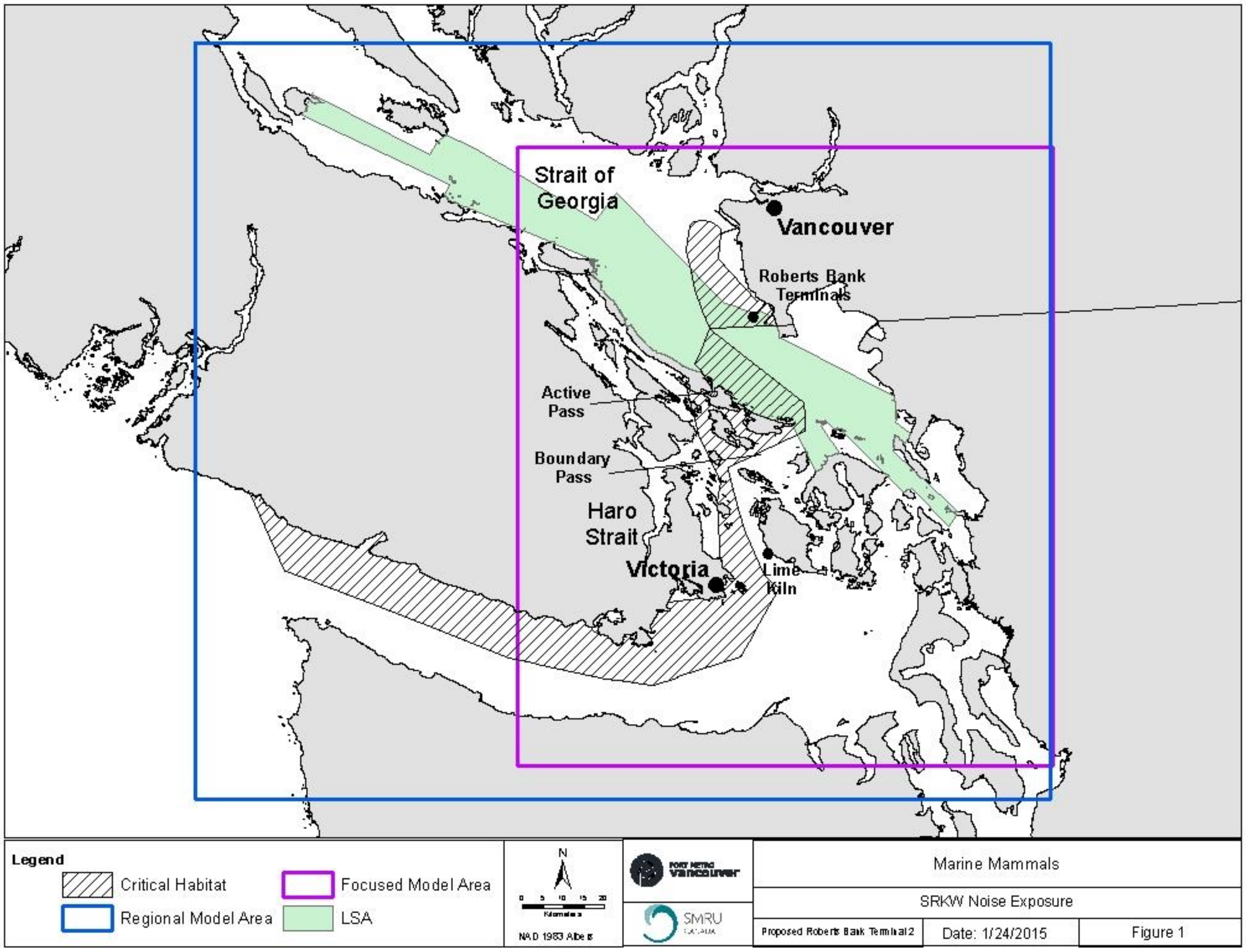
Scenario	Year	Description
S1	2012	Existing commercial vessel traffic.
S2	2030	Future commercial vessel traffic with no new projects except RBT2, and future incremental vessel traffic associated with RBT2 (includes existing and expected conditions)[1].
S3	2030	Future commercial vessel traffic due to certain and foreseeable projects without RBT2, or incremental vessel traffic associated with RBT2 (includes existing and expected conditions).
S4	2030	Future commercial vessel traffic due to certain and foreseeable projects ,with RBT2, incremental shipping traffic associated with RBT2 (includes existing and expected conditions)

[1]Expected conditions between 2012 and 2030 include no new projects but increases in vessel traffic at Westshore and Deltaport terminals i.e., DTRRIP.

3.1.2 Study Area

The regional model area was restricted to the waters surrounding the Project and most of SRKW critical habitat in Canadian and U.S. waters, including the entire U.S. summer core use area (DFO 2011, NOAA 2008). The modelled regional study area was a 184 km by 208 km rectangle, covering 14,750 km^2 of water (**Figure 1**). The focused model was designed to provide a higher temporal and spatial resolution to the estimates of exposure of SRKWs to underwater noise from commercial vessel traffic than the regional study. Results are also presented for the Local Study Area (LSA) to provide context of the relative amount of noise exposure of SRKW to commercial vessel traffic occurring in an area where Project effects could occur (**Figure 1**).

Figure 1 Extent of the Regional and Focused Model Areas



3.1.3 Temporal Scope

Due to seasonal variability of habitat use by SRKW (Hemmera 2014), and seasonal changes in sound transmission (JASCO 2014), the regional model was divided into summer and winter periods. Month-long noise exposure estimates were based on Vessel Traffic Operational Support System (VTOSS) data from January 2010 and July 2010 (adjusted to 2012 vessel activity levels) (JASCO 2014), and were combined with effort-corrected SRKW relative density from 2001 through 2011 (Hemmera 2014).

3.1.4 Inputs to the Regional Model

3.1.4.1 Modelled Ship Noise Data

Four acoustic model scenarios were developed based on one year (2010) of VTOSS data (JASCO 2014) and updated manually to reflect 2012 traffic. The VTOSS dataset aggregates data from radar and the Automatic Identification System (AIS) and includes information on vessel type, location, track, size, and speed. Vessels were categorised into 14 classes based on their length and type. Noise outputs were modelled based on ship location, track, and speed, and the geoacoustic properties relevant to that location and season (JASCO 2014). The commercial vessel traffic underwater noise model generated month-long average broadband noise levels in each of the 800 m x 800 m grid cells in the regional model area. Existing underwater noise levels during summer and winter (January and July) and for the three future development scenarios are provided in **Table 2**. Broadband, rather than audiogram-weighted noise model outputs, were used to estimate underwater noise exposure as these were determined to be more appropriate noise exposure metrics for SRKW (SMRU 2014a).

3.1.4.2 SRKW Relative Density

SRKW sighting data were collected off the southern coast of B.C. and the northern coast of Washington by two opportunistic observer sightings networks: the Canadian-based B.C. Cetacean Sightings Network (BCCSN 2014; www.wildwhales.org), and the U.S.-based OrcaMaster (http://hotline.whalemuseum.org/). These networks provide data on cetacean sightings by researchers, whale watch operators, lighthouse keepers, and the general public. The two datasets were first combined and corrected for spatio-temporal duplicates. Opportunistic datasets are typically limited in their ability to assess habitat use because information on the distribution of observer effort is often lacking. The sightings were corrected for observer effort based on an effort model developed by the Vancouver Aquarium (Hemmera 2014), and the effort-corrected sightings were used as inputs into the noise exposure model. SRKW sightings were split into summer (May through September) and winter (October through April) seasons.

3.2 DATA ANALYSIS

The regional model analyses were conducted in ArcGIS (version 10.1) and Matlab (version 2012a). The JASCO (2014) noise files contained broadband noise levels in 800 m x 800 m grid cells, which were resampled into 200 m x 200 m grids. A widely used method for estimating the spatial distribution of animals is called kernel density estimation (e.g., Hobbs et al. 2005, Urian et al. 2009, Williams et al. 2013). The method creates a smoothed two-dimensional surface from point observations by creating a distance-based spatial average, where the distance over which the surface is smoothed is the kernel radius, or bandwidth (Worton 1989). SRKW relative densities were calculated with a 4.5 km kernel smoothing radius and applied to 200 m x 200 m grid cells. To compare between seasons and scenarios, the underwater noise layers and SRKW relative density layers were normalised to have values between 0 and 1 (i.e., by subtracting the overall minimum value from the value of a given grid cell and dividing by the range). Spatial overlays were generated for the four scenarios by multiplying the noise and SRKW density values for each grid cell and then normalising the overlays to create an index that could be used to identify areas of SRKW-noise overlap. These methods are the same as those used by Erbe et al. (2014) except that the current study covers a smaller area with higher spatiotemporal resolution. The absolute change between the existing and future scenarios in each grid cell was calculated by subtracting the normalised value for existing conditions from the value for each development scenario.

3.3 STUDY RESULTS

The results of the regional study component of the SRKW Underwater Noise Exposure and Acoustic Masking Study are presented for summer and winter periods below.

3.3.1 Summer Exposure

The estimated exposure of SRKW to underwater noise from commercial vessel traffic during the summer is shown for the regional study area in **Figure 2** for S2, and for S1, S3, S4 in **Appendix A** (**Figure A** 3**, Figure A** 4, and **A 5** respectively). It is important to note that all these data layers have been normalised and thus range from 0 to 1. Noise exposure was predicted to be highest in Haro Strait with intermediate noise exposure levels predicted in Boundary Pass and Active Pass. The largest change in summer noise exposure from existing (S1) to S2 would occur in Haro Strait, Boundary Pass, and the approaches to RBT2 (**Figure 3**), while the largest noise exposure changes from existing conditions (S1) to S3 and S4 would occur in those same areas, and the southern entrance to Rosario Strait (**Appendix A**: **Figure A 6** and **Figure A 7**). It is important to note that the figures that depict change (e.g., **Figure 3**) are scaled differently than the noise exposure figures (e.g., **Figure 2**). The figures that depict change depict absolute change in normalised exposure from S1 to future development scenarios. Since the absolute change is small, the scale of the figures needed to be adjusted. The median change (see **Section 3.2**) from existing conditions to all the development scenarios was zero, reflecting a large number of grid cells in the regional study area for which noise exposure did not change. The maximum change was also small and the change was spread between 22 and 39% of the model grid cells (**Table 3**).

Figure 2 Summer Normalised Regional Noise Exposure for RBT2 Vessel Traffic and Incremental Vessel Traffic associated with RBT2 (S2)

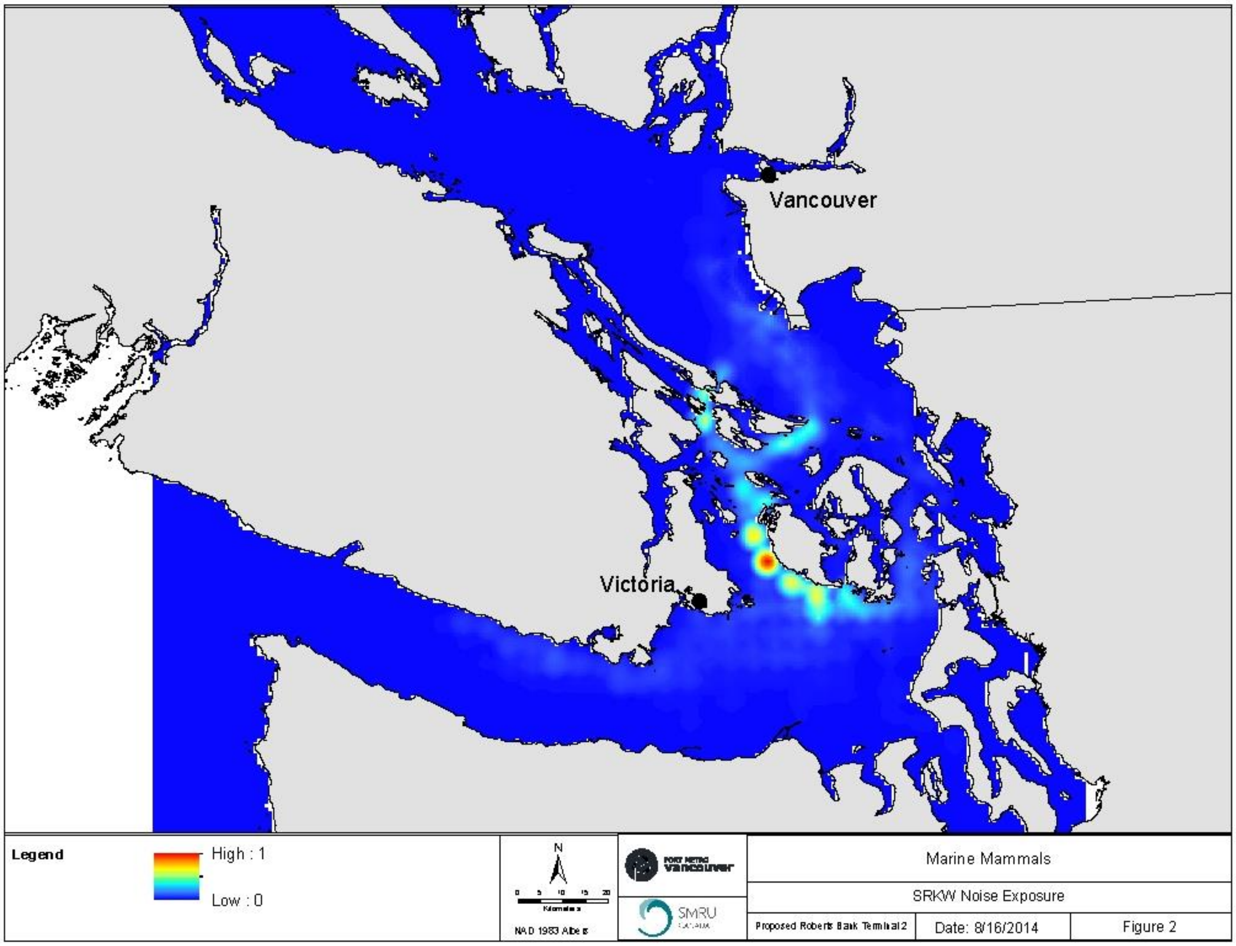


Figure 3 Summer Change in Normalised Noise Exposure from Existing Conditions (S1) to RBT2 Vessel Traffic and Incremental Vessel Traffic Associated with RBT2 (S2)

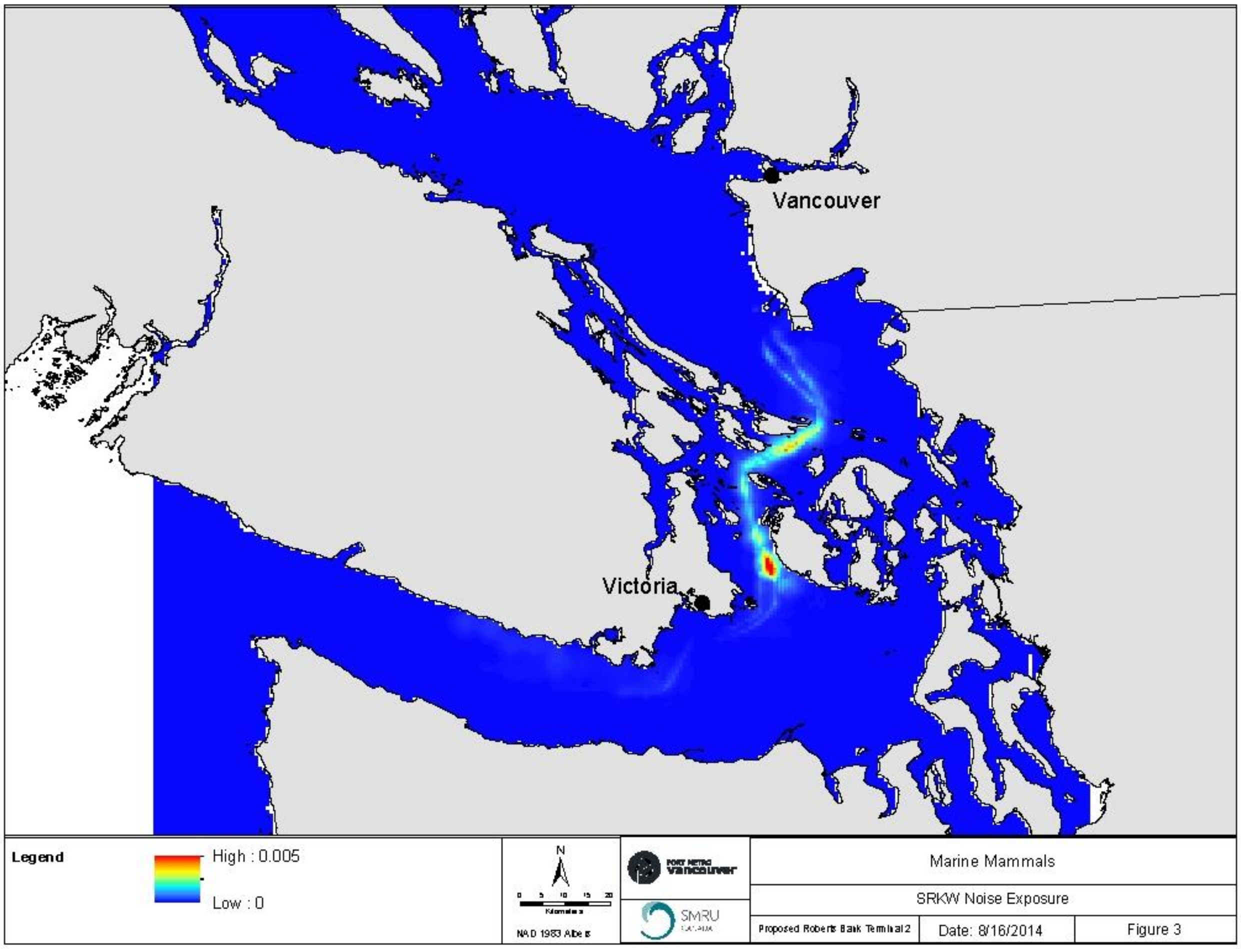


Table 3 Maximum Normalised Change from Existing Conditions to Future Development Scenarios and the Percent of Grid Cells with Change During Summer

Change from S1 to	S2	S3	S4
Maximum	0.0092	0.0111	0.0147
Percent of grid cells with change	22%	38%	39%

3.3.2 Winter Exposure

Like the summer exposure estimates, the winter exposure estimates were similar across scenarios; however, normalised exposure levels during winter were lower than those predicted for the summer (**Figure 4** (S2) and **Appendix A: Figure A 8, Figure A 9** and **Figure A 10** (i.e., S1, S3, S4, respectively). In addition, the change in winter exposure levels from existing conditions to all three development scenarios was concentrated in Haro Strait and Boundary Pass (**Figure 5** and **Appendix A: Figure A 11** and **A12**). The median change (see **Section 3.2**) from existing conditions in winter to the three future development scenarios was zero and the maximum change was less than that of the summer scenarios (**Table 4**). The percent of model grid cells with change from existing conditions to the three future development scenarios ranged from 25 to 29%.

Figure 4 Winter Normalised Regional Noise Exposure for RBT2 Vessel Traffic and Incremental Vessel Traffic Associated with RBT2 (S2)

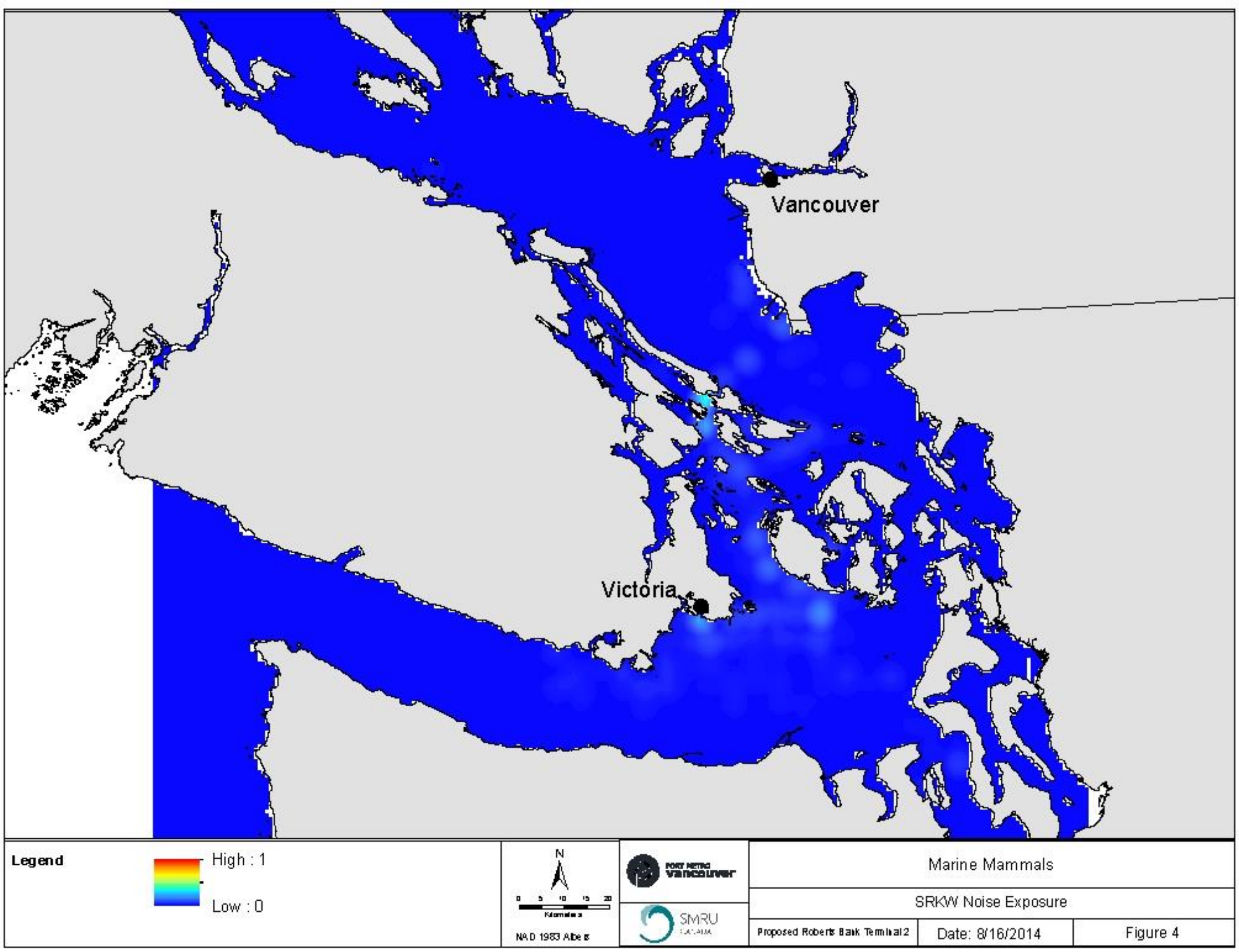


Figure 5 Winter Change in Normalised Noise Exposure from Existing Conditions (S1) to RBT2 Vessel Traffic and Incremental Vessel Traffic Associated with RBT2 (S2)

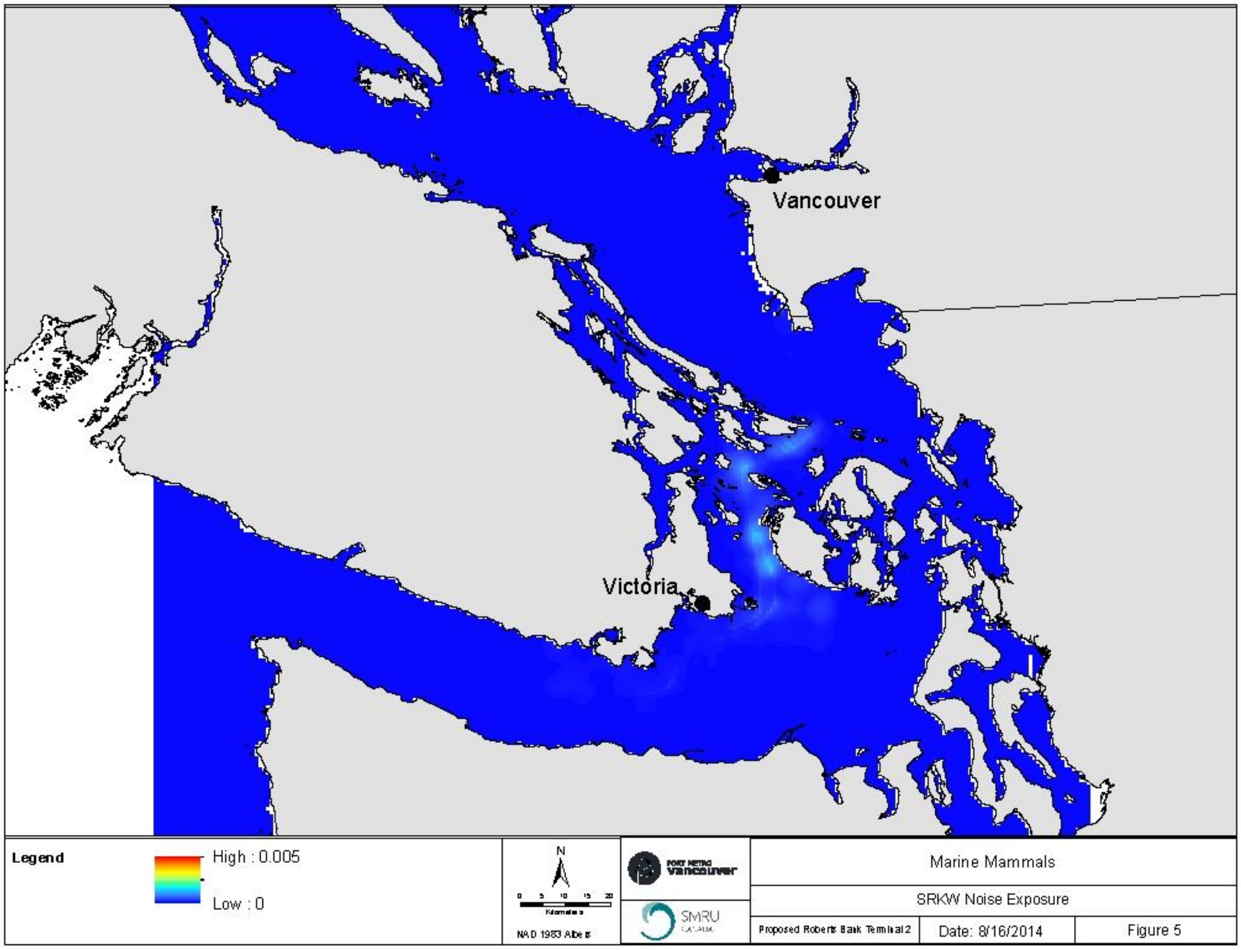


Table 4 Maximum Normalised Change from Existing Conditions to Development Scenarios and the Percent of Grid Cells with Change During Winter

Change from S1 to	S2	S3	S4
Maximum	0.0013	0.0015	0.0023
Percent of grid cells with change	25%	29%	29%

4.0 FOCUSED NOISE EXPOSURE STUDY

This section presents the methods and results for the focused study component of the SRKW Noise Exposure Study.

4.1 METHODS

The spatial and temporal scope, and model inputs of the focused study component are presented below.

4.1.1 Overview and Scenarios

The focused model was designed to provide a higher temporal and spatial resolution to the estimates of exposure of SRKWs to underwater noise from commercial vessel traffic than the regional study. The objective was to identify SLs every five minutes that occur from commercial vessels in the area and how this underwater noise might affect behaviour of SRKW.

The focused model is a simulation study designed to evaluate the impact of fine-scale underwater noise inputs from commercial vessel traffic on SRKW behaviour over one year. Individual stochasticity (i.e., randomness) in the SRKW population was accounted for by assigning behavioural responses following probability distributions. For each five-minute time window, low-severity and moderate-severity behavioural responses were assigned to individual SRKWs based on dose-response curves (SMRU 2014a). Potential occurrence of masking of echolocation clicks (when behavioural responses were not predicted) was also estimated for each SRKW using high frequency (50 kHz) noise levels based on a masking model developed by Au et al. (2004). Behavioural impacts were estimated on individual SRKWs distributed in pods J, K, and L at a 200 m x 200 m spatial scale over 365 days. The simulation was repeated 1,000 times to incorporate uncertainty around behavioural response to noise.

Like the regional model, the focused model examined four underwater noise scenarios (see **Table 2**). Results of this study were intended to evaluate disturbances of individual animals due to short-term exposures to underwater noise from commercial vessel traffic within the study area (**Figure 1**) and provide population-level estimates of noise exposure.

4.1.2 Study Area

The study area for the focused model is a 130 km x 150 km region located within the boundaries of the regional model study area and covering 9,150 km^2 of water (**Figure 1**). Exposure estimates were also reported for a local study area (LSA), which covered the maximum spatial extent of Project-related noise originating from within Port Metro Vancouver jurisdiction. Part of the LSA was not included in the focused model area due to lack of SRKW sightings there.

4.1.3 Temporal Scope

The focused noise exposure estimates were based on two representative 24-hour periods of VTOSS data collected on Tuesday, January 19, 2010 and Friday, July 16, 2010. Days were selected based on the most complete temporal coverage of the VTOSS data (i.e., 1,422 minutes in winter and 1,440 minutes in summer; full coverage of 24 hours is 1,440 minutes). More details on the underwater noise modelling can be found in JASCO 2014. Underwater noise estimates were combined with SKRW relative density data from 2001 through 2011.

4.1.4 Inputs to the Focused Model

The five inputs to the focused noise exposure estimates included two estimates of ship noise (i.e., broadband SPL and 1/3-octave noise levels at 50 kHz), SRKW relative density, behavioural response, and masking (see **Table 5** and subsequent sections). Estimates of boat and whale watch vessel noise were not included as they were not included in VTOSS used to conduct underwater noise modelling (see **Section 3.1.4 Inputs to the Regional Model** and JASCO (2014). With underwater noise from small vessel traffic not included in the VTOSS data, including whale-watching, the focused model may underestimate behavioural responses and acoustic masking during all scenarios. However, with or without the inclusion of whale-watching vessels, the absolute contribution of underwater noise by the Project and predicted behavioural responses and acoustic masking will not change.

Table 5 Summary of Inputs to the Focused Noise Exposure Model

Input	Source	Brief Overview
Broadband Noise Levels	Regional Commercial Vessel Traffic Noise Modelling Study (JASCO 2014)	At 5-minute resolution, the maximum predicted broadband sound pressure levels (SPL) from JASCO's 1-minute predictions for each of the 200 m x 200 m grid cells. Broadband underwater noise estimates were provided for: A) A day in summer with complete coverage over time (288 5-minute windows); and B) A day in winter with incomplete coverage over time (286 5-minute windows) due to missing, broken or incomplete tracks or track segments.
SRKW Effort Corrected Sightings	Southern Resident Killer Whale Network Sighting Synthesis Study (Hemmera 2014)	Effort corrected SRKW sighting data split by summer and winter and transformed into probability estimates of relative density.
Behavioural Dose Response Curves	Determination of Behavioral Effect Noise Thresholds for Southern Resident Killer Whales Study (SMRU 2014a)	Dose-response curves (with 95% CI) that predict the probability behavioural responses from SRKW.

Input	Source	Brief Overview
Echolocation Masking	Potential for Masking of Southern Resident Killer Whale Calls and Echolocation Clicks due to Underwater Noise (**Appendix C**)	An acoustic masking model that predicts when masking of SRKW echolocation clicks is likely to occur (in cases where no behavioural response had been predicted) and the duration of the occurrence.
50 kHz Noise Levels	Regional Commercial Vessel Traffic Noise Modelling Study (JASCO 2014)	1/3-octave noise levels at 50 kHz in 1-minute intervals for each of the 200 m x 200 m grid cells. 50 kHz underwater noise estimates were provided for: A) A day in summer with complete coverage over time (1,440 minutes); and B) A day in winter with incomplete coverage over time (1,422 minutes) due to 15 minutes of missing ship data.

4.1.4.1 Modelled Ship Noise

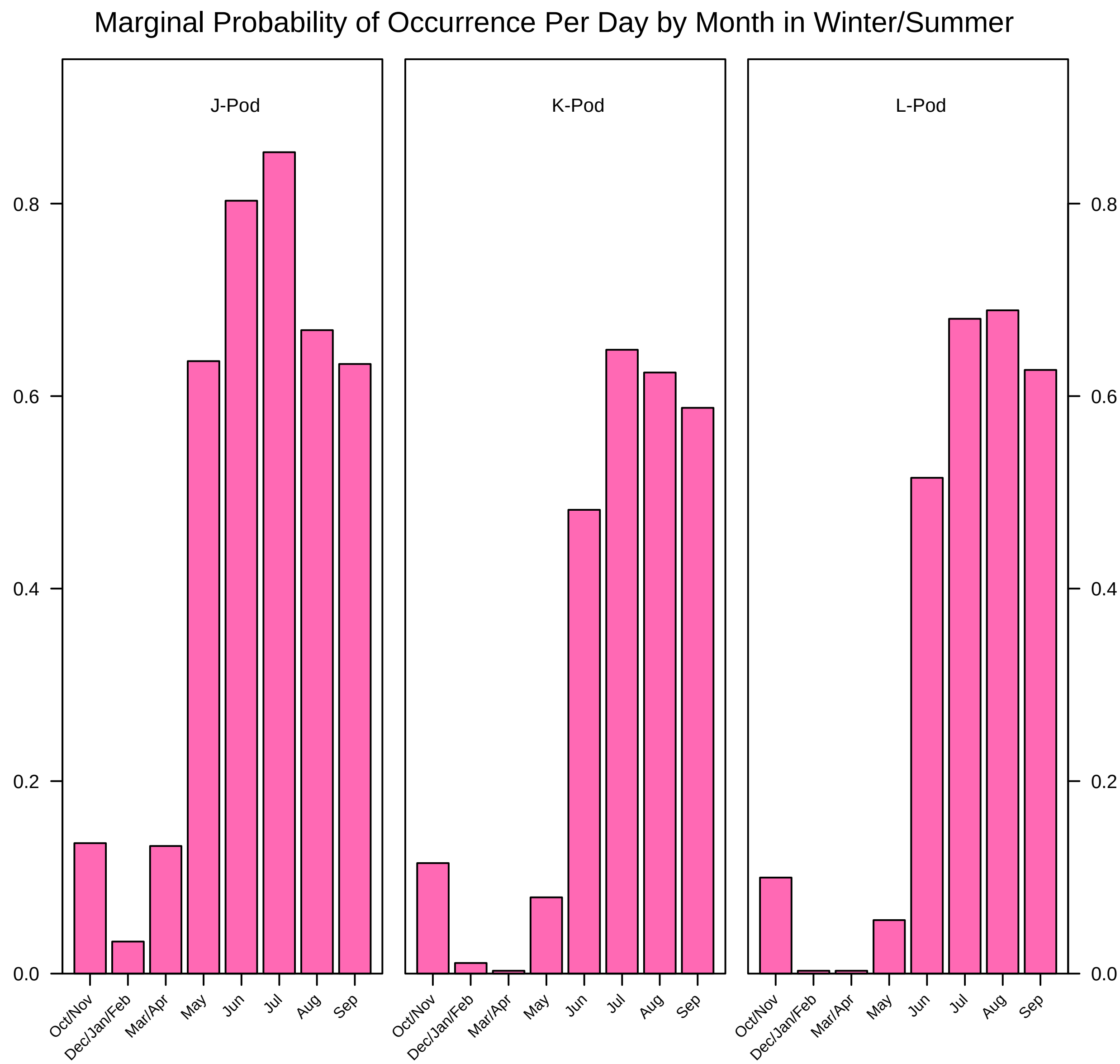
The same gridded vessel noise propagation model used for the regional noise exposure study was used to estimate the finer scale resolution of the focused noise exposure study. Details of the underwater noise propagation models and how broadband and 1/3-octave band estimates were calculated can be found in JASCO 2014. Noise outputs were produced at 1-minute intervals in each 200 m x 200 m grid cell for the 24-hours of a representative day in summer and winter. The 1,440 (summer), and 1,422 (winter) 1-minute broadband files were consolidated into 288 and 286, 5-minute files by taking the maximum (i.e., most conservative) 1-minute values for each grid cell in that 5-minute period. This step was done to increase the focused model run speed, and was expected to have little impact on noise exposure estimates.

As documented in JASCO 2014, sound speed profiles of underwater noise vary by time of year. Based on this information, the January noise outputs of the focused model were applied to November through March (i.e., 151 ‘winter’ days of the year) while the July noise outputs were applied to April through October (i.e., 214 ‘summer’ days). For each of 365 days, these broadband noise levels were used to estimate the level of noise exposure an individual SRKW would experience on a given day over the spatial extent of the study area. The 15 minutes of missing data in the winter model were compensated for by infilling with the median response to the remaining 1,422 minutes of JASCO model noise levels. Limitations of the VTOSS dataset used to generate noise level within the focused noise exposure study area are further discussed in JASCO 2014.

4.1.4.2 SRKW Relative Density

The same sightings data were used for the focused noise exposure model as in the regional model. The dataset was queried for relative occurrences of pods J, K, and L, and the seasonal association between each of the three pods (i.e., the seasonal association suggests J, K, and L are more likely to be together in the summer), and how these distributions changed over the year. Months that had little observer effort, and thus less reliable sightings data, were consolidated to avoid bias in the simulation; therefore, relative densities and seasonal associations were split into eight ‘month’ periods as follows: October/November; December/January/February; March/April; May; June; July; August; and September (**Figure 6**).

Figure 6 Monthly Marginal Probabilities of Occurrence of Pods J, K, and L

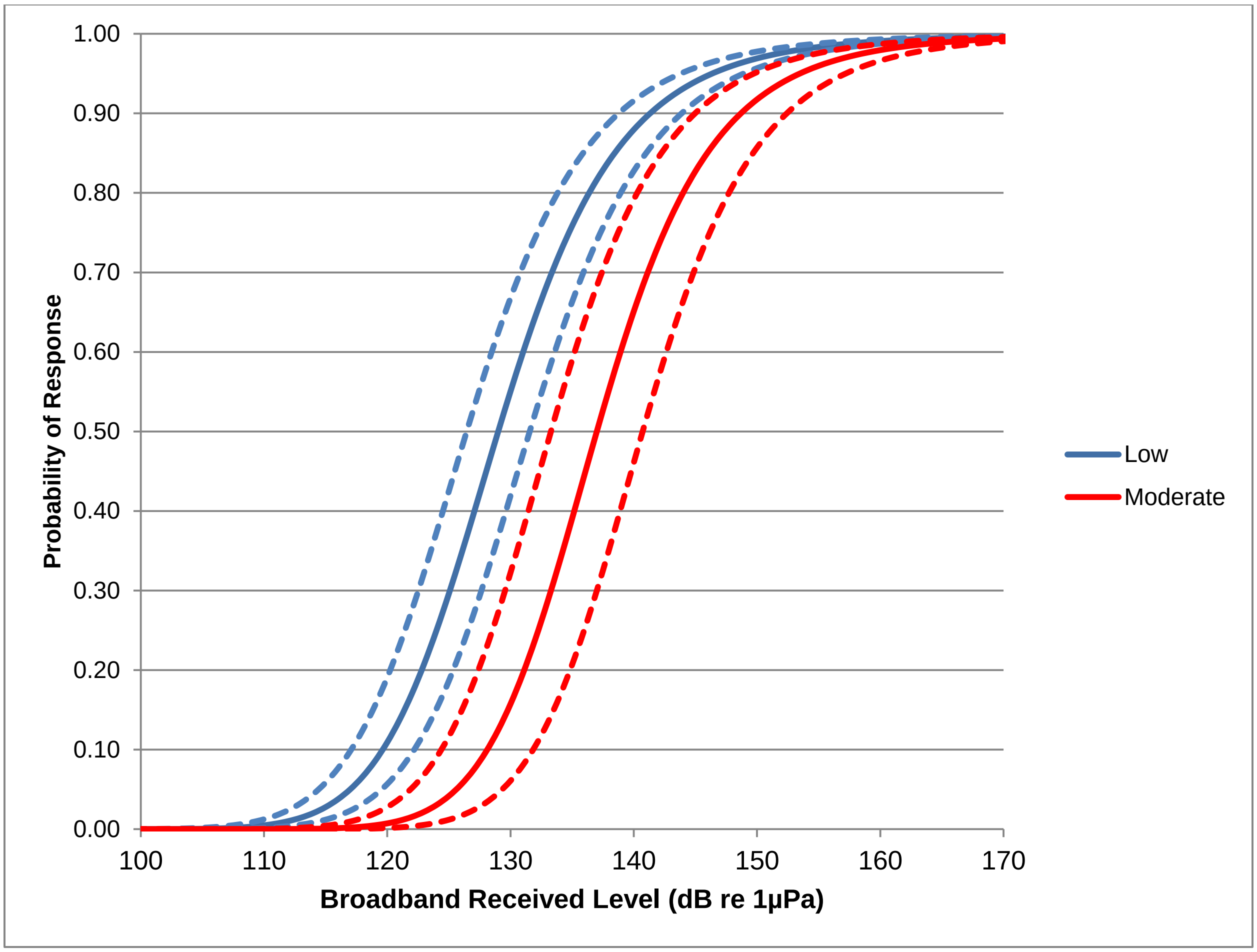


For each day of each ‘month’ in the 365-day simulation, whales were placed in the model space based on the monthly probability of a pod being present, the probability of being co-located with another pod, and the relative spatial density of each pod as determined by the effort-corrected sightings data. When pods were placed in the model space, they were spread to a maximum of 4.5 km from the pod centroid in order to match the kernel smoothing radius applied to the SRKW density surface and allow for SRKWs to experience different noise levels within a pod.

4.1.4.3 Dose-Response Curves

Three datasets containing interactions between killer whales and ships were analysed to develop dose-response curves as part of the Project environmental studies (see SMRU 2014a for details), which predicted the probability of a low-severity and moderate-severity behavioural response for a given broadband noise level (**Figure 7**). High-severity behavioural responses were not predicted to occur as a result of underwater noise produced by the Project or regional commercial vessel traffic, as SRKW were considered highly unlikely to approach to within a few metres of the vessels propellers.

Figure 7 Low-Severity and Moderate-Severity Behavioural Dose-response Curves with Corresponding 95% Confidence Intervals (dotted lines) from SMRU 2014a



4.1.4.4 Echolocation Masking and 50 kHz Noise Levels

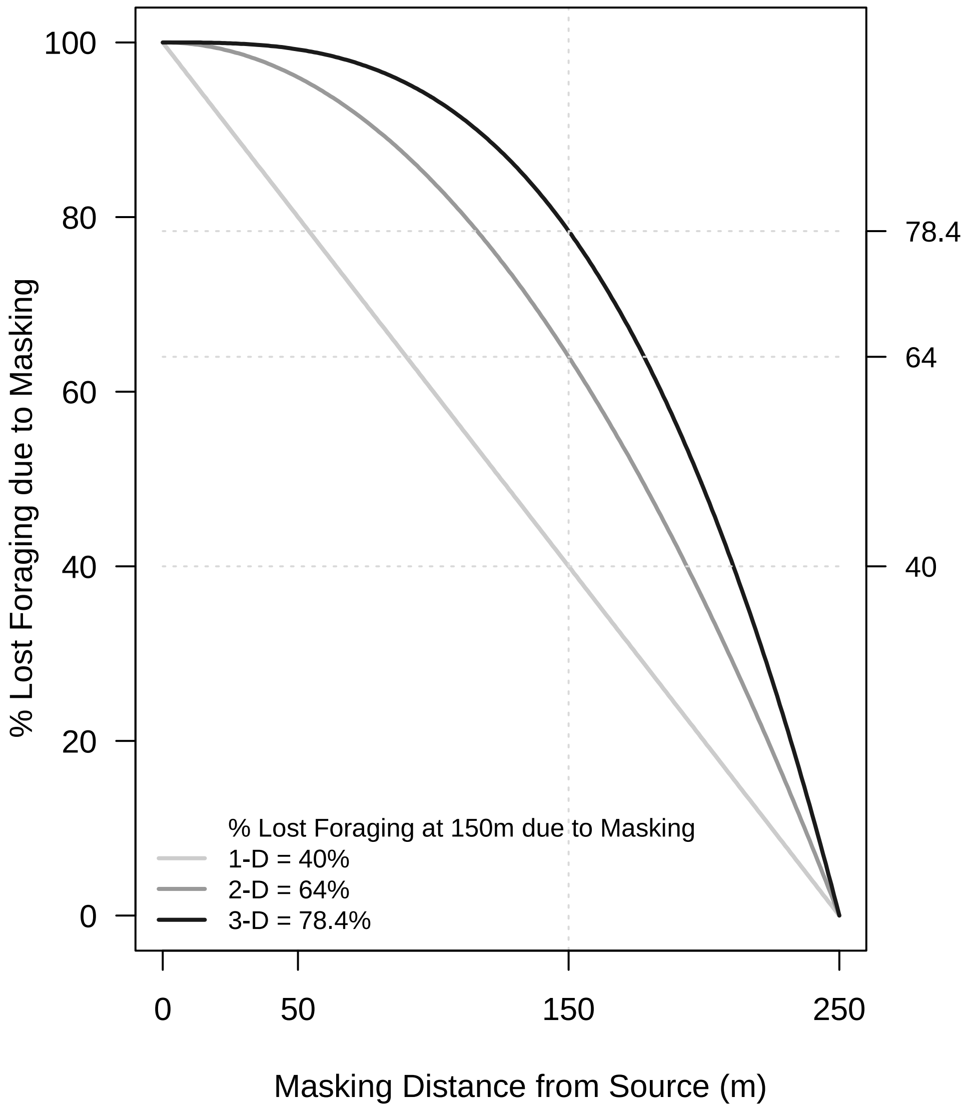
Methods for estimating the distance over which SRKW might use echolocation in the current study were developed by Au et al. (2004) and reviewed in **Appendix C: Potential for Masking of Southern Resident Killer Whale Calls and Echolocation Clicks due to Underwater Noise**. This simple model with masking estimated at a single frequency was chosen because so much is unknown about masking of echolocation clicks. A more complex model would have been less intuitive to follow and would have relied on a number of assumptions. Fifty kHz was used by Au et al. (2004) for their masking model because it is near the centre of the frequency distribution of often bimodal killer whale echolocation clicks. The masking model takes into account the amplitude of killer whale clicks, transmission loss, how much of the click echoes off preferred salmon prey (known as the target strength), and killer whale hearing.

The appropriate 1/3 octave band was extracted from JASCO 2014 and converted to 50 kHz power spectral density (PSD) levels, and compared to the echo level at successively larger distances from the modelled killer whale. The distance at which the 50 kHz PSD noise level is no longer less than the echo level is considered the masking distance (see **Appendix C**).

Since there is uncertainty in whether to measure changes in echolocation by distance, area, or volume, masking was calculated as a proportional loss function in 1, 2, and 3-dimensional listening space. The maximum echo distance was inferred from data collected from a hydrophone located at Lime Kiln State Park, Washington (see **Appendix C**) and was set for 1, 2, and 3-dimensions at a distance of 250 m, area of 7,009 m^2, and volume of 210,885 m^3, respectively. The area and volume were based on the distance of 250 m and a half power beam width angle of 13^o (this is the angle at which the echolocation click amplitude drops by 3 dB). This angle was estimated using a formula developed by Au et al. (1999) which uses the diameter of the sound source, and its frequency, to estimate the beam width angle. An example of proportion of lost foraging, as a result of masking using these three metrics, is presented in **Figure 8**, which provides an example where masking starts at a distance of 150 m. Under this scenario, the estimated percent loss of foraging would be 40, 64, and 78.4% for 1, 2, and 3-D metrics, respectively.

Masking was only assumed to occur if an SRKW individual had not already experienced a low-severity or moderate-severity behavioural response in that 5-minute period, as these behaviour changes were assumed to cause a complete loss of foraging opportunity (i.e., equivalent to complete masking).

Figure 8 Loss Function for Masking Echolocation Clicks as a Function of Distance from the SRKW



4.2 DATA ANALYSIS

The simulation model was implemented using the following steps:

Step 1: For a given day, up to three pods of 80 SRKW were randomly distributed across the study area (SRKW population was estimated at 80 individuals at time of modelling). Pods were selected (yes/no) each day based on a Bernoulli trial similar to the flipping of a biased coin where the bias is the probability of occurrence of pod J, K, and L for the month in which that day occurs. If a pod was selected for the given day, the location of the pod 'centroid' was proportional to the relative measure of occurrence derived from the kernel-smoothed SRKW relative density (Hemmera 2014). Occurrence of each pod was assumed independent, but if more than one pod was randomly chosen, the location of the pods was based on the probability of seasonal association. The probability of association between pods was implemented through the use of a copula function that linked the marginal distributions of pods J, K, and L. A copula is a function that joins several outcome variables described by a multivariate distribution to their one dimensional marginal distribution functions (e.g., Nelsen 2006). Pods are located once for a 24-hour period, so that location was constant over all time windows during the 24-hour period.

Step 2: For each of the (up to three) centroid locations, 'killer whales' associated with each pod were spatially distributed in a density-weighted kernel of 4.5 km, which is the same kernel bandwidth used with the sightings data for the kernel-smoothed density. The 4.5 km radius was selected to allow variability in the sound exposure, and behavioural response, of whales within the same pod, and approximate the potential spatial spread of SRKWs in the wild.

Step 3: For one 5-minute time window (t) and each whale present, the model determined the received SL from the broadband and 50 kHz noise datasets.

Step 4: For time window (t) and each whale present, there will be a probability of low and moderate severity response to the broadband measure of noise at that location according to the dose response curve. This was calculated with two Bernoulli random variables that were generated with probability of low and moderate behavioural response proportional to the dose-response curve, with uncertainty generated according to the confidence intervals (CI) around those curves (**Figure 7**). This procedure generated either a 0 (corresponding to no response) or 1 (corresponding to a response) for low and moderate severity responses, and resulted in a record for that time window of whether each whale exhibited a low or moderate behavioural response. If a whale exhibited both a low and moderate response, only the moderate response was counted (to avoid double counting).

Step 5: For time window (t) and each whale present, the 1-, 2-, and 3-dimension loss functions shown in **Figure 8** were used to determine the horizontal distance at which masking occurs for each whale on the surface. The proportion of 250 m lost due to masking was calculated and translated to proportion of minutes lost to foraging. For example, if there was a 50% loss in foraging distance, then there was a loss of 2.5 minutes of foraging (i.e., 50% of the 5-minute time window). If the whale already had a behavioural response in that period, then no masking was calculated (to avoid double counting).

Step 6: For time window (t) and each whale present, it was determined if the broadband noise level was <110 dB re 1 μPa [10 Hz to 63 kHz], which was chosen to provide a simple metric for the change in relatively quiet periods that might be important to SRKW life functions. The cut-off at 110 dB was based on it being at the low end of the dose-response curves and that noise levels when ships are present are louder than 110 dB re 1 μPa [20 Hz to 96 kHz] 90% of the time (SMRU et al. 2014).

When Steps 4, 5, and 6 are complete, the model provides for that time window (t) and each of (up to) 80 whales as follows:

- 0 or 1 for low-severity behavioral disturbance if moderate severity = 0;
- 0 or 1 for moderate-severity behavioral disturbance;
- Proportion of time lost due to masking (in 1, 2 and 3-D) if both severity responses = 0; and
- Proportion of time that the whale was in ‘quiet’ conditions (<110 dB re 1 μPa) [10 Hz to 63 kHz].

Step 7: Simulation for each 5-minute time window was repeated in a 24-hour period, where T = 286 in winter, and 288 in summer. At the end of each ‘day’ the simulation provides 286 or 288 measures of the four outputs above, which are summarised for that day and passed to the outcome array as follows:

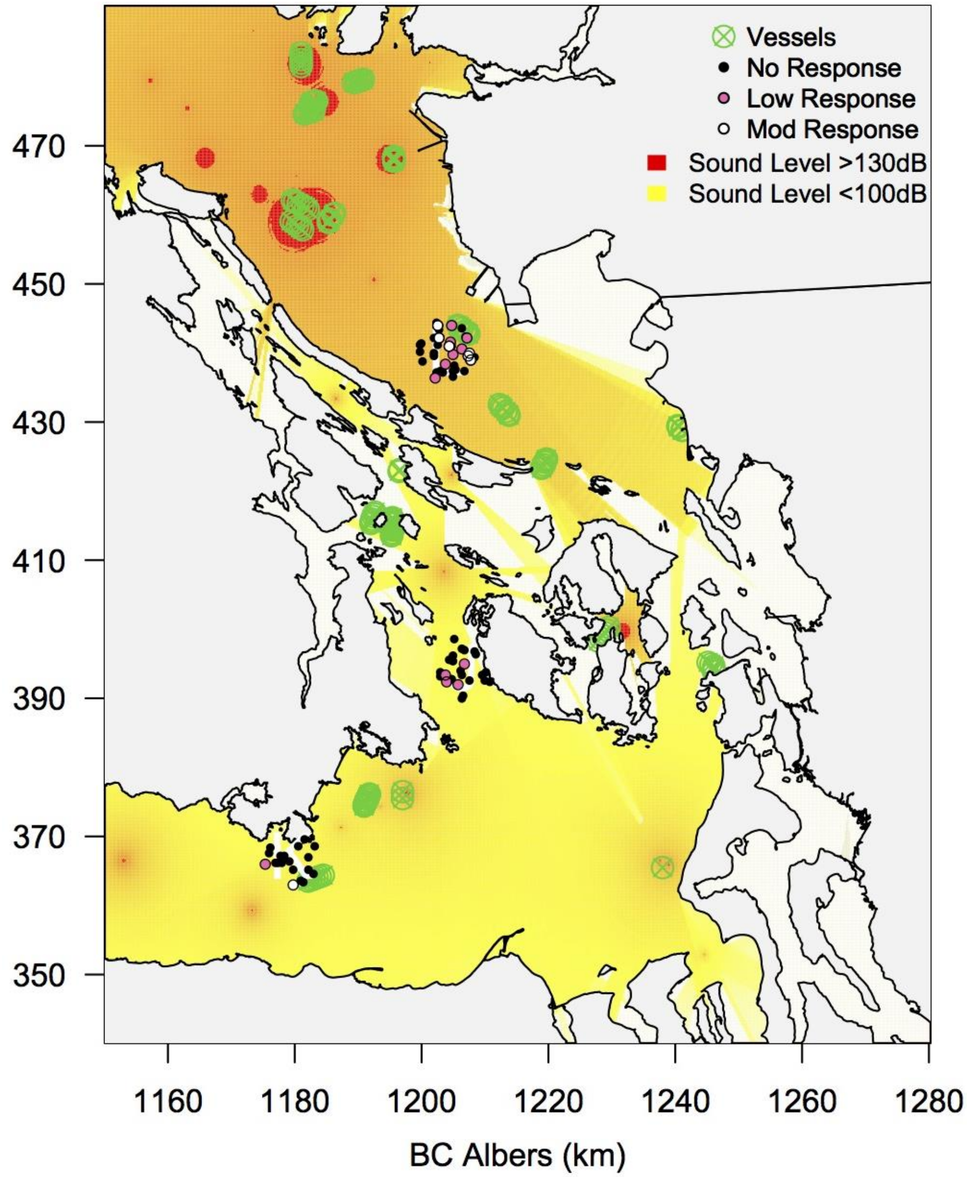
- Number of low-severity behavioural responses per day;
- Number of moderate-severity behavioural responses per day;
- Number of any additional minutes of foraging time lost due to masking; and
- Proportion of the day that whales on the surface experienced noise levels <110 dB re 1 μPa [20 Hz to 96 kHz].

Step 8: Simulation was repeated for 365 days. Probability of pod occurrences change with each ‘month’ according to **Figure 6**, and the SRKW density surface changes from Winter (October 1st to April 30th) to Summer (May 1st to September 30th) on Day 212 (May 1st). Noise files and vessel locations from the January dataset were used from November 1st to March 31st, while the July dataset was used from April 1st to October 31st (see **Section 4.1.4.1**).

Step 9: Simulation was repeated for 1,000 iterations to get a distribution of all simulation outcomes.

Figure 9 is an illustration of how the focused noise exposure model works. In this figure, whale locations are shown in each of the three pods for a single 5-minute period, at ‘00:00 – 00:05’, in winter. The larger green points indicate vessels locations in the study area during this time window, while smaller solid dots are SRKWs. Note the whales are distributed in ‘pods’, but there are some with low-severity behavioural responses, some with moderate-severity behavioural responses, and some with no behavioural response. The map is projected in BC Albers.

Figure 9 Illustration of How the Focused Noise Exposure Model Works



4.3 STUDY RESULTS

The results of the focused study component of the SRKW Noise Exposure Study are presented below.

4.3.1 Behavioural Response

Based on the SRKW sightings data (see **Section 4.1.4.2**), the simulations placed SRKW in the model space on 182 days of the year. Split by pod (J, K, and L), the whales were placed a median of 151, 97, and 101 days respectively in the focused study area, and 26, 17, and 18 days respectively in the LSA, resulting in a median of 1,482 low-severity behavioural responses and 624 moderate-severity behavioural responses per year per whale in the focused model area during the existing conditions (S1) scenario

(**Table 6**). The range in the 95% CI within each scenario is larger than the average difference between the four scenarios such that the CIs of all four scenarios overlap, which implies no significant differences between scenarios. On average, an increase of 74 low-severity (5.0% increase) and 26 moderate-severity (4.2% increase) responses per year per whale are estimated from existing conditions (S1) to the existing conditions plus RBT2 (and incremental shipping associated with the Project) only scenario (S2) (**Table 7**). Estimates of the median number of low-severity and moderate-severity behavioural responses per year per whale split by pod for each scenario are presented in **Table A1** to **A4**. The median (and 95% CI) low-severity and moderate-severity responses per day per whale split by month for each scenario are illustrated in **Appendix A: Figure A 13** (focused model area) and **Figure A14** (LSA). As expected, behavioural responses are estimated to be higher during summer months because more whales are present within the study areas.

Table 6 Median (95% CI) Low-Severity and Moderate-Severity Behavioural Reponses per Year per Whale by Scenario and Study Area

Area	Response Type	S1	S2	S3	S4
Focused Model Area	LOW	1,482 (1,082; 2,680)	1,556 (1,144; 2,801)	1,555 (1,141; 2,802)	1,587 (1,167; 2,855)
Focused Model Area	MOD	624 (417; 1,100)	650 (438; 1,141)	642 (432; 1,129)	657 (444; 1,154)
LSA	LOW	207 (76; 478)	217 (81; 495)	214 (80; 490)	220 (82; 501)
LSA	MOD	90 (28; 205)	94 (30; 212)	92 (29; 208)	95 (30; 214)

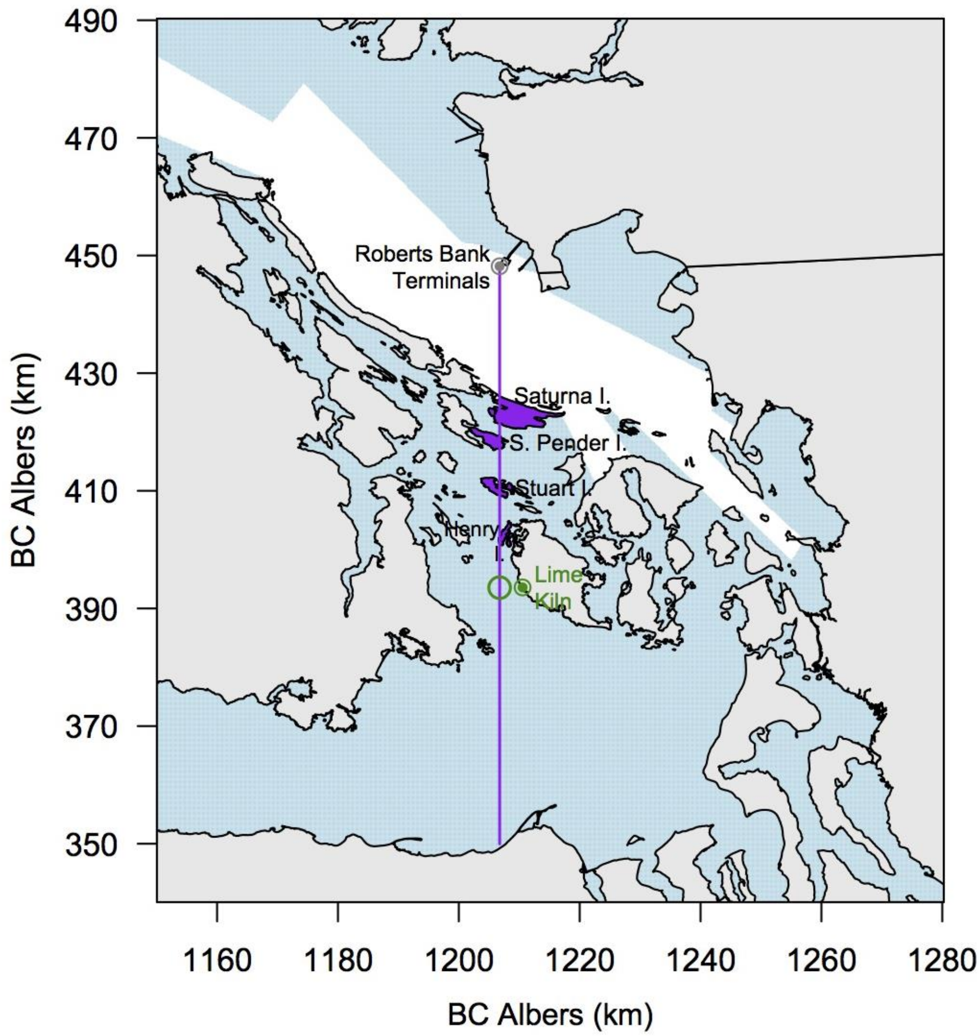
Table 7 Average Expected Increase in Low-Severity and Moderate-Severity Behavioural Responses per Year per Whale from Existing Conditions to Development Scenarios by Study Area

Area	Response Type	S1 to S2	S1 to S3	S1 to S4
Focused Model Area	LOW	74	73	105
Focused Model Area	MOD	26	18	33
LSA	LOW	10	7	13
LSA	MOD	4	2	5

In order to provide more insight into the factors driving the results and to provide more spatial context, model inputs (i.e., broadband noise levels and SRKW relative density) and results (i.e., number of behavioural responses) were extracted along a north-south transect from Roberts Bank terminals, through Haro Strait to the Olympic Peninsula (**Figure 10**) and plotted for each scenario in **Appendix A: Figure A15** to **A18**. In all scenarios, the median underwater noise levels during winter are higher than in the summer, but the upper summer 95% distribution is often higher than the winter equivalent. The highest underwater noise levels were near Roberts Bank terminals and just south of Lime Kiln in Haro Strait. Noise levels do not differ greatly across the four scenarios. The relative density of SRKW is highest in Haro Strait (centred on Lime Kiln) and substantially higher during the summer than the winter. Model inputs result in the number of summer behavioural responses being much higher than the number of winter behavioural responses. Furthermore, the summer responses are concentrated in Haro Strait, especially in the vicinity of Lime Kiln, while the winter responses are more evenly spread across space.

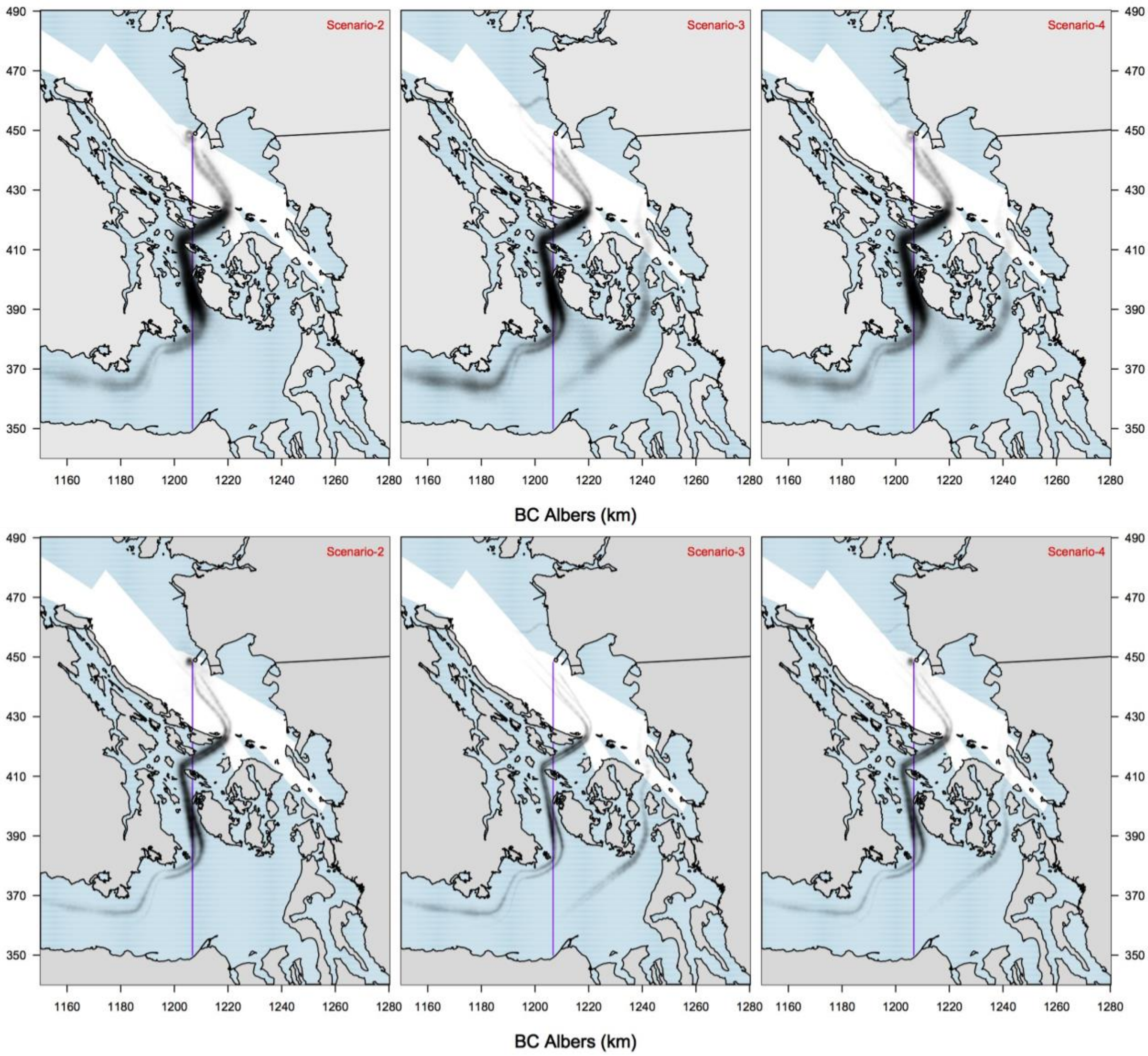
To provide further spatial context to the change in behavioural responses from existing conditions to the development scenarios, only those behavioural responses that occurred during the future development scenarios, but not the existing conditions, were plotted in **Figure 11**. These locations are from the combined 1,000 simulations for each of the scenarios, with the existing locations removed. The darker the area, the more behavioural responses at that location. The plots therefore, show the location of behavioural responses due to the development scenarios, but should not be used to estimate the number of behavioural responses. As per the regional model, the focused model shows that change is concentrated in Haro Strait, Boundary Pass, and the approaches to Roberts Bank terminals in S2, and that S3 and S4 change is concentrated in these same areas plus the southern approaches of Rosario Strait.

Figure 10 Low-Severity and Moderate-Severity Behavioural Responses along North-south Transect



Note: The LSA sub-area is shown in white. Note the location of the Roberts Bank terminals at the north end of the transect line, the location of the Lime Kiln lighthouse, and the adjacent point along the transect line.

Figure 11 Change in Low-Severity and Moderate-Severity Behavioural Responses During each Development Scenario from Existing Conditions



Note: top panel is low-severity behavioural responses, bottom panel is moderate-severity behavioural responses

4.3.2 Masking

Masking was assumed to co-occur with all incidences of predicted behavioural response, but in cases where a behavioral response was not predicted, the time period of additional potential masking was calculated. Thus, these data do not represent total masking minutes, but simply a residual result in addition to behavioural responses. As expected, the three metrics of masking (1-D, 2-D, and 3-D) resulted in successively increasing estimates of the duration of masking (**Appendix A: Figure A 19**). Because it is not known which of these metrics is most appropriate, results for the 3-D metric are reported as these are the highest and most conservative. The number of minutes of masking per year per whale under the four scenarios is reported in **Table 8**. There are slight increases in the amount of masking from existing conditions (S1) to the development scenarios, but little difference in masking between development scenarios (Note: Estimates are based on a proportional reduction of echolocation volume converted to an equivalent proportion of each 5-minute simulation time window and only calculated during simulation time windows without low or moderate behavioural responses).

Table 8 Median Number of Minutes of Additional Masking per Year per Whale and the Equivalent Percentage of a Year for each Scenario

Measure	Area	S1	S2	S3	S4
Masking Minutes per Year per Whale	Focused Model Area	1,966	2,198	2,221	2,244
Masking Minutes per Year per Whale	LSA	385	446	450	454
Percent of Year with Masking	Focused Model Area	0.37%	0.42%	0.42%	0.43%
Percent of Year with Masking	LSA	0.07%	0.08%	0.09%	0.09%

4.3.3 Quiet Time

The percent of time that an SRKW is expected to be exposed to broadband ship noise levels <110 dB is highly variable as it depends on the relative location of SRKW and ships; however, there is a consistent reduction in this number from existing conditions to development scenarios (**Table 9**). Note that model estimates of underwater sound less than 110 dB did not include small vessel traffic, including whale-watching vessels and likely overestimates the amount of quiet time during all scenarios.

Table 9 Median Percent Time SRKW Were Exposed to <110 dB Broadband Noise in Simulation Model by Scenario and Area

Area	S1	S2	S3	S4
Focused Model Area	61.93%	59.50%	57.41%	56.84%
LSA	75.23%	72.80%	71.53%	71.06%

5.0 DISCUSSION

A discussion of the major results arising from the SRKW Underwater Noise Exposure study and data gaps are provided below.

5.1 DISCUSSION OF KEY FINDINGS

This study used two approaches, regional and focused models, to estimate SRKW noise exposure to commercial vessel noise under four scenarios. Spatial and temporal trends of input data to these models and their outputs were consistent. Winter noise data inputs to the regional and focused noise exposure models were on average higher than the summer noise levels and the highest noise levels were found along shipping lanes. Conversely, SKRW density was much higher in the summer than in the winter, and SRKW density was highest in Haro Strait. These inputs and the noise exposure models resulted in the following key findings:

- Noise exposure estimates were higher in summer than in winter, which is driven by changes in SRKW seasonal density;
- Development scenarios increased expected noise exposure for SRKW;
- Change in noise exposure from existing conditions to S2 was small (4.2 to 5.0% increase, see **Table 6**) and focused in Haro Strait, Boundary Pass, and approaches to RBT2;
- Change in noise exposure from existing conditions to S3 and S4 was small (2.9 to 4.9% and 5.3 to 7.1% increases respectively, see **Table 6**)) and focused in Haro Strait, Boundary Pass, the approaches to RBT2, and the southern approaches of Rosario Strait;
- A large amount of variability in noise exposures occurred such that the variability in noise exposure within scenarios was larger than the difference in noise exposure between scenarios (see **Table 6**). Variability in underwater noise exposure was driven by the relative proximity of SRKW and ships; and
- Noise exposure was highest near shipping lanes.

Like Erbe et al. (2014), the regional noise exposure model identified areas of noise exposure; however, the model expanded upon this approach. Firstly, the regional model was conducted on seasonal data that allowed comparison of results across seasons, which was important given the seasonal difference in underwater noise propagation and SRKW density. In addition, the regional model was conducted for four scenarios to estimate change in noise exposure from existing conditions to future development scenarios. What is clear is that these models do not estimate variability of behavioural response and masking because they use long-term noise monthly averages as their input.

The focused model overcame these shortcomings by implementing a simulation model that incorporated underwater noise inputs over short time periods, thus allowing for variability in noise inputs, and behavioural response and masking. This approach is important because behavioural response and masking are driven by noise extremes, not averages. This study was able to estimate the statistical distribution of low-severity and moderate-severity behavioural responses and acoustic masking, which have been provided as inputs to the Population Consequences of Disturbance (PCoD) model in order to model population-level effects of the four scenarios used in this study on the SRKW population (see SMRU 2014b).

5.2 DATA GAPS AND LIMITATIONS

Models are by necessity, simplifications of reality and are completely reliant on the quality of their inputs; therefore, any limitations in model inputs, including noise inputs, SRKW relative density, behavioural response curves, or masking could affect results of the noise exposure models. Limitations are described in the relevant technical data reports (i.e., Hemmera 2014, SMRU 2014a, JASCO 2014). As mentioned in **Section 5.1**, the regional model was applied to long-term monthly averages and depicts average change in underwater noise exposure across the regional model area, but does not provide information on variability. Estimates of variability can be found in the focused model.

Variability was included in the focused model by using underwater noise inputs in short time windows (5-minutes), and distributing SRKW in the model space by using the seasonal relative kernel density, monthly probability of occurrence, and a 4.5 km smoothing function. Where possible, any uncertainty in inputs to the focused model was incorporated (e.g., dose-response curves), to ensure that uncertainty was accounted for. To simplify model development, SRKW were placed in the model space at one location during an entire 24-hour model run. SRKW do not spend 24-hours in a single location; however, by using the SRKW probability of occurrence, SRKW kernel density, and by iterating the model 1,000 times for each day of the year, the focused model accurately approximated actual spatial and temporal distribution of SRKW with a much simpler set of code.

In addition, underwater noise from whale watch vessel noise were not included in this model as they were not included in VTOSS and therefore, the model may underestimate behavioural responses and acoustic masking during all scenarios. However, with or without the inclusion of whale-watching vessels, the absolute contribution of underwater noise by the Project and predicted behavioural responses and acoustic masking will not change.

Despite these limitations, the SRKW Underwater Noise Exposure Study can be used to inform a future effects assessment for the Project and regional commercial vessel traffic. This is because, all data inputs (and their assumptions) were kept the same across all scenarios, except for the commercial vessel noise inputs, which varied by scenario. Therefore, any over- or under-estimate of behavioural response, or masking, would be present in all the scenario results, and the relative changes across scenarios would remain fairly constant.

6.0 CLOSURE

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8.0 STATEMENT OF LIMITATIONS

This report was prepared by SMRU Canada Ltd., based on desktop studies conducted by SMRU Canada Ltd., for the sole benefit and exclusive use of Hemmera and Port Metro Vancouver. The material in it reflects SMRU Canada Ltd.'s best judgment in light of the information available to it at the time of preparing this Report. Any use that a third party makes of this Report, or any reliance on or decision made based on it, is the responsibility of such third parties. SMRU Canada Ltd. accepts no responsibility for damages, if any, suffered by any third party as a result of decisions made or actions taken based on this Report.

SMRU Canada Ltd. has performed the work as described above and made the findings and conclusions set out in this Report in a manner consistent with the level of care and skill normally exercised by members of the environmental science profession practicing under similar conditions at the time the work was performed.

This Report represents a reasonable review of the information available to SMRU Canada Ltd. within the established Scope, work schedule and budgetary constraints. The conclusions and recommendations contained in this Report are based upon applicable legislation existing at the time the Report was drafted. Any changes in the legislation may alter the conclusions and/or recommendations contained in the Report. Regulatory implications discussed in this Report were based on the applicable legislation existing at the time this Report was written.

In preparing this Report, SMRU Canada Ltd. has relied in good faith on information provided by others as noted in this Report, and has assumed that the information provided by those individuals is both factual and accurate. SMRU Canada Ltd. accepts no responsibility for any deficiency, misstatement or inaccuracy in this Report resulting from the information provided by those individuals.

APPENDIX A
Figures

Figure A 1 Summer SRKW Sightings Locations, SRKW Critical Habitat, Regional Model Area, Focused Model Area, and LSA

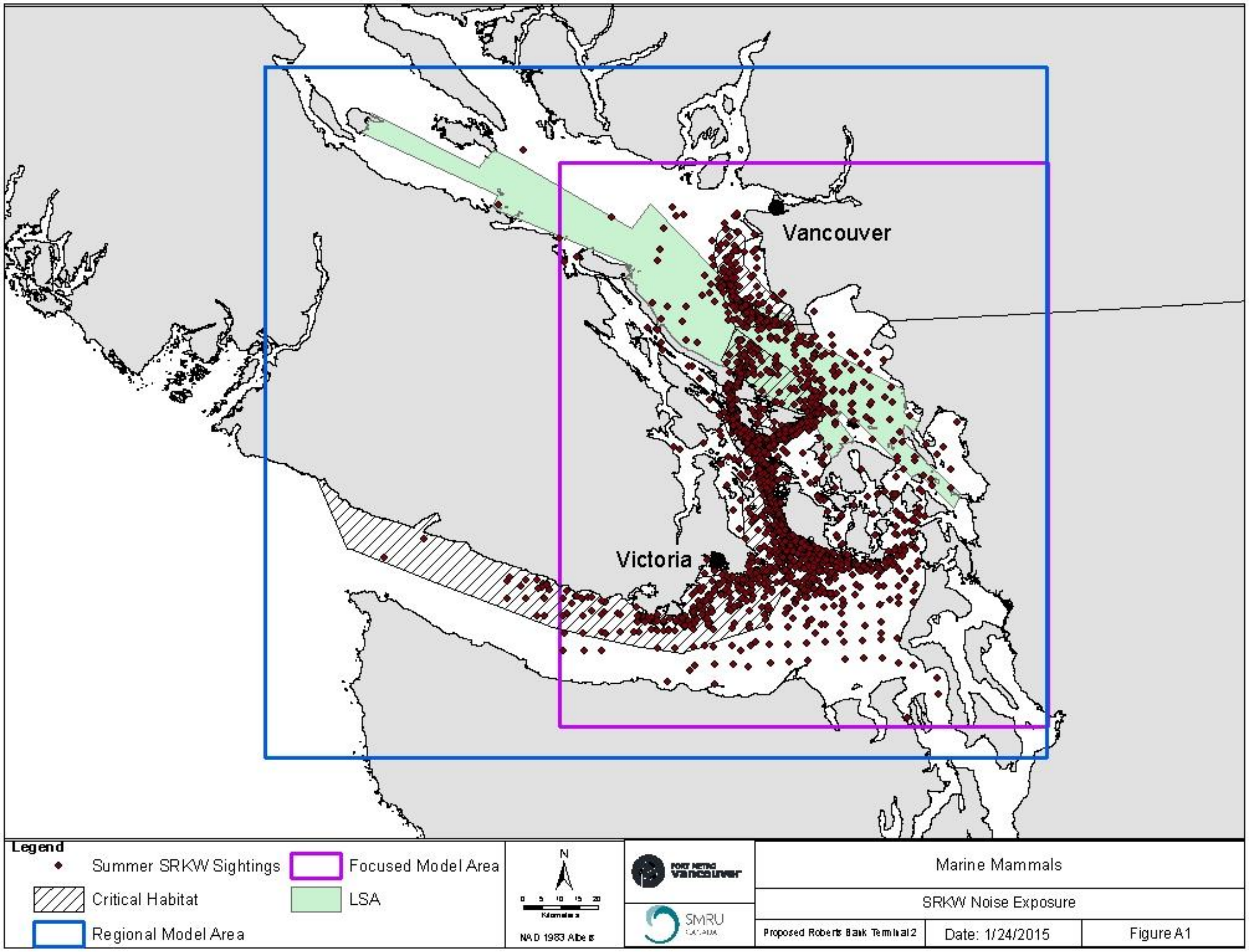


Figure A 2 Winter SRKW Sightings Locations, SRKW Critical Habitat (Canadian and US), Regional Model Area, Focused Model Area, and LSA

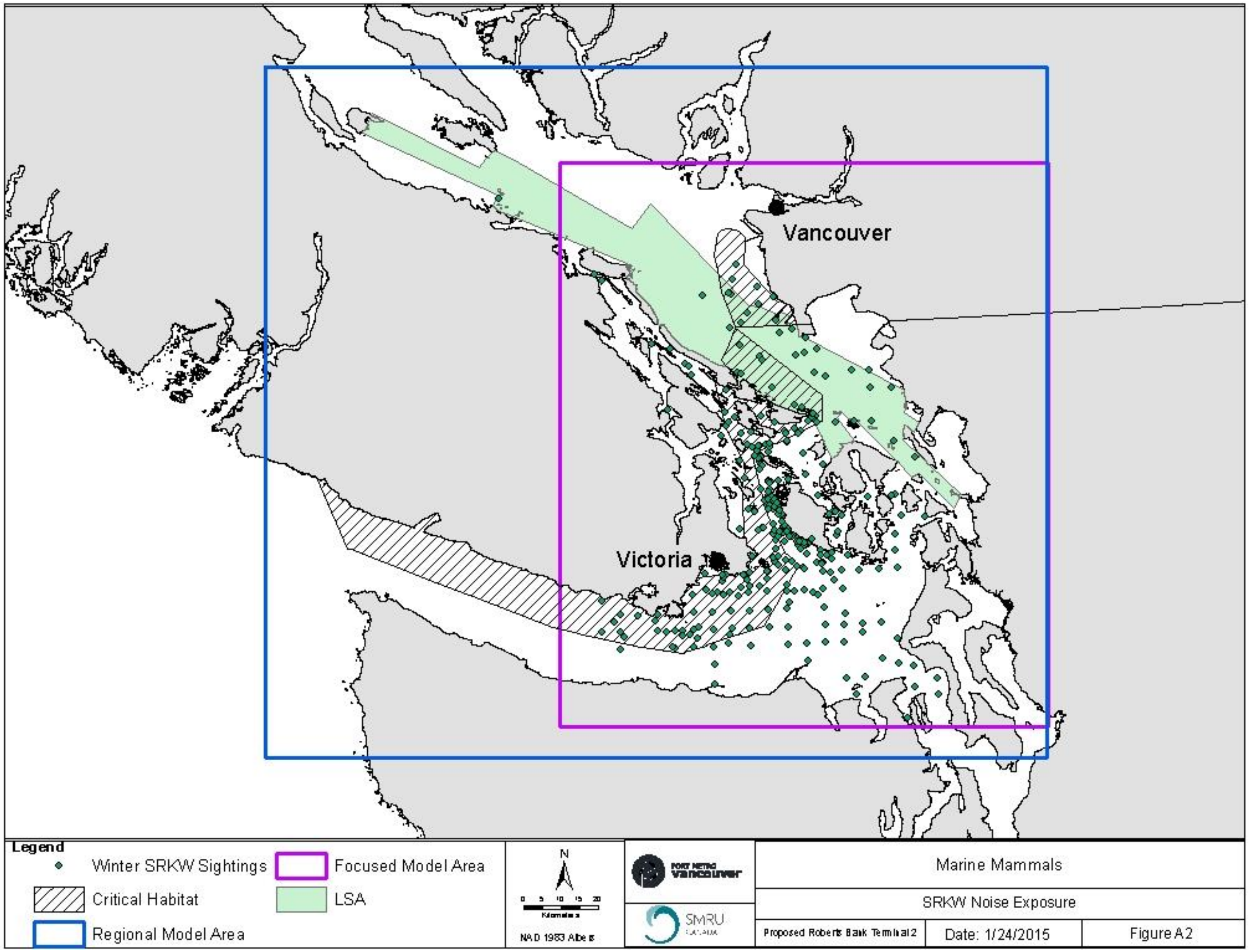


Figure A 3 Summer Normalised Regional Noise Exposure for Existing Conditions (S1)

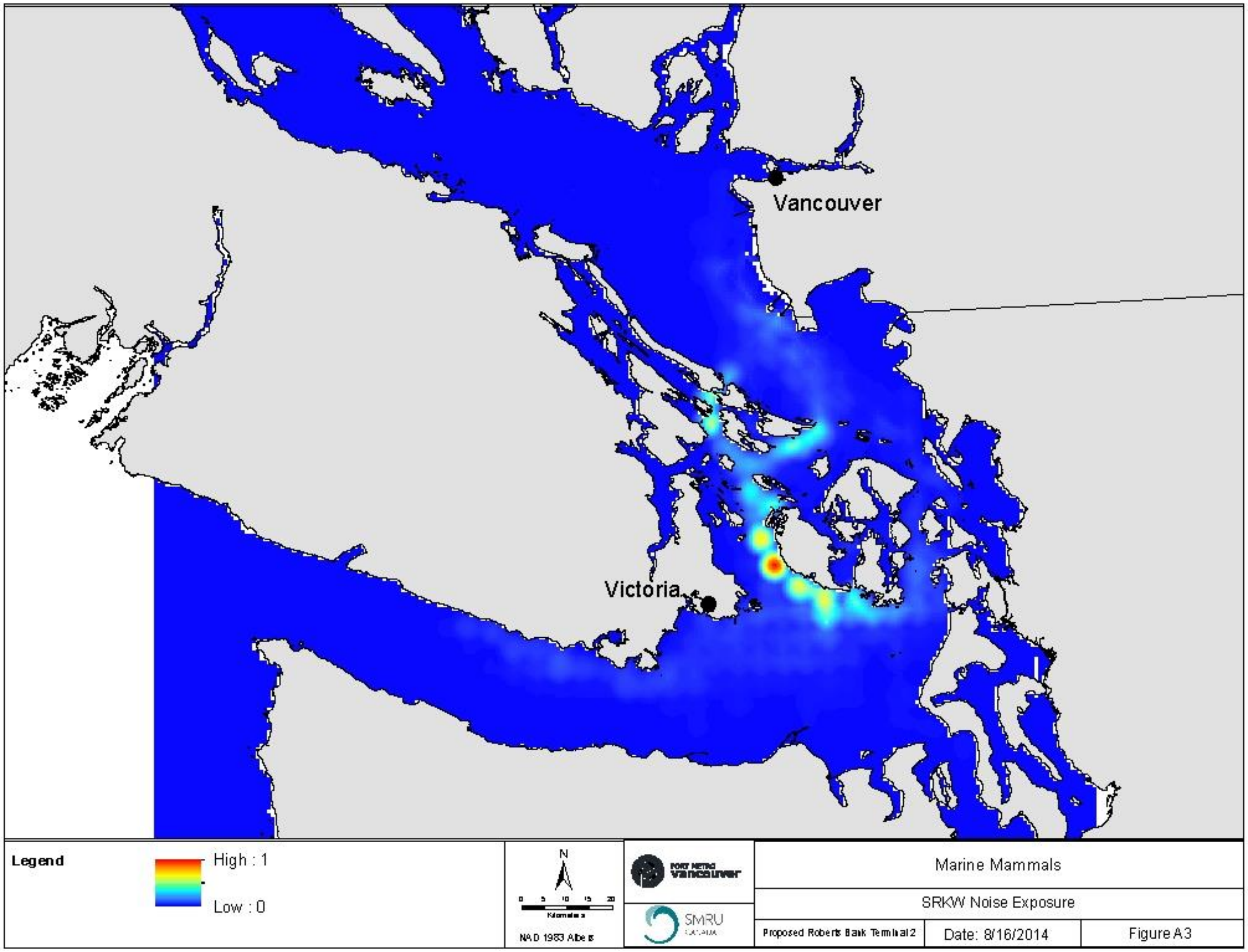


Figure A 4 Summer Normalised Regional Noise Exposure for S3

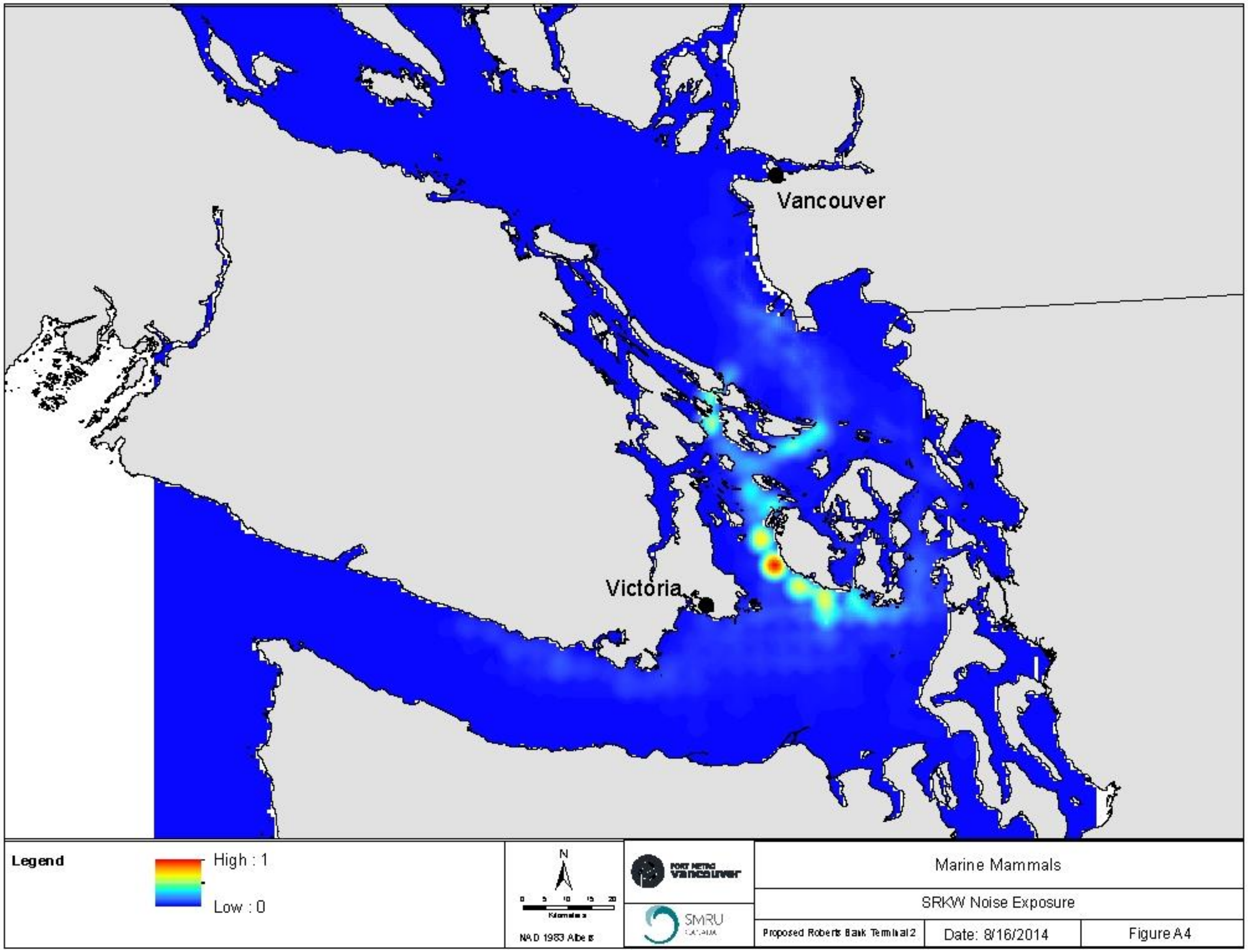


Figure A 5 Summer Normalised Regional Noise Exposure for S4

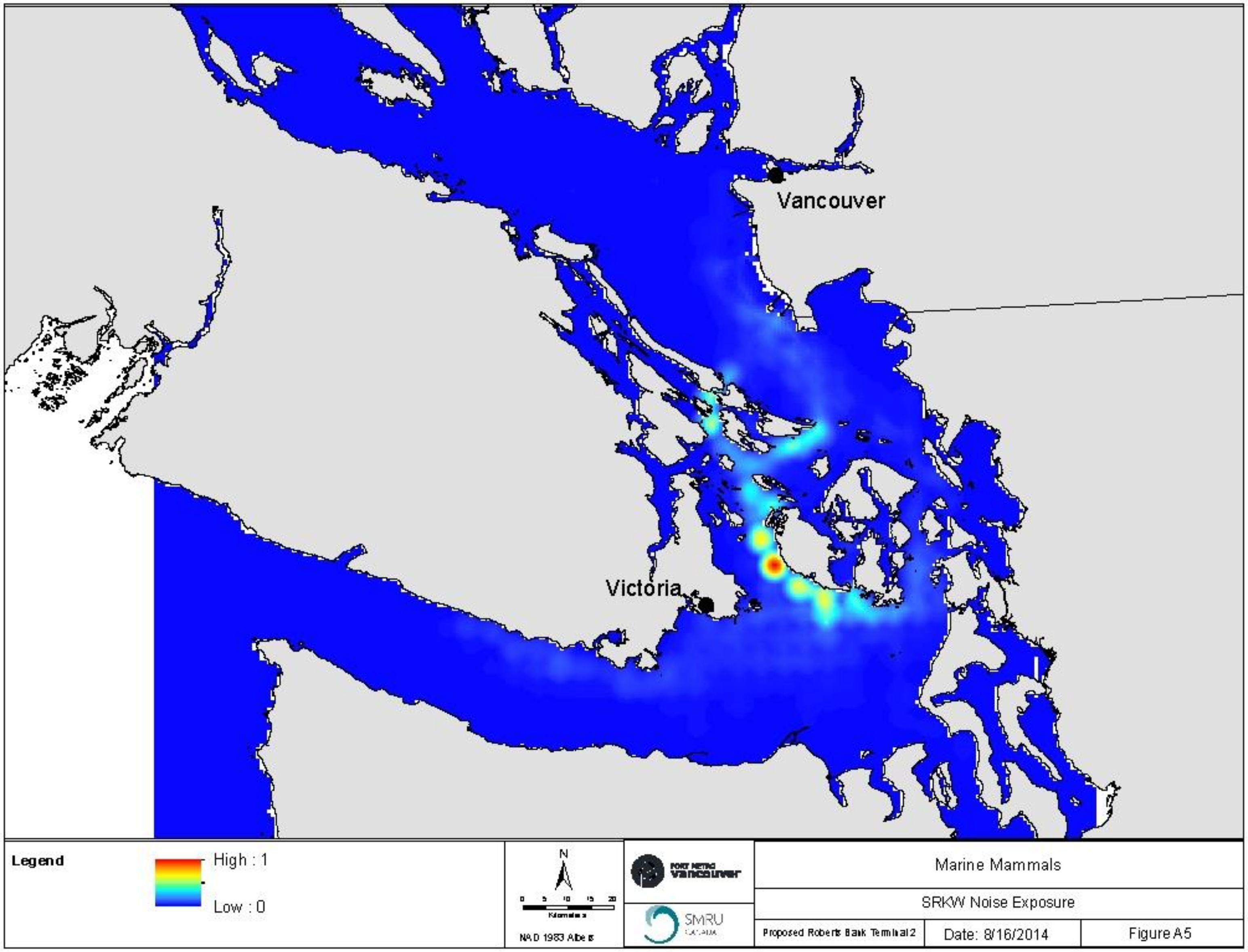


Figure A 6 Summer Change in Normalised Noise Exposure from S1 to S3

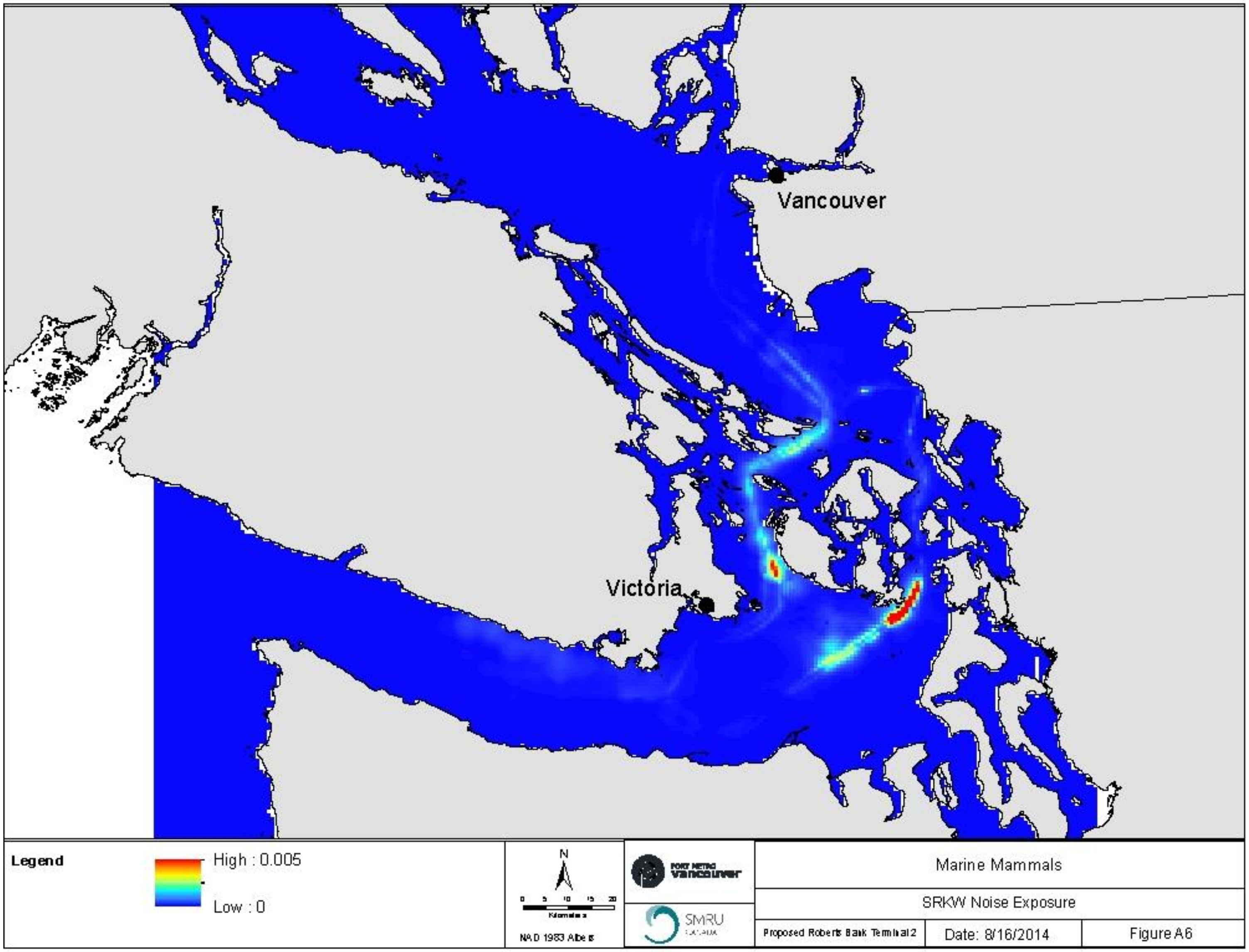


Figure A 7 Summer Change in Normalised Noise Exposure from S1 to S4

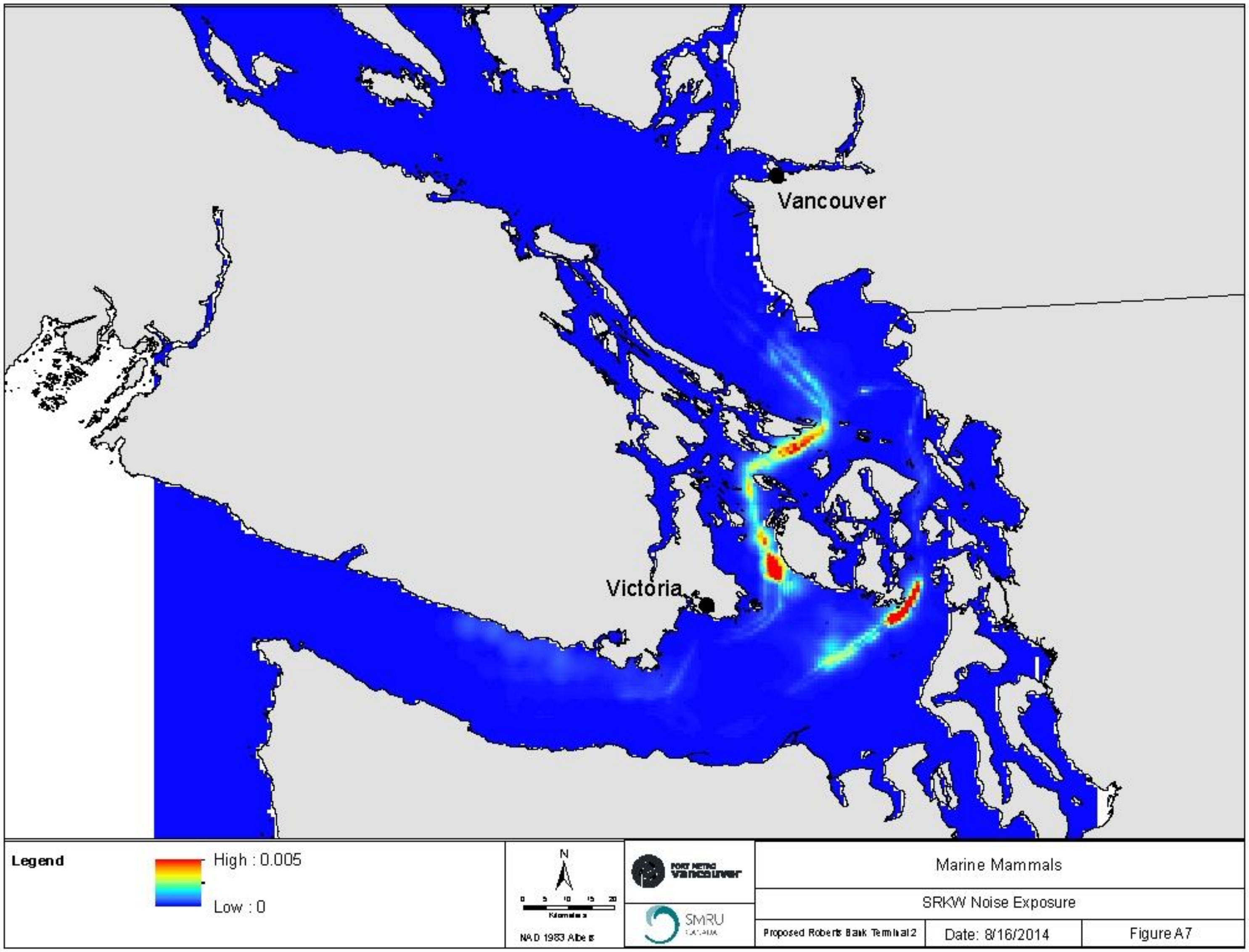


Figure A 8 Winter Normalised Regional Noise Exposure for S1

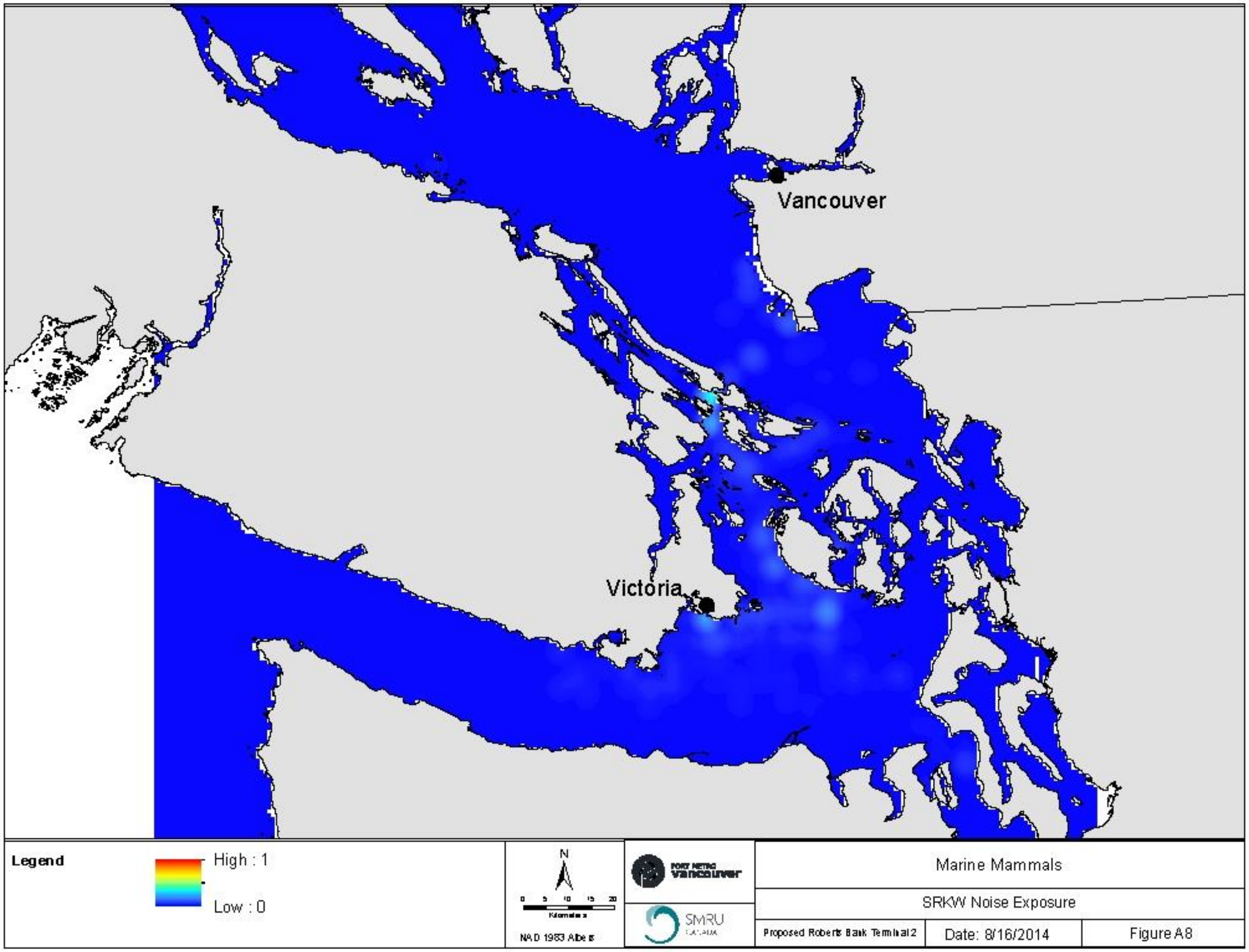


Figure A 9 Winter Normalised Regional Noise Exposure for S3

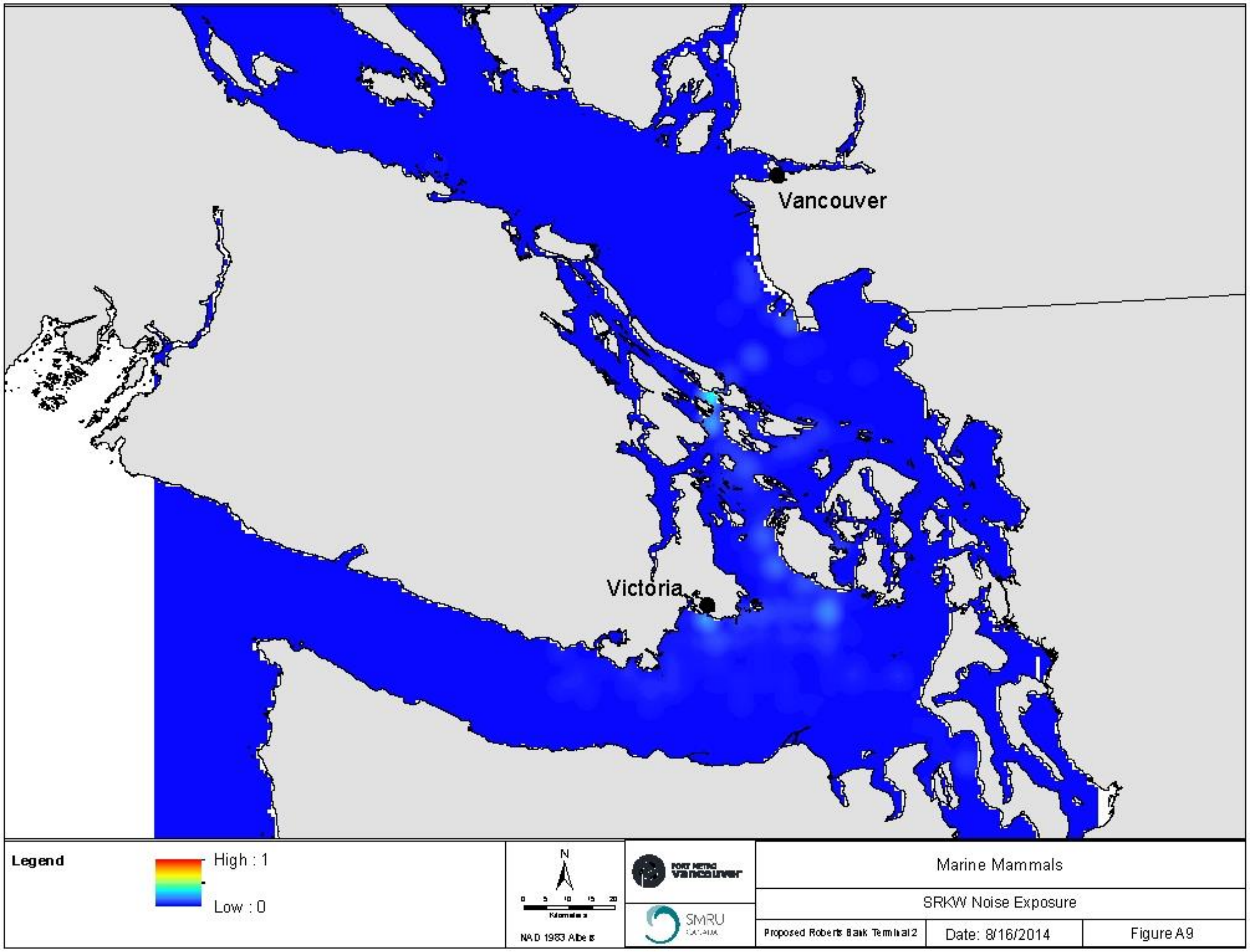


Figure A 10 Winter Normalised Regional Noise Exposure for S4

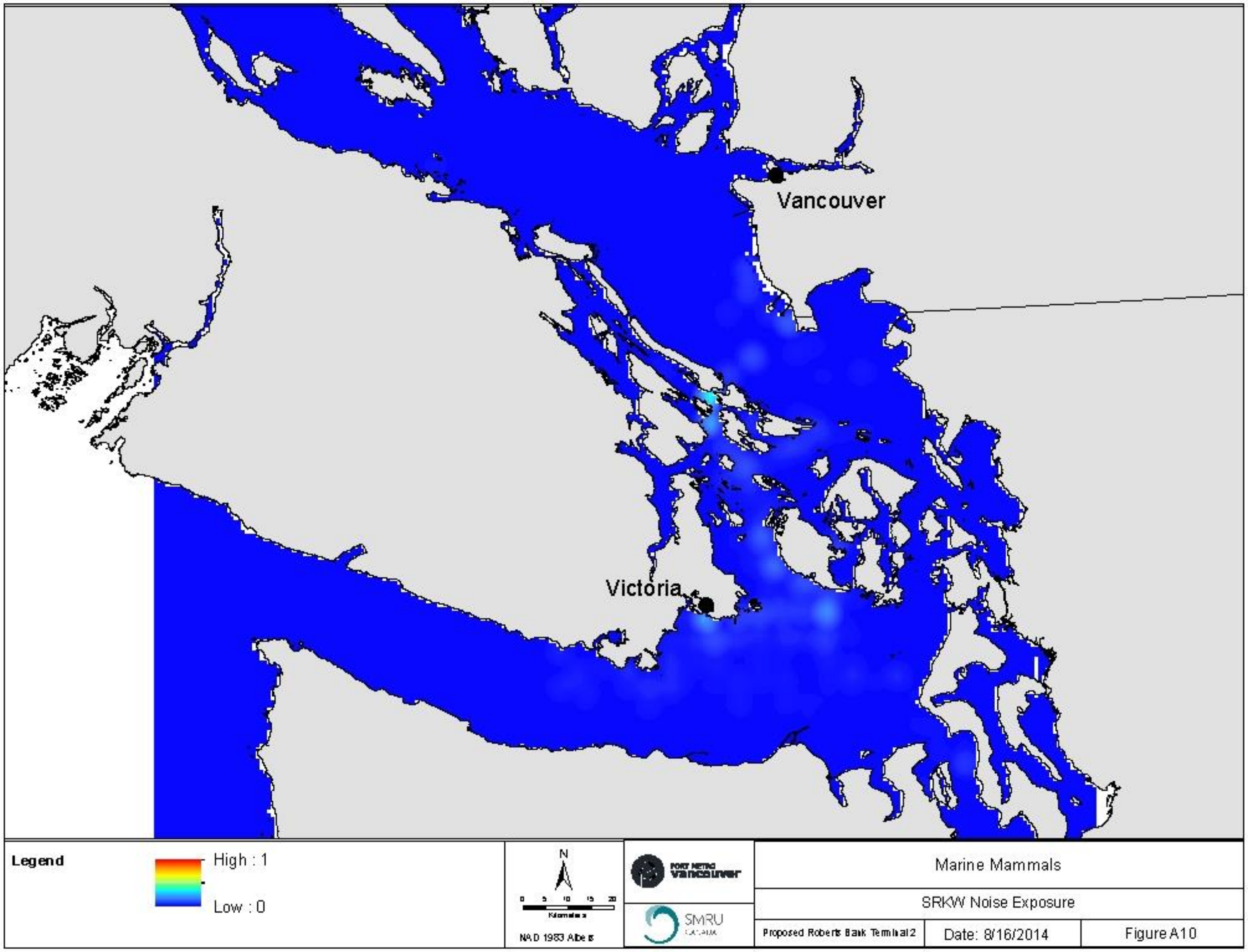


Figure A 11 Winter Change in Normalised Noise Exposure from S1 to S3

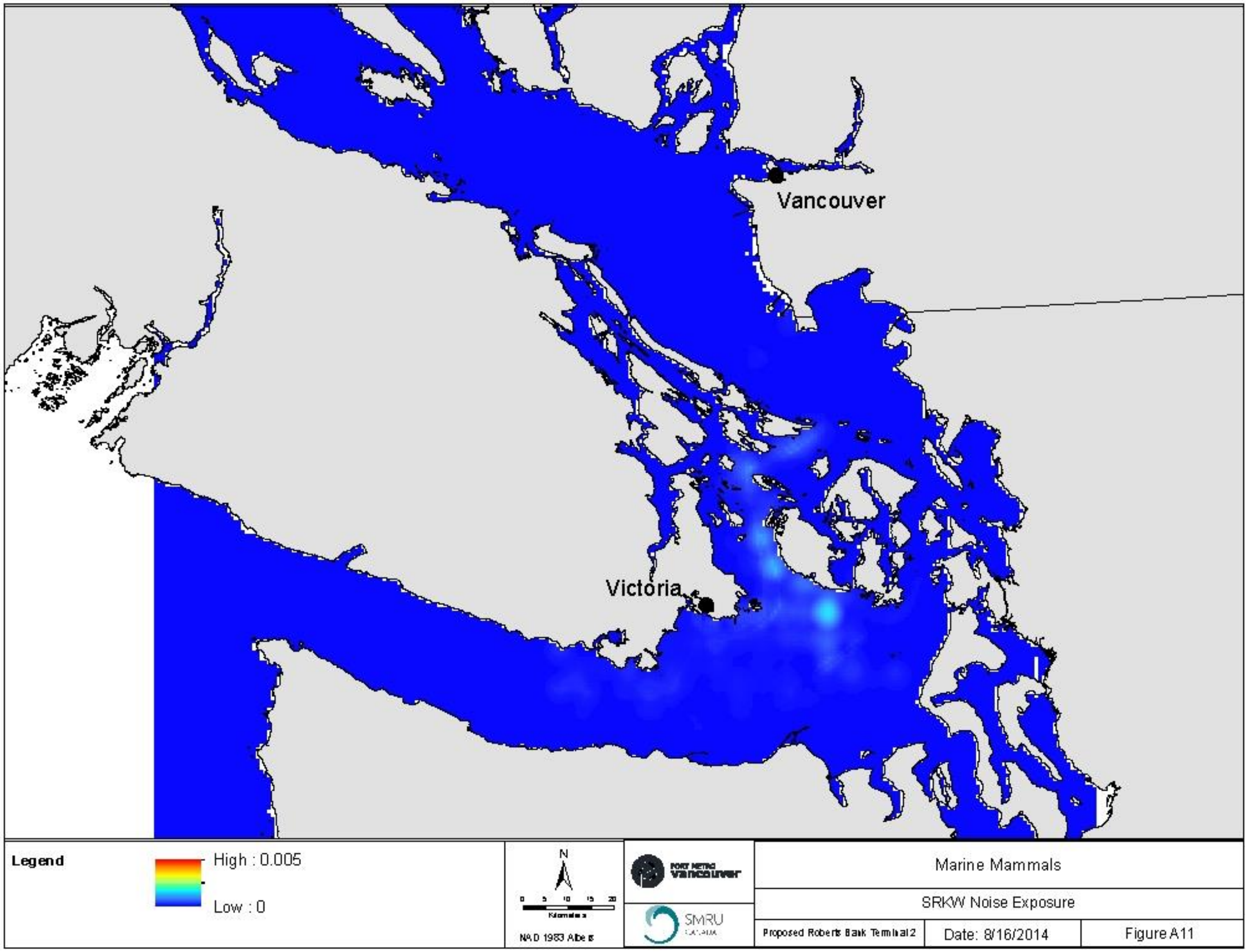


Figure A 12 Winter Change in Normalised Noise Exposure from S1 to S4

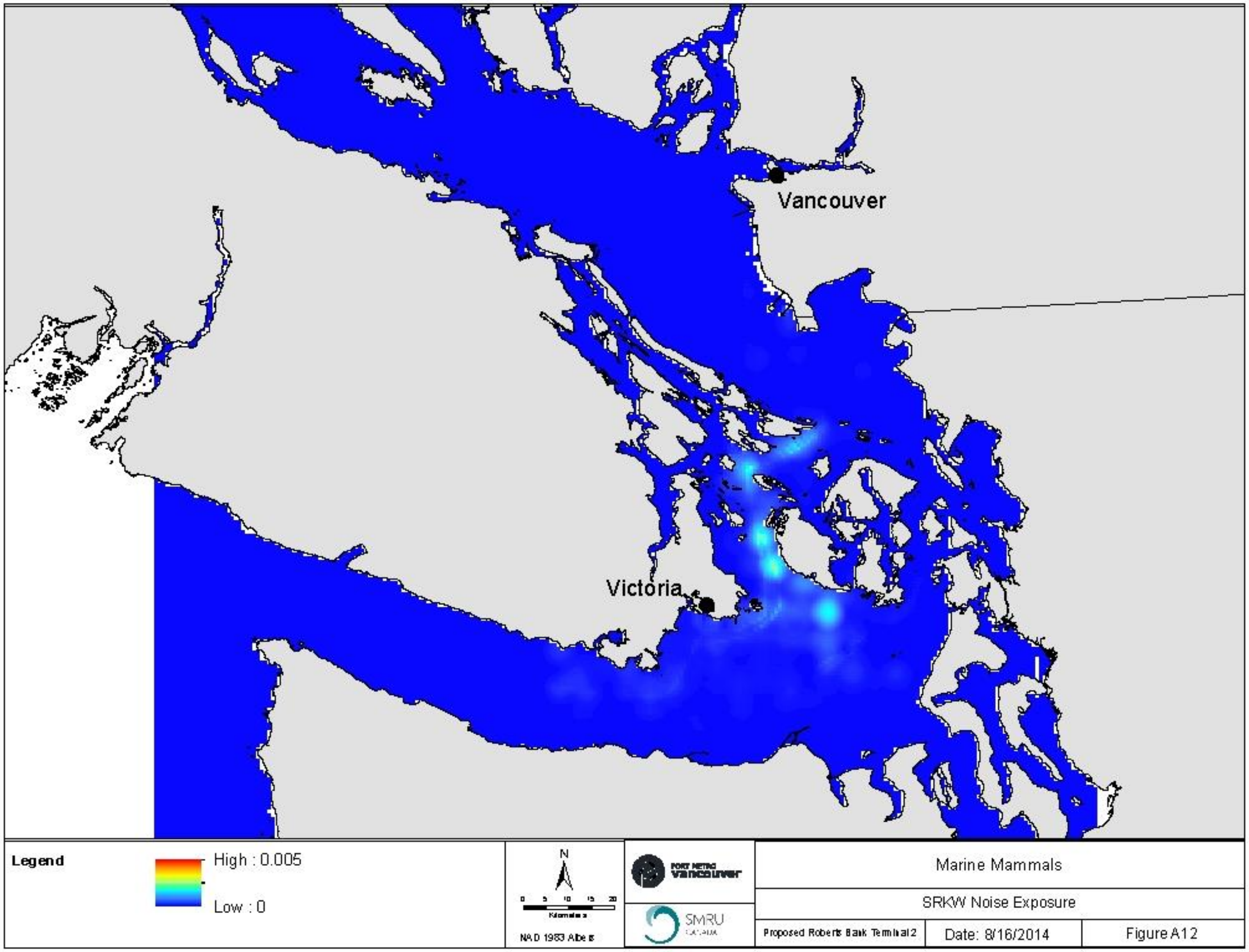
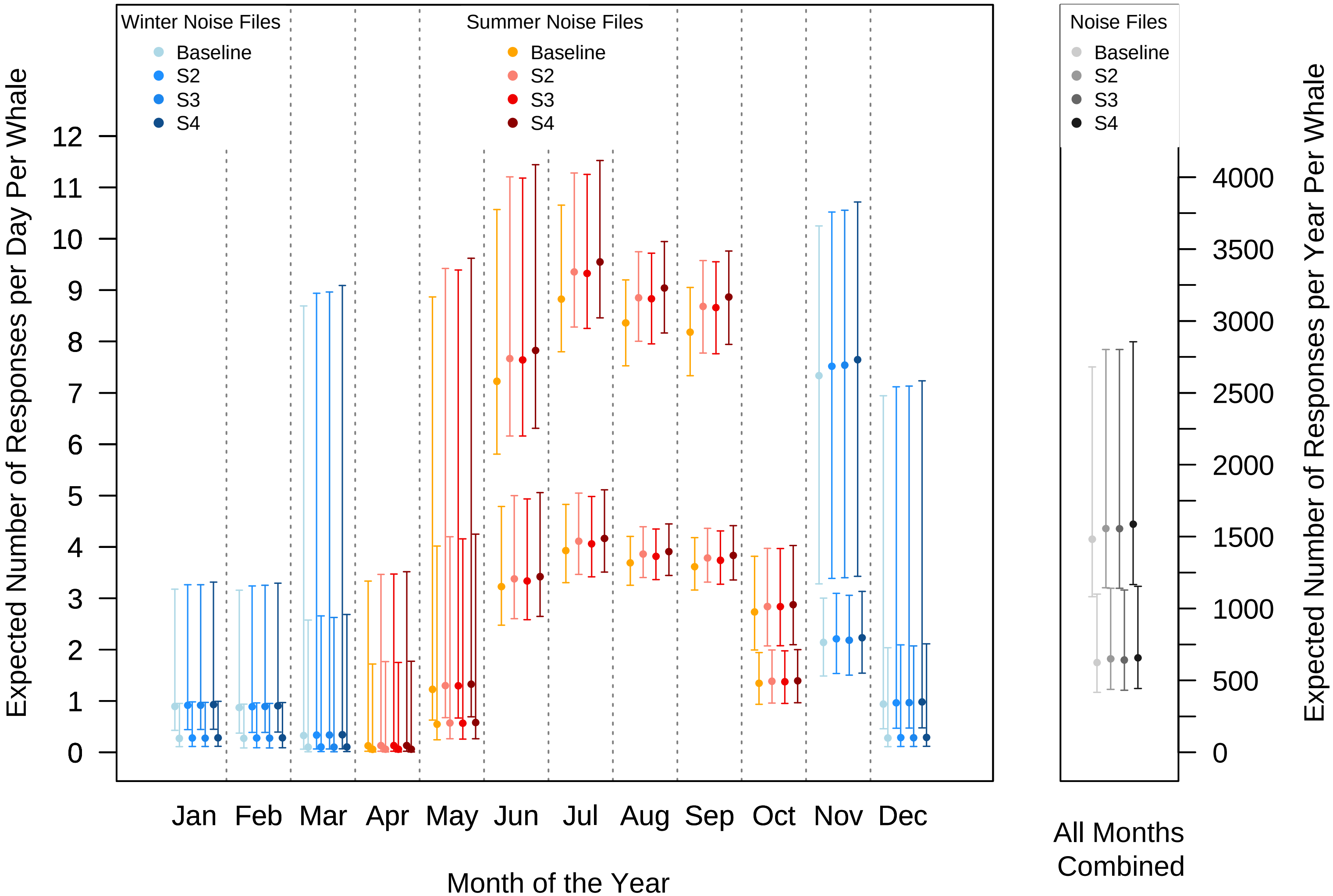
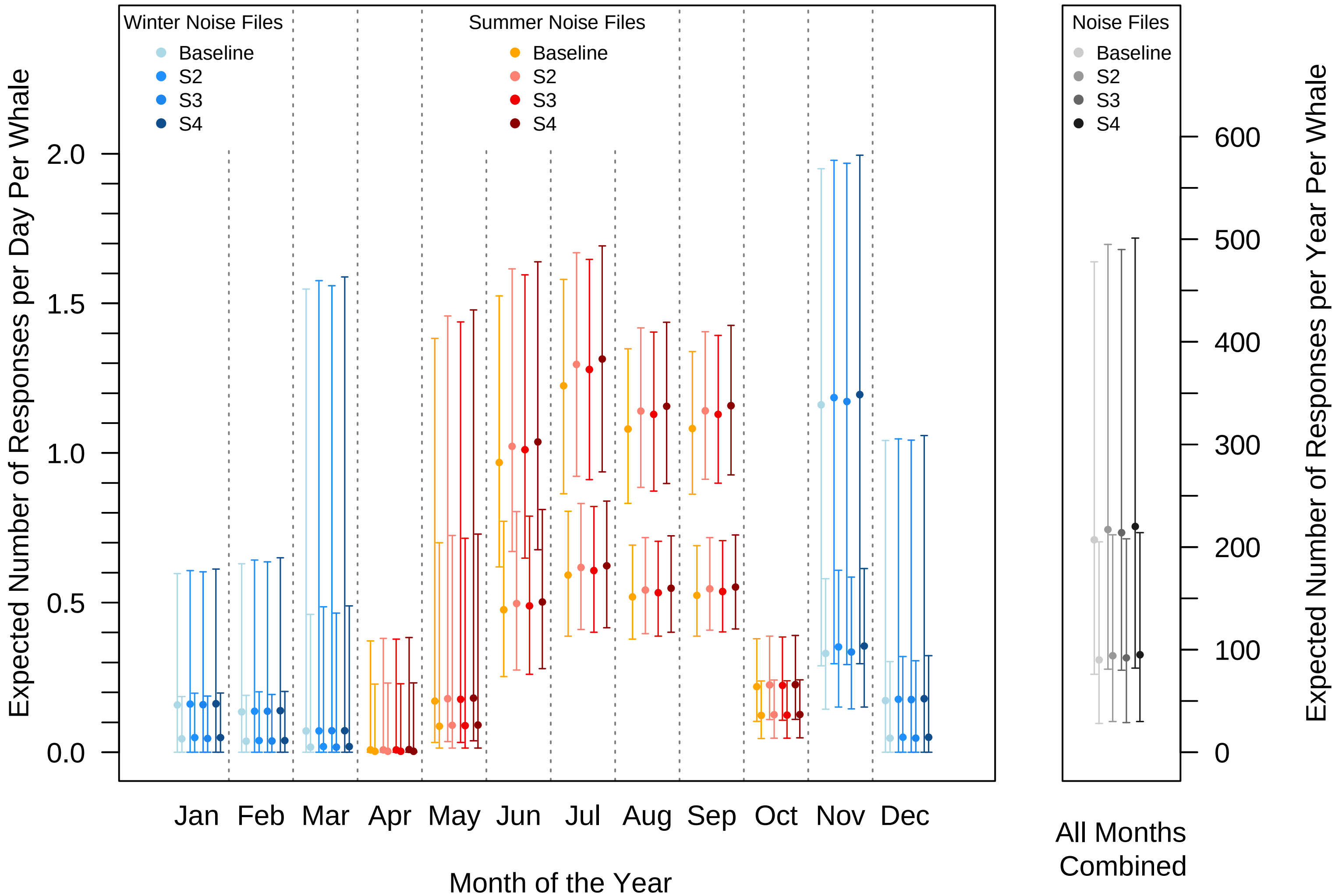


Figure A 13 Monthly Median Values and 95% CI for Low and Moderate Behavioural Responses per Day per Whale in the Focused Study Area



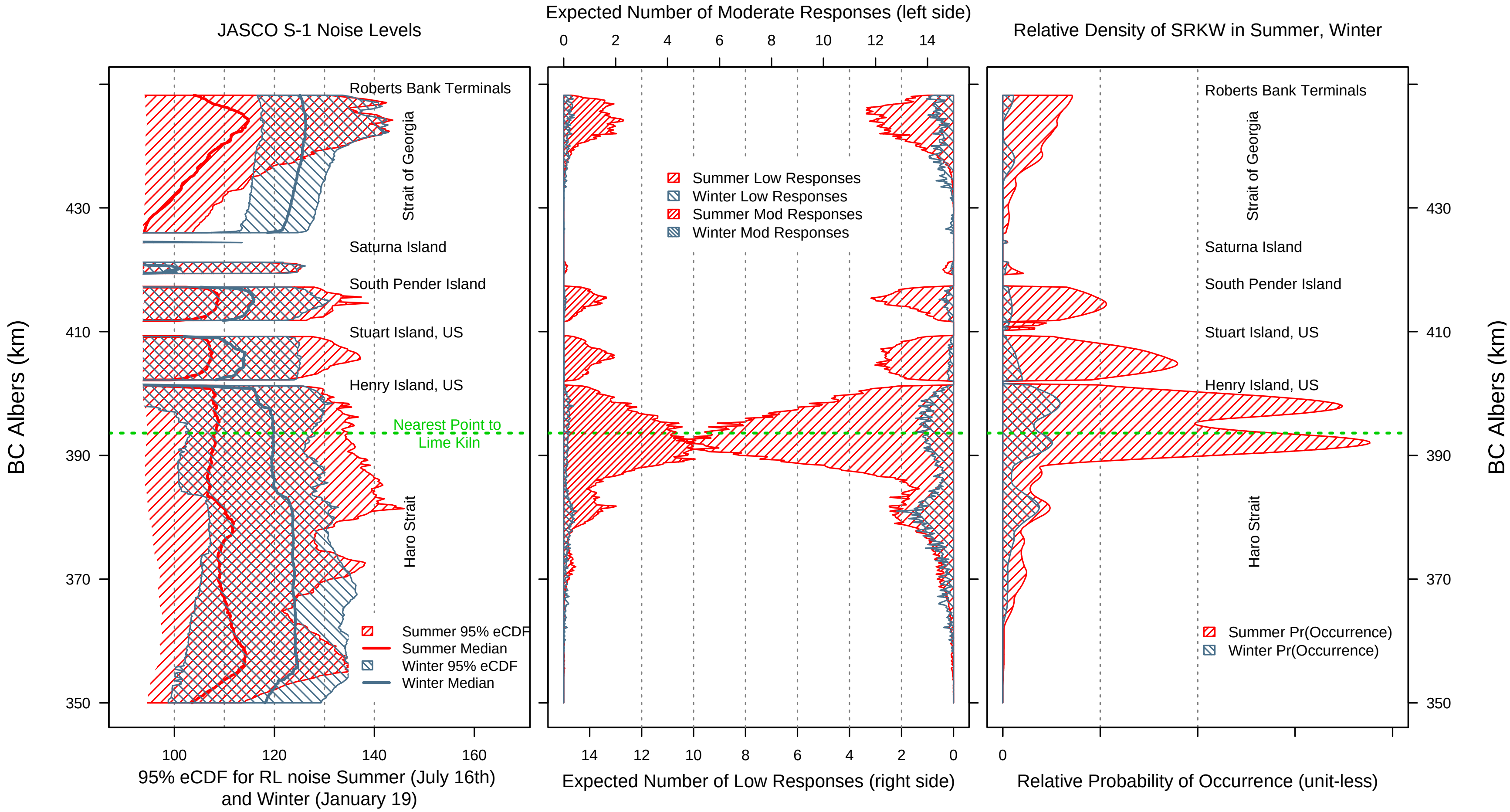
Note: Values are plotted in pairs with the low-severity behavioural responses having higher numbers of responses per day. The right panel sums the number of behavioural responses expected to occur for a randomly selected SRKW over 365 days.

Figure A 14 Monthly Median Values and 95% CI for Low-Severity and Moderate-Severity Behavioural Responses per Day per Whale in the LSA



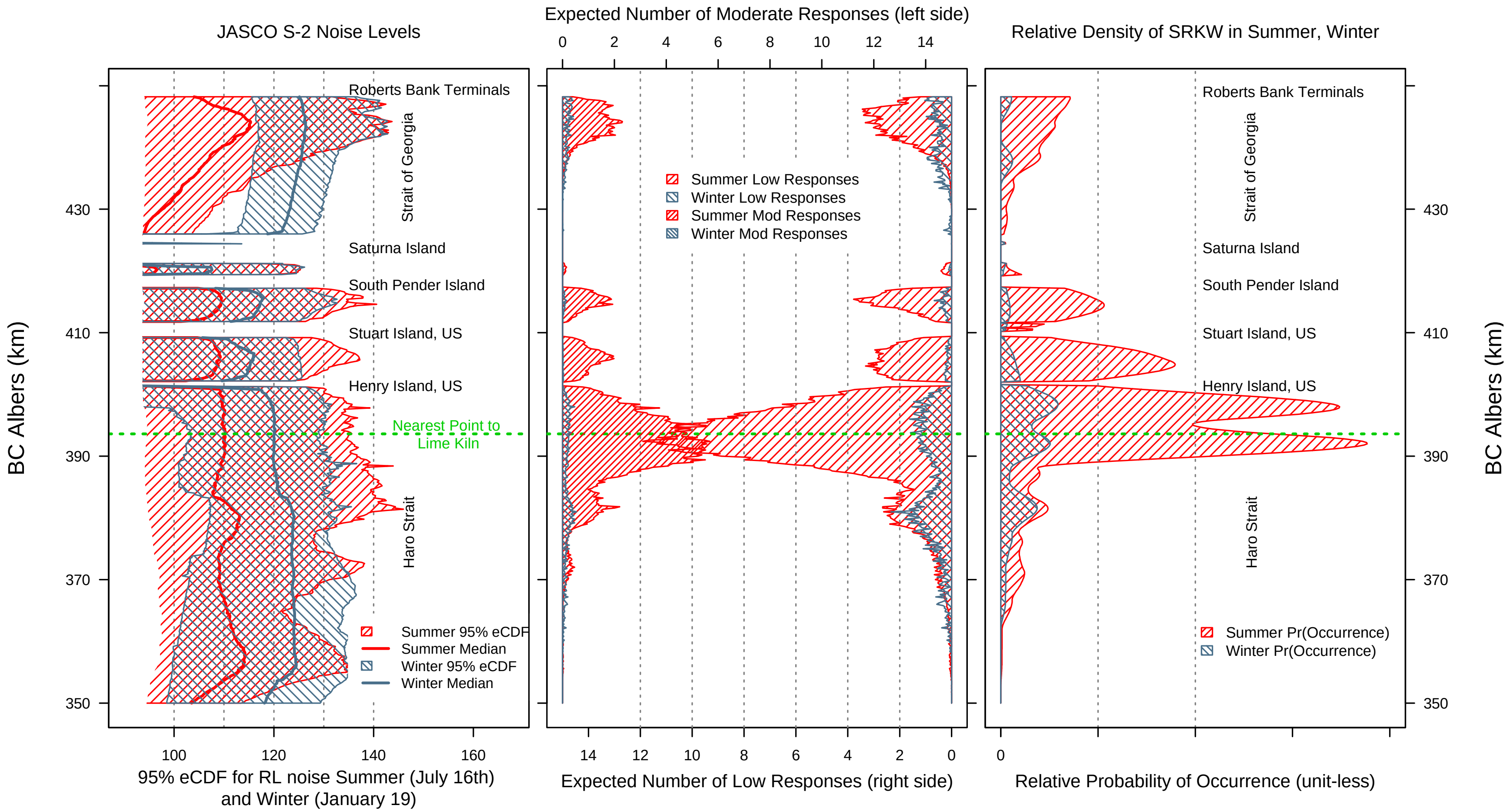
Note: Values are plotted in pairs with the low-severity behavioural responses having higher numbers of responses per day. The right panel sums the number of behavioural responses expected to occur for a randomly selected SRKW over 365 days.

Figure A 15 S1 Outputs Along the Transect Line in Figure 10



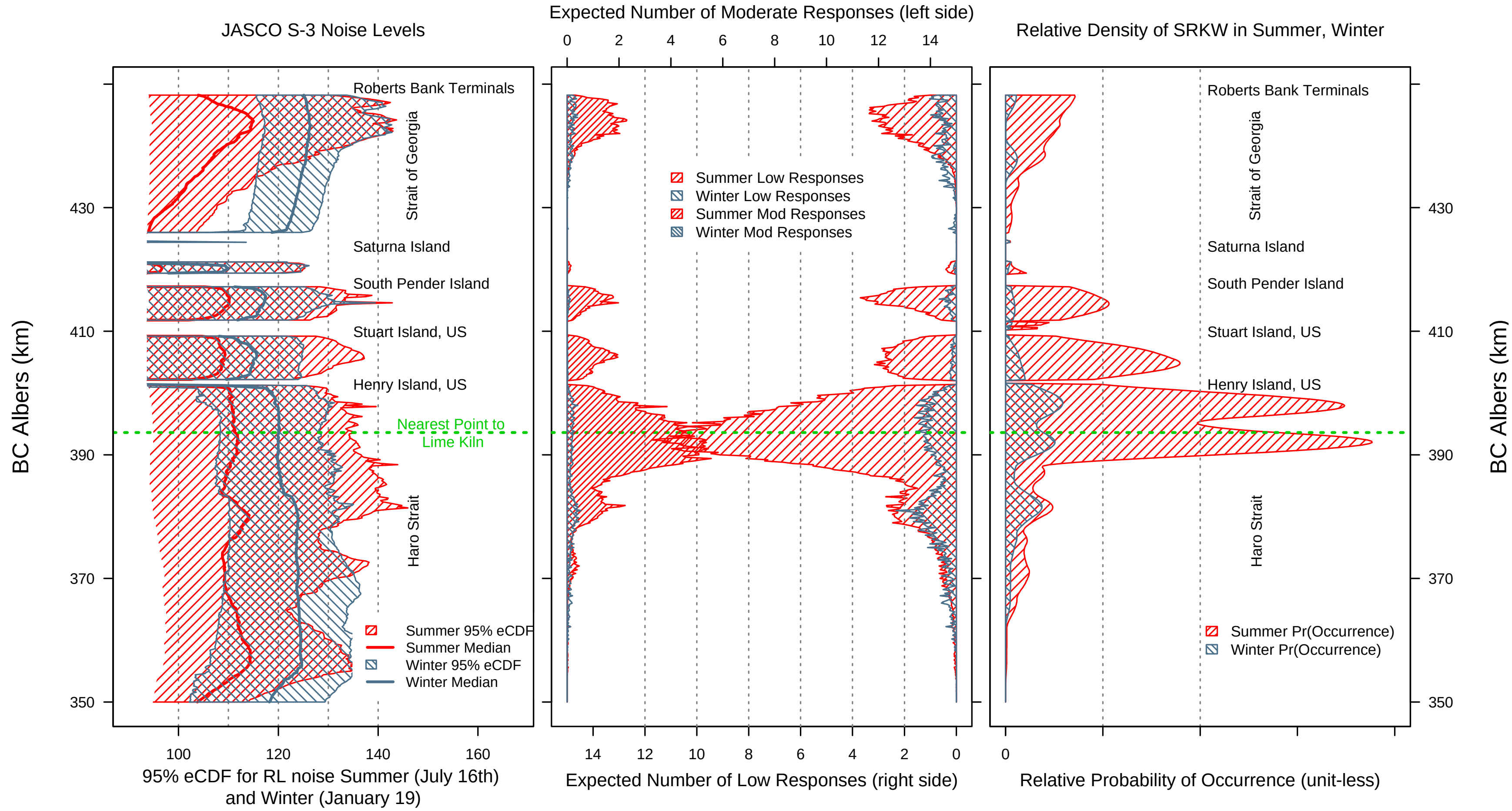
Note: The left panels compare the median (bold line) and ventral 95% distribution of JASCO broadband noise files for summer and winter. The right panels compare the relative probability that an SRKW will be at that Location in summer relative to winter. The central panels show the number of low (right axis) and moderate (left axis) behavioural responses that occurred along the transect during the summer and winter.

Figure A 16 S2 Outputs Along the Transect Line in Figure 10



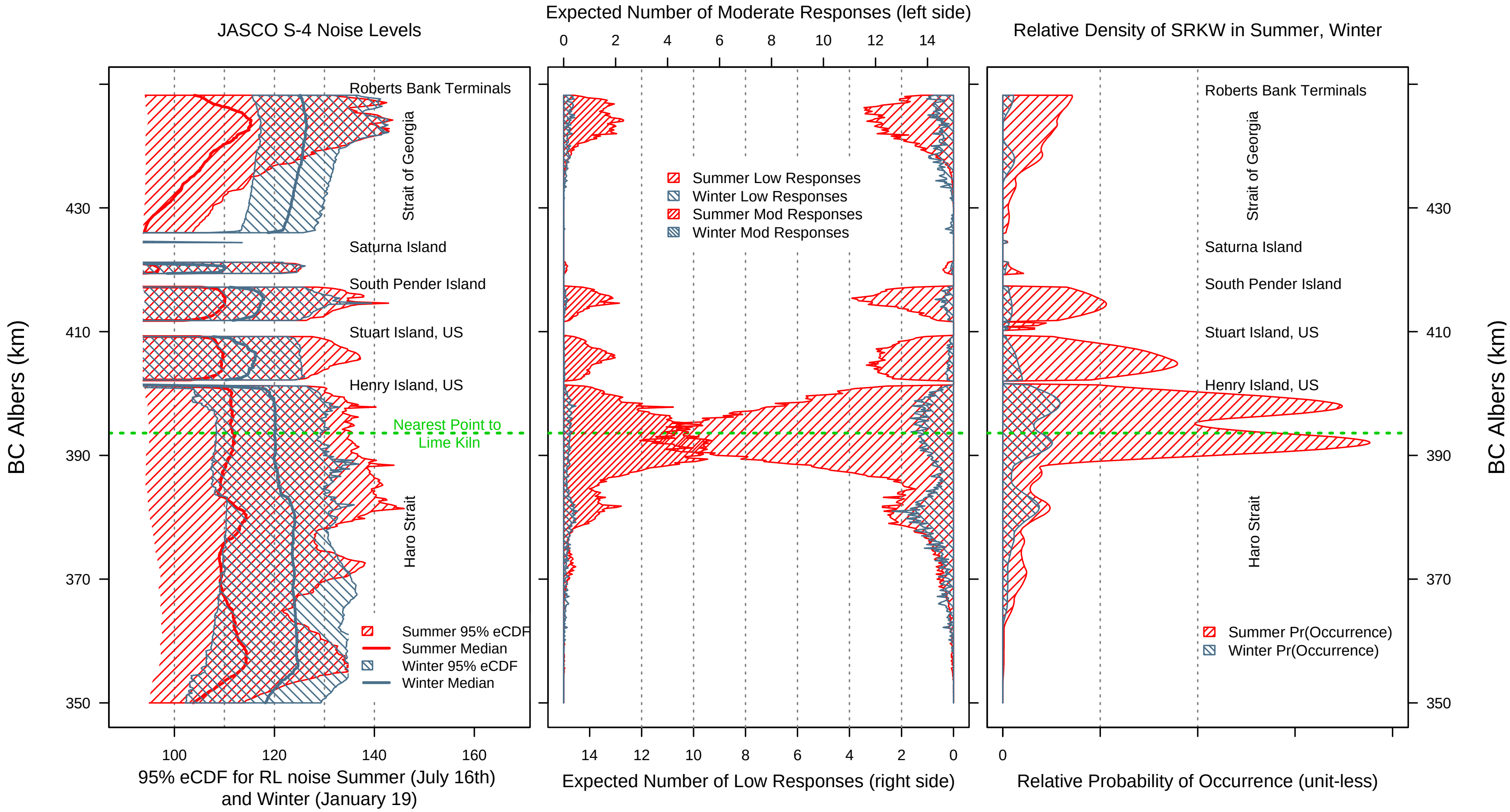
Note: The left panels compare the median (bold line) and central 95% distribution of JASCO broadband noise files for summer and winter. The right panels compare the relative probability that an SRKW will be at that location in summer relative to winter. The central panels show the number of low-severity (right axis) and moderate-severity (left axis) behavioural responses that occurred along the transect during the summer and winter.

Figure A 17 S3 Outputs Along the Transect Line in Figure 10



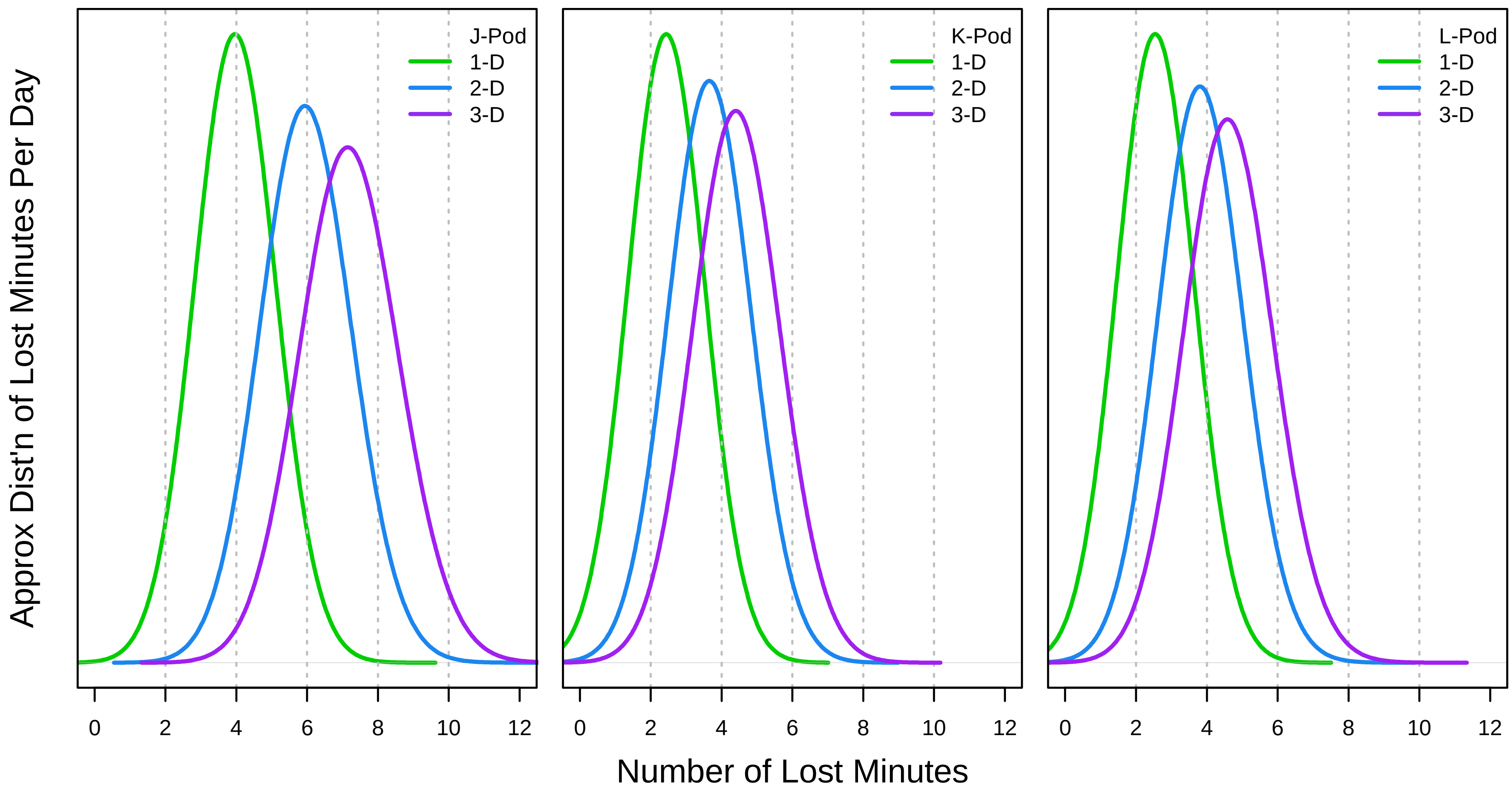
Note: The left panels compare the median (bold line) and central 95% distribution of JASCO broadband noise files for summer and winter. The right panels compare the relative probability that an SRKW will be at that Location in summer relative to winter. The central panels show the number of low-severity (right axis) and moderate-severity (left axis) behavioural responses that occurred along the transect during the summer and winter.

Figure A 18 S4 Outputs Along the Transect Line in Figure 10



Note: The left panels compare the median (bold line) and central 95% distribution of JASCO broadband noise files for summer and winter. The right panels compare the relative probability that an SRKW will be at that location in summer relative to winter. The central panels show the number of low-severity (right axis) and moderate-severity (left axis) behavioural responses that occurred along the transect during the summer and winter.

Figure A 19 **Distributions of the Number of Estimated Foraging Minutes Lost to Additional Masking per Day per Whale Using 1, 2, and 3-D measures for Pods J, K, and L under Existing Conditions Scenario**



APPENDIX B

Tables

Table A 1 Median Levels (and 95% CI) of Low-Severity and Moderate-Severity Behavioural Response per Year per Whale during S1 by Pod and Study Area

Area	Response Type	J	K	L
Focused Model Area	LOW	2,353 (1,920; 2,805)	1,358 (1,043; 1,725)	1,385 (1,063; 1,733)
Focused Model Area	MOD	930 (726; 1,169)	557 (394; 754)	573 (411; 767)
LSA	LOW	306 (139; 556)	173 (67; 356)	177 (70; 341)
LSA	MOD	127 (52; 237)	76 (24; 163)	78 (25; 159)

Table A 2 Median Levels (and 95% CI) of Low-Severity and Moderate-Severity Behavioural Response per Year per Whale during S2 by Pod and Study Area

Area	Response Type	J	K	L
Focused Model Area	LOW	2,467 (2,019; 2,931)	1,429 (1,101; 1,805)	1,457 (1,126; 1,817)
Focused Model Area	MOD	969 (760; 1,213)	581 (414; 782)	598 (432; 796)
LSA	LOW	319 (149; 574)	182 (73; 367)	186 (75; 352)
LSA	MOD	133 (56; 246)	79 (25; 168)	82 (27; 164)

Table A 3 Median Levels (and 95% CI) of Low-Severity and Moderate-Severity Behavioural Response per Year per Whale during S3 by Pod and Study Area

Area	Response Type	J	K	L
Focused Model Area	LOW	2,464 (2,016; 2,929)	1,427 (1,100; 1,806)	1,454 (1,123; 1,816)
Focused Model Area	MOD	957 (749; 1,200)	574 (408; 773)	591 (426; 788)
LSA	LOW	316 (146; 569)	180 (72; 364)	184 (73; 351)
LSA	MOD	130 (54; 240)	78 (25; 165)	80 (26; 162)

Table A 4 Median Levels (and 95% CI) of Low-Severity and Moderate-Severity Behavioural Response per Year per Whale during S4 by Pod and Study Area

Area	Response Type	J	K	L
Focused Model Area	LOW	2,514 (2,060; 2,984)	1,457 (1,124; 1,840)	1,486 (1,149; 1851)
Focused Model Area	MOD	980 (769; 1225)	588 (420; 789)	605 (438; 803)
LSA	LOW	323 (151; 580)	184 (74; 370)	188 (76; 356)
LSA	MOD	134 (57; 248)	80 (26; 169)	83 (27; 165)

Table A 5 Median (95% CI) Low-Severity Behavioural Reponses per Month per Whale by Scenario for the Focused Model Area

Area	Month	S1	S2	S3	S4
Focused Model Area	JAN	0.89 (0.43; 3.18)	0.92 (0.44; 3.26)	0.92 (0.44; 3.26)	0.93 (0.45; 3.31)
•	FEB	0.87 (0.37; 3.16)	0.89 (0.39; 3.24)	0.89 (0.39; 3.25)	0.90 (0.39; 3.30)
•	MAR	0.33 (0.06; 8.69)	0.34 (0.07; 8.94)	0.34 (0.07; 8.96)	0.34 (0.07; 9.09)
•	APR	0.13 (0.02; 3.33)	0.13 (0.02; 3.47)	0.13 (0.02; 3.47)	0.13 (0.02; 3.52)
•	MAY	1.23 (0.63; 8.87)	1.30 (0.68; 9.42)	1.29 (0.67; 9.39)	1.33 (0.69; 9.62)
•	JUN	7.22 (5.81; 10.57)	7.67 (6.16; 11.21)	7.64 (6.16; 11.18)	7.83 (6.31; 11.44)
•	JUL	8.82 (7.80; 10.66)	9.35 (8.28; 11.28)	9.33 (8.25; 11.25)	9.55 (8.46; 11.52)
•	AUG	8.36 (7.52; 9.20)	8.85 (8.00; 9.75)	8.83 (7.95; 9.72)	9.04 (8.17; 9.95)
•	SEP	8.18 (7.33; 9.05)	8.68 (7.77; 9.58)	8.66 (7.76; 9.55)	8.87 (7.94; 9.76)
•	OCT	2.73 (1.99; 3.82)	2.84 (2.07; 3.97)	2.84 (2.07; 3.97)	2.88 (2.09; 4.03)
•	NOV	7.33 (3.28; 10.25)	7.52 (3.39; 10.52)	7.54 (3.40; 10.55)	7.65 (3.43; 10.72)
•	DEC	0.94 (0.46; 6.94)	0.96 (0.47; 7.12)	0.97 (0.47; 7.13)	0.98 (0.48; 7.24)
•	ALL MONTHS COMBINED	1482 (1082; 2680)	1556 (1144; 2801)	1555 (1141; 2802)	1587 (1167; 2855)

Table A 6 Median (95% CI) Moderate-Severity Behavioural Reponses per Month per Whale by Scenario for the Focused Model Area

Area	Month	S1	S2	S3	S4
Focused Model Area	JAN	0.27 (0.11; 0.95)	0.28 (0.11; 0.98)	0.28 (0.11; 0.97)	0.28 (0.11; 0.99)
•	FEB	0.27 (0.08; 0.94)	0.28 (0.09; 0.96)	0.28 (0.09; 0.95)	0.28 (0.09; 0.97)
•	MAR	0.10 (0.01; 2.57)	0.10 (0.01; 2.66)	0.10 (0.01; 2.63)	0.10 (0.01; 2.68)
•	APR	0.06 (0.00; 1.72)	0.06 (0.00; 1.77)	0.06 (0.00; 1.75)	0.06 (0.00; 1.77)
•	MAY	0.55 (0.24; 4.02)	0.57 (0.26; 4.20)	0.57 (0.26; 4.16)	0.58 (0.26; 4.25)
•	JUN	3.23 (2.48; 4.79)	3.38 (2.60; 5.00)	3.34 (2.58; 4.93)	3.42 (2.65; 5.06)
•	JUL	3.93 (3.30; 4.83)	4.11 (3.46; 5.05)	4.06 (3.42; 4.98)	4.17 (3.51; 5.11)
•	AUG	3.69 (3.25; 4.21)	3.86 (3.40; 4.39)	3.82 (3.36; 4.35)	3.91 (3.44; 4.45)
•	SEP	3.62 (3.16; 4.18)	3.78 (3.31; 4.36)	3.74 (3.27; 4.31)	3.83 (3.36; 4.41)
•	OCT	1.34 (0.93; 1.94)	1.38 (0.96; 1.99)	1.37 (0.95; 1.97)	1.39 (0.96; 2.00)
•	NOV	2.14 (1.48; 3.00)	2.21 (1.53; 3.09)	2.18 (1.50; 3.06)	2.23 (1.54; 3.13)
•	DEC	0.28 (0.11; 2.04)	0.29 (0.11; 2.09)	0.28 (0.11; 2.07)	0.29 (0.12; 2.11)
•	ALL MONTHS COMBINED	624 (417; 1100)	650 (438; 1141)	642 (432; 1129)	657 (444; 1154)

Table A 7 Median (95% CI) Low-Severity Behavioural Reponses per Month per Whale by Scenario for the LSA

Area	Month	S1	S2	S3	S4
LSA	JAN	0.16 (0.00; 0.60)	0.16 (0.00; 0.61)	0.16 (0.00; 0.60)	0.16 (0.00; 0.61)
•	FEB	0.14 (0.00; 0.63)	0.14 (0.00; 0.64)	0.14 (0.00; 0.64)	0.14 (0.00; 0.65)
•	MAR	0.07 (0.00; 1.55)	0.07 (0.00; 1.58)	0.07 (0.00; 1.56)	0.07 (0.00; 1.59)
•	APR	0.01 (0.00; 0.37)	0.01 (0.00; 0.38)	0.01 (0.00; 0.38)	0.01 (0.00; 0.38)
•	MAY	0.17 (0.03; 1.38)	0.18 (0.04; 1.46)	0.18 (0.03; 1.44)	0.18 (0.04; 1.48)
•	JUN	0.97 (0.62; 1.53)	1.02 (0.67; 1.62)	1.01 (0.65; 1.60)	1.04 (0.68; 1.64)
•	JUL	1.22 (0.86; 1.58)	1.30 (0.92; 1.67)	1.28 (0.91; 1.65)	1.31 (0.94; 1.69)
•	AUG	1.08 (0.83; 1.35)	1.14 (0.88; 1.42)	1.13 (0.87; 1.40)	1.16 (0.90; 1.44)
•	SEP	1.08 (0.86; 1.34)	1.14 (0.91; 1.41)	1.13 (0.90; 1.39)	1.16 (0.93; 1.43)
•	OCT	0.22 (0.10; 0.38)	0.22 (0.11; 0.39)	0.22 (0.11; 0.39)	0.23 (0.11; 0.39)
•	NOV	1.16 (0.29; 1.95)	1.19 (0.30; 1.98)	1.17 (0.29; 1.97)	1.20 (0.30; 2.00)
•	DEC	0.17 (0.00; 1.04)	0.18 (0.00; 1.05)	0.18 (0.00; 1.04)	0.18 (0.00; 1.06)
•	ALL MONTHS COMBINED	207 (76; 478)	217 (81; 495)	214 (80; 490)	220 (82; 501)

Table A 8 Median (95% CI) Moderate-Severity Behavioural Reponses per Month per Whale by Scenario for the LSA

AREA	MONTH	S1	S2	S3	S4
LSA	JAN	0.04 (0.00; 0.19)	0.05 (0.00; 0.20)	0.05 (0.00; 0.19)	0.05 (0.00; 0.20)
•	FEB	0.04 (0.00; 0.19)	0.04 (0.00; 0.20)	0.04 (0.00; 0.19)	0.04 (0.00; 0.20)
•	MAR	0.02 (0.00; 0.46)	0.02 (0.00; 0.49)	0.02 (0.00; 0.47)	0.02 (0.00; 0.49)
•	APR	0.00 (0.00; 0.23)	0.00 (0.00; 0.23)	0.00 (0.00; 0.23)	0.00 (0.00; 0.23)
•	MAY	0.09 (0.01; 0.70)	0.09 (0.01; 0.72)	0.09 (0.01; 0.72)	0.09 (0.01; 0.73)
•	JUN	0.48 (0.25; 0.77)	0.50 (0.27; 0.80)	0.49 (0.26; 0.79)	0.50 (0.28; 0.81)
•	JUL	0.59 (0.39; 0.80)	0.62 (0.41; 0.83)	0.61 (0.40; 0.82)	0.62 (0.42; 0.84)
•	AUG	0.52 (0.38; 0.69)	0.54 (0.40; 0.72)	0.53 (0.39; 0.71)	0.55 (0.40; 0.72)
•	SEP	0.52 (0.39; 0.69)	0.55 (0.41; 0.72)	0.54 (0.40; 0.71)	0.55 (0.41; 0.73)
•	OCT	0.12 (0.05; 0.24)	0.13 (0.05; 0.24)	0.12 (0.05; 0.24)	0.13 (0.05; 0.24)
•	NOV	0.33 (0.14; 0.58)	0.35 (0.15; 0.61)	0.34 (0.14; 0.58)	0.36 (0.15; 0.61)
•	DEC	0.05 (0.00; 0.30)	0.05 (0.00; 0.32)	0.05 (0.00; 0.31)	0.05 (0.00; 0.32)
•	ALL MONTHS COMBINED	90 (28; 205)	94 (30; 212)	92 (29; 208)	95 (30; 214)

APPENDIX C

Potential for Masking of Southern Resident Killer Whale Calls and Echolocation Clicks due to Underwater Noise

ROBERTS BANK TERMINAL 2

TECHNICAL DATA REPORT

Marine Mammals

Potential for Masking of Southern Resident Killer Whale Calls and Echolocation Clicks due to Underwater Noise

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December 2014

TECHNICAL REPORT/TECHNICAL DATA REPORT DISCLAIMER

The Canadian Environmental Assessment Agency determined the scope of the proposed Roberts Bank Terminal 2 Project (RBT2 or the Project) and the scope of the assessment in the Final Environmental Impact Statement Guidelines (EISG) issued January 7, 2014. The scope of the Project includes the project components and physical activities to be considered in the environmental assessment. The scope of the assessment includes the factors to be considered and the scope of those factors. The Environmental Impact Statement (EIS) has been prepared in accordance with the scope of the Project and the scope of the assessment specified in the EISG. For each component of the natural or human environment considered in the EIS, the geographic scope of the assessment depends on the extent of potential effects.

At the time supporting technical studies were initiated in 2011, with the objective of ensuring adequate information would be available to inform the environmental assessment of the Project, neither the scope of the Project nor the scope of the assessment had been determined.

Therefore, the scope of supporting studies may include physical activities that are not included in the scope of the Project as determined by the Agency. Similarly, the scope of supporting studies may also include spatial areas that are not expected to be affected by the Project.

This out-of-scope information is included in the Technical Report (TR)/Technical Data Report (TDR) for each study, but may not be considered in the assessment of potential effects of the Project unless relevant for understanding the context of those effects or to assessing potential cumulative effects.

EXECUTIVE SUMMARY

The Potential for Masking of Southern Resident Killer Whale Calls and Echolocation Clicks due to Underwater Noise Study was conducted as part of an environmental program for the proposed Roberts Bank Terminal 2 Project (Project or RBT2) to inform a future effects assessment for the Project. The Project, part of Port Metro Vancouver's Container Capacity Improvement Program, is a proposed new three-berth marine container terminal located at Roberts Bank in Delta B.C. The objective of this study is to estimate the distance over which masking of southern resident killer whale (SRKW, *Orcinus orca*) social calls and echolocation clicks may occur. This study uses a simplified and clear model to investigate masking scenarios for SRKW social calls and estimate masking effects.

Underwater noise can affect marine mammals hearing and partially or completely reduce an individual's ability to effectively communicate, detect important predator, prey, and/or conspecific signals, and/or detect important environmental features associated with spatial orientation (Clark et al., 2009 for a review) and is referred to as auditory masking. Five information sources were used to inform the model of masking of SRKW social signals including: 1) SRKW social call from recordings at Lime Kiln State Park, Washington (herein referred to as Lime Kiln) on July 17, 2012 with source levels (SLs) adjusted for underwater noise levels; 2) mean underwater noise SLs from three classes of container ships of different lengths; 3) received underwater noise levels of the container ship *Zim LA* (334 m long) recorded in Haro Strait; 4) SLs of dredging and vibro-densification activities recorded during Deltaport Third Berth construction at Roberts Bank; and 5) underwater noise levels recorded at Lime Kiln in Haro Strait. The ships and dredges included in modeling were selected as examples of potential noise sources during operations and construction of the Project.

The SRKW social call masking model clearly illustrated distances at which masking started to decrease active space (i.e. the estimated maximum range of call detection distance). Under typical noise conditions with container and other commercial vessel traffic recorded at Lime Kiln in Haro Strait, active space of social calls varied from ~1.5 to 2.1 km. When other noise sources were modelled (e.g., vibro-densifier noise sources during construction activities from the derrick barges *Hayward* and *Pennine*), active space of social calls varied from ~2.3 to 9.5 km. The estimated larger active space during construction activities than during noise recorded at Lime Kiln is due to the larger frequency overlap between noise conditions at Lime Kiln and SRKW social calls. The noise recorded at Lime Kiln comes from many sources. The modelling of active space during construction activity only included noise from those specific activities. The results during construction activities are comparable with active space reported by other researchers for short range killer whale social calls (Miller 2006). The container ship *Zim LA* started to decrease active space of social calls at 1.5 km from a killer whale, while for more typical container ships, active space was predicted to be reduced at around 1 km. Construction activities including dredging and vibro-densification piling were not predicted to reduce active space until whales were within 500 m.

SRKW echolocation clicks during the presence and absence of container ships were modelled using underwater recordings of 1) the container ship *Zim LA* up to its closest point of approach at ~800 m, 2) a smaller (290 m) container ship *Hanjin Marseilles* recorded in Haro Strait, and 3) typical underwater noise levels recorded at Lime Kiln over three summer months in 2012. The maximum range of echolocation click detection during the *Zim LA* transit was 330 m while the ship was >2.5 km away, but detection range dropped to as low as 60 m when the ship was 800 m away. In contrast, noise from the *Hanjin Marseilles* resulted in a detection range of 310 m when it was 1 km away from a killer whale and as low as 19 m at 100 m. Under typical noise conditions at Lime Kiln, the estimated maximum range of echolocation click detection varied between 60 and 250 m. The lower predicted echolocation click ranges at Lime Kiln in summer were because of the numerous small boats passing the site that generate more high frequency noise than ships.

The model predicted that most container ships would reduce social call and echolocation click detection distances when they are ~1 km away from a killer whale. Ships with high amplitude noise, such as the *Zim LA*, would start to reduce detection distance of social calls at ~1.5 km and echolocation clicks at ~2.5 km. Reduction of detection distance from ship noise is predicted to be a short duration event, since container ships transiting at approximately 20 knots travel 2.5 km in ~4 minutes, 1.5 km in ~2.5 minutes, and 1 km in ~1.5 minutes.

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GLOSSARY

Active Space	the range up to which signals can be perceived by other members of the species.
Audiograms	a plot of frequency versus detection limit or hearing threshold
Broadband	acoustic term indicating that the amplitude measurements were made over a large frequency range.
Critical ratio	a measure of the difference between signal intensity and noise power spectral density (both at the same frequency) necessary for signal detection.
Critical bandwidth	a measure of the width of the auditory filter within which masking can occur
Masking	a partial or complete reduction of an individual's ability to acoustically detect predator, prey, and/or conspecific signals.
Lombard effect	an increase in SL caused by an increase in RL that was first documented in humans by Etienne Lombard, a French otolaryngologist, in 1911.
Power spectral density	acoustic term indicating the power of a sound in 1 Hz frequency bins. Often reported in units of dB re 1µPa^2/Hz.
Received Level	acoustic term indicating the amplitude of a sound measured or modelled at some distance greater than 1 m from the source of the sound. Often reported in units of dB re 1µPa.
root-mean-square	acoustic term indicating the manner in which the sound amplitude is measured over time. An rms measure allows for average amplitude to be calculated for continuous sounds.
Signal to noise ratio	the ratio of the amplitude of a signal to the amplitude of the background noise. In general, a higher SNR make it easier for the signal to be detected.
Source Level	acoustic term indicating the amplitude of a sound measured or modelled at a standardised 1 m from the source of the sound. Usually reported in units of dB re 1µPa at 1 m.
Stereotyped calls	calls that can be easily classified into a call type used commonly by many whales in the population.

ACRONYMS

AEP	Auditory evoked potential
CB	Critical bandwidth
CR	Critical ratio
KW	Killer whale
RBT2	Roberts Bank Terminal 2
RL	Received level
RMS	Root mean square
SL	Source level
SNR	Signal to noise ratio
SRKW	Southern resident killer whale

1.0 INTRODUCTION

This section provides project background information for the Potential for Masking of Southern Resident Killer Whale Calls and Echolocation Clicks due to Underwater Noise Study.

1.1 PROJECT BACKGROUND

The Roberts Bank Terminal 2 Project (RBT2 or the Project) is a proposed new three-berth marine terminal at Roberts Bank in Delta, B.C. that could provide 2.4 million TEUs (twenty-foot equivalent unit containers) of additional container capacity annually. The Project is part of Port Metro Vancouver's Container Capacity Improvement Program, a long-term strategy to deliver projects to meet anticipated growth and demand for container capacity to 2030.

Port Metro Vancouver, through its consultant Hemmera, has retained SMRU Canada Ltd. to undertake marine mammal studies related to the Project. This technical data report describes a study undertaken to assess potential auditory **masking** of relevant southern resident killer whales (SRKW, *Orcinus orca*) signals (i.e., echolocation clicks and social calls) from potential Project construction, and operational underwater noise.

1.2 MASKING OF SOCIAL CALLS AND ECHOLOCATION CLICKS STUDY OVERVIEW

Port Metro Vancouver is undertaking field and desktop studies to assess potential effects of RBT2-related acoustic disturbance to SRKW. Of the marine mammal species known to inhabit Roberts Bank and the Strait of Georgia, SRKW are of particular cultural (public and Aboriginal groups) and economic (tourism) value. In addition, SRKWs are provincially Red-listed and are listed federally as Endangered under Schedule 1 of the *Species at Risk Act* (*SARA*). SRKW are considered Endangered because of their small population size (i.e., 78 individuals; Center for Whale Research 2015), low reproductive rate, and potential anthropogenic threats (e.g., environmental contaminants, physical and acoustic disturbance, decreased availability and quality of prey) (DFO 2011).

The RBT2 project is located within federally designated SRKW critical habitat; therefore, Project-related underwater noise (e.g., ship approach and berthing and construction activity) that could disrupt natural behaviours and acoustically mask SRKW feeding (i.e., echolocation) and communication (i.e., social calls) are being studied. Masking effects have been identified as a concern and a data gap by the SRKW Technical Advisory Group (Compass 2013).

A review of available information and state of knowledge on masking in odontocetes (i.e., toothed whales, and dolphins) that hear 'mid-frequency' sounds (e.g., killer whales) was completed to identify key approaches to addressing data gaps and areas of uncertainty relevant to the Project. A SRKW social call and echolocation click masking model was developed to inform the assessment of potential Project-related masking effects under differing noise source scenarios.

This technical data report describes the study findings for the key component identified from this gap analysis. Study component, major objective, and a brief overview are provided in **Table 1**.

Table 1 Masking Project - Study Components and Major Objectives

Component	Major Objective	Brief Overview
Masking Model	To assess the potential for signal (i.e., social call and echolocation click) masking from underwater noise produced during Project-related activities and regional commercial vessel traffic	Describe the extent of signal masking from underwater noise from RBT2 activities and commercial vessel traffic near the proposed RBT2 site and regionally.

2.0 REVIEW OF AVAILABLE LITERATURE AND DATA

Odontocetes rely on sound for life processes, including foraging, navigation, mating, and social interactions (Holt 2008). This section reviews pertinent hearing and masking literature from the extensive body of research conducted in the past 48 years.

2.1 HEARING SENSITIVITY

Predicting potential effects of noise on a species requires an understanding of its hearing abilities. To create **audiograms** (i.e., a plot of frequency versus detection limit or hearing threshold), researchers determine an individual animal's hearing sensitivity by measuring its responses to pure tones. In research on captive animals, responses to acoustic stimuli have been documented in two primary ways: 1) from behavioural responses by trained animals, which allow the experimenter to determine which tones were heard; or 2) through neurophysiological responses, called auditory evoked potentials (AEP), measured from small voltages generated by neurons in the auditory system.

Auditory capabilities likely vary between individuals depending on factors such as age, gender, and history of noise exposure. Therefore, behavioural audiograms, which are limited to captive and/or trained animals, provide a limited sample size for modelling purposes. Recent development of non-invasive AEP techniques, which record responses from the skin surface via electrodes, provide a rapid way to test hearing (Nachtigall et al. 2007), allow for increased sample sizes, and broaden our understanding of odontocete hearing sensitivity.

Since the first behavioural audiogram on a bottlenose dolphin (*Tursiops truncatus*) (Johnson 1966), hearing sensitivity has been investigated for a wide range of odontocete species (reviewed in Nachtigall et al. 2007, Mooney et al. 2012), including killer whales (Hall and Johnson 1972, Szymanski et al. 1999). **Annex A: Table A-1** summarises audiogram data to date for odontocete species in the mid-frequency cetacean functional hearing grouping defined by Southall et al. (2007). Killer whales have their best sensitivity range from approximately 12 to 60 kHz and at 20 kHz have the lowest threshold of mid-frequency cetaceans measured to date (**Annex A: Table A-1**).

2.2 CRITICAL BANDWIDTH AND CRITICAL RATIO

Critical bandwidth (CB) and **Critical ratio** (CR) are two important metrics of auditory masking that measure different properties of hearing (Fletcher 1940). The CB measures the width of the auditory filter within which masking can occur. Integrating noise intensity across this CB and comparing it to signal intensity determines if a signal is audible. The CR measures the difference between signal intensity and noise **power spectral density** (both at the same frequency) necessary for signal detection. In other words, CB is a measure of how loud a signal needs to be above the background or ambient noise before it can be detected. Acoustic masking occurs when a signal is received at a level below the CR in relation to the background noise, such that the signal is essentially 'hidden' within the background noise. The cut-off frequencies for CBs are defined as the frequencies where masking effects are 3 dB below the effect at the centre of the CB; however, the actual bands in which the ear integrates noise, and where masking occurs, are much wider. CBs also tend to be asymmetrical. For example, a noise that is one octave lower than a pure tone signal will have a stronger masking effect than the same amount of noise one octave higher than the tone's frequency. For killer whales, Bain and Dahlheim (1994) found that low frequency noise could mask tones two octaves higher. In addition, loud sounds can affect a wider range of frequencies than quieter sounds.

The CR and CB are calculated in different ways, but both estimate how background noise masks a signal. The CR is an indirect estimate of CB because it measures thresholds of tones masked by **broadband** white noise, which is a sound with constant power spectral density (Lemonds et al. 2011). Only one noise bandwidth is required to calculate CR, compared to CB, and data collection is simpler; therefore, CR is the most widely used masking metric for marine mammals (Mooney et al. 2012). Nevertheless, CB remains the more accurate measure of masking of pure tone signals.

The CR has been measured in a few captive odontocetes (Johnson et al. 1989, Erbe 2008), but not killer whales. Bain and Dahlheim (1994) reported unpublished CR data for captive killer whales that ranged from about 20 dB at 10 kHz to 40 dB at 80 kHz. Overall, available studies indicate that odontocete CRs range from 17 to 20 dB below 1 kHz to about 40 dB at approximately 100 kHz (Holt 2008), and suggest that, at higher frequencies, signal level must exceed the background noise level by a greater amount in order to be heard by the receiver.

The CBs estimated from CRs using the equal-power assumption are often well below 1/6th of an octave in the frequency range 1 to 80 kHz, and increase above 1/6th of an octave above 80 kHz (Richardson et al. 1995). Erbe (2008) reported *Fletcher Critical Bandwidths* (calculated from CRs assuming equal power) of either 1/5th or 1/11th of an octave for sounds <2 kHz for a beluga; however, direct measures of CB by Lemonds et al. (2000) and Finneran et al. (2002) found broader bandwidths of ~1/6th of an octave at 20 kHz in bottlenose dolphins and a beluga whale. With some exceptions (e.g., 1/12th octave bands used by Erbe 2002), studies of masking in delphinids (oceanic dolphins) tend to use 1/3rd octave bands; therefore, this study used CBs of 1/3rd octave for estimating call masking.

2.3 MASKING MODELS

Few studies have estimated masking effects on the functional range of odontocete signals in their natural environment, termed the **active space** of signals. Masking effects on active space, and zones of acoustic masking, have been studied for only a few mid-frequency odontocetes species including belugas, short-finned pilot whales (*Globicephala macrorhynchus*), bottlenose dolphins**,** and killer whales (Erbe and Farmer 2000a, b; Jensen et al. 2009; Bain and Dahlheim 1994; Erbe 2002; Au et al. 2004; Miller 2006;

Griffin and Bain 2006; Holt 2008, Crystal et al. 2011) (see **Annex A: Table A-2**). The concept of active space of signals is extremely useful to understand how masking effects can shape the spatial structure of social groups and the vocal behaviour of individuals (Jensen et al. 2012). Masking by noise effectively reduces the active space of a signal, since the receiver would have to be closer to the signaler as ambient noise increases to maintain the same signal-to-noise ratio (SNR) as without the noise. The active space of a signal is largely a function of its **source level** (SL), the levels of background noise, and the transmission loss of the signal between the signaler and the receiver (Miller 2006). Species-specific active space is determined by physical, behavioural, and ecological factors (reviewed in Jensen et al. 2012).

2.3.1 Killer Whale Calls

SRKW produce a rich repertoire of calls which typically range from 1 to 15 kHz and above (Ford 1989, Holt 2008). Calls appear to be used to mediate social interactions and maintain group cohesion (Ford, 1989, Ford 1991). A number of models have contributed to predicting the noise-induced decrease in active space of killer whale calls (Bain and Dahlheim 1994, Miller 2006, Erbe 2002, Veirs and Veirs 2011, Williams et al. 2013). Most of these studies used three inputs to estimate active space: 1) hearing sensitivity; 2) CB; and 3) the source level of calls. Most of these models used audiograms from Szymanski et al. (1999) or some variant of them for hearing sensitivity in killer whales. The choice of CB varied considerably between studies. Erbe (2002) and Veirs and Veirs (2011) used 1/12th octave bands, Miller (2006) used 1/3rd octave bands, and Williams et al. (2013) used two frequency bands from 1.5 to 3.5 kHz and 5 to 12 kHz (see **Annex A: Table A-2** for details). The SLs of calls used by the studies also varied, but none included vocal compensation in SL (see **Section 2.4**). Likewise, different models used different sound sources including different sea states, boat noise, and ship noise. In general, all previous models assumed that masking occurs when the level of the signal drops below the background noise levels or the whales hearing sensitivity in each CB. For this study, the Miller (2006) model was determined to be the clearest, simplest and conservative, and was used with a small modification (see **Methods** section).

In the Miller (2006) model, a call was assumed to be detectable by another killer whale when the **received level** in at least one 1/3rd octave band exceeded the hearing threshold of killer whales or was 6 dB below the background noise level, whichever was greater. To estimate active space, Miller modelled two background noise scenarios of long-range and short-range calls. In sea state zero conditions (no wind, waves or anthropogenic noise), the mean active space of 'long-range' calls was between 10 and 16 km, while the mean active space of 'short- range' calls was between 5 and 9 km. At sea state six (wind speed of 27 to 33 knots, wave height of 4 m), the estimated active space was reduced by 74% and 81%, for long and short-range call types, respectively.

2.3.2 Killer Whale Echolocation Clicks

Echolocation clicks are very short, broadband sounds that are used for navigation and prey location. Killer whale clicks have center frequencies of 50 kHz and a frequency range from 8 to 80 kHz (Au et al. 2004; Holt 2008). SRKW can only detect prey that are close enough for the whales to hear the echoes above ambient noise (Au et al. 2004). The echolocation active space of an animal is smaller than its call active space because the transmission loss for the echoing click at a given distance is twice that of a social call, especially for high frequency echolocation clicks subject to more scattering and absorption losses than lower frequency social calls (Clark et al. 2009). As ambient noise increases, the range of the clicks decreases, as does the ability of the whales to detect the faint echoes, which could result in fewer prey items being detected and consumed.

Some studies have suggested that vessel noise might impair the ability of odontocetes, including killer whales, to forage using echolocation (e.g., Bain and Dahlheim 1994, Bain et al. 2006, Aguilar Soto et al. 2006). However, the effects of noise on the echolocation abilities of odontocetes are difficult to predict with uncertainty regarding frequency overlap between the echolocation click and ambient noise. Since both echolocation clicks and noise are broadband, measuring noise within specific CBs as has been done for calls that are tonal in nature (see **Section 2.2**), may not be appropriate. Earlier studies assumed that all frequencies were equally important for a whale to detect prey. Although our understanding of echolocation processes is improving (Madsen et al. 2005, Nachtigall and Supin 2008, Au et al. 2010, Li et al. 2011), a better understanding of the parts of the echo that are most important for prey detection and differentiation is needed.

Despite these uncertainties, studies have estimated the effects of masking on killer whale echolocation (Bain 2002, Au et al. 2004, Holt 2008, Griffin and Bain 2006, Veirs and Veirs 2011) (see **Annex A: Table A-2** for details). As an example of an echolocation click masking model, Au et al. (2004), predicted levels of northern resident killer whale (NRKW) echolocation click masking by modelling parameters of recorded echolocation clicks, such as center frequency, noise levels, and **root-mean-square** (RMS) bandwidth. The RMS bandwidth is a measure of the frequency width of a spectrum about the centre frequency (in this case, 35 to 50 kHz). Au et al. (2004) used RMS bandwidth to overcome the difficulty of estimating the overlap between broadband click and noise discussed earlier. Other inputs were the modelled target strength (i.e., the amplitude loss in echolocation click from reflecting off a target) of SRKW's primary prey chinook salmon (*Oncorhynchus tshawytscha*), ambient noise levels, and the Szymanski et al. (1999) killer whale hearing thresholds. Au et al. (2004) estimated chinook detection distances of at least 100 m under quiet conditions and up to sea state four (wind speed 16 to 20 knots, wave height 2 m) conditions. However, under moderately heavy rain conditions, and therefore louder background noise levels, the estimate of detection distance dropped to 40 m.

2.4 VOCAL COMPENSATION STRATEGIES

To improve signal detection in noisy environments, marine mammals, including odontocetes, exhibit a variety of noise-induced vocal modifications as compensation strategies, including longer calls, louder calls, increasing call rate, shifting the frequency of the call outside the noise band, and waiting to call until the noise decreases (Tyack 2008a). Masking can be tested directly in captive species (e.g., Erbe 2008), but since this is difficult for wild populations, masking can be modelled or inferred from evidence of masking compensation. **Annex A: Table A-3** summarises evidence of vocal compensation in mid-frequency odontocetes, including killer whales, exposed to masking anthropogenic sounds.

A simple compensation mechanism involves timing the signal production to minimise overlap with the interfering noise, but this has not yet been documented in marine mammals (reviewed in Tyack 2008a, 2008b). Another vocal response to noise, known as the **Lombard effect**, refers to the tendency of a human or animal to raise the SL of their vocalisations in a noisy environment (Brumm and Zollinger 2011). Holt et al. (2009) first described the Lombard effect in SRKW with one call type (S1) and then expanded their analyses to further call types (Holt et al. 2011). The author's S1 data showed that SRKW increased their call SL by 1 dB with every 1 dB increase in background noise levels. Although the Lombard response sustains a favourable SNR and maintains a given active space, there is an upper noise level beyond which a whale is no longer able to compensate. The SRKW Acoustic Detection Study (SMRU 2014a) also documented the Lombard response in SRKW at lower and higher noise levels.

Other vocal compensation strategies documented in odontocetes include shifting signal frequency, and increasing signal repetition rate (redundancy) and duration (Lesage et al. 1999, Buckstaff 2004). When call duration is compared over decades, increased duration of SRKW pulsed calls has been documented, which may be to overcome the masking effects of assumed increasing vessel noise (Foote et al. 2004, Wieland et al. 2010). Holt et al. (2009) and the SRKW Acoustic Detection Study (SMRU 2014a) measured call duration and background ambient noise measurements, and found no significant change in the call duration in relation to background noise.

In addition to modifying the parameters of communication calls to overcome noise interference, belugas, bottlenose dolphins, and false killer whales (*Pseudorca crassidens*) appear to be able to adapt their echolocation strategy to increased noise by modifying the amplitude, spectral, and temporal parameters of echolocation clicks (Hotchkin and Parks 2013, Au et al. 1982, Au et al. 1985). These strategies have yet to be documented in killer whales, therefore only the Lombard effect was included in the masking model in this study.

2.4.1 Energetic Costs of Vocal Compensation

Several studies have reported varying energetic costs of sound production (e.g., a 0.5% increase in metabolic rates of bottlenose dolphins; Jensen et al. 2012), whereas at least one study suggested that some dolphins use passive listening to avoid the energetic costs of echolocation (e.g., Gannon et al. 2005). To resolve these contrasting points of view, Noren et al. (2013) measured oxygen consumption in two captive bottlenose dolphins during rest, while vocalising at low to moderate levels, and during a recovery period. Percentage oxygen data showed that dolphins incur a measurable, but variable and relatively small, metabolic cost during production of two types of sounds. The mean metabolic rates measured during vocal periods were 1.2 times the resting values, although results varied widely by individual and trial. Interestingly, there was a positive linear relationship between mean vocalisation duration and the metabolic cost of the vocal period, but not between vocalisation rate and metabolic cost. This data suggested that longer sounds (e.g., whistles) may be more costly because a dolphin must sustain higher air pressure levels in the nasal cavity, which requires more muscular energy (Noren et al. 2013). Although a small metabolic cost for vocal compensation is possible in SRKW, these costs were not included in masking models in this study.

3.0 METHODS

Descriptions of the spatial and temporal scopes, and study methods of the Potential for Masking of SRKW Calls and Echolocation Clicks from Construction and Operation Noise Study are provided below.

3.1 STUDY AREA

Underwater noise from Project-related construction and terminal activity and regional commercial traffic vessel was modelled at Roberts Bank (JASCO 2014b) and regionally in the Salish Sea (JASCO 2014a). The ships and dredges included in modeling were selected as examples of potential noise sources during operations and construction of the Project.

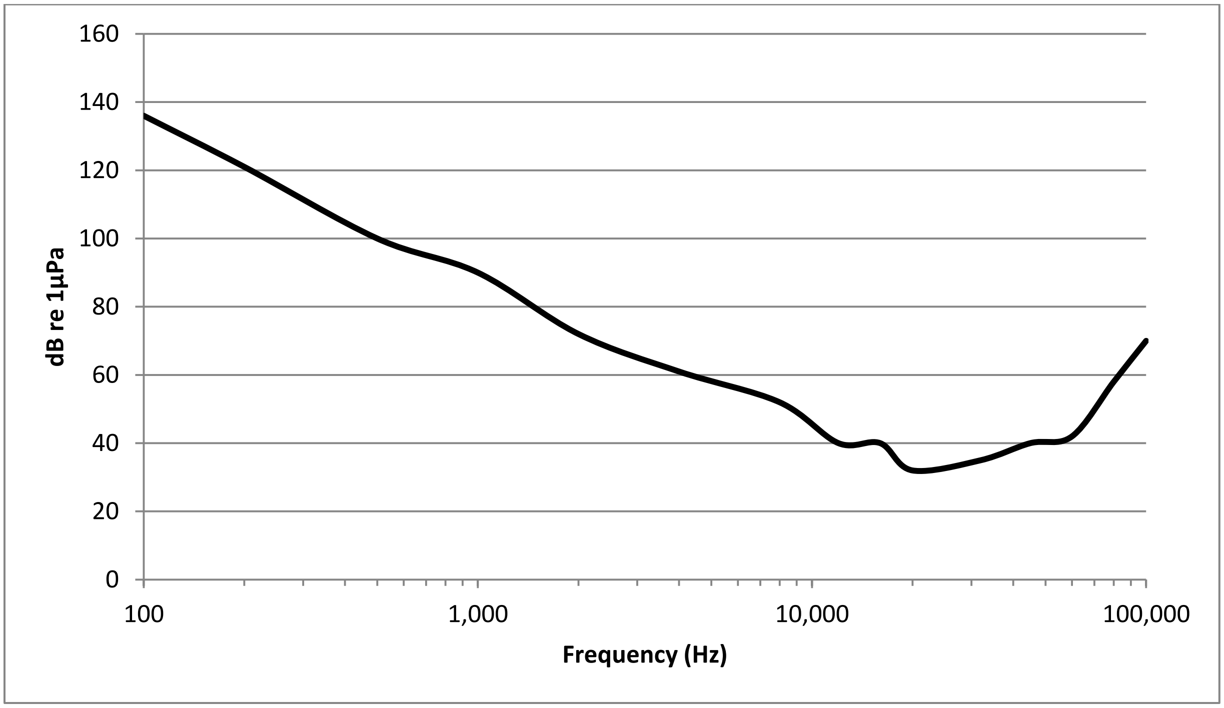
3.2 TEMPORAL SCOPE

This study modelled potential SRKW call and echolocation click masking using the most current and locally pertinent data. Model inputs included ambient noise, regional commercial traffic noise, Project-related operational (e.g., approach and berthing) and construction noise (e.g., vibratory piling, vibro-densification, dredging), and SRKW calls and clicks. Underwater noise data was collected at Lime Kiln and Roberts Bank during the summer of 2012 (see SMRU et al. 2014). Received levels of a large (334 m long) container ship *Zim LA* were recorded in Haro Strait on June 8, 2013 and other ship noise was recorded from August 3, 2011 to August 10, 2013 at Lime Kiln. Further ship noise data for Haro Strait were extracted from data collected in 2006 (Hildebrand et al. 2006). Dredge and vibro-densifier recordings were made on April 5, 2007 and August 18, 2007, respectively, during Deltaport Third Berth construction activities (Zykov et al. 2007, Warner and Zottenberg 2008). The SRKW S1 call used in modelling was recorded on July 17, 2012 at Lime Kiln.

3.3 STUDY METHODS

In order to use the most sensitive audiogram levels (i.e., most precautionary) and extend measurements down to low frequencies produced by vessels, this study used a composite audiogram (**Figure 1**) for killer whales that incorporated the most sensitive levels reported by Szymanski et al. (1999) and Hall and Johnson (1972). In addition, extrapolations by Erbe (2002) were used for levels <500 Hz (Erbe used an average from studies of beluga whales (*Delphinapterus leucas*), bottlenose dolphins (*Tursiops truncatus*) and Pacific white-sided dolphins (*Lagenorhynchus obliquidens*), and the Szymanski et al. (1999) level at 1 kHz was decreased from 105 to 90 dB re 1µPa to smooth the curve and deal with assumed effects from pump noise in captivity.

Figure 1 Composite Audiogram for Killer Whales Used in this Study



No new data were required for this study since the modelling utilised previously collected data from earlier RBT2 studies or Deltaport Third Berth construction monitoring. Study data sources are listed in **Table 2** and details of data collection methodology can be found in those studies.

Table 2 Sources of Data for Masking Model Inputs

Model Input	Recording Equipment	Sampling Rate (kHz)	Study
Haro Strait noise	Reson TC4032/MOTU Traveler	192	SMRU et al. 2014
Roberts Bank noise	AMAR	96	SMRU et al. 2014
Zim LA received levels	AMAR	128	Hemmera et al. 2014
Hanjin Marseilles	Reson 4033/Avisoft Recorder	500	Hildebrand et al. 2006
Ship noise	Reson TC4032/MOTU Traveler	192	Hemmera et al. 2014
Dredge	Reson TC4032/Sound Devices 722	96	Zykov et al. 2007
Vibro-densification	Reson TC4032/Sound Devices 722	96	Warner and Zottenberg 2008

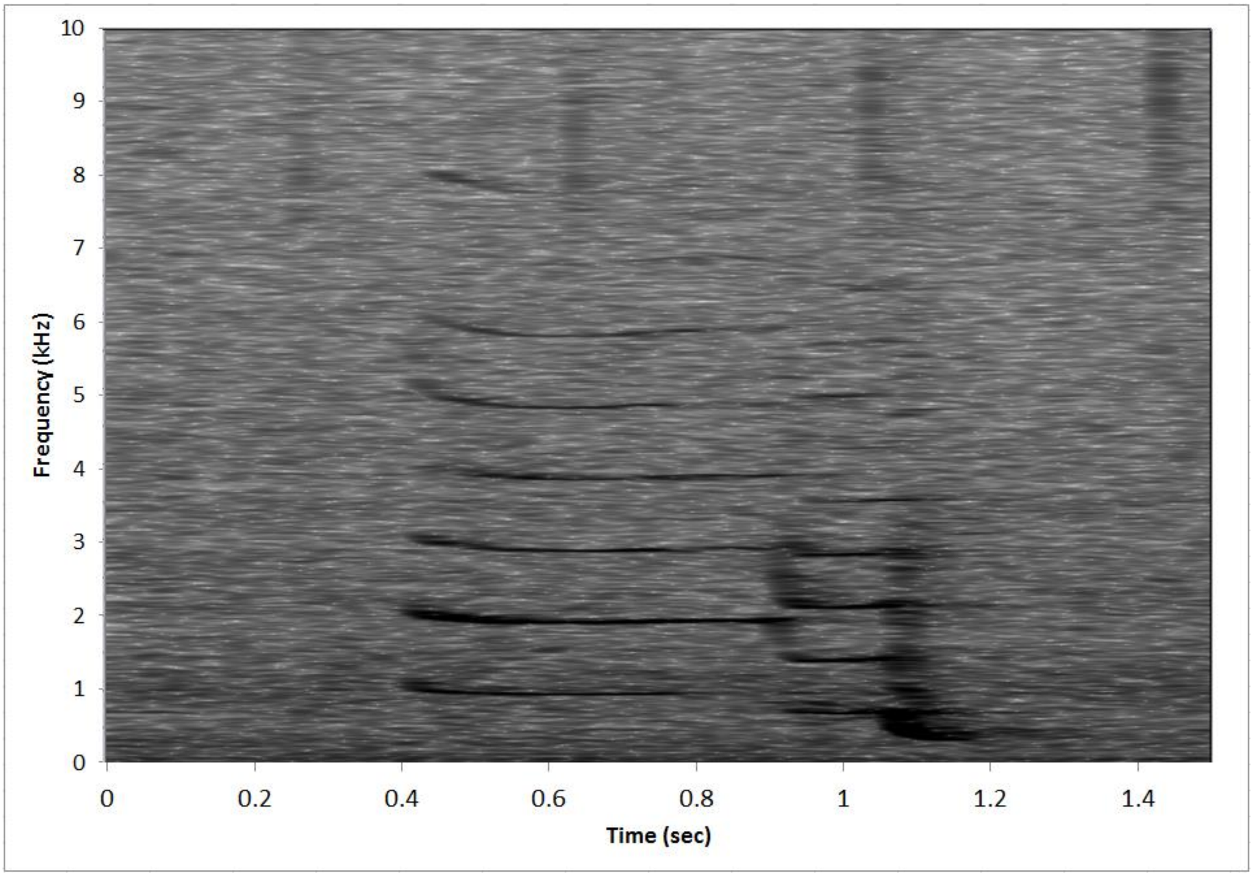
3.4 DATA ANALYSIS

Given the various noise sources and signals, all of which are varying in time and space, and how these noises are perceived by the individual animal, modelling effects of masking in the wild is challenging. Therefore, this study used simple models developed by Miller (2006) for calls and Au et al. (2004) for echolocation clicks (see **Section 2**). Although these models only provide an initial estimate of masking ranges, the models generate measures of the likely magnitude of change in masking range due to different masking noise and estimates of the distance at which different noise sources are likely to start masking SRKW calls and clicks.

3.4.1 Modelling Call Masking

Following the model methodology developed by Miller (2006), 1/3 octave band levels were used for all input data. SRKW produce a number of different **stereotyped calls**; however, only the S1 stereotypical call, commonly produced by J pod, was used in modelling (**Figure 2**). For the current study, the Miller (2006) model was updated with the inclusion of the Lombard effect since the effect had been documented by Holt et al. (2009, 2011) and (SMRU 2014b: Appendix C). Therefore, the SL of each modelled call was defined based on the underwater noise level the hypothetical animal would have been exposed to. Holt et al. (2009) reported S1 call SLs 47.5 dB higher than background noise levels, therefore this **signal to noise ratio** was maintained up to a maximum SL of 174 dB re 1μPa, the maximum SL reported for SRKW by Holt et al. (2009).

Figure 2 Spectrogram of the S1 Call Recorded at Lime Kiln and Used in the Masking Model (Sample rate: 192 kHz. FFT: 4096. Hanning window)



3.4.2 Modelling Echolocation Masking

Given the uncertainties in predicting echolocation click masking, the Au et al. (2004) model was chosen for use in this study due to its simplicity and clarity. Au et al. (2004) found that killer whales decreased the SL of their echolocation as they moved closer to a target. Au et al. (2004) modelled the target strength of a chinook salmon and derived the following equation to estimate the echo level of a 50 kHz echolocation click:

$$EL = 128.633 - 20\log R - 2 \propto R - 0.410\,\theta + 0.006\,\theta^2$$

Where:

- α is the absorptions loss of a 50 kHz signal (~0.016 dB/m);
- *R* is the range in meters; and
- θ is angle between the killer whale and the chinook salmon (in this case 60º was used to approximate a killer whale swimming just below the surface echolocating on a chinook salmon at deeper depth).

In addition, the noise level experienced by a killer whale was estimated by Au et al. (2004) using the following equation:

$$NL = N_0 + BW - DI$$

Where:

- N_0 is the noise spectral density at 50 kHz;
- *BW* is the received bandwidth, estimated at 46 dB; and
- *DI* is the directivity index, estimated to be 21 dB.

From these two equations, it was possible to determine at what ranges the echo level remained above the noise level under various noise conditions at Lime Kiln and as container ships approached.

4.0 RESULTS

This section presents the main findings of the modelling conducted in the current study.

4.1 CALL MASKING RESULTS

The underwater noise conditions measured at Lime Kiln are representative of noise conditions in Haro Strait during the summer and include noise from a number of different sources. Louder noise periods are driven largely by anthropogenic noise sources (boats and ships) while quieter periods might include wind, tidal currents or rain noise (SMRU et al. 2014). Noise levels at Lime Kiln never exceeded 126.5 dB re 1µPa, therefore the SL of calls were never limited (i.e., they never reached the maximum of 174 dB re 1µPa. See **Section 3.4.1**) and there was no decrease in call detection range. Given the nature of the 1/3 octave noise levels at Lime Kiln, calls were modelled to be audible at larger ranges during louder periods (**Table 3**) since most of the noise level increase was at frequencies <4 kHz. Under all these noise conditions, the S1 call harmonic at 4 kHz was the last harmonic detectable.

Table 3 Estimated Maximum Detection Distance of SRKW Calls Under Various Noise Conditions Measured at Lime Kiln

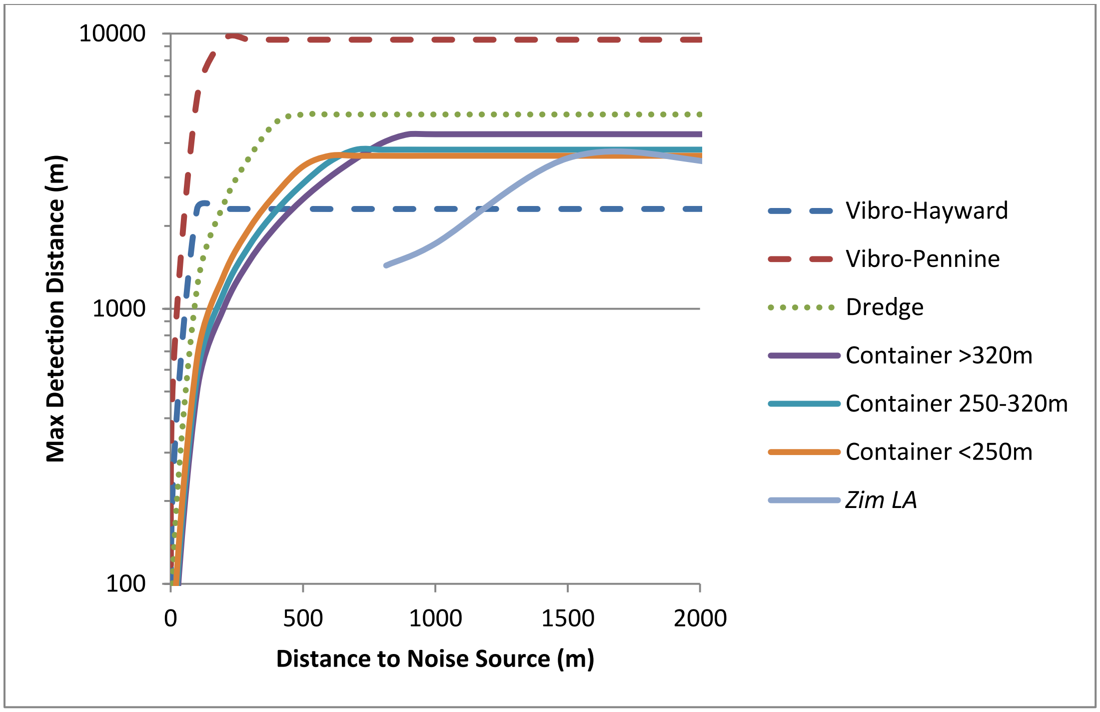
Noise Percentile (%)	Max Range (m)
95	1,497
75	1,686
50	1,613
25	1,743
5	2,108

Note: Percentiles indicate amount of data that exceeds that level (i.e., 95% of noise data was higher amplitude than this level).

Modelling of construction noise sources (i.e., the derrick barges *Hayward* and *Pennine* recorded during Deltaport Third Berth construction) resulted in very different maximum detection ranges of 9,500 and 2,300 m, respectively (**Figure 3**), due to a combination of the difference in the slope of their 1/3 octave spectrum levels and overall SLs (**Figure 4**). Call detection ranges for both barges only decreased substantially when the hypothetical whale was within ~200 m. Dredge noise modelling suggested that call detection ranges started to drop when the whale was within 500 m of the operating dredge.

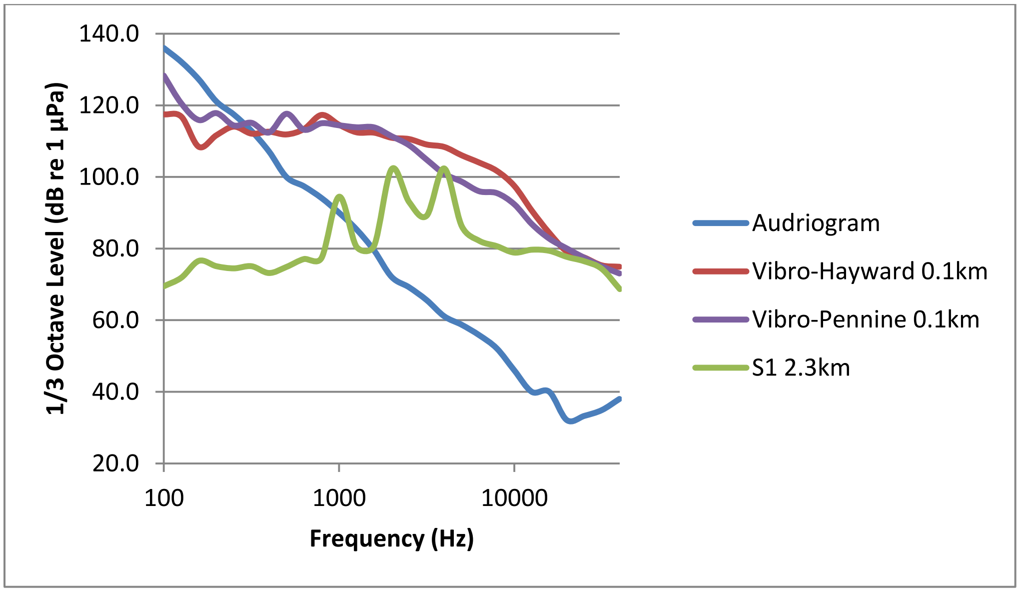
Operational noise modelling (i.e., ships transiting) resulted in detection distances of calls around container ships that had between 3,600 and 4,300 m of detection distance, which dropped when most container ships were within ~1 km. For the *Zim LA*, this drop started at ~1.5 km. For most container ships, the detection distance of a call from an animal 200 m from the ship was still ~1.2 km. At the closest point of approach of the *Zim LA* (800 m), maximum detection distance was modelled to be ~1.4 km.

Figure 3 **Plot of Distance to Various Noise Sources and Maximum Detection Distance of Calls**



Note: The vibro-densifiers and dredging are considered construction noise. The container ships (including *Zim LA*) are transiting vessels and considered operation noise.

Figure 4 **Plot of SRKW Audiogram, Received Level at the Calling Whale from the Vibro-densifiers on the Derrick Barges *Hayward* and *Pennine* at 0.1 km, and Received Level of the Listening Whale at 2.3 km from the Calling Whale**



4.2 ECHOLOCATION MASKING RESULTS

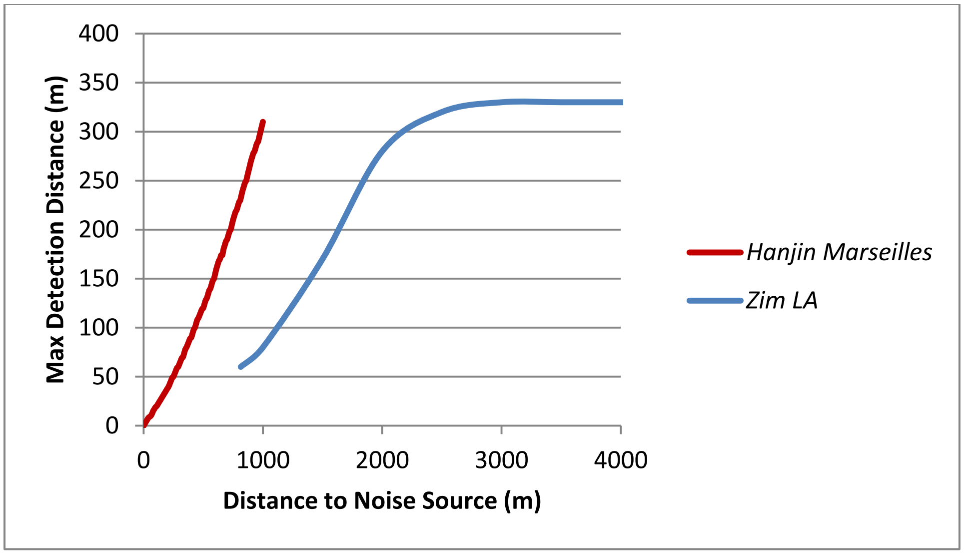
Maximum echolocation click detection distance between a whale and a chinook salmon under typical noise conditions at Lime Kiln was estimated to be between 60 and 250 m (**Table 4**). Again, these noise conditions include noise from a number of different sources, with the loudest periods driven by boat and ship noise. Under noise conditions recorded in Haro Strait with the approach of the *Zim LA* (i.e., operational noise), maximum echolocation detection distance was estimated to remain at ~330 m until the ship was 2,500 m away, and then decreased to 60 m at the closest point of approach (~800 m; **Figure 5**). It was not possible to model detection ranges for other ships in Haro Strait or vibro-densifiers and dredges (i.e., construction noise) at Roberts Bank, as data were not available for those noise sources at 50 kHz. Hildebrand et al. (2006) estimated the SL for a 290 m long container ship, the *Hanjin Marseilles*, in Haro Strait. Using their results at 50 kHz, received levels were calculated at ranges up to 1 km and echolocation masking was modelled. At 1 km, the echolocation click detection distance was estimated to be 310 m (**Figure 5**). Beyond this distance, echolocation click detection range is not likely to increase further given background noise levels at 50 kHz.

Table 4 Estimated Maximum Detection Distance of Echolocation Clicks Under Various Underwater Noise Conditions at Lime Kiln

Noise Percentile (%)	Max Range (m)
95	250
75	230
50	210
25	180
5	60

Note: Percentiles indicate amount of data that exceeds that level
(i.e. 95% of noise data was higher amplitude than this level).

Figure 5 Plot of Range to the *Zim LA* and *Hanjin Marseilles* and Maximum Detection Distance of Echolocation Clicks



5.0 DISCUSSION

A discussion of the major results arising from the Potential for Masking of SRKW Calls and Echolocation Clicks from Underwater Noise Study and data gaps are provided below.

5.1 DISCUSSION OF KEY FINDINGS

This study updated masking models of killer whale calls by including vocal compensation (i.e., the Lombard effect) given evidence that SRKW utilise the Lombard effect to compensate for changes in background noise levels. Typical estimated maximum detection distances for calls in this study varied from 1.5 km to almost 10 km, similar to estimates of short range calls provided by Miller (2006). Differences in noise levels and frequency drove the large range in estimated maximum detection distances. If the Lombard effect (i.e., an increase in call SL as noise RL increases) had not been included in the call masking model, detection distances would have been smaller during louder noise periods.

The call masking model suggests that call detection range does not start to decrease until a whale is <500 m from dredging or vibro-densification (construction) activities and that at 200 m detection distance of a call would still be at least 2 km. For most container ships (operational activities), call detection distance was not expected to decrease until the whale was within 1 km of the ship. For the container ship *Zim LA*, which generated a high level of underwater noise, call detection distance decreased when the whale was within 1.5 km. Masking from container ships may occur at larger distances than construction activities (e.g., vibro-densification and dredging), but will occur over short periods of time (e.g., a typical container ship in Haro Strait travels at 20 knots, covering 1.5 km in ~2.5 minutes. Ships that travel at slower speeds create less noise (Hemmera et al. 2014), and will have a reduced range at which masking occurs.

To model echolocation click masking distance, this study followed the methods of Au et al. (2004). Au et al. (2004) did not report maximum echolocation detection range due to uncertainty over the distances at which killer whales might use echolocation to locate prey. At sea state four, the authors found that an echolocation click would be detectable by a whale at 100 m and that this distance would be reduced to 40 m under moderately heavy rain conditions. This current study also could not determine the maximum distance that a killer whale would echolocate for prey, but did conclude that the maximum masking distance under typical (5th through 95th percentile) noise conditions at Lime Kiln ranged from 60 to 250 m. Typical noise conditions at Lime Kiln included both natural and anthropogenic noise sources.

Modelling dredging and vibro-densification (construction activities) masking of echolocation clicks was not possible because the data did not include recordings at a high enough frequency (i.e., 50 kHz). Modelling of echolocation click masking of the container ship *Zim LA* (operational activities) suggested that noise from this ship starts to reduce echolocation detection distance at ~2.5 km, which is 1 km further than when it starts to reduce call masking detection distance. For more typical container ships, such as the *Hanjin Marseille*, ship noise is likely to start decreasing echolocation click detection at ~1 km.

Masking of echolocation clicks could potentially result in increased expended energy or decreased caloric intake by SRKW individuals in a population that may be limited by prey availability. Potential effects of echolocation click masking are estimated in the SRKW Noise Exposure Study (SMRU 2014c) and SRKW Population Consequence of Disturbance Study (SMRU 2014d).

5.2 DATA GAPS AND LIMITATIONS

This study used simple models to generate initial relative estimates of detection distance under various underwater noise scenarios and provides a basis for assessing Project-related effects and cumulative effects from commercial vessel traffic. Given data gaps in the literature and current knowledge, the model could not include all the complexities of moving whales and noise sources in a dynamic marine environment. The simple masking model benefits from being easy to understand and provides a standardised input into the SRKW Noise Exposure Study (SMRU 2014c). In SMRU 2014c, the masking estimates are kept constant across scenarios, with only the noise scenarios being changed. The goal of the SRKW Noise Exposure Study (SMRU 2014c) and SRKW Population Consequence of Disturbance Study (SMRU 2014d) is to estimate the relative effect of different scenarios, thus the absolute value of masking estimates is not crucial. The relative estimates of masking from this project are therefore useful in assessing Project-related and cumulative effects of underwater noise on SRKW.

6.0 CLOSURE

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8.0 STATEMENT OF LIMITATIONS

This report was prepared by SMRU Canada Ltd., based on desktop studies conducted by SMRU, for the sole benefit and exclusive use of Hemmera and Port Metro Vancouver. The material in it reflects SMRU's best judgment in light of the information available to it at the time of preparing this Report. Any use that a third party makes of this Report, or any reliance on or decision made based on it, is the responsibility of such third parties. SMRU accepts no responsibility for damages, if any, suffered by any third party as a result of decisions made or actions taken based on this Report.

SMRU has performed the work as described above and made the findings and conclusions set out in this Report in a manner consistent with the level of care and skill normally exercised by members of the environmental science profession practicing under similar conditions at the time the work was performed.

This Report represents a reasonable review of the information available to SMRU within the established Scope, work schedule and budgetary constraints. The conclusions and recommendations contained in this Report are based upon applicable legislation existing at the time the Report was drafted. Any changes in the legislation may alter the conclusions and/or recommendations contained in the Report. Regulatory implications discussed in this Report were based on the applicable legislation existing at the time this Report was written.

In preparing this Report, SMRU has relied in good faith on information provided by others as noted in this Report, and has assumed that the information provided by those individuals is both factual and accurate. SMRU accepts no responsibility for any deficiency, misstatement or inaccuracy in this Report resulting from the information provided by those individuals.

ANNEX A

Tables

Table A-1 Audiogram Data for Mid-frequency Odontocetes (grouping as defined in Southall et al. 2007), Revised and Updated from Mooney et al. (2012)

Note: Since the definition of "best sensitivity range" (i.e., frequency range of best hearing) can vary between studies, it is defined here as the region within 20 dB of the lowest hearing threshold (after Mooney et al. 2012).

Species	*n*	Hearing Range (kHz)	Best Sensitivity Range (kHz)	Maximum Sensitivity (lowest threshold)	Method	References	Comments
Bottlenose dolphin, *Tursiops truncatus*	1	0.75 - 150	7 - 130	65 (40.8 dB)	Behaviour	Johnson (1966,1967)	
	42	10 - 150	10 - 80	Variable	AEP	Houser and Finneran (2006)	Varied depending on sex and age – Hearing loss between 20 and 30 yrs. Best sensitivity range is a mean for all ages.
	1	10 - 150	10 - 130	50 (60 dB)	Behaviour AEP	Schlundt et al. (2007)	Compared relative hearing thresholds obtained with AEP and behavioural techniques. Differences generally within ±5 dB. Data presented here are averages of both measurements. Sharp upper frequency cut-off beyond 130 kHz.
	1	40 - 140	40 - 120		Behaviour	Lemonds et al. (2011)	Frequencies lower than 40 kHz were not tested. Pure tone sensitivity nearly flat from 40 to 120 (averaging 45 dB).
Pacific bottlenose dolphin, *Tursiops truncatus gilli*	1	2 - 135	25 - 110	25 (47 dB) 50 (46 dB)	Behaviour	Ljungblad et al. (1982)	This study reported two frequencies with maximum sensitivity.
	13	10 - 150	20 - 130	40 (60.7 dB ± 4.8)	AEP	Houser et al. (2008)	Mean AEP thresholds were 15 to 20 dB higher than behavioural thresholds for frequencies below 80 kHz. The two oldest animals (17 and 18 yrs) showed reduced hearing.
Killer whale, *Orcinus orca*	1	0.5 - 31	5 - 30	15 (35 dB)	Behaviour	Hall and Johnson (1972)	Authors caution that thresholds below 10 kHz may have been noise limited.
	2	4 - 100	12 - 60	20 (34 dB)	Behaviour	Szymanski et al. (1999)	
	2	1 - 100	16 - 45	20 (37 dB)	AEP	Szymanski et al. (1999)	Same animal tested as preceding study.
Beluga, *Delphinapterus leucas*	2	1 - 130	15 - 110		Behaviour	White et al. (1978)	
	3	0.125 - 8	4 - 8		Behaviour	Awbrey et al. (1988)	Low frequency hearing data. Did not establish upper limit.
	1	8 - 128	32 - 108	54 (54.6 dB)	AEP	Klishin et al. (2000)	

Species	*n*	Hearing Range (kHz)	Best Sensitivity Range (kHz)	Maximum Sensitivity (lowest threshold)	Method	References	Comments
	2	2 - 130	14 - 90	50 (43 dB)	Behaviour	Finneran et al. (2005)	One subject exhibited hearing loss above 37 kHz.
	1	8 - 128	22 - 90	50 (43.9 dB)	AEP	Mooney et al. (2008)	
False killer whale, *Pseudorca crassidens*	1	2 - 115	16 - 64	64 (40 dB)	Behaviour	Thomas et al. (1988)	
	1	4 - 45	7 - 27 (*16 - 24)	20 (69 dB)	Behaviour	Yuen et al. (2005)	The animal believed to have suffered hearing loss. * Authors define range of best sensitivity as 10 dB from lowest threshold.
	1	4 - 45	6.7 - 27 (*16 - 22.5)	22.5 (80.9 dB)	AEP	Yuen et al. (2005)	Same animal tested as preceding study, believed to have suffered hearing loss. * Authors define range of best sensitivity as 10 dB from lowest threshold.
Risso's dolphin, *Grampus griseus*	1	1.6 - 110	4 - 80	8 and 16 (63.7, 63.8 dB)	Behaviour	Nachtigall et al. (1995)	
	1	4 - 150	8 - 108	32, 64 and 90 kHz (≤ 50 dB)	AEP	Nachtigall et al. (2005)	The animal was an infant. The best sensitivity measured was 20 dB lower than reported by Nachtigall et al. (1995).
Tucuxi, *Sotalia fluviatilis guianensis*	1	4 - 135	16 - 105 (*64 - 105)	85 (50 dB)	Behaviour	Sauerland and Dehnhardt (1998)	* Authors define range of best sensitivity as 10 dB from maximum sensitivity.
Pacific white-sided dolphin *Lagenorhynchus obliquidens*	1	100 Hz - 140 kHz	4 - 128	64 (64 dB)	Behaviour	Tremel et al. (1998)	Below 1 kHz, hearing sensitivity dropped at a rate of ≈ 43 dB per kHz. Authors note that thresholds at the mid-frequencies could have been masked by ambient pool noise.
Striped dolphin, *Stenella coeruleoalba*	1	0.5 - 160	32 - 120 (*29 - 123)	64 (42 dB)	Behaviour	Kastelein et al. (2003)	* Authors define range of best sensitivity as 10 dB from maximum sensitivity.
Gervais' beaked whale, *Mesoplodon europaeus*	1	5 - 80	40 - 80	80 (85 dB)	AEP	Cook et al. (2006)	Subject was a stranded juvenile. Authors were unable to test frequencies higher than 80 kHz due to equipment sampling rate limitations. They reported only the lowest SPLs for which an AEP was detected at each frequency.
	1	20 - 90	20 - 80	40 (90 dB)	AEP	Finneran et al. (2009)	Subject was a stranded animal. The observed range of hearing encompasses the reported

Species	*n*	Hearing Range (kHz)	Best Sensitivity Range (kHz)	Maximum Sensitivity (lowest threshold)	Method	References	Comments
							frequency range of echolocation clicks for beaked whales (Johnson et al. 2006)
Blainville's beaked whale, *Mesoplodon densirostris*	1	5.6 - 160	40 - 50	50 (48.9 dB)	AEP	Pacini et al. (2011)	Stranded animal. Best sensitivity range partially overlaps with the frequency-modulated upsweep used during echolocation.
White beaked dolphin, *Lagenorhynchus albirostris*	2	16 - 181	32 - 128	45 (45.3 dB)	AEP	Nachtigall et al. (2008)	Wild caught and temporarily restrained animals in foam-lined plastic chamber with no background noise. No measurements made for frequencies <16 kHz.
Long-finned pilot whale, *Globicephala melas*	1	4 - 100	11.2 - 50	40 (53.1 dB)	AEP	Pacini et al. (2010)	Subject was a rehabilitated juvenile.
Short-finned pilot whale, *Globicephala macrorhync-hus*	2	10 - 100	40 - 56	40 (78 dB)	AEP	Schlundt et al. (2011)	AEP reported here for one of the two individuals, a healthy captive adult female. The second whale was a stranded juvenile with severe hearing loss.
Rough- toothed dolphin, *Steno bredanensis*	14*	10 - 120	Unclear	40 (≈ 65 dB)	AEP	Mann et al. (2010)	Subjects were stranded. Four out of 14 showed hearing loss. *Results reported for only one normal hearing subject.
Pygmy killer whale, *Feresa attenuata*	2	5 - 120	20 - 60	40 (≈ 55 dB)	AEP	Montie et al. (2011)	Rehabilitated stranded animals. Amikacin sulfate treatment may have caused low frequency hearing loss in one of the whales. The authors note that the audiogram of the pygmy killer whale was most similar to the audiogram of the killer whale by Szymanski et al. (1999).
Indo-Pacific humpback dolphin, *Sousa chinensis*	1	11.2 - 128	20 - 120	45 kHz (47 dB)	AEP	Li et al. (2012)	The subject was 13 years old.

Table A-2 Studies of Zone of Masking and Effects on the Active Space of Signals for Mid-frequency Odontocetes (grouping as defined in Southall et al. 2007)

Species	Field, Lab or Model	Masking Noise	Masking Noise Characteristics	Signal Characteristics	Results Relevant to Masking	Authors
Killer whale, *Orcinus orca*	Lab and Model	Simulated vessel noise and white noise	White noise: 500 Hz to 5 kHz band Vessel noise ≤ 20dB	**Lab:** Pure tone, KW call N32, and echolocation click train. **Propagation model:** Assumed SLs of KW calls 180 dB re 1µPa @ 1m for high-freq. components and 150 dB for low-frequency components.	**Masking of pure tones:** observed at frequencies ≤ 20 kHz. **Masking of whale sounds by vessel noise:** Little masking effect if low levels of noise and KW vocalisation contain significant high frequency energy. **Sound propagation model:** At 10 km only HFC audible. 20dB increase in noise would reduce detection ranges to 5 km. **Directional effects of masking:** masking sources needed to be 4 to 40 dB more intense to mask natural calls when located to the side or behind an animal.	Bain and Dahlheim (1994)
	Model	Whale watching vessels	Frequency structure and vessel source levels estimated from Richardson et al. (1995) and Erbe (2001). Received levels from one vessel 100 m to the side of the whale: 105 to 110 dB re 1 µPa – Power spectral densities: 70 to 80 dB re 1 µPa2 / Hz at 20 kHz.	Echolocation click (from Miller 2000).	Detection range and detection efficiency impaired. Small increases in detection thresholds (3 dB) resulted in large increases on the proportion of detectable prey items. A 60 dB increase in noise relative to low ambient noise levels corresponded to ≈ 30-fold decrease in detection range. The consequences of active space reduction depended on foraging tactic.	Bain (2002)
	Model	Vessel noise	**1/12th octave band analysis**. Center frequencies: 100 Hz to 20.3 kHz. SLs ranged from 145 to 169 dB re 1µPa @ 1m, increasing with speed.	One KW pulsed call S1 – RL between 105 and 124 dB re 1 µPa.	Predicted masking range: 14 km for boats operating at 51 km/hour, 1 km for boats cruising at low speeds of 10 km/hour.	Erbe (2002)

Species	Field, Lab or Model	Masking Noise	Masking Noise Characteristics	Signal Characteristics	Results Relevant to Masking	Authors
	Model	Ambient noise	Used RMS bandwidth instead of CB. Sea state 4: RMS noise spectral level at 50 kHz = 35 dB re 1 μPa^2/Hz. Rain fall of 3 mm/hr: RMS noise spectral level at 50 kHz = 42 dB .	Echolocation clicks. Center frequencies: 45 to 80 kHz, bandwidths: 35 to 50 kHz, SLs: 195 t0 224 dB re 1 µPa. Bimodal spectra with peak in center frequency at 50 kHz. Echo levels reflecting off a salmon: 78 to 104 dB re 1 µPa, depending on salmon depth and horizontal range.	Chinook detection distances: ≥ 100 m under quiet conditions up to sea state 4, 40 m under louder background noise levels of heavy rain.	Au et al. (2004)
Killer whale, *Orcinus orca*	Model	Ambient noise levels in sea states zero and six.	**1/3rd octave band analysis.** **Sea state zero:** 44 dB re 1µPa^2/Hz at 1 kHz to 20 dB at 20 kHz. **Sea state six:** NL 26 dB higher.	759 stereotyped calls, 60 variable calls, 24 whistles. Estimated SLs in the 1 to 20 kHz band (re 1 µPa @ 1 m): whistles: 140.2 dB variable: 146.6 dB stereotyped: 152 dB	**In sea state zero:** Active space of 10 to 16 km for long range calls with high frequency component; 5 to 9 km for short range sounds. whistles: 6.4 ± 2.4 km Variable calls: 7.8 ± 3.7 km **Active space reduction in sea state six:** Stereotyped calls: 74% to 81% variable calls: 83% whistles: 93%	Miller (2006)
	Model	Acoustic environ-ment of SRKW summer habitat during whale watching.	RLs (dB RMS // 1µPa) = 106 min, 128 median, 146 max.	Echolocation clicks (parameters as in Au et al. 2004).	Average active space reduction due to increased ambient noise levels: 64 to 90% - Average annual decrease in foraging space with increased noise levels: 15 to 20%. Change in foraging efficiency depends on foraging tactics.	Griffin and Bain (2006)
	Model	Vessel noise	SLs at 50 kHz dB re 1 µPa^2/Hz : Cruising speed: 93 - 111 db Power speed: 92 - 107 dB	Echolocation clicks – Analysis only considers one frequency of 50 kHz.	Predicted maximum horizontal detection ranges for a KW at the surface echolocating on chinook salmon 65 m deep in Haro Strait ambient noise was 400 m. Reductions in echolocation range for cruising and power up speeds up to 400 m to the whales ranged from 38 to 100%	Holt (2008) Source levels from Hildebrand et al. (2006).

Species	Field, Lab or Model	Masking Noise	Masking Noise Characteristics	Signal Characteristics	Results Relevant to Masking	Authors
	Model	Commer-cial ships	SLs = 133 to165 dB re 1µPa @	KW discrete calls.	20% of all shipping activity produced noise loud enough to mask killer whale vocalisations; a speed limit of 20 knots did not create any reduction in masking, a 15 knots speed limit reduced the occurrence of masking by 30%, and a 10 knots limit reduced masking by 100%.	Crystal et al. (2011)
	Model	Commer-cial ships	**1/12th octave band analysis.** Ambient noise broadband SL estimate: 91 dB re 1 µPa (long-term minimum).	S1 call (SL estimates of 150 dB re 1µPa@ 1 m) and clicks (SL estimates of 200 dB re 1µPa@ 1m).	Haro straight ships may reduce communication space by 94 to 98% when abeam foraging space by 58 to 89% when abeam. Larger impact on communication due to lower call SL and more ship noise near 7 kHz than 20 kHz.	Veirs and Veirs (2011)
Beluga whale, *Delphinapt-erus leucas*	Model	Bubbler system and propeller cavitation noises from ice-breaker ship.	**1/12th octave band analysis.** **Bubbler noise:** most energy < 5 kHz, median SL 192 dB re 1µPa @ 1 m 100 Hz-20 kHz. **Cavitation:** broad band median SL of 197 dB re 1µPa @ 1 m 100 Hz-22 kHz.	Typical beluga vocalisation, relatively low in frequency.	Masking of communication signals predicted within 14 to 71 km range depending on noise source (bubbler or cavitation) and water depth. Propeller cavitation accounts for the long-range effects.	Erbe and Farmer (2000)
Bottlenose dolphin, *Tursiops truncatus*	Field and model	Vessel noise	**1/3rd octave band analysis.** Five 1/3rd octave bands with center frequencies from 4 to 10 kHz.	Fundamental whistle contour.	Small vessels traveling at 5 knots in shallow water can reduce communication range of dolphin by 26%.	Jensen et al. (2009)
Short-finned pilot whales, *Globicepha-la macrorhyn-chus*	Field and model	Vessel noise	**1/3rd octave band analysis.** Nine 1/3rd octave bands with center freq. from 2 to 12.5 kHz. Back-calculated RMS source levels for vessels in the 2-12.5 kHz band: 132 to 146 dB re 1 µPa RMS at 1 m	Fundamental whistle contour	Small vessels traveling at 5 knots in deep water can reduce the communication range of pilot whales by 58%.	Jensen et al. (2009)

Table A-3 Evidence of Vocal Compensation Strategies in Mid-frequency Odontocetes (grouping as defined in Southall et al. 2007) Exposed to Masking Anthropogenic Sounds

Species	Noise Type	Type of Vocal Modification	Signal Studied	Details	Authors
Beluga whales, *Delphinapterus leucas*	Ambient noise	Bandwidth shift Frequency shift Amplitude	Echolocation clicks	The same beluga shifted the frequency and intensity of its echolocation clicks after it was moved to a habitat with an increase in 12 to 17 dB re 1 µPa ambient noise level. Peak frequencies shifted from 40 to 60 kHz to 100 to 120 kHz. Bandwidth shifted from 15 to 25 kHz to 20 to 40 kHz. Signal intensities increased by 18 dB re 1 µPa.	Au et al.(1985)
	Vessel traffic	Frequency shift Redundancy	Communication calls	Upward shifts in the mean frequencies of calls in the presence of low frequency vessel noise in the St. Lawrence estuary (from 3.6 kHz prior to exposure, to 5.2 to 8.8 kHz in the presence of vessels), and an increase in the repetition of specific calls.	Lesage et al. (1999)
	Vessel traffic	Amplitude (Lombard response)	Communication calls	The levels of the four common vocalisation types selected for analysis increased as a function of vessel noise level.	Scheifele et al. (2005)
Killer whales, *Orcinus orca*	Vessel traffic	Duration	Pulsed calls	KW in three pods increased the duration of one prominent call type by 15% following a large increase in whale-watching boats. (Challenged by evidence in Holt et al. 2009)	Foote et al. (2004)
	Vessel traffic	Amplitude (Lombard response)	Pulsed calls	The source level of 1 SRKW call type increased by ≈ 1 dB for every 1 dB increase in background noise levels. Vessel traffic was correlated with background noise levels.	Holt et al. (2009)
	Vessel traffic	Duration	Pulsed calls	The mean duration of 14 of 21 SRKW call types recorded in two periods, 1978 to 1983 and 2005 to 2006, significantly increased (mean increase of 54%).	Wieland et al. (2010)
	Background noise level	Amplitude (Lombard response)	Pulsed calls	A significant positive relationship was found between source levels and noise levels for the seven analysed call types for which there was a sufficient sample size.	Holt et al. (2011)

Species	Noise Type	Type of Vocal Modification	Signal Studied	Details	Authors
Bottlenose dolphins, *Tursiops truncatus*	White noise	Click rate	Echolocation clicks	The average number of clicks per trial during a target detection task increased with increasing masking noise level, up to a ceiling (77 dB *re* 1 μPa^2/Hz noise level) after which the number of clicks decreased with further increases in the noise level.	Au et al. (1982)
	Recreational boats	Rate	Whistles	Higher whistle rate at onset of noise than during or after exposure. Whistle rate for boat presence was also significantly greater than when no boats were present.	Buckstaff (2004)
Indo-Pacific bottlenose dolphin, *Tursiops aduncus*	Ambient noise (boat traffic)	Frequency shift	Whistles	In habitats with less ambient noise, dolphins produced more modulated whistles at varying frequencies. In a noisier habitat, whistles were lower in frequency with fewer frequency modulations.	Morisaka et al. (2005)
False killer whale, *Pseudorca crassidens*	*Long-line saver* acoustic deterrent device	Not measured	Echolocation clicks	Initially, the presence of the device reduced the whale's echolocation performance (detection of a target in the broadband, complex noise) to chance levels, but performance improved to 85% conceivably due to adaptation of echolocation technique to overcome the masking of the echoes.	Mooney et al. (2009)

APPENDIX 14-C

Southern Resident Killer Whale Population Consequence of Disturbance Model

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PROPOSED ROBERTS BANK TERMINAL 2

TECHNICAL REPORT

Marine Mammals

Southern Resident Killer Whale Population Consequences of Disturbance

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December 2014

SMRU CANADA

Technical Report/Technical Data Report Disclaimer

The Canadian Environmental Assessment Agency determined the scope of the proposed Roberts Bank Terminal 2 Project (RBT2 or the Project) and the scope of the assessment in the Final Environmental Impact Statement Guidelines (EISG) issued January 7, 2014. The scope of the Project includes the project components and physical activities to be considered in the environmental assessment. The scope of the assessment includes the factors to be considered and the scope of those factors. The Environmental Impact Statement (EIS) has been prepared in accordance with the scope of the Project and the scope of the assessment specified in the EISG. For each component of the natural or human environment considered in the EIS, the geographic scope of the assessment depends on the extent of potential effects.

At the time supporting technical studies were initiated in 2011, with the objective of ensuring adequate information would be available to inform the environmental assessment of the Project, neither the scope of the Project nor the scope of the assessment had been determined.

Therefore, the scope of supporting studies may include physical activities that are not included in the scope of the Project as determined by the Agency. Similarly, the scope of supporting studies may also include spatial areas that are not expected to be affected by the Project.

This out-of-scope information is included in the Technical Report (TR)/Technical Data Report (TDR) for each study, but may not be considered in the assessment of potential effects of the Project unless relevant for understanding the context of those effects or to assessing potential cumulative effects.

EXECUTIVE SUMMARY

The Roberts Bank Terminal 2 Project (RBT2 or Project) is a proposed new three-berth marine terminal at Roberts Bank in Delta, B.C. that could provide 2.4 million TEUs (twenty-foot equivalent units) of additional container capacity annually. The Project is part of Port Metro Vancouver's (PMV) Container Capacity Improvement Program, a long-term strategy to deliver projects to meet anticipated growth in demand for container capacity to 2030.

The Southern Resident Killer Whale (SRKW, *Orcinus orca*) Technical Advisory Group (TAG) considered two approaches to model population consequences of underwater noise and disturbance to SRKWs, and reported both options to Hemmera and PMV. Hemmera requested that SMRU Canada Ltd. develop a statistical model to estimate the population consequences of disturbance (PCoD) effects of underwater noise from regional commercial vessel traffic on the endangered population of SRKW. Various marine mammal PCoD models have been developed by members of a U.S. Office of Naval Research-funded Working Group. Lead research scientists from that Working Group are involved in this study. The PCoD framework links changes in individual behaviour and physiology due to acoustic disturbance, to the consequences to health, vital rates and, ultimately, the population (NRC 2005). For SRKW, data limitations limited the ability to construct, fit, and estimate transfer functions for every link of the PCoD framework. However, a simplified version of the PCoD approach was developed that uses a data-based relationship between lost foraging opportunities of chinook salmon (*Oncorhynchus tshawytscha*) and vital rates of SRKW to predict the effects of an individual's response to acoustic disturbance.

Three factors were taken into account when constructing the model scenarios: 1) spatial areas of interest; 2) the manner in which SRKW search for prey (i.e., in one, two or three dimensions); and 3) four development scenarios of acoustic disturbance occurring within the areas of interest (i.e., existing commercial vessel traffic; RBT2 and incremental traffic associated with RBT2 (including existing and expected traffic); future projects (including existing and expected traffic) without RBT2 or incremental traffic associated with RBT2; and RBT2 and incremental traffic associated with RBT2 plus other future projects (including existing and expected traffic). Thus, 24 scenarios were run based on a combination of spatial areas, prey searching patterns, and development scenarios to investigate lost foraging opportunities as a result of acoustic disturbance.

Data from the SRKW Underwater Noise Exposure and Acoustic Masking Report (SMRU 2014a) were used to parameterise the PCoD model. That study used estimates of SRKW density, predictive models of underwater noise, SRKW-specific behavioural underwater noise thresholds, and an underwater noise masking model to calculate: 1) the number of potential acoustic disturbances; and 2) the additional proportion of time that SRKW echolocation clicks may be masked by noise under the disturbance scenarios. The total amount of lost foraging time due to disturbance and masking was then calculated using the information provided in the SRKW Underwater Noise Exposure and Acoustic Masking Study

(SMRU 2014a) and the estimates of behavioural response durations provided in this report. The proportional decrease in foraging time was then assumed to be equivalent to a similar proportional decrease in the availability of chinook salmon, because changes in chinook availability have previously been shown to be correlated with SRKW's birth and death rates (i.e., "vital rates") in a predictable way. The resulting estimates of changes in vital rates of individual SRKW were incorporated into a stochastic population model which was used to calculate the growth rate of the population under the different factor combinations, and the relative difference in population size between existing conditions and future development scenarios.

It was estimated that there are 19.1 days (27,507 mins) of potential lost foraging time due to combined behavioural disturbance and masking per whale per year under existing conditions at a regional scale. This number increased by 1,446 minutes (5.3%) to 20.1 days under the RBT2 and incremental traffic associated with RBT2 scenario. In comparison, across the local study area, only 2.75 days of foraging were lost due to existing conditions and 2.90 days under the RBT2 and incremental traffic associated with RBT2 scenario. Little variation in the PCOD model estimates of vital rates, and the population growth rate and size was observed across the factor combinations, primarily because the predicted reductions in foraging time under the different acoustic disturbance scenarios were small compared to the existing acoustic levels (lost foraging time across one year varied from 5.2 to 5.6% across regional scenarios). Although the hypothesised manner in which SRKW search for prey did affect the total number of minutes lost to masking, in the absence of a behavioural response, this was a relatively small proportion of the total lost foraging time. Despite low statistical power, the underlying SRKW population model developed for this PCoD predicted a slowly increasing growth rate of 1% per annum (95% CI -3% to +4%) under existing conditions. Model results predict very similar growth rates for forecast increases in commercial vessel traffic noise.

In order to model the reduction in foraging time due to behavioural disturbance from underwater noise, assumptions had to be made on the duration of each behavioural response. Empirical data on the durations of killer whales behavioural responses to anthropogenic noise sources are limited; therefore, duration of responses were estimated by re-analysing a digital acoustic tag (DTAG) dataset of northern resident killer whales.

The development of a full PCoD framework was reviewed by the SRKW Technical Advisory Group while exploring various options to assess population level effects. Data limitations and data gaps were identified particularly within linkages needed for a full PCoD framework between body condition, health and vital rates. Instead, a more simplified approach was used to capture current biological knowledge to forecast how the population is likely to respond to acoustic disturbance.

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List of Acronyms

CONCEAL	Chronic ocean noise: cetacean ecology and acoustic habitat loss
FMA	Focused model area
LSA	Local study area
NRC	National Research Council
PCAD	Population consequences of acoustic disturbance
PCoD	Population consequences of disturbance
PMV	Port Metro Vancouver
RBT2	Roberts Bank Terminal 2 Project
SRKW	Southern resident killer whale
TR	Technical report

1.0 INTRODUCTION

1.1 PROJECT BACKGROUND

The Roberts Bank Terminal 2 Project (RBT2 or the Project) is a proposed new three-berth marine terminal at Roberts Bank in Delta, B.C. that could provide 2.4 million TEUs (twenty-foot equivalent unit containers) of additional container capacity annually. The Project is part of Port Metro Vancouver's (PMVs) Container Capacity Improvement Program, a long-term strategy to deliver projects to meet anticipated growth and demand for container capacity to 2030.

Port Metro Vancouver has retained Hemmera to undertake environmental studies related to the Project. This technical report (TR) describes the results of the Population Consequences of Disturbance (PCoD) study conducted by SMRU Canada Ltd. on behalf of Hemmera.

1.2 SOUTHERN RESIDENT KILLER WHALE PCOD OVERVIEW

This TR describes the findings from a study to predict the PCoD for southern resident killer whales (SRKW, *Orcinus orca*) from underwater noise produced by the Project and regional commercial vessel traffic. PCoD is a conceptual framework that links disturbance induced behavioural and physiological changes to population-level effects via changes in health and vital rates (NRC 2005). **Figure 1** depicts the relationship of this study to other Project studies. Study components, major objectives, and a brief overview are provided in **Table 1**.

Figure 1 Relationship of this study to other Project studies. DTAG = Digital Acoustic Recording Tag

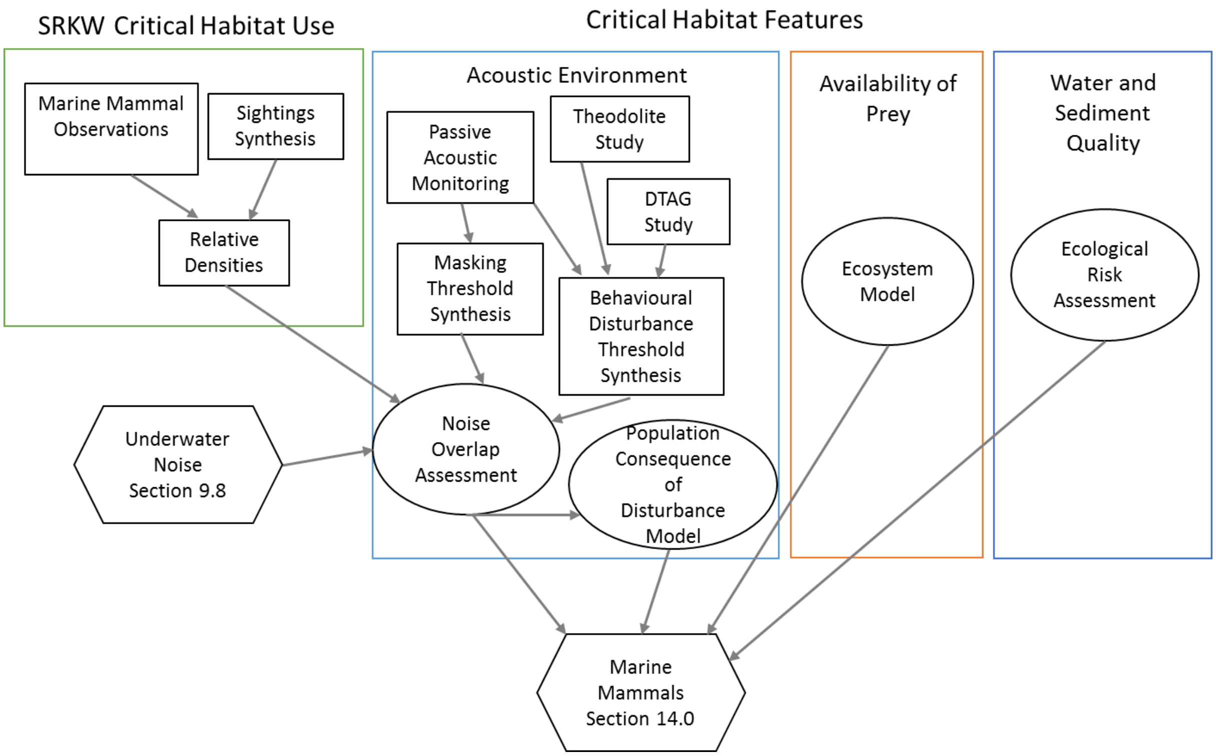


Table 1 SRKW Population Consequences of Disturbance Study Components and Major Objectives

Representative Species	Major Objective	Brief Overview
1) SRKW (*Orcinus orca*)	• Develop a model to estimate the population consequences of acoustic disturbance (PCoD) on SRKW in relation to four development scenarios	• Estimate the effect of existing underwater noise from commercial vessel traffic on SRKW survival, fecundity and growth rate. • Estimate changes in SRKW survival, fecundity, and growth rate from existing acoustic conditions as a result of underwater noise produced by commercial vessel traffic during three development scenarios. • Calculate the relative change in SRKW growth rate and population size between the existing conditions and future development scenarios.

The primary objective of this study was to determine how acoustic disturbance from commercial vessels within a pre-defined area of the Salish Sea might affect the SRKW population. In recent years, expansion of marine developments has led to a global concern among environmental scientists and policy makers about the potential effects of anthropogenic noise on the marine ecosystem. Effects on marine mammals are of particular concern because noise has the potential to cause behavioural changes, range displacement, communication interference, decreased foraging efficiency, hearing damage, and physiological stress (Tyack 2008).

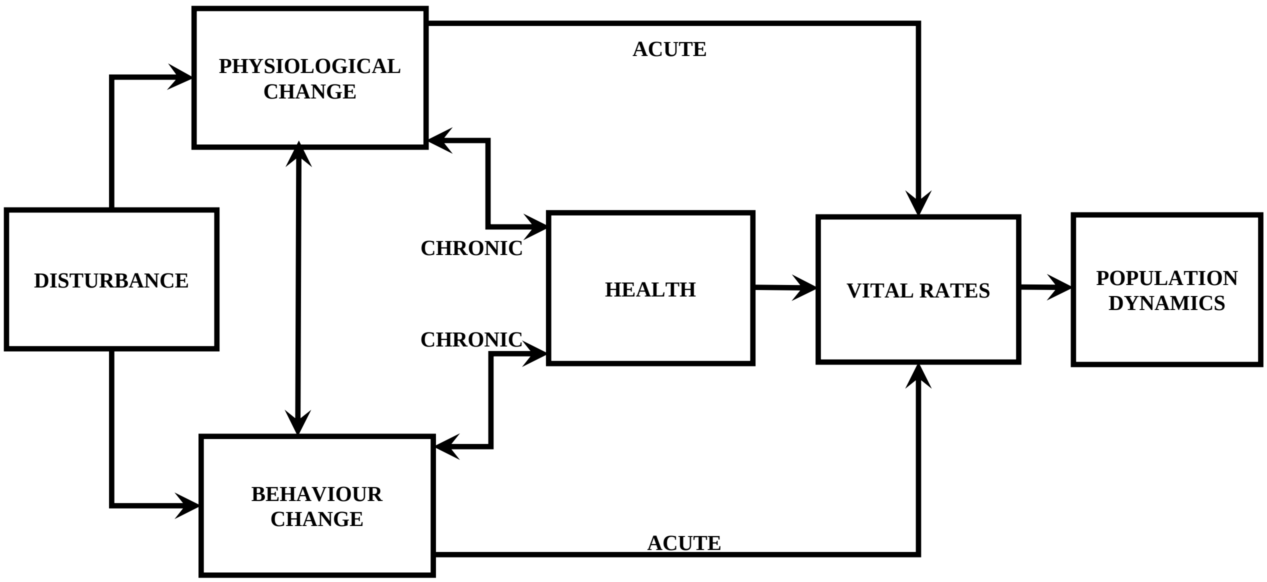
South resident killer whales are Endangered under the *Species at Risk Act (SARA)* and the proposed Project is located within federally designated critical habitat for the species in Canada. Noise, availability of prey, and environmental contaminants have been identified as factors potentially threatening population recovery; however, the relative influence of these factors on SRKW population dynamics is not fully understood. This study aims to quantify the relationship between underwater noise and feeding success of SRKW on their principal prey, chinook salmon (*Oncorhynchus tshawytscha*), and predict potential effects on the population.

2.0 REVIEW OF EXISTING LITERATURE AND DATA

Anthropogenic noise has the potential to negatively affect the marine environment. Cetaceans, in particular have the potential to be affected by underwater noise, either directly by changes in vital rates, or indirectly through changes in behaviour and physiology. A U.S. National Research Council working group (NRC 2005) made one of the first attempts to address these issues, outlining a conceptual framework for the population consequences of acoustic disturbance (PCAD). Some aspects of the framework, such as the relationship between vital rates and population effects, were well understood at the time (e.g., Caswell 2001), but others, such as the relationship between behaviour change and 'life functions', required more study (NRC 2005). When the PCAD framework was developed, one of the main impediments to its application was a lack of statistical tools and computational power; however, advances in these areas since the report was published have enabled researchers to begin applying the PCAD framework to marine species.

To date, the PCAD framework has been applied to elephant seals (*Mirounga* sp.) (New et al. 2014, Schick et al. 2013), coastal bottlenose dolphins (*Tursiops truncatus*) (New et al. 2013a, Pirotta et al. 2014), North Atlantic right whales (*Eubalaena glacialis*) (Schick et al. 2013), and beaked whales (family *Ziphiidae*) (New et al. 2013b). These case studies have demonstrated PCAD's potential and have led to further developments in the framework, expanding it to include anthropogenic and environmental disturbance, physiological and behavioural effects, and ultimately leading to the formulation of the PCoD framework (**Figure 2;** New et al. 2014). The PCoD framework is also able to distinguish between disturbances that have an acute, immediate effect on vital rates, and disturbances that have a chronic effect on vital rates through individual health, which is the primary route through which indirect impacts on vital rates take place. Any alterations in vital rates can lead to predictable changes in the dynamics of the population of interest (New et al. 2014). The PCoD framework is applicable to a much wider range of species and disturbances than the original PCAD framework. As a result, it is better suited to an analysis of the long-term consequences of short-term changes in behaviour or physiology in response to disturbance (New et al. 2014).

Figure 2 The Conceptual Model of the PCoD Framework, Linking Disturbance to Changes in Behaviour and Physiology, Health, Vital Rates, and Population Dynamics (based on Figure 6 in New et al. 2014)



The PCoD approach requires a large amount of data if the model is to be completely parameterised. Although SRKWs are extremely well studied at the population level, much of the information required for a full PCoD model (e.g., the number of days a whale can sustain itself without feeding, the link between prey intake [or fasting] and body condition, and the point at which poor body condition leads to loss of pregnancy or death) is lacking.

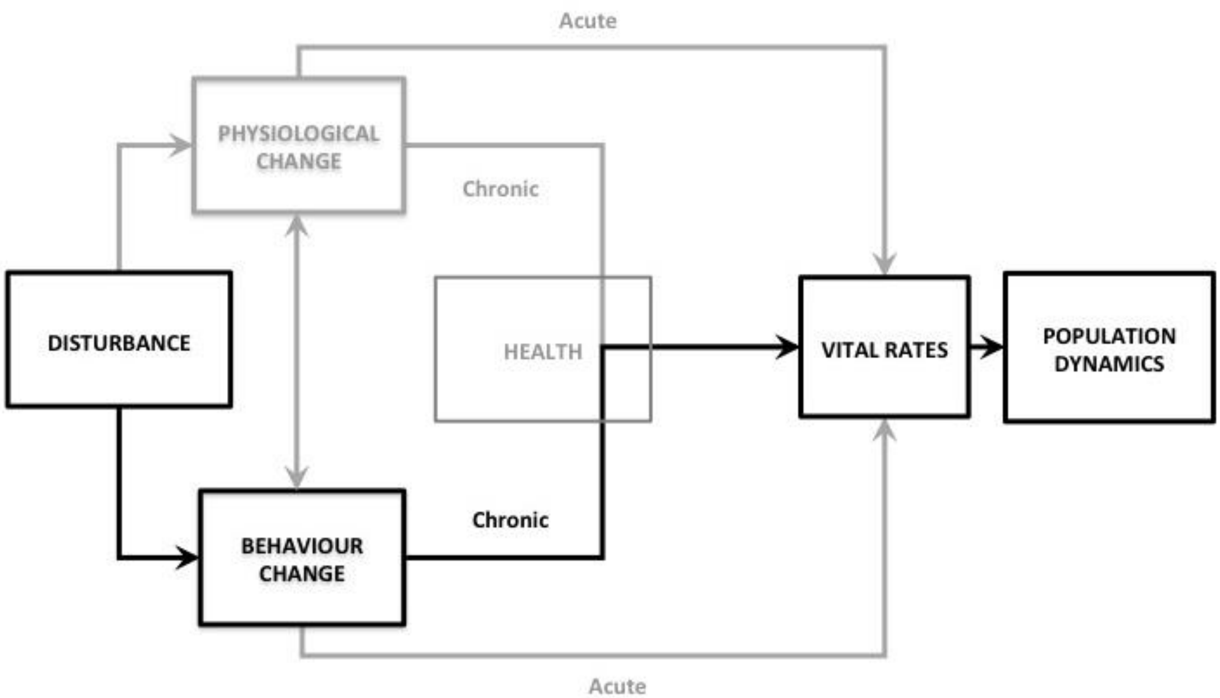
In 2010, a European Union-funded project called CONCEAL (Chronic ocean noise: cetacean ecology and acoustic habitat loss) was launched to develop management recommendations on allowable harm limits of disturbance for data-poor cetacean species. [1]

The CONCEAL program modelled the relationships between variability in prey abundance and dynamics of generic populations of fin, humpback, and killer whales. CONCEAL was used as a starting point, but updated with case-specific information on how Northeast Pacific resident killer whale populations have responded, historically, to inter-annual variability in abundance of chinook salmon, the whales' preferred prey (Ward et al. 2009, Ford et al. 2010). Published and emerging research has shown that when chinook salmon abundance is low, the probability of a killer whale surviving to the next year declines (Ford et al. 2010, Ward et al. 2013), as does the probability that a female of a given age will produce a calf (Ward et al. 2009; Ward et al. 2013). Disturbance outputs from the SRKW Noise Exposure and Acoustic Masking Study (SMRU 2014a) were used to estimate how much SRKWs would reduce their time spent feeding, or how often underwater noise from commercial vessel traffic would mask their echolocation clicks. By

[1] Dr. Rob Williams, Prof. Philip Hammond, Dr. Len Thomas and Dr. Chris Clark developed the CONCEAL project, and, although currently unpublished, data were kindly provided to undertake this PCOD approach.

combining this information with demographic information on the SRKW population, it was possible to predict how reproductive and survival probabilities would be affected by increased noise exposure and, consequently, how population growth rate would be affected under future acoustic scenarios of commercial vessel traffic relative to current existing conditions. This study therefore embraces a state of the art framework (PCoD), but uses a simplified version termed PCoD-Lite that takes account of the data gaps described above (**Figure 3**).

Figure 3 The Conceptual Model Underpinning the PCoD-Lite Framework

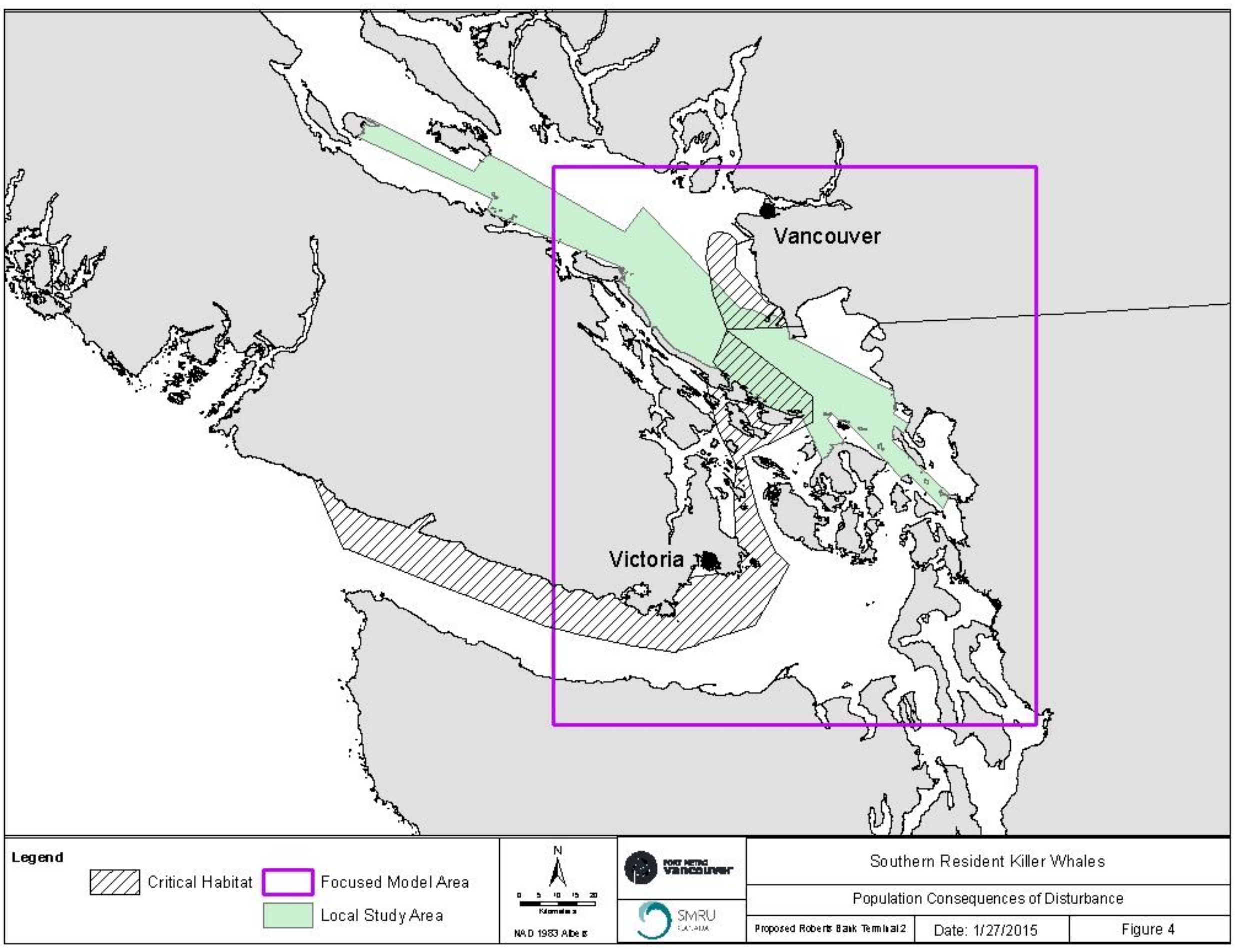


3.0 METHODS

3.1 STUDY AREA

The study area is bordered by the Strait of Georgia and Burrard Inlet in the north and the Juan de Fuca Strait in the south, and encompasses SRKW high use habitat areas. Two areas were considered, the focused model area (FMA) and the local study area (LSA) (**Figure 4**), as defined by the maximum estimated zone of audibility of Project noise for SRKW (Construction and Terminal Activity Underwater Noise Modelling Study (JASCO 2014a)). Given noise modelling constraints (Regional Commercial Vessel Traffic Underwater Noise Modelling Study (JASCO 2014b)) and the geographical extent of data on SRKW relative density (SRKW Network Sighting Synthesis (Hemmera 2014a), results were only possible for the section of the LSA within the FMA. As such, while it is possible to model the entire FMA, the same is not true for the LSA. The north-east end of the LSA, which is outside of the FMA (**Figure 4**), is not included in this modelling process.

Figure 4 The Location of the Study Areas in the Salish Sea.



3.2 TEMPORAL SCOPE

Within the study areas (**Section 3.1**), acoustic disturbance from underwater noise produced by commercial vessels may occur at any point over a 24-hour time period. Each 24-hour period was discretised into 288, 5-minute intervals within the boundaries shown in **Figure 4**, and an underwater noise behavioural effect threshold and acoustic masking model (SMRU 2014a) was used to determine whether a low-severity response, moderate-severity response, or echolocation masking, had occurred within that time interval. Based on the behavioural effects threshold studies (SMRU 2014b), high-severity behavioural responses were not expected to occur as a result of underwater noise produced by regional commercial vessel traffic. The total number of each response type was then summed within each day over the course of one year, which was divided into two seasons: summer and winter.

The total number of acoustic responses per day was used as the input into the PCoD model framework described in **Section 3.3.2.** The PCoD model was then used to investigate the effect of different scenarios of acoustic disturbance and spatial coverage on the survival, fecundity, and population growth rate of the SRKW population after a year of disturbance. The potential effects of different disturbance scenarios on population growth rate over a 20-year time period were estimated assuming that average underlying demographic rates did not change over this period. This time period was considered to be the most appropriate for estimating population growth rate, given the small number of breeding age females in the population and the long inter-birth interval.

3.3 STUDY METHODS

3.3.1 Disturbance Scenarios

This study investigated four scenarios (**Table 2**) within the two spatial extents (FMA and LSA, **Figure 4**).

Table 2 The Four Acoustic Impact Scenarios Considered as Part of the Population Consequences of Disturbance Study

Scenario	Year	Description
(S1)	2012	Existing commercial vessel traffic.
(S2)	2030	Future commercial vessel traffic with no new projects except RBT2, and future incremental vessel traffic associated with RBT2 (includes existing and expected conditions) [1].
(S3)	2030	Future commercial vessel traffic due to certain and foreseeable projects without RBT2, or incremental vessel traffic associated with RBT2 (includes existing and expected conditions).
(S4)	2030	Future commercial vessel traffic due to certain and foreseeable projects, with RBT2, incremental shipping traffic associated with RBT2 (includes existing and expected conditions).

[1]Expected conditions between 2012 and 2030 include no new projects but increases in vessel traffic at Westshore and Deltaport terminals i.e. Deltaport Terminal Road and Rail Improvement Project (DTRRIP).

For each of the four scenarios, the effects of three hypotheses regarding the way that SRKW search for prey were examined (Bain et al. 2014): 1) if SKRW know that chinook salmon are found along a certain depth contour, they may search along a line (i.e., in one dimension); 2) if they know that prey are somewhere on the seabed, they may search in two dimensions; and 3) if they have no information on where the fish are, they may search in three dimensions. Uncertainty in basic killer whale foraging ecology has the potential to increase uncertainty in any estimates of the effects of acoustic masking; therefore, the effects of all three prey-searching hypotheses on results from the scenarios were examined. As a result, 24 factor by scenario combinations, which accounted for the two spatial areas, four development scenarios, and three prey search hypotheses, were investigated (**Table 3**). Two temporal scales were considered when investigating the total and proportion of foraging time lost to disturbance: 1) the time spent by the SRKW within the LSA or FMA; and 2) the total amount of time within a year.

Table 3 The 24 Factor by Scenario Combinations Used for Estimating the Population Consequences of Disturbance on the SRKW Population

Area	Scenario	Search
LSA	S1	1D
		2D
		3D
	S2	1D
		2D
		3D
	S3	1D
		2D
		3D
	S4	1D
		2D
		3D
FMA	S1	1D
		2D
		3D
	S2	1D
		2D
		3D
	S3	1D
		2D
		3D
	S4	1D
		2D
		3D

Given the large number of combinations, only results for the LSA with the three-dimensional search hypothesis are presented in the main text. They can be considered as a 'worst' case scenario because searching in three dimensions results in the highest percentage loss of foraging time due to masking (SMRU 2014a). Results for the FMA and all other LSA factor by scenario combinations are reported in **Appendix A**.

3.3.2 Implementing the PCoD framework

The PCoD framework demonstrates how acoustic disturbance can affect individual behaviour and physiology, and the consequences of these changes for health, vital rates and, ultimately, population dynamics (**Figure 1**). Components of the framework are linked by transfer functions, which may be modeled mechanistically or phenomenologically. Behavioural change can also affect physiology, and vice versa. A full application of the PCoD framework to SRKW was not possible given current data limitations. As recommended by the SRKW Technical Advisory Group, a simplified version of the PCoD approach (i.e., utilising aspects of the model framework developed for the CONCEAL project) was used to predict the potential effects of any loss of foraging opportunities on an individual's vital rates due to its response to acoustic disturbance. Since SRKW survival and reproduction are known to be correlated with chinook availability (Ward et al. 2009, Ford et al. 2010, Ward et al. 2013), effects of acoustic disturbance on vital rates were modelled by reducing the value of an index indicating the availability of chinook salmon to an individual.

The modified PCoD framework was implemented in the statistical programming language R (R Core Team 2014) and was designed to use model outputs from SMRU 2014a (**Table 4**). Required inputs to the PCoD model are the total number of low- and moderate-severity behavioural responses to disturbance from underwater noise experienced by each individual over the course of one year, and an estimate of the additional amount of time an individual experienced echolocation masking if no disturbance was predicted. No high-severity behavioural responses were expected and low- and moderate-severity responses were the only responses investigated (SMRU 2014a).

Table 4 Required Inputs for the PCoD Framework Used for SRKW

Input	Source	Brief Overview
Number of behavioural disturbances	**SRKW Noise Exposure and Acoustic Masking Study** (SMRU 2014a)	Focused model provides total number of low- and moderate-severity behavioural disturbances from underwater noise experienced by each individual whale over the course of one year, and any additional time each individual loses foraging opportunities to acoustic masking of echolocation calls. Results are presented in the focused model area and the local study area.
Number of days individuals spend in the areas of interest	**SRKW Noise Exposure and Acoustic Masking Study** (SMRU 2014a)	Focused model provides total number of days each individual whale spends in the FMA and LSA over the course of one year

Input	Source	Brief Overview
Behavioural response durations	• **Appendix B:** DTAG re-analysis: assessing behavioral response durations for killer whales[1]	The northern resident killer whale DTAG dataset was analysed to determine killer whale behavioral response duration to large ships. Two analytical approaches were used; i) randomisation tests; and ii) B-spline regression analysis. The estimated response durations used in the PCOD model are as follows: 1) low-severity response - 5 minutes; and 2) moderate-severity response - 25 minutes.
Chinook Index	Ward et al. 2009, Ward et al. 2013	For each simulation, a value from the Chinook Technical Committee's West Coast Vancouver Island index for chinook is randomly generated from a uniform distribution whose end points are the highest and lowest values for the index observed between 1970 and 2010.

[1] Dr. John Ford, Mr. Graeme Ellis, and Dr. Volker Deecke collected this DTAG data as part of their ongoing studies on NRKW and kindly provided the data for this study.

3.3.3 Data Analysis

Simulations from the SMRU 2014a considered a SRKW population of 80 individuals[2] (Center for Whale Research 2014) divided into three pods (J, K and L), but did not assign individuals to a specific gender or age class. Because these classes have different survival and reproductive rates (Ward et al. 2009, Ford et al. 2010, Ward et al. 2013), the simulated individuals needed to be assigned to appropriate classes in order to model the population dynamics accurately. Each individual was randomly assigned to an age category (i.e., calf, juvenile, adult male, reproductively active adult female, senescent female) using probabilities calculated from the current composition of the SRKW population (Center for Whale Research 2014), and then assigned an age appropriate to that category. This assignment was restricted to ensure there were no motherless calves and there was at least one representative of each age class, barring calves, in each pod. A different age structure was assigned at the beginning of each simulation, along with a value for the chinook salmon index (*I*). The known age and class structure of the population was not used, to make it clear that this was not an attempt to predict the fate of each individual SRKW. The value of the chinook salmon index was drawn from a uniform distribution with a limit of 0.4 and 1.3 (Ward et al. 2009), since all known values for the index were considered equally likely. The index was assumed to represent the average prey available to the SRKW population given all existing levels of acoustic disturbance not included in the scenarios (JASCO 2014b). As a result, the simulated chinook index was assumed to account for all disturbances outside the LSA or FMA spatial coverage, and any small boats, such as recreational vessels or whale-watching vessels that would have been operating in the LSA or FMA. Although current vital rates for the SRKW population were assumed to reflect the historical effects of small boat noise during the time period covered by Ward et al. (2009, 2013) analyses, these effects may have increased or decreased over the time which has not been accounted for. This is discussed in **Section 5.2**. In addition, while SRKW do consume other prey items, chinook salmon is their primary food

[2] At the time of modelling, the SRKW population was 80 individuals (Center for Whale Research 2014)

source and the only one for which a relationship with their vital rates has been estimated. As a result, while the foraging time lost to disturbance would also impact the SRKW intake of alternate prey, any potential impact is not accounted for in our models.

Once the initial population age structure was defined, the total time each individual was affected by acoustic disturbance was calculated. First, the behavioural response to acoustic disturbance was evaluated. Based on a reanalysis of DTAG data (SMRU 2014b), it was assumed that an individual exhibiting a low-severity behavioural response would be affected for 5 minutes and an individual exhibiting a moderate-severity behavioural response would be affected for 25 minutes. **Appendix B** provides a summary of the rationale and justification for these behavioural response durations. The total amount of time individual *i* was affected by disturbance within a year (B_i) was calculated as:

$$B_i = \sum_{t=1}^{T_i} (5L_{i,t} + 25M_{i,t}), \qquad \textbf{(3.3.1)}$$

Where:

$L_{i,t}$ and $M_{i,t}$ are the number of low-severity and moderate-severity behavioural responses for individual *i* on day *t*, respectively, and T_i is the total number of days an individual has spent in the spatial area of interest.

The total amount of time an individual's ability to detect prey was masked ($K_{i,t}$) depended on how the foraging hypothesis was modelled and was calculated directly as part of the noise exposure simulations (SMRU 2014a); therefore, the total time an individual was affected by acoustic disturbance (D_i) was:

$$D_i = B_i + \sum_{t=1}^{T_i} K_{i,t} \qquad \textbf{(3.3.2)}$$

Equations 3.3.1 and **3.3.2** represent the final steps in determining the effects of the first transfer function in the modified PCoD framework, linking disturbance to a change in behaviour and physiology (**Figures 2** & **3**).

The next transfer function in the PCoD framework would normally predict the effect of changes in behaviour and physiology on an individual's health; however, development of this transfer function is not currently feasible. Instead, the relationship between chinook salmon abundance and SRKW vital rates was used as a proxy for the relationship between time lost due to disturbance (D_i, **Equation 3.3.2**) and these rates. As a result, the PCoD framework was modified to link D_i directly to individuals' vital rates (**Figure 3**). Ward et al. (2009, 2013) and Ford et al. (2010) have shown that SRKW survival and fecundity are correlated with the value of the chinook index (*I*); therefore, the percentage of total time lost due to acoustic disturbance was assumed to result in an equivalent reduction in *I*. The chinook index for each individual (C_i) was therefore estimated as:

$$C_i = \left[1 - \left(\frac{D_i}{T_{a,i}}\right)\right] I \qquad \textbf{(3.3.3)}$$

Where: $T_{a,i}$ is the duration (in minutes) of the temporal scale under consideration. The relevant temporal scales were: 1) the total amount of time each individual spent in the LSA; 2) the total amount of time each individual spent in the FMA; and; 3) the total number of minutes within a year.

The chinook index (C_i) was used to calculate individual vital rates using **Equations 3.3.4** and **3.3.6**. The activity time budget of each individual was assumed to be the same whether it was in the FMA, LSA, or outside these areas, which allowed the use of (1 - D_i / $T_{a,i}$) as an estimate of the proportional loss in time spent foraging.

The survival probability for each individual was calculated using the relationship from Ward et al. (2013):

$$\phi_i = \frac{\exp(\alpha_{0,G,A} + \alpha_1 P_i + \alpha_2 W_i)}{1 + \exp(\alpha_{0,G,A} + \alpha_1 P_i + \alpha_2 W_i)}, \qquad \textbf{(3.3.4)}$$

Where: $\alpha_{0,G,A}$ is a gender and age class specific intercept; and P_i is a dummy variable that takes the value one if an individual is a member of L Pod, but is zero otherwise.

Ward et al. (2013) used a different chinook index (W in **Equation 3.3.4**) when estimating the effect of the salmon on SRKW survival than that used by Ward et al. (2009) to estimate the effect of salmon on SRKW fecundity (**Equation 3.3.6**). Since the chinook index (I) at the start of each simulation is drawn from the index used by Ward et al. (2009), it was necessary to estimate the value for W based on the available chinook index, C (**Equation 3.3.3**), which was done via a linear model whose response variable was the observed values of the Ward et al. (2013) index and whose explanatory variable was the observed values for the Ward et al. (2009) index. Using this model, the index of available chinook for the survival model, W_i, is calculated as:

$$W_i = \gamma_0 + \gamma_1 C_i \qquad \textbf{(3.3.5)}$$

Where: the γ parameters are the slope and intercept from the linear regression.

The survival for a calf was calculated using the maternal value for W_i, based on the assumption that the calf would have the same response to disturbance as its mother and that the mother would decrease provisioning of her calf in direct proportion to the decrease in prey availability she experienced.

The fecundity rate (λ_i) for each surviving female of breeding age without a calf in the previous time period was calculated using Ward et al.'s (2009) relationship:

$$\lambda_i = \frac{\exp\left(\beta_0 + \beta_{pop} + \beta_1 A_i + \beta_2 A_i^2 + \beta_3 A_i^3 + \beta_4 A_i^4 + \beta_5 C_i\right)}{1 + \exp\left(\beta_0 + \beta_{pop} + \beta_1 A_i + \beta_2 A_i^2 + \beta_3 A_i^3 + \beta_4 A_i^4 + \beta_5 C_i\right)}, \quad \textbf{(3.3.6)}$$

Where: A is the female's age, C_i is the index of available chinook, β_{pop} is an offset specific to the SRKW population, the remaining β parameters relate A_i and C_i to the individual's fecundity; and λ_i is equal to zero for all non-reproductive age classes and mothers with calves.

The mean demographic rates for the population were calculated as:

$$\begin{aligned} \Phi &= \sum \phi_i / 80 \\ \Lambda &= \sum \lambda_i / 80, \end{aligned} \quad \textbf{(3.3.7)}$$

Where: Φ and Λ are the mean population-level survival and fecundity rates, respectively, for the year under consideration.

The annual population growth rate (1 + R) (henceforth referred to as `population growth rate`) was defined in terms of the discrete net growth rate (R) and was calculated as:

$$1 + R = \mathrm{F} + \mathrm{L}. \quad \textbf{(3.3.8)}$$

When calculating the potential effects of acoustic disturbance on the growth rate of the SRKW population, density dependence was not explicitly incorporated into the model. Although there is evidence that density dependence occurs in resident killer whale populations (Olesiuk et al. 2005, Ward et al. 2013), those analyses also indicate that the population was close to its carrying capacity at the time the analyses were undertaken. Because the PCoD model requires a prey-demography linkage, this information was prioritised over `forcing` the population to behave in a way that conforms to a specific density dependence relationship. The choice was made to focus on the salmon-demography relationships, given the primary concern of the study was to predict how underwater noise could affect population dynamics by disrupting feeding.

Demographic stochasticity was incorporated by allowing the number of surviving adults (S_y) and calves born (F_y) in year y to vary according to a binomial distribution where N is equal to the population size in year y (N_y) and p is the survival and fecundity rates calculated in **Equation 3.3.7**; therefore, population size in year y+1 was modelled as:

$$\begin{aligned} S_y &\sim Bin(N_y, \Phi) \\ F_y &\sim Bin(N_y, \Lambda) \\ N_{y+1} &= S_y + F_y, \end{aligned} \qquad \textbf{(3.3.9)}$$

The size of the SRKW population in each of the 20 years under different scenarios of disturbance was obtained by applying **Equation 3.3.9** over 20 time steps.

Uncertainty in the number of behavioural disturbances and number of days individual SRKW spend in the spatial areas of interest was incorporated into the PCoD model by applying **Equations 3.3.1** to **3.3.9** to all 1,000 simulations that were outputs from SMRU 2014a allowing for an estimation of uncertainty in the model`s parameter values.

4.0 RESULTS

Outputs from SMRU 2014a (**Table 4**) were used as inputs for the modified PCoD framework. The modelling approach described in **Section 3.4** was applied to all 1,000 simulations that formed these outputs; therefore, for each of the 24 factor by scenario combinations, there were 1,000 estimates, which were used to calculate the means and 95% confidence intervals of the vital rates, population growth rate, and population size in 20 years.

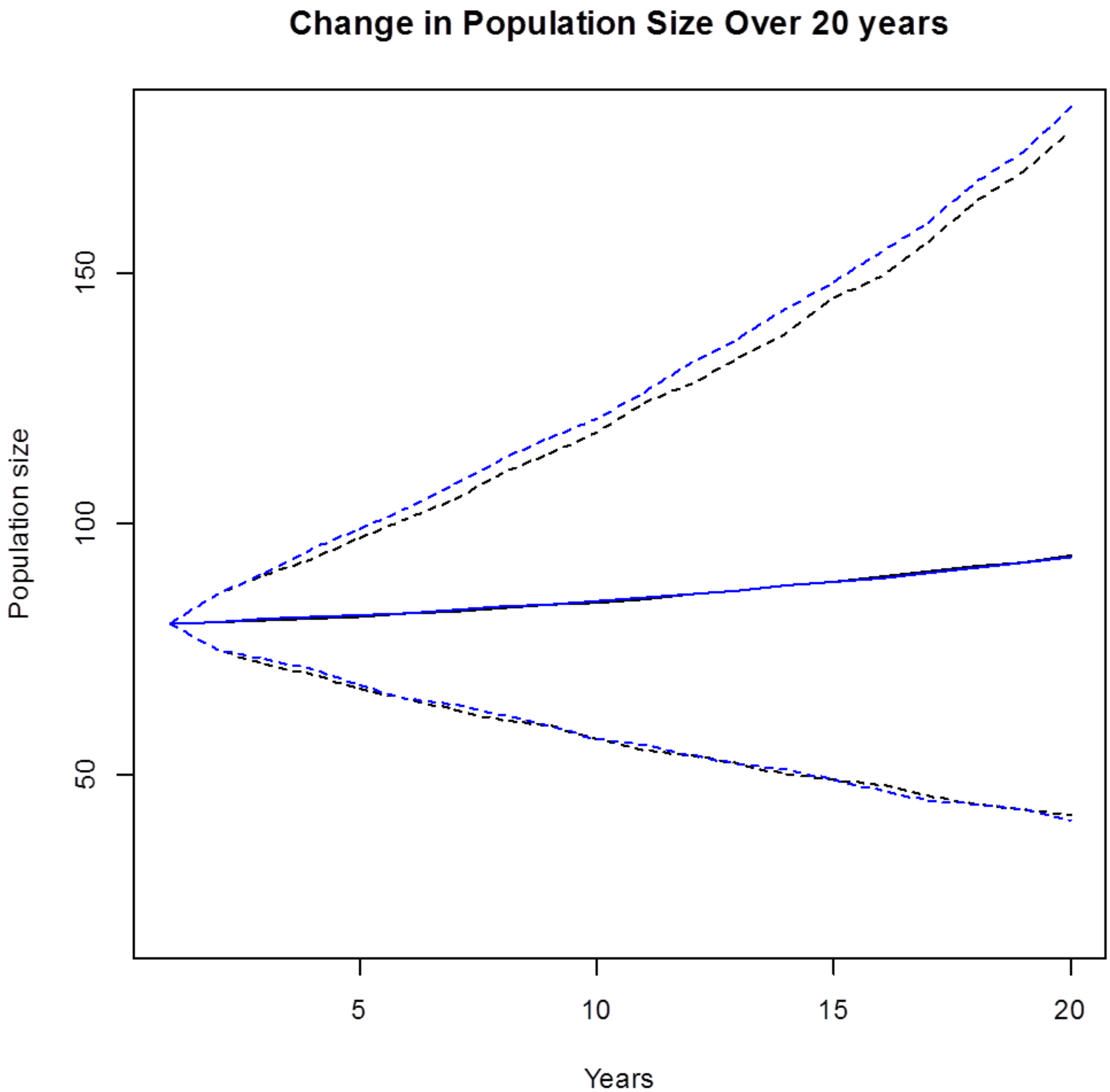
Results from the three development scenarios for the LSA, assuming a three-dimensional search pattern (i.e., the worst case scenario), are compared in **Table 5**. Results from the remaining scenarios are provided in (**Appendix A: Tables 7 to 11**). Absolute values for estimates of survival (Φ), fecundity (Λ), and population growth rate (1 + *R*) are presented, and estimates of the relative difference in *R* between the disturbance and existing condition scenarios and for the predicted SRKW population in 20 years (N_{20}), are provided. A negative value indicates that the absolute value was less than that obtained with the existing conditions scenario. A plot of the predicted population size (with CI) is presented in **Figure 5** under existing commercial vessel traffic and future Project plus incremental shipping traffic associated with the Project based on the effects of disturbance in the LSA and assuming 3D prey search. The average amount of time an individual is exposed to acoustic disturbance is presented in **Table 6**, along with the percentage of lost foraging time within the LSA and over the course of one year.

Table 5 Estimates of the Effects of Disturbance in the LSA on SRKW over the Time Period of One Year, Assuming Three-dimensional Prey Search Behaviour.

Disturbance scenario	Survival (Φ)	Fecundity (Λ)	Population Growth Rate (1+*R*)	Relative Change in Population Growth Rate (Δ1+*R*)	Relative Change in Population size (ΔN_{20})
S1	0.97 (0.95, 0.99)	0.034 (0.016, 0.058)	1.01 (0.97, 1.04)	-	-
S2	0.97 (0.95, 0.99)	0.034 (0.017, 0.058)	1.01 (0.97, 1.04)	0.0 (-0.03, 0.03)	0.0 (-0.43, 0.75)
S3	0.97 (0.95, 0.99)	0.034 (0.016, 0.058)	1.01 (0.97, 1.04)	0.001 (-0.03, 0.02)	0.0 (-0.45, 0.77)
S4	0.97 (0.95, 0.99)	0.034 (0.016, 0.059)	1.01 (0.97, 1.03)	0.001 (-0.03, 0.03)	0.0 (-0.45, 0.83)

Note: The table provides the mean values for survival (Φ), fecundity (Λ), and population growth rate (1+*R*). In addition, it provides estimates of the difference between 1+*R* for the predicted (S2, 3, 4) and Existing Conditions (S1) scenarios and for the forecast difference in SRKW population size in 20 years (N_{20}) relative to the values of the Existing Conditions scenario. A negative value indicates that the absolute value was less than that obtained with the Existing Conditions scenario. All values are provided with their 95% confidence intervals.

Figure 5 Estimates of Population Size (solid lines) Under Existing Conditions Based on the Effects of Disturbance in the LSA Over the Course of a Year and Assuming 3D Prey Search. Dashed lines are the 95% CI.



Note: (S1; black) and Project + Incremental Shipping Traffic Associated with the Project Scenario (S2; blue); the confidence interval is larger for S2 likely due to higher variability in the model inputs for S2.

More than 13% of foraging time in the LSA was lost under existing conditions (S1). At most, there was a 6.0% increase in lost foraging time under the other disturbance scenarios (S2, 3, 4), based on an absolute increase of 0.8% from 13.4% to 14.2%. Over the course of a year, as opposed to just the time spent in the LSA, there was a maximum of 6.7% increase in lost foraging time due to disturbance, resulting from an absolute increase from 0.75% to 0.80% (**Table 6**). The estimates of population survival, fecundity and population growth rate were similar across all three disturbance scenarios within the LSA, assuming a three-dimensional prey search pattern (**Table 5**). A similar consistency was observed in the estimates of the change in population growth rate and relative change in population size (**Table 5 and Figure 5**). The same pattern was observed for the remaining LSA and FMA scenarios when results from the other prey-search hypotheses were compared (**Appendix A: Tables 7 to 11**).

Table 6 The Average Amount of Foraging Time Lost per Individual Due to Acoustic Disturbance, and the Associated Proportional Reduction in Foraging Time within the LSA and over the Course of One Year, Assuming Three-dimensional Prey Search Behaviour. All Values are Provided with their 95% Confidence Intervals.

Disturbance scenario	Lost Time per Individual (min)	Proportional Reduction in Foraging Time in the LSA	Proportional Reduction in Foraging Time Over the Course of One Year
S1	3961 (3229, 5455)	0.134 (0.126, 0.145)	0.0075 (0.006, 0.01)
S2	4173 (3407, 5742)	0.141 (0.133, 0.152)	0.0079 (0.007, 0.011)
S3	4100 (3346, 5635)	0.138 (0.131, 0.149)	0.0078 (0.006, 0.011)
S4	4211 (3437, 5791)	0.142 (0.134, 0.154)	0.0080 (0.007, 0.011)

5.0 KEY FINDINGS

5.1 DISCUSSION OF KEY FINDINGS

This study calculated the number of lost foraging minutes based on behavioural responses and masking due to commercial traffic noise to input into a simulated population model that also varied inter-annual salmon availability and natural demographic variability. The model was run across 20 years for a population size of 80. Existing conditions were predicted to reduce the total number of minutes of foraging by 19.1 days (27,507 minutes) per animal per year when in the FMA and 2.8 days (3,961 minutes) when in the LSA. The inclusion of the RBT2 and incremental vessel traffic associated with RBT2, in addition to existing and expected conditions (S2), increased the foraging time lost by approximately 5.3% resulting in 20.1 days lost in the FMA (17.3% reduction of foraging time in the FMA) and 2.9 days in the LSA (14.1% reduction of foraging time in the LSA). When integrating lost foraging time for each study area across an annual period the time lost represents no more than 5.6% per animal for the FMA and 0.8% per animal for the LSA under any scenario. There was no discernable difference between PCoD predictions of vital rates, growth rate, or population size between the four acoustic development scenarios after a 20 year period, reflecting both relatively small differences in number of vessels between scenarios, but also reflecting wide confidence limits around the model predictions. The potential loss of foraging time within the FMA under any of the scenarios is considered large and does not include small (whale-watching) vessels or the disturbance and potential lost foraging caused by periods of disturbance accumulated outside of the study area. The potential cumulative effects of RBT2 construction noise and other anthropogenic stressors have not been considered within this model. However, the absolute difference in lost foraging between scenarios is small and the average estimate of population trajectory does not seem to differ from the current population trajectory.

The key reason for the indistinguishability between population trajectories under different scenarios is due to the small differences in acoustic noise levels under the different scenarios. As reported in the JASCO 2014b, existing commercial vessel traffic density in the FMA is high and the addition of vessels under the Project and incremental traffic associated with Scenario 2 (S2) adds approximately 0.6 dB to existing noise levels in the shipping lanes. While behavioural responses and masking are driven by noise maxima during the passing of individual ships and not monthly noise averages, the monthly average does give a sense of the magnitude of change in noise under these scenarios.

Estimates of population growth rates (**Table 5**) imply that under the existing scenario of commercial vessel traffic, the SRKW population is likely to be stable or slowly increasing; therefore, although the future acoustic scenarios do not appear to worsen the situation for SRKW, the future status of the population remains uncertain. The SRKW population is small and incapable of rapid population growth; therefore, it is particularly vulnerable to the effects of demographic stochasticity (random events). While the future increase in commercial vessel traffic is not expected to cause a statistically significant worsening of the population trajectory, declines in the populations are possible within the range of the 5% and 95% confidence intervals.

Under existing conditions the model predicts a slightly increasing population trajectory. The relatively small predicted increase in foraging time lost per year across scenarios makes no apparent difference in the population trajectory or survival and fecundity rates. However, in years of poor chinook abundance, the population growth rate can naturally shift from a slight increase to a slight decline (Ward et al. 2013). As a result, the 5.3% reduction in foraging time from S1 to S2 in the FMA over the course of a year may have a larger impact in low chinook abundance years, particularly if they occur in succession. While the stochasticity in chinook abundance is accounted for in the simulations by varying each year randomly, a stable trend in chinook abundance over time is being assumed. If this is incorrect, the population's vital rates may change in ways not predicted here. Further, SRKW are a small, isolated population making them more vulnerable to demographic stochasticity, especially since the confidence intervals on growth rate allow for the possibility of a declining population (**Table 5 and Figure 5**). Masking of echolocation clicks, which was modelled in three different ways (i.e., in one, two or three dimensions), also had no discernable effect on relative estimates of the PCoD. Even with the highest level of masking, additional loss of foraging opportunities due to masking in the absence of a behavioural response was predicted to be only a small percentage of the overall time lost in comparison to the time lost from low-severity or moderate-severity behavioural disturbances, simply because the animals are expected to be exposed to this disturbance for relatively small parts of their range and year.

5.2 DATA GAPS AND LIMITATIONS

In terms of analytical methods, the PCoD model presented here encompasses a state of the art framework (PCoD). It is one of the few PCoD models to have been developed in which the effects of disturbance on behaviour, and behavioural change on vital rates has been quantified.

The model does, however, rely on the quality of the data inputs; many of which also have strong assumptions of their own (Hemmera 2014a, SMRU 2014a and JASCO 2014b). Therefore, while the model itself is robust, the results should be interpreted in the larger context of the limitations imposed by the different components of its structure and inputs (e.g., any limitations on the number of days individuals spend in the area of interest, the data used to parameterise the PCoD transfer functions, the number and duration of behavioural disturbances, and the existing noise data, could affect the results). These factors are described in more detail below.

Killer Whale Sightings Data

Killer whale seasonal distribution was based on an integration of B.C. and U.S. opportunistic sightings data using correction factors that reflected differences in sighting effort (Hemmera 2014a). Effort-corrected sightings data provide a minimum estimate of the amount of time that SRKW could be exposed to commercial vessel noise regionally; therefore, the amount of time SRKW spend in the spatial areas of interest may be underestimated, which would lead to an underestimation of the PCOD as well.

Similarly, there is some evidence that the activity budget of SRKW varies spatially, such that SRKWs make long range transits from their core area in order to spend time feeding on seasonally available prey (Ashe et al. 2010, Beneze et al. 2011). The robustness of model conclusions to violation of these assumptions was tested by increasing the effect of disturbance and masking on the chinook Index by 50 and 100% (**Table 13**). These changes did not alter the conclusion that the effects of acoustic disturbance on population growth rate are likely to be very small. For more information on the data input limitations please refer to SMRU 2014a.

PCOD Transfer Functions

Unlike previous applications of the PCoD framework (e.g.-2013, New et al. 2013a, b, 2014; Pirotta et al. 2014), the approach used for SRKW does not include a transfer function linking disturbance-mediated behavioural changes to the health of individual animals, because the necessary information is not currently available. Instead, the model uses available information on the relationship between prey abundance and population dynamics as a proxy for the relationship between behavioural change and vital rates. However, the central assumption of this approach (that the effects of lost foraging time on vital rates are the same as an equivalent reduction in chinook salmon abundance) may not reflect biological reality due to the potential consumption of alternative prey resources (Ford and Ellis 2006, Hanson et al. 2010). That central assumption is effectively the same as assuming that SRKW feed entirely on chinook while they are in the area of Interest. It will lead to an over-estimate of the effects of disturbance if SRKW are actually feeding on prey with a lower net energy value than chinook during this time, but will under-estimate the effect if they are feeding on prey with a higher net energy value. The available evidence suggests that chinook are the most valuable prey for SRKW. For, example, killer whales ignore large schools of pink salmon to capture chinook salmon (Ford and Ellis 2006).

The models used to link changes in the chinook index to survival and fecundity (**Equations 3.3.4** and **3.3.6**) used different chinook indices as covariates; therefore, the relationship between these covariates needed to be modelled (**Equation 3.3.5**), adding additional uncertainty not fully captured by the confidence intervals for the survival estimates. In addition, neither of these models for survival and fecundity captures the full range of potential variation in the parameters themselves. Other models, such as forcing the intercepts through zero, were considered but these were deemed inappropriate because there has been no year since 1970 when no chinook returned to the Salish Sea. Therefore, information on killer whale demography when chinook abundance is zero is not available. Current models for survival and fecundity (Ward et al. 2009, 2013) predict relatively high survival even when chinook abundance is naturally low, possibly because SRKW switch to feeding on chum salmon or other species (Ford and Ellis 2006).

Behavioural Response Durations

Estimates of the amount of foraging time lost due to disturbance depended critically on the value used for the duration of a moderate-severity behavioural response to acoustic disturbance, because these responses have the greatest influence on the estimates of the percentage of lost foraging time. The duration estimate was based on a very limited sample of data from a small number of killer whales fitted with DTAGs. Any bias in this estimate would generate a similar bias in the estimate of lost foraging time. A larger sample size from more killer whales would undoubtedly increase confidence in the value used for this parameter; however, model results did not change significantly when disturbance effect was increased by 50 and 100% suggesting that the model itself is not very sensitive to the duration of behavioural response.

Accurate Representation of Existing Conditions

Finally, there are other aspects of the proposed RBT2 expansion by PMV that could affect SRKW vital rates that were not included in the PCoD model. These include, but are not limited to, underwater noise associated with the construction period, increased exposure to chemical pollutants (e.g., Ross et al. 2000, Hall et al. 2006, Hemmera 2014b) due to dredging during construction, small-vessel noise not included in the JASCO noise models (JASCO 2014b), and an increased risk of vessel collision. Chemical pollutants were not included in the PCoD model because the necessary link functions were not available. The construction noise and small-vessel noise were not included due to the complexity of adding them to the models.

The VTOSS database on which JASCO 2014b is based is known to underestimate the contribution of small boats to anthropogenic ocean noise levels (Erbe et al. 2012). Small whale watching boats that produce higher-frequency noise, with greater potential to mask echolocation clicks, and that follow whales rather than move randomly with respect to whale movements are not included in the noise modelling and this exclusion from modelling could result in a substantial underestimation of its contribution to existing conditions. We have assumed that any effects of whale watching boat noise on SRKW vital rates during the time period covered by the Ward et al. (2009, 2013) analyses of the relationship between these rates and the chinook index are reflected in the average values for the vital rates. However, any trend in these effects, for example as a result of an increase in whale watching boat traffic during the time period, would not be captured by this assumption and could result in an underestimation of their current impact on SRKW.

6.0 CLOSURE

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8.0 STATEMENT OF LIMITATIONS

This report was prepared by SMRU Canada Ltd., based on desktop studies conducted by SMRU, for the sole benefit and exclusive use of Hemmera and Port Metro Vancouver. The material in it reflects SMRU's best judgment in light of the information available to it at the time of preparing this Report. Any use that a third party makes of this Report, or any reliance on or decision made based on it, is the responsibility of such third parties. SMRU accepts no responsibility for damages, if any, suffered by any third party as a result of decisions made or actions taken based on this Report.

SMRU has performed the work as described above and made the findings and conclusions set out in this Report in a manner consistent with the level of care and skill normally exercised by members of the environmental science profession practicing under similar conditions at the time the work was performed.

This Report represents a reasonable review of the information available to SMRU within the established Scope, work schedule and budgetary constraints. The conclusions and recommendations contained in this Report are based upon applicable legislation existing at the time the Report was drafted. Any changes in the legislation may alter the conclusions and/or recommendations contained in the Report. Regulatory implications discussed in this Report were based on the applicable legislation existing at the time this Report was written.

In preparing this Report, SMRU has relied in good faith on information provided by others as noted in this Report, and has assumed that the information provided by those individuals is both factual and accurate. SMRU accepts no responsibility for any deficiency, misstatement or inaccuracy in this Report resulting from the information provided by those individuals.

APPENDIX A

Tables

Table 7 Estimates of the Effects of Disturbance in the LSA on SRKW over the Time Period of One Year, Assuming Two-dimensional Prey Search Behaviour

Disturbance Scenario	Survival (Φ)	Fecundity (Λ)	Population Growth Rate ($1+R$)	Relative Change in Population Growth Rate ($\Delta 1+R$)	Relative Change in Population size (ΔN_{20})
S1	0.97 (0.95, 0.99)	0.034 (0.016, 0.058)	1.01 (0.97, 1.04)	-	-
S2	0.97 (0.95, 0.99)	0.034 (0.017, 0.058)	1.01 (0.97, 1.04)	0.0 (-0.03, 0.03)	0.0 (-0.44, 0.82)
S3	0.97 (0.95, 0.99)	0.034 (0.016, 0.058)	1.01 (0.97, 1.04)	0.001 (-0.03, 0.02)	0.0 (-0.46, 0.83)
S4	0.97 (0.95, 0.99)	0.034 (0.016,0.059)	1.01 (0.97, 1.04)	0.001 (-0.03, 0.03)	0.0 (-0.48, 0.82)

Note: The table provides the mean values for survival (Φ), fecundity (Λ), and population growth rate ($1+R$). In addition, it provides estimates of the difference between $1+R$ for the predicted (S2, 3, 4) and Existing Conditions (S1) scenarios and for the forecast difference in SRKW population size in 20 years (N_{20}) relative to the values of the Existing Conditions scenario. A negative value indicates that the absolute value was less than that obtained with the Existing Conditions scenario. All values are provided with their 95% confidence intervals.

Table 8 The Estimates of the Effects of Disturbance in the LSA on SRKW over the Time Period of One Year, Assuming One-dimensional Prey Search Behaviour

Disturbance Scenario	Survival (Φ)	Fecundity (Λ)	Population Growth Rate ($1+R$)	Relative Change in Population Growth Rate ($\Delta 1+R$)	Relative Change in Population size (ΔN_{20})
S1	0.97 (0.95, 0.99)	0.034 (0.016, 0.058)	1.01 (0.97, 1.04)	-	-
S2	0.97 (0.95, 0.99)	0.034 (0.017, 0.058)	1.01 (0.97, 1.04)	0.0 (-0.03, 0.03)	0.0 (-0.46, 0.80)
S3	0.97 (0.95, 0.99)	0.034 (0.016, 0.058)	1.01 (0.97, 1.04)	0.001 (-0.03, 0.02)	0.0 (-0.46, 0.78)
S4	0.97 (0.95, 0.99)	0.034 (0.016, 0.059)	1.01 (0.97, 1.04)	0.001 (-0.03, 0.03)	0.0 (-0.46, 0.75)

Note: The table provides the mean values for survival (Φ), fecundity (Λ), and population growth rate ($1+R$). In addition, it provides estimates of the difference between $1+R$ for the predicted (S2, 3, 4) and Existing Conditions (S1) scenarios and for the forecast difference in SRKW population size in 20 years (N_{20}) relative to the values of the Existing Conditions scenario. A negative value indicates that the absolute value was less than that obtained with the Existing Conditions scenario. All values are provided with their 95% confidence intervals.

Table 9 The Estimates of the Effects of Disturbance in the FMA on SRKW over the Time Period of One Year, Assuming One-dimensional Prey Search Behaviour.

Disturbance Scenario	Survival (Φ)	Fecundity (Λ)	Population Growth Rate ($1+R$)	Relative Change in Population Growth Rate ($\Delta 1+R$)	Relative Change in Population size (ΔN_{20})
S1	0.97 (0.95, 0.98)	0.033 (0.016, 0.056)	1.0 (0.97, 1.04)	-	-
S2	0.97 (0.95, 0.98)	0.033 (0.016, 0.057)	1.0 (0.97, 1.04)	0.0 (-0.03, 0.03)	0.0 (-0.44, 0.81)
S3	0.97 (0.95, 0.98)	0.033 (0.016, 0.054)	1.0 (0.97, 1.04)	0.001 (-0.03, 0.02)	0.0 (-0.45, 0.75)
S4	0.97 (0.94, 0.98)	0.03 (0.015, 0.051)	1.0 (0.96, 1.03)	0.001 (-0.03, 0.02)	0.0 (-0.52, 0.57)

Note: The table provides the mean values for survival (Φ), fecundity (Λ), and population growth rate ($1+R$). In addition, it provides estimates of the difference between $1+R$ for the predicted (S2, 3, 4) and Existing Conditions (S1) scenarios and for the forecast difference in SRKW population size in 20 years (N_{20}) relative to the values of the Existing Conditions scenario. A negative value indicates that the absolute value was less than that obtained with the Existing Conditions scenario. All values are provided with their 95% confidence intervals.

Table 10 The Estimates of the Effects of Disturbance in the FMA on SRKW over the Time Period of One Year, Assuming Two-dimensional Prey Search Behaviour

Disturbance Scenario	Survival (Φ)	Fecundity (Λ)	Population Growth Rate ($1+R$)	Relative Change in Population Growth Rate ($\Delta 1+R$)	Relative Change in Population size (ΔN_{20})
S1	0.97 (0.95, 0.98)	0.033 (0.016, 0.055)	1.0 (0.97, 1.04)	-	-
S2	0.97 (0.95, 0.98)	0.033 (0.016, 0.056)	1.0 (0.97, 1.04)	0.0 (-0.03, 0.03)	0.0 (-0.44, 0.79)
S3	0.97 (0.95, 0.98)	0.033 (0.016, 0.054)	1.0 (0.97, 1.04)	0.001 (-0.03, 0.02)	0.0 (-0.45, 0.73)
S4	0.97 (0.94, 0.98)	0.03 (0.015, 0.051)	1.0 (0.96, 1.03)	0.001 (-0.03, 0.02)	0.0 (-0.50, 0.61)

Note: The table provides the mean values for survival (Φ), fecundity (Λ), and population growth rate ($1+R$). In addition, it provides estimates of the difference between $1+R$ for the predicted (S2, 3, 4) and Existing Conditions (S1) scenarios and for the forecast difference in SRKW population size in 20 years (N_{20}) relative to the values of the Existing Conditions scenario. A negative value indicates that the absolute value was less than that obtained with the Existing Conditions scenario. All values are provided with their 95% confidence intervals.

Table 11 The Estimates of the Effects of Disturbance in the FMA on SRKW over the Time Period of One Year, Assuming Three-dimensional Prey Search Behaviour

Disturbance Scenario	Survival (Φ)	Fecundity (Λ)	Population Growth Rate ($1+R$)	Relative Change in Population Growth Rate ($\Delta 1+R$)	Relative Change in Population size (ΔN_{20})
S1	0.97 (0.95, 0.98)	0.033 (0.016, 0.055)	1.0 (0.97, 1.04)	-	-
S2	0.97 (0.95, 0.98)	0.033 (0.016, 0.056)	1.0 (0.97, 1.04)	0.0 (-0.03, 0.03)	0.0 (-0.43, 0.82)
S3	0.97 (0.95, 0.98)	0.033 (0.015, 0.054)	1.0 (0.97, 1.04)	0.0 (-0.03, 0.02)	0.0 (-0.46, 0.74)
S4	0.97 (0.94, 0.98)	0.03 (0.015, 0.051)	1.0 (0.96, 1.03)	0.0 (-0.03, 0.02)	0.0 (-0.52, 0.60)

Note: The table provides the mean values for survival (Φ), fecundity (Λ), and population growth rate ($1+R$). In addition, it provides estimates of the difference between $1+R$ for the predicted (S2, 3, 4) and Existing Conditions (S1) scenarios and for the forecast difference in SRKW population size in 20 years (N_{20}) relative to the values of the Existing Conditions scenario. A negative value indicates that the absolute value was less than that obtained with the Existing Conditions scenario. All values are provided with their 95% confidence intervals.

Table 12 The Average Amount of Time Lost per Individual due to Disturbance, as well as the Associated Proportional Reduction in Foraging Time within the FMA and over the Course of One Year, Assuming Three-dimensional Prey Search Behaviour. All Values are Provided with their 95% Confidence Intervals.

Disturbance Scenario	Lost Time per Individual (min)	Proportional Reduction in Foraging Time in the FMA	Proportional Reduction in Foraging Time Over the Course of One Year
S1	27507 (22402, 37860)	0.164 (0.158, 0.174)	0.052 (0.043, 0.072)
S2	28953 (23605, 39784)	0.173 (0.167, 0.183)	0.055 (0.045, 0.076)
(S3	28756 (23441, 39533)	0.172 (0.166, 0.182)	0.055 (0.045, 0.075)
S4	29375 (23952, 40368)	0.175 (0.169, 0.186)	0.056 (0.046, 0.077)

Table 13 The Estimates of the Effects of Disturbance in the LSA on SRKW over the Time Period of One Year, Assuming Three-dimensional Prey Search Behaviour and 1.5 and 2 times the Estimated Levels of Disturbance for the Project and Baseline Acoustic Scenarios.

Disturbance Scenario	Survival (Φ)	Fecundity (Λ)	Population Growth Rate (1+R)	Relative Change in Population Growth Rate (Δ1+R)	Relative Change in Population size (ΔN_{20})
1.5 x S1	0.97 (0.95, 0.99)	0.034 (0.016, 0.058)	1.01 (0.97, 1.04)	-	-
1.5 x S2	0.97 (0.95, 0.99)	0.034 (0.017, 0.059)	1.01 (0.97, 1.04)	0.0 (-0.03, 0.03)	0.0 (-0.45, 0.77)
2 x S1	0.97 (0.95, 0.99)	0.034 (0.017, 0.059)	1.01 (0.97, 1.04)	-	-
2 x S2	0.97 (0.95, 0.99)	0.034 (0.016, 0.056)	1.01 (0.97, 1.04)	0.001 (-0.03, 0.03)	0.0 (-0.48, 0.75)

Note: The table provides the mean values for survival (Φ), fecundity (Λ), and population growth rate (1+R). In addition, it provides estimates of the difference between 1+R for the predicted (S2, 3, 4) and Existing Conditions (S1) scenarios and for the forecast difference in SRKW population size in 20 years (N_{20}) relative to the values of the Existing Conditions scenario. A negative value indicates that the absolute value was less than that obtained with the Existing Conditions scenario. All values are provided with their 95% confidence intervals.

APPENDIX B

Supplementary Material

DTAG re-analysis: assessing behavioral response durations for killer whales

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Summary

As part of the development of the southern resident killer whale (SRKW) PCoD project, SMRU Canada Ltd. re-analysed the northern resident killer whale (NRKW) DTAG dataset to determine killer whale moderate behavioral response durations to large vessel presence. Six ship and killer whale 'interactions' were re-analysed using the DTAG data used in the Determination of Behavioral Effect Noise Thresholds for Southern Resident Killer Whales Study (SMRU 2014). Dr. John Ford, Mr. Graeme Ellis, and Dr. Volker Deecke collected this DTAG data as part of their ongoing studies on NRKWs. Response variables included vocalisation rate, breathing rate, depth, heading change, and jerk (i.e., rate of change of acceleration). Given the small size of this dataset and large amount of variability within the data, two analytical approaches were used to determine behavioural response durations from the six ship and killer whale 'interactions': i) randomisation tests; and ii) B-spline regression analysis. Randomisation tests, which were used to determine whether a significant response by the killer whale group occurred, grouped data into 5-minute time bins during and after the ship interaction. The number of sequential 5-minute time bins where a response occurred was summed to give the behavioral response duration. B-spline regression analysis was applied to 1-minute means of the raw data during and after the vessel interaction. The B-spline regression estimated the start and end time of the 'disturbed' behavior (i.e., the behavioral response duration). There was good agreement between both analytical approaches for the duration of a moderate behavioral response to be 25 minutes.

Introduction

On behalf of PMV, Hemmera requested that SMRU Canada Ltd. assess the population consequences of disturbance (PCoD) of the proposed RBT2 expansion by Port Metro Vancouver and regional commercial vessel traffic on SRKW. This work involved estimating the number of potential acoustic disturbances per year using vessel noise modelling provided by JASCO (JASCO 2014) and SRKW-specific behavioural thresholds developed by SMRU Canada Ltd. (SMRU 2014) from previous reports. Dose-response curves were created for low-severity (severity scores 2 to 3) and moderate-severity (severity scores 4 to 6) behavioural responses, as described by Southall et al. (2007). Low-severity responses are considered brief and low level reactions that, in isolation, are unlikely to have any more than a weak impact on vital rates. Moderate-severity responses are higher level, last longer, and are more likely to affect foraging, reproductive behaviour, and subsequent vital rates. One of the defining features of the severity scale is that the severity of the response depends upon its duration in relation to the exposure duration (Southall et al. 2007). A behavioral response that continues beyond the end of an exposure is considered more severe (scores ≥5) than a response that stops when the exposure stops.

In order to model the population consequences of behavioural disturbances experienced by individual killer whales to ships, some assumptions on the duration of each behavioural disturbance were required. These assumptions are important in future modelling, as they will have a direct impact on calculations of the reduction in foraging time due to ship disturbance. Unfortunately, empirical data on the durations of behavioural changes of killer whales to anthropogenic noise sources are limited; therefore, using the severity scale definition to help inform assumptions about durations for low- and moderate-severity responses, the following approaches were used: i) low-severity changes in behaviour last the duration of the exposure period (i.e., 5 minutes, as per the temporal resolution of the SRKW PCoD noise overlap model); and ii) the duration of moderate-severity behavioural responses will be estimated using empirical data on killer whale behavioural responses to ships.

For this analysis, the DTAG dataset from SMRU 2014 that contains six interactions between killer whales and ships was used. In all six interactions, animals exhibited a moderate behavioural response (severity scores 5 and 6). Re-analysis of the six interactions from the DTAG dataset determines the duration of the moderate behavioural responses shown by the killer whales to ships, and informs the underlying assumptions used to derive inputs parameters for the PCoD model and interpretations of the model output.

Methods

Tag deployments were identified where an interaction occurred between a ship and a group of killer whales and the duration of the interaction (referred to as ‘during’) and the duration of the time period ‘before’ and ‘after’ the ‘interaction’ were determined (see **Table 1**).

Table 1 Tag Deployments where Killer Whales had an 'Interaction' with a Ship, the Closest Distance between the Group and the Ship and the Duration of the 'before', 'during' and 'after' Ship Interaction Periods Used in the Analysis

Tag	Ship Type	Closest Approach (m)	Before Duration (minutes)	During Duration (minutes)	After Duration (minutes)
oo09_235	tug + barge	400	35	53	35
oo09_244	Cruise Ship	200	20	20	20
oo11_245	tug then cruise ship	100 (cruise ship)	60	126	60
oo11_245	commercial fishing transport ship	100	35	26	35
oo11_267	cruise ship	900	23	36	23
oo11_267	cruise ship	2000	35	20	35

For each tag, there are six response variables, including vocalisation rate (i.e., social sounds and echolocation clicks), depth, breathing rate, heading change, and jerk (i.e., rate of change of acceleration) of the tagged animal. The temporal resolution of the depth, heading change, and jerk data was 200 m, although the data were then grouped into 1-minute means to account for some of the temporal autocorrelation. Mean vocalisation rate and breathing rate were calculated per 1-minute time bin.

Only those variables where a behavioural response had been determined to occur in the original analysis (Table B1 in SMRU 2014 DTAG report) were taken forward for analysis here.

Statistical Analysis

Given the small sample size of this dataset, two analytical approaches were used to increase confidence in the results.

i) Randomisation Tests

In the original D-Tag data in SMRU 2014, a randomisation test was used to determine if the percentage change in mean rate for each of the response variables between the 'before', 'during' and 'after' time periods occurred at above chance levels. This analysis determined whether there was a difference in rate between the time periods, and thus if a behavioural response occurred, but did not determine the duration of the behavioural response.

These same time series data were re-analysed in 5-minute time bins. An increment size of five minutes was selected to match the temporal resolution of the noise overlap model used to predict behavioral responses for the SRKW PCoD model. Randomisation tests were used to determine whether a significant response by the killer whale group occurred in each consecutive 5-minute time bin during and after the large vessel interaction. One 5-minute time period was chosen at random from the 'before' time period, which acted as a control. The same 5-minute period was chosen for each of the six variables within each tag deployment. The control period was compared to each consecutive 5-minute period from the beginning of the 'during' time period to the end of the 'after' time period. As with the previous DTAG analysis (see SMRU 2014), the mean percentage change in rate was calculated between each 5-minute time period and the control 5-minute time period. Data were then randomised between the 'before', 'during', and 'after' time periods and a new percentage change in rate was calculated for each time period comparison.

A random distribution of 10,000 new 'percentage change in rate' values was calculated under the null hypothesis that the observed 'percentage change in rate' was due to chance. The observed 'percentage change in rate' was then compared to the random distribution, and a p-value was calculated by counting the number of random 'percentage changes in rate' that were at least as big as the one observed, and translating this count to a p-value by dividing the count by 10,000. Only observed rates of change with p-values <0.05 were considered significantly different from the random distribution and were treated as an occurrence of a behavioural response. If a behavioural response occurred, the 5-minute period was scored a 1 and if the mean rate of change did not occur at above chance levels then the 5-minute period scored a 0 (no behavioural response).

ii) B-spline Regression Analysis

The second analytical approach estimated the start and end times of the 'disturbed' behaviour by fitting linear B-splines to 1-minute means of the raw data for each response variable. This approach uses an objective method to identify the transition points between undisturbed and disturbed behaviour ('before' to 'during'), and the return to normal behaviour ('during' to 'after'). The analysis proceeds by fitting linear B-splines to segments of data that are connected by knot locations, such that the fitted function is linear over each of the data segments. The knot locations are model parameters that identify the break points in the linear B-spline segments and signify the locations of behavioural transitions. The analysis allowed for discontinuity in the fitted B-spline regression line at the knots, to allow for abrupt changes in behaviour as opposed to a smooth transition between behaviours. The analysis solved for the optimal knot locations by minimising the sums of squared residuals. The knot locations were constrained to sensible limits. The first knot location (i.e, start of the behavioural response) must occur after the 'before' time period has ended, and the second knot location (i.e., end of the behavioural response) must occur after the first knot location by at least 5 minutes. This is because a moderate response is assumed to last at least 5 minutes (i.e., longer than the exposure period).

Results

i) Randomisation Tests

Results from the randomisation test analysis are presented in **Table 2**. The data show if a behavioural response occurred (1) or no behavioural response occurred (0) in each 5-minute time bin from the beginning of the ‘during’ time period to the end of the ‘after’ time period. Given the variability in the dataset, some assumptions had to be made on the start and end point of the behavioural response when 1’s and 0’s are dispersed across the time series.

For the purpose of this study, it was assumed that the longest period of consecutive 1’s represented the moderate behavioural response (i.e., the time period where the strongest response occurred); therefore, the sum of those 5-minute periods is deemed to be the behavioural response duration.

The behavioural response durations for each variable for each tag deployment are presented in **Table 3**. The average behavioural response durations are presented across tags (right column) and across variables (bottom row). For the purpose of the SRKW PCOD model, the average behavioural response duration across all tags and variables was 25 minutes for the randomisation test analysis.

Table 2 The Results (raw data) from the Randomisation Test between each Consecutive 5-minute Time Period in the 'during' and 'after' Time Period for the Ship Interaction and the 5-minute Control Time Period from 'before' the Ship Interaction

Tag	Social sounds	Clicks	Depth	Breathing Rate	Heading Change	Jerk
oo09_235	00000111110111000	00100101111101101	01001110110011111	NA	00100001000010101	01010110111111111
oo09_244	11111111	00000010	00010101	NA	11111111	00001000
oo245_P1 *	0000000000000001101 11111111111111111	00000000001111001011 00011111111110011	NA	NA	00001010111011011100 11000001000011000	10100000000101111001 0011111011111001
oo245_P2	0000100000111	1100100000000	1101001110111	0000010000001	0111001111111	1001001110111
oo267_P1	111111111111 **ψ**	111111111111 **ψ**	0011011111111	0011000001011	1011111111111	0010011101111
oo267_P2	10110000011	10000010100	01110000100	01100000011	00110000000	00000000111

Note: A **0** is where the response was not significantly different from the control period and a **1** is where the response is significantly different from the control period and is found to be a behavioural response. NA – these variables were not significant in the original analysis – Table B1 in SMRU 2014 DTAG report– and thus were not taken forward for further analysis.

***** During this interaction, the animals were observed resting and travelling slowly. They then increased surface activity during the vessel interaction, with lots of tail slaps and breaches, followed by increased vocal activity and socialising after the interaction. The elevated vocalisation rate may be due to socialising; however, these data are still included in the calculation of the average behavioural response duration.

Ψ During this interaction, the vocalisation rate was high in the 'before' period and then the animals went completely silent in the 'during' and 'after' period. Due to the inflated zeros and smaller time bins (5-minute periods), the randomisation tests did not class the % change (100%) as significant (although it was strongly significant in the original analysis where larger time bins were used i.e. '*'before'* & *'during'*, '*during'* & *'after',* and *'before'* & '*after'* time periods). Due to this striking change in vocal behaviour, and it's original classification as a moderate behavioural response, all the 5-minute time periods in the 'during' and 'after' phase have been classified as a 1 (i.e., a behavioural response). It should be noted that the inclusion of this data did not significantly change the final result (see **Table 3**).

Table 3 The Duration of the Behavioural Response for each Variable for each Tag Deployment, as well as the Average Response Duration across each Variable and each Tag, as Calculated using Randomisation Tests

Tag	Social sounds	Clicks	Depth	Breathing Rate	Heading Change	Jerk	Response duration by tag
oo09_235	00000111110111000 response: 5 x 5mins	00100101111101101 response: 5 x 5mins	01001110110011111 response: 5 x 5mins	NA	00100001000010101 response: 1 x 5mins	01010110111111111 response: 9 x 5mins	Average: 5 x 5
oo09_244	11111111 response: 8 x 5mins	00000010 response: 1 x 5mins	00010101 response: 1 x 5mins	NA	11111111 response: 8 x 5mins	00001000 response: 1 x 5mins	Average: 4 x 5
oo245_P1	0000000000000000110 111111111111111111 response: 18 x 5mins	000000000011110010 110001111111111001 1 response: 10 x 5mins	NA	NA	0000101011101101110 011000001000011000 response: 3 x 5mins	1010000000001011110 010011111011111001 response: 5 x 5mins	Average: 9 x 5
oo245_P2	0000100000111 response: 3 x 5mins	1100100000000 response: 2 x 5mins	1101001110111 response: 3 x 5mins	0000010000001 response: 1 x 5mins	0111001111111 response: 7 x 5mins	1001001110111 response: 3 x 5mins	Average: 3 x 5
oo267_P1	111111111111 response: 12 x 5mins **ψ**	111111111111 response: 12 x 5mins **ψ**	0011011111111 response: 8 x 5mins	0011000001011 response: 2 x 5mins	1011111111111 response: 11 x 5mins	0010011101111 response: 4 x 5mins	Average: 8 x 5
oo267_P2	10110000011 response: 2 x 5mins	10000010100 response: 1 x 5mins	01110000100 response: 3 x 5mins	01100000011 response: 2 x 5mins	00110000000 response: 2 x 5mins	00000000111 response: 3 x 5mins	Average: 2 x 5
Response duration by data stream	Average: 8 x 5	Average: 5 x 5	Average: 4 x 5	Average: 2 x 5	Average: 5 x 5	Average: 4 x 5	

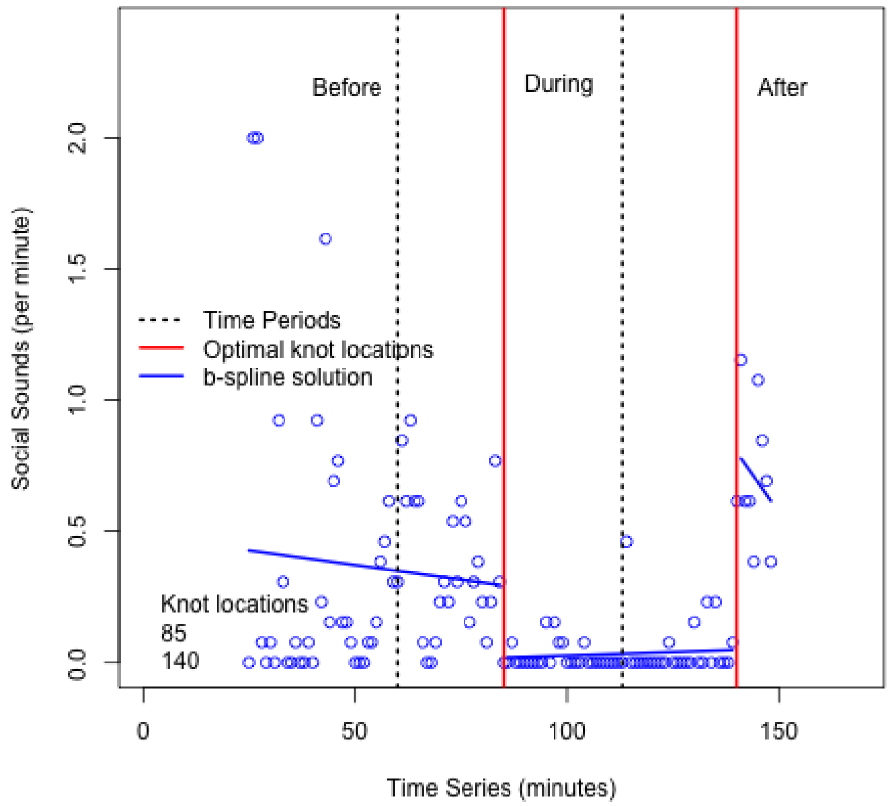
Note: NA – these Variables were not Significant in the Original Analysis – Table B1 in SMRU 2014 – and thus were not Taken Forward for Further Analysis.

Ψ The inclusion of this data did not significantly change the final result. The mean behavioural response duration across all tags and variables is 25 minutes (with this data included) and 21.5 minutes (with these two durations set as 0 minutes). As the smallest temporal resolution of the SRKW PCOD noise overlap model is 5 minutes, the moderate behavioural response duration would have to be rounded up to 25 minutes, which is the mean duration when this data is included.

ii) B-spline regression analysis

The B-spline regression analysis estimated the start and end time of the behavioral response (i.e., the time in minutes between the 2 knots) by fitting B-splines to each variable against the time series data. The predicted line for the linear B-spline regression for social sounds for one tag deployment is shown in **Figure 1**.

Figure 1 The Optimal Knot Locations Fitted within a B-spline Regression Framework to Vocalisation Rate (social sounds) against the Time Series Data ('before', 'during' and 'after' shown by dashed lines) for Tag oo09_235



Note: Knot Locations are shown in red, with absolute time of each knot shown in the bottom left hand corner. B-splines fitted to the data are shown in blue.

The behavioural response durations for each variable for each tag deployment are presented in **Table 4**. The average behavioural response durations are presented across tags (right column) and across variables (bottom row). For the purpose of the SRKW PCOD model, the average behavioural response duration across all tags and variables was 25 minutes.

Table 4 The Behavioural Response Durations for each Variable for each Tag Deployment, and the Average Response Duration across each Variable and each Tag, as Calculated using B-spline Regression Analysis

Tag	Social sounds	Clicks	Depth	Breathing Rate	Heading Change	Jerk	Response duration by tag
oo09_235	K1 = 85 K2 = 140 response: 55 mins	K1 = 69 K2 = 74 response: 5 mins	K1 = 69 K2 = 75 response: 6 mins	NA	K1 = 62 K2 = 77 response: 15 mins	K1 = 70 K2 = 75 response: 5 mins	Average: 17 mins
oo09_244	K1 = 29 K2 = 38 response: 9 mins	K1 = 40 K2 = 53 response: 13 mins	K1 = 40 K2 = 50 response: 10 mins	NA	K1 = 35 K2 = 46 response: 11 mins	K1 = 37 K2 = 45 response: 8 mins	Average: 10 mins
oo245_P1	K1 = 471 K2 = 496 response: 25 mins	K1 = 409 K2 = 480 response: 71 mins	NA	NA	K1 = 479 K2 = 516 response: 37 mins	K1 = 372 K2 = 529 response: 157 mins	Average: 73 mins
oo245_P2	K1 = 570 K2 = 575 response: 5 mins	K1 = 567 K2 = 572 response: 5 mins	K1 = 580 K2 = 603 response: 23 mins	K1 = 580 K2 = 598 response: 18 mins	K1 = 580 K2 = 601 response: 21 mins	K1 = 554 K2 = 602 response: 48 mins	Average: 20 mins
oo267_P1	K1 = 77 K2 = 83 response: 6 mins	K1 = 76 K2 = 101 response: 25 mins	K1 = 89 K2 = 113 response: 24 mins	K1 = 76 K2 = 98 response: 22 mins	K1 = 84 K2 = 120 response: 36 mins	K1 = 92 K2 = 106 response: 14 mins	Average: 21 mins
oo267_P2	K1 = 329 K2 = 356 response: 27mins	K1 = 311 K2 = 361 response: 50 mins	K1 = 329 K2 = 335 response:6 mins	K1 = 325 K2 = 342 response:17 mins	K1 = 320 K2 = 342 response:22 mins	K1 = 325 K2 = 342 response:17 mins	Average: 23 mins
Response duration by data stream	Average: 21 mins	Average: 28 mins	Average: 14 mins	Average: 19 mins	Average: 24 mins	Average: 42 mins	

Note: NA – these variables were not significant in the original analysis – Table B1in SMRU 2014 DTAG report – and thus were not taken forward for further analysis. K1 represents the first knot location (the time [in minutes] at which the behavioural response started) and K2 represents the second knot location (behavioural response ends).

Discussion

The duration of a moderate-severity behavioral response was estimated by re-analysing DTAG data from SMRU 2014 using two analytical approaches. Both approaches had a mean response duration (across all tags and variables) of 25 minutes. Although the two types of analyses were effective at highlighting time periods where a strong change in behavior occurred, both have their limitations. A large amount of variability is inherent in the DTAG data, largely due to the small amount of data available (i.e., six large vessel 'interactions' across four tags) and the relatively short behavioral time periods ('before', 'during', and 'after').

In the B-spline regression analysis, the large variance in measures of behavior meant that for many variables there were no differences in the regression slopes between the behavioral periods. The location of the first knot (i.e., start time of the behavioral response) appeared to be consistent across all the data and appeared to match the change in pattern evident in the summary plots of raw data; however, the location of the second knot (i.e., end time of the behavioral response) did not always reflect this pattern. For these cases, the time between the two knot locations was short (Table 4) and perhaps do not reflect the true behavioral response duration. The regression approach forced the B-spline linear function to have two knot locations. Although this assumption worked well for some of the tag data (**Figure 1**), this was not the case for all the variables. In particular, there is no certainty that the time series of behavior data has actually returned to 'undisturbed' or baseline behavior. Likewise, it is assumed that the behavior observed prior to the arrival of the large vessel is correctly classified as 'undisturbed'; however, a larger dataset would be required to identify 'undisturbed' to 'disturbed' behavior with a greater degree of confidence.

Variability in the data also led to the randomisation test analyses revealing a dispersion of 1 (behavioral response) and 0 (no behavioral response) 5-minute time bins across the behavioral time periods (**Table 2**), which required an assumption that the longest period of consecutive 1's represented the time period where the strongest behavioural response occurred. Although 5-minutes was chosen to match the temporal resolution of the SRKW noise overlap model, 5-minutes may not have been a suitable time window to capture a good estimate of 'behaviour'.

Conclusion

Absolute response durations for the different variables are expected to show substantial variation between tags or vessel 'interactions' due to differences between individuals, behavioural context during sound exposure, exposure history, and sound source type. A level of disparity was also observed in the response durations for individual variables from the two analysis types (**Table 3** and **4**). Nonetheless, the overall mean behavioral response duration of 25 minutes was calculated for both approaches. The SRKW PCoD model (SRKW Population Consequence of Disturbance Model TR) requires one estimated value for behavioral response duration (across all variables and all tags). Given that these data are the only empirical information available, there is some reassurance that both approaches showed strong agreement in the overall mean duration of 25 minutes.

Recommendation for Inclusion in PCOD Model

For the SRKW PCoD model, a duration of 5 minutes is recommended for a low-severity behavioural response and 25 minutes is recommended for a response of moderate-severity.

The sensitivity of the PCoD model output to these parameters can be analysed by reducing or increasing the response duration.

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APPENDIX 14-D

Changes in Polychlorinated Biphenyl (PCB) Exposures of Southern Resident Killer Whales Associated with RBT2 Disposal at Sea

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ROBERTS BANK TERMINAL 2
TECHNICAL REPORT

Changes in Polychlorinated Biphenyl (PCB) Exposures of Southern Resident Killer Whales Associated with RBT2 Disposal at Sea

Prepared for:
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Technical Report / Technical Data Report Disclaimer

The Canadian Environmental Assessment Agency determined the scope of the proposed Roberts Bank Terminal 2 Project (RBT2 or the Project) and the scope of the assessment in the Final Environmental Impact Statement Guidelines (EISG) issued January 7, 2014. The scope of the Project includes the project components and physical activities to be considered in the environmental assessment. The scope of the assessment includes the factors to be considered and the scope of those factors. The Environmental Impact Statement (EIS) has been prepared in accordance with the scope of the Project and the scope of the assessment specified in the EISG. For each component of the natural or human environment considered in the EIS, the geographic scope of the assessment depends on the extent of potential effects.

At the time supporting technical studies were initiated in 2011, with the objective of ensuring adequate information would be available to inform the environmental assessment of the Project, neither the scope of the Project nor the scope of the assessment had been determined.

Therefore, the scope of supporting studies may include physical activities that are not included in the scope of the Project as determined by the Agency. Similarly, the scope of supporting studies may also include spatial areas that are not expected to be affected by the Project.

This out-of-scope information is included in the Technical Report (TR)/Technical Data Report (TDR) for each study, but may not be considered in the assessment of potential effects of the Project unless relevant for understanding the context of those effects or to assessing potential cumulative effects.

EXECUTIVE SUMMARY

The Roberts Bank Terminal 2 Project (RBT2 or the Project) is a proposed new three-berth marine terminal at Roberts Bank in Delta, B.C. The Project is part of PMV's Container Capacity Improvement Program, a long-term strategy to deliver projects to meet anticipated growth in demand for container capacity to 2030. This technical report describes the results of PCB food web bioaccumulation modelling in southern resident killer whale (*Orcinus orca*; SRKW) critical habitat to assess the risk of increased exposures to PCBs due to sediment re-suspension and disposal at sea (DAS) as a result of the proposed construction of the RBT2 Project.

Resident killer whales (*Orcinus orca*) inhabiting coastal waters of B.C. are at the top of the marine food web and are long-lived. Thus, they are at risk of accumulating high concentrations of polychlorinated biphenyls (PCBs) and other persistent bioaccumulative and toxic contaminants. The southern population of resident killer whales (SRKW) are listed as Endangered under the Canadian *Species at Risk Act* (*SARA*). The Resident Killer Whale Recovery Strategy has identified persistent organic pollutant exposures as a conservation threat to population recovery (Fisheries and Oceans Canada 2011). PCB exposures are relevant to SRKW population recovery because adverse health effects in these top predators have been largely attributed to this class of chemicals (Ross et al. 2000). The Recovery Strategy identifies the southern Strait of Georgia, including Roberts Bank, as SRKW critical habitat. This report assesses the ecological risk to SRKWs associated with potential changes in PCB exposure as a possible result of the construction of the RBT2 Project.

Sediments in resident killer whale critical habitat contain PCBs that can enter and biomagnify through marine food webs to high trophic level killer whales. Modelling was performed to predict changes in SRKW PCB exposure potential due to marine discharge of sediment in the candidate RBT2 disposal at sea site. This site- and project-specific evaluation builds on an earlier modelling assessment that examined the issue of dredging and DAS in the context of killer whale health (Alava et al. 2012, Lachmuth et al. 2010). The model predicts accumulated concentrations in SRKW after lifetime exposure, based on the distribution of PCBs in sediments, the water column, and biota.

The food-web model predicts that, under existing conditions, the uptake from surface sediments and trophic transfer could result in PCB concentrations in SRKW tissues that exceed conservative health effect thresholds for PCBs in marine mammals.The model further predicts that incremental changes in the exposures of SRKWs to PCBs will be very low to negligible for Project-related construction activities when compared to existing conditions. An increase in PCB SRKW tissue concentratons of 0.00003% for males and 0.00002% for females is predicted for an affected seabed area from at-sea disposal during Project construction that is predicted (from dispersion modelling) to result in sediment deposition to a thickness of ≥ 0.1 mm over an area of 196.7 km^2. An increase of 0.000003% for both males and females is predicted based on the predicted area of seabed with deposition to a thickness ≥ 1.0 mm over an area of 21.6 km^2.

Based on the weighted-area approach for RBT2 pre- and post- disposal scenarios, the increase in risk in terms of predicted PCB concentrations for both male and female SRKW is sufficiently low that it is deemed to be negligible. The predicted increase in exposure is very small in comparison with both the known variability of tissue concentrations for killer whale individuals (male or female) and the precision with which changes in exposure can be predicted or measured.

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1.0 INTRODUCTION

This section provides Project background information, key study objectives, and an overview of resident killer whale critical habitat and polychlorinated biphenyls (PCBs).

1.1 PROJECT BACKGROUND

The Roberts Bank Terminal 2 Project (RBT2 or Project) is a proposed new three-berth marine terminal at Roberts Bank in Delta, B.C. that could provide 2.4 million TEUs (twenty-foot equivalent units) of additional container capacity annually. The Project is part of Port Metro Vancouver's (PMV) Container Capacity Improvement Program, a long-term strategy to deliver projects to meet anticipated growth in demand for container capacity to 2030.

Port Metro Vancouver has retained Hemmera to undertake environmental studies in support of an effects assessment for the Project. This technical report describes the results of PCB food web bioaccumulation modelling in southern resident killer whale (*Orcinus orca*; SRKW) critical habitat to assess the risk of increased exposures to PCBs due to sediment re-suspension and disposal at sea (DAS) as a result of the proposed construction of the RBT2 Project. This site- and project-specific evaluation builds on an earlier modelling assessment that examined the issue of dredging and DAS in the context of killer whale health (Alava et al. 2012, Lachmuth et al. 2010).

1.2 STUDY OBJECTIVES

This study characterises the PCB-related toxicological risk to killer whales associated with sediment disposal and disturbance activities for the RBT2 Project. Some of the sediments proposed for use in the Project (i.e., a portion of Fraser River navigational channel maintenance dredgate) or that occur in the tug basin dredge prism exhibit a total PCB concentration greater than 12 to 200 pg/g (dry weight basis: dw). This sediment concentration range (based in turn on the range available estimates of PCB concentrations in SRKW tissue that are thought to result in reproductive impairment) has been nominated as a sediment threshold beyond which PCB-related risks to SRKW cannot be ruled out without further study (Lachmuth et al. 2010). The overall objective of this technical report is to ensure that sufficient information is available to inform an effects assessment for the Project. The risk assessment is based on a site- and Project-specific application of the PCB bioaccumulation model that was previously developed for the assessment of risks associated with ocean disposal in resident killer whale critical habitat in general (Lachmuth et al. 2010, Alava 2011, Alava et al. 2012). Using site-specific environmental variables, available empirical data, and knowledge of killer whale ecology, the Alava et al. (2012) model was adapted to estimate PCB trophic transfers from suspended or bed sediments to SRKW.

Specifically, this report provides:

1. A site-specific application of an existing toxicokinetic/trophodynamic PCB food web bioaccumulation model for SRKW critical habitat, and assessment of pre- and post-disposal scenarios at the proposed RBT2 DAS sites/affected seabed; and
2. A comparison of model-predicted PCB concentrations in SRKW with existing toxicity thresholds for marine mammals.

1.3 RESIDENT KILLER WHALES AND CRITICAL HABITAT

Three ecotypes of killer whales (i.e., genetically distinct and socially isolated) inhabit coastal waters of B.C. These ecotypes are termed resident, transient, and offshore (Ford et al. 1998, 2000). Two populations of resident killer whales have been identified: northern residents (NRKW), often found in the waters off mid-Vancouver Island (B.C.) north to southeastern Alaska; and southern residents, which frequent the waters off southeast Vancouver Island and neighbouring Washington State (**Figure 1-1**) (Ford et al. 1998, 2010). Under the federal *Species at Risk Act* (*SARA*), SRKWs are listed as endangered and NRKWs are listed as threatened (Government of Canada 2010a, b). Major threats/stressors to the SRKW population include sub-optimal prey availability, contaminant (especially PCB) exposures, vessels, and noise (NOAA 2014).

Critical habitat, defined under *SARA* Section 2 as “the habitat that is necessary for the survival or recovery of a listed wildlife species and that is identified as the species’ critical habitat in a Recovery Strategy or in an action plan for the species” has been identified for both resident killer whale populations (**Figure 1-2**) (Government of Canada 2010a, b). For SRKW critical habitat in B.C. waters, this includes Haro Strait, Boundary Pass and adjoining areas in the Strait of Georgia, and Juan de Fuca Strait (**Figure 1-2**) (Fisheries and Oceans Canada 2008; Ford 2006, 2010; Krahn et al. 2007). Due to the importance of critical habitat for SRKW life functions of foraging, mating, socialising, and resting, there are implications for the disposal of potentially contaminated materials into these areas (Lachmuth et al. 2010). Under *SARA* Section 58, critical habitat is legally protected from destruction, and advice supported by existing science is needed to justify management decisions that could affect resident killer whale critical habitat.

1.4 PCBS IN FRASER RIVER AND ROBERTS BANK SEDIMENTS OF RELEVANCE TO RBT2

The RBT2 Project will require the transfer of dredgate from the dredge basin or from stockpiled Fraser River maintenance dredgate to the new terminal footprint, as fill, with approximately 625,000 m^3 of fine textured sediment that will not settle out and will be discharged to the adjacent marine environment (Hemmera 2014a). The Project will also entail the disposal at sea of an additional 128,000 m^3 of fine sediment that will accumulate at the base of the dredge basin dredge prism as a result of upward expulsion during vibro-replacement seabed densification, as well as 164,000 m^3 of dredgate from the expandion of the tug basin. The total estimated volume of sediment requiring disposal at sea is 917,000 m^3.

As discussed in Hemmera (2014a), the vast majority of the total volume of discharged sediments will have very low to negligible (non-measurable) PCB concentrations. In particular, dredged material from the dredge basin contains PCBs in the top ca. 0.5 to 2 m of seabed (depending on specific location) in the range of non-detectable levels to 45 pg/g dw total PCBs, while deeper sediments are expected to contain negligible PCBs since they were deposited prior to the intitiaton of PCB production. The fines resulting from vibro-replacement will also have negligible PCB concentrations, since they will originate from the sediments at depths below the base of the dredge prism. Cumulatively, this material, containing negligible concentrations of PCB, will account for an estimated 82% of the mass of sediment that will require disposal at sea.

Sediment PCB concentrations in the surficial sediments of the tug basin dredge prism were as high as 655 pg/g dw, while that portion of the tug basin dredge prism deeper than approximately 0.75 m below the seabed is expected to have negligible PCB concentrations based on the history of sediment deposition.

Finally, maintenance dredgate from the navigational channel of the lower Fraser River that will be beneficially reused to construct the RBT2 terminal was observed to have bulk sediment concentrations of total PCBs in the range of 3.6 to 246 pg/g dw. This material will comprise the vast majority of terminal fill (8,300,000 m^3). It is expected that the vast majority of Fraser River dredgate (>99%) will be retained within the terminal footprint owing to its coarse sandy nature; however, it cannot be precluded that a small amount of fines in the fill material will remain in suspension and be included as part of the material to be disposed at sea. Furthermore, the finer textured fractions (silts and clays) within the bulk sediment that are less likely to settle out contain substantially higher PCB concentrations than the bulk sediments in general (Hemmera 2014a).

As discussed in Hemmera (2014a), the existing PCB concentrations in the surficial sediments of the Fraser River delta foreslope at Roberts Bank in the vicinity of the candidate DAS sites were observed to be in the range from less than detection fo 20 pg/g dw.

Figure 1-1 SRKW Sightings and Encounters in British Columbia and Washington State waters (Ford and Ellis 2006)

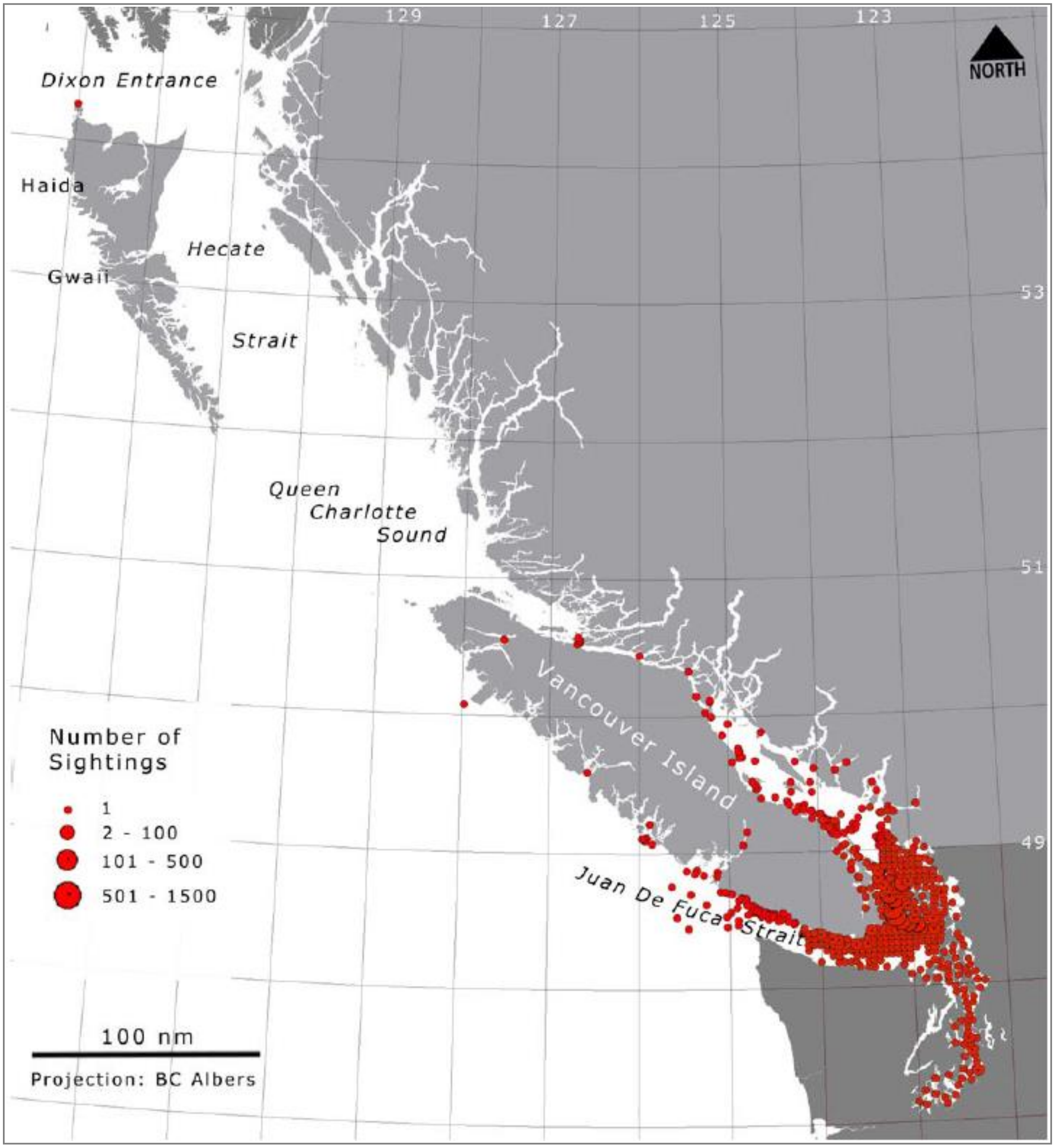
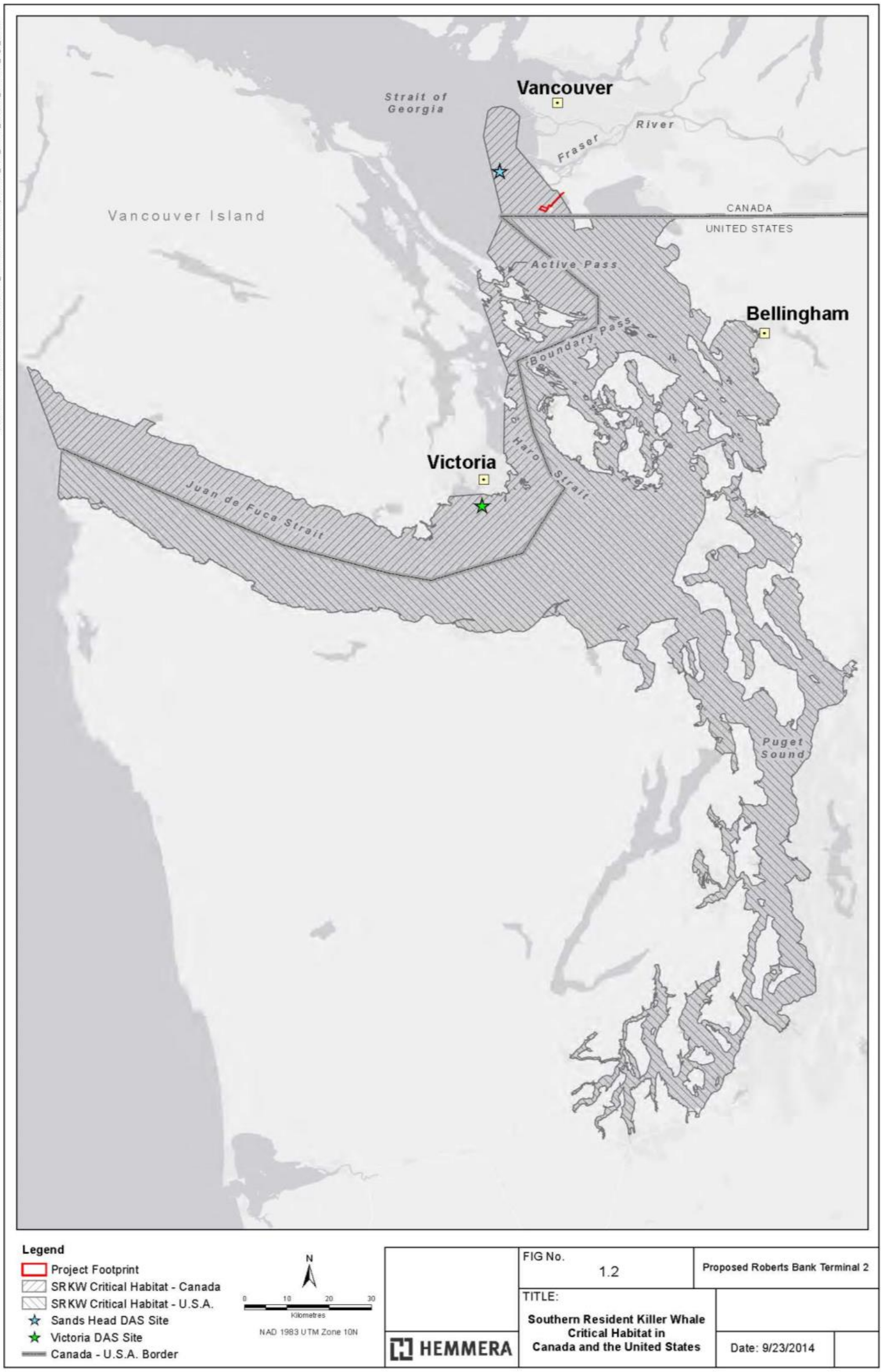


Figure 1-2 SRKW Critical Habitat in Canada and the United States



Note: The hatched area in the U.S. waters depicts the approximate critical habitat under the *U.S. Endangered Species Act*.

1.5 PCBs and Adverse Effects on Killer Whales

While the regulatory bans on PCB production and use worldwide have resulted in declining PCB concentrations in many ecosystems, high concentrations continue to be detected in high trophic level species, including resident killer whales and harbour seals. PCBs are persistent, toxic, bioaccumulative, and ubiquitous in the environment. PCBs are characterised by their high chemical stability, low water solubility and low volatility. Soils and sediments tend to act as environmental reservoirs of PCBs. Major factors that influence sediment dispersal and sedimentation accumulation rates – and thus the fate of sediment-associated contaminants - include natural processes such as tidal currents, wind-generated waves and currents, and gravity flows, and human influences such as dredging and marine discharges (Hill et al. 2008).

Exposures to PCBs have been linked with adverse health effects in marine mammals. According to Ross et al. (2000), adult NRKW and SRKW range among the most PCB-contaminated marine mammals in the world, with total PCB concentrations in tissues ranging from 9,300 to 146,000 $\mu g \cdot kg^{-1}$ lipid weight. Detected PCB concentrations in resident killer whales exceed thresholds for the onset of adverse health effects determined for other marine mammals. These threshold concentrations range from 10,000 to 77,000 $\mu g \cdot kg^{-1}$ PCB in blubber or liver (Hall et al. 2006, Kannan et al*.* 2000, Reijnders 1986, Ross et al*.* 1996).

PCB concentrations in killer whales are highly variable and are influenced by age, sex, reproductive status, and birth order (Ross et al. 2000, Ylitalo et al*.* 2001). Newborn calves have very low contaminant loads, but this rapidly changes during transfer of contaminant load via lipid rich milk. The contaminant load is especially high for first-born calves (Ylitalo et al*.* 2001). One-year-old killer whale calves tend to be the most contaminated members of the population, and as killer whales grow and switch to a fish diet, their PCB concentration is diluted (Ylitalo et al*.* 2001). PCB concentrations in adult male killer whales continue to increase, whereas adult females transfer (offload) their contaminant burden to their offspring (Ylitalo et al*.* 2001).

1.6 Strait of Georgia Food Webs

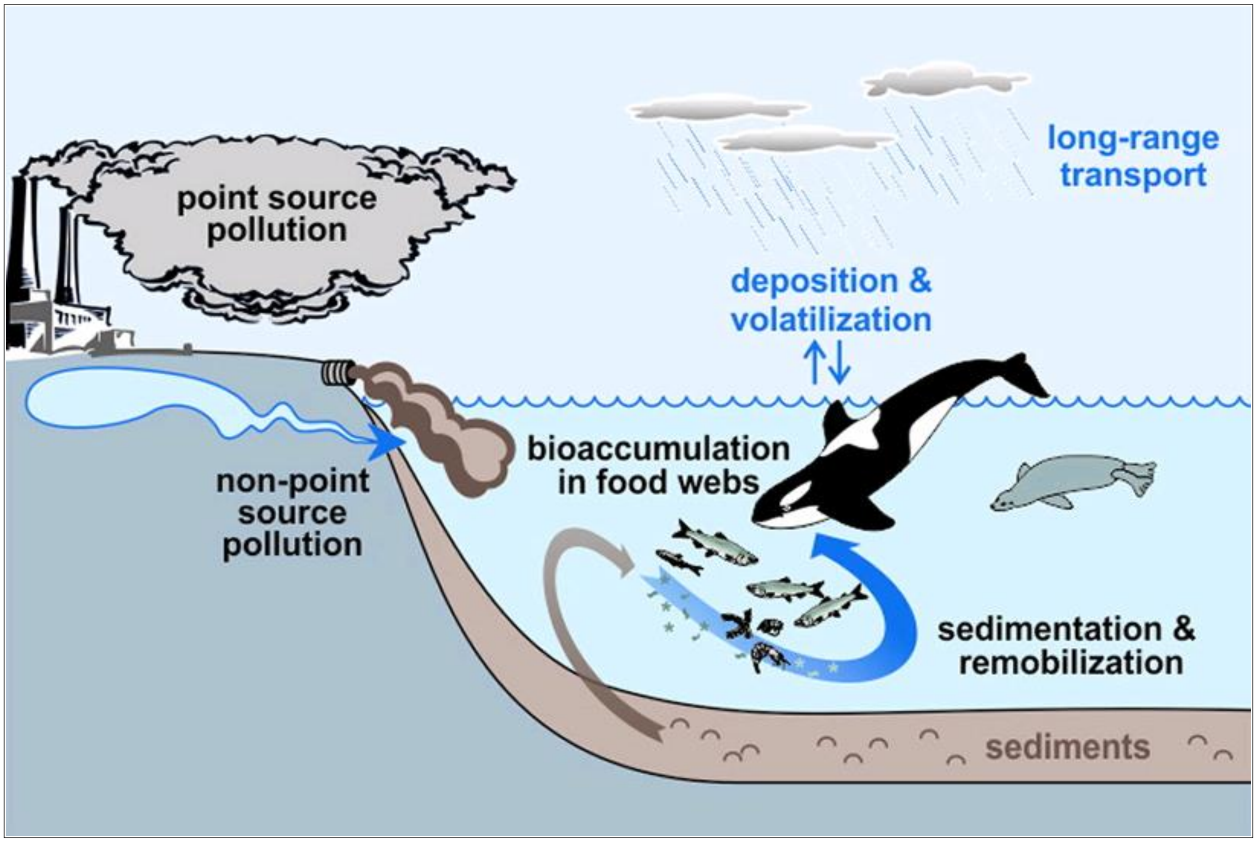
The Strait of Georgia is a semi-enclosed basin (approximately 6,900 km^2) situated between Vancouver Island and southern mainland B.C. As well as being a rich and diverse ecosystem, the Strait of Georgia supports a spectrum of human activities including marine vessel traffic, shipping and port-related industry, fishing, pulp and paper mills, and highly urbanised and densely populated areas such as Metro Vancouver. Sedimentation accumulation rates range from less than 1 g cm^{-2} yr^{-1} in the northern Strait of Georgia to much greater than 3 g cm^{-2} yr^{-1} (relatively high rates) near the mouth of the Fraser River (Johannessen et al. 2008b). The marine sediments in this region provide a record of historical contamination, including:

- Trace elements such as lead and mercury (Johannessen et al*.* 2005, Long et al. 2005, Macdonald et al. 1991);

- Industrial by-products such as dioxins and furans (Long et al. 2005, Macdonald et al. 1992);
- PCBs, and the flame retardant polybrominated diphenyl ethers (PBDEs) (Ikonomou et al*.* 2002, Rayne and Ikonomou 2002, Johannessen et al*.* 2008a, Long et al. 2005);
- Polycyclic aromatic hydrocarbons (Long et al*.* 2005, Yunker and Macdonald 2003); and
- Industrial detergents (Shang et al*.* 1999).

Long-lived marine mammals including killer whales can accumulate very high concentrations of PCBs through their diet. Their primary prey item is the high lipid/high energy chinook salmon (*Oncorhynchus tshawytscha*), comprising approximately 71% of their 97% salmonid diet (Ford et al. 2010, Ford 2006). Because PCBs and related compounds are persistent and bioaccumulative, their behaviour in aquatic systems is often described in terms of food web structure. **Figure 1-3** depicts the fate of persistent organic contaminants in the environment and biota. Although PCBs are declining in the marine environment of the SRKW critical habitat adjacent to B.C. and reaching steady state in biota over time, as observed in harbour seal pups (Ross et al. 2013), they continue to biomagnify in regional food webs (Cullon et al. 2005, 2012; Alava et al. 2012), and therefore, remain a major toxicological concern for killer whales.

Figure 1-3 Sources of Persistent Organic Pollutants, including PCBs, to Marine Mammals (adapted from Alava et al. 2012)



1.7 DISPOSAL AT SEA

Environment Canada (EC) oversees ocean disposal activities under the terms of the *Canadian Environmental Protection Act* (CEPA). There are fifteen DAS sites in coastal B.C. in addition to a site at Roberts Bank, which has been permitted as a DAS site for various previous port expansion projects on Roberts Bank. Within SRKW critical habitat are two routinely used DAS sites: Sand Heads and Victoria (Environment Canada 2006) (**Figure 1-2**). The Roberts Bank site is also located in SRKW critical habitat; however, it has been used only infrequently. Additional disposal sites are located outside the boundaries of SRKW critical habitat, but within their general habitat range (e.g., Point Grey DAS site; Environment Canada 2006).

Tidal currents at the Roberts Bank candidate DAS site cause a predominant northward drift along the Fraser River delta fore slope. Mean flood tide velocities exceed 1.2 $m \cdot s^{-1}$ (Meulé 2005), so deposited material will move in a predominantly northward direction with the tidal current (Phil Hill, Geological Survey of Canada, personal communication, 2010, in Lachmuth et al. 2010). During peak current velocities, re-suspension of fine sand from the sea floor occurs to depths as great as 90 m (Kostaschuk et al. 1995). Substantial sediment accumulation occurs at the river mouth, which can exceed 1 $m \cdot yr^{-1}$ (Hill 2012), resulting in episodic subsea slope failures (McKenna et al. 1992).

The sediments at the Roberts Bank site are sandy, and the site is located in a sand wave field that has a high sedimentation rate, that experiences little if any bioturbation (Phil Hill, Geological Survey of Canada, personal communication, 2010, in Lachmuth et al. 2010). The Roberts Bank DAS site is located below minus 40 m CD and is not used routinely or considered available other than for port development at Roberts Bank (Sean Standing, Environment Canada, personal communication, 2010, in Lachmuth et al. 2010). The most recent disposal at the site occurred in 2008, with deposition of 118,663 m^3 of material (Sean Standing, Environment Canada, personal communication, 2010, in Lachmuth et al. 2010).

2.0 MODEL INPUTS AND STUDY AREA

The following section describes model inputs and the model's spatial and temporal scope.

2.1 GENERAL CRITERION FOR MODEL APPLICATION TO KILLER WHALES

The model presented in this study was used to predict PCB concentrations in chinook salmon and SRKW for pre- and post-Project DAS scenarios (see **Section 2.13**), using empirical measurements of sediment PCB concentrations prorated for the percent time SRKW are foraging in their B.C. and WA critical habitat. Extensive descriptions of the existing food-web model, particulars of development and optimisation, and relevant assumptions are found in Alava et al. (2012) and Lachmuth et al. (2010), and therefore, not described in detail here.

2.2 PHYSICO-CHEMICAL PROPERTIES OF PCBS

Like most food-web-based, trophic transfer models, the predictions of contaminant concentrations in one biotic or abiotic compartment from another assumes equilibrium partitioning that is largely driven by the intrinsic physico-chemical properties of the substances of interest. PCBs comprise a complex mixture made of a subset of 209 possible individual molecules (or "congeners") that are unique by virtue of the number of chlorines attached to either of the two phenyl rings and their specific position relative to the carbon-carbon bond between the two rings. The degree of chlorination, from a possible one to ten chlorines, is a useful proxy for physico-chemical properties such as volatility, tendency to partition into lipids, and other hydrophobic phases [as described by the octanol-water partition co-efficient (K_{OW}) or organic carbon-water partition co-efficient (K_{OC})], tendency to partition lipids into air [(as described by the octanol-air partition coefficient, K_{OA})], or tendency to move from water in the dissolved phase to air (as described by the Henry's Law constant). Higher chlorinated PCB congeners are less volatile, less water soluble, more lipophilic, and generally more persistent than lower chlorinated congeners.

A summary of PCB congener octanol-water (Log K_{OW}) and octanol-air (Log K_{OA}) partition coefficients used in the model areas is provided in **Appendix A: Table A-1**. The tables contain the freshwater-based K_{OW} at the mean ambient water temperature of the area of interest. These were used to calculate the saltwater-based K_{OW} values based on the approach of Xie et al. (1997), and further used to determine the PCB distribution between fish and water in the areas of interest. Freshwater-based K_{OW} values at 37.5°C were used to describe partitioning between lipids and aqueous media (e.g., urine) in killer whales. Also included in the table are K_{OA} values corrected to 37.5°C (killer whale body temperature), used in the calculation of PCB transfer during killer whale respiration.

2.3 PCB INPUTS TO RESIDENT KILLER WHALE HABITAT

Of 209 theoretically possible PCB congeners (ATSDR 2000), 136 have been detected in killer whales in B.C. (Ross et al. 2000) – although the major portion of marine mammal PCB body burdens is attributable to a much smaller subset of these. Properties of individual congeners vary, with different distributions in the environment, different levels of toxicity, and environmental persistence half-lives ranging from a few years to a hundred years. PCBs can enter the Strait of Georgia via atmospheric deposition, urban runoff, sewage outfalls, ground water, watersheds such as the Fraser River, and smaller tributaries.

2.3.1 Sediment-Associated PCBs in SRKW Critical Habitat

It is important to capture the distribution of PCB congeners in the environment in the model as these contaminants are widespread in the Strait of Georgia (**Figure 2-1**). Empirical studies have detected a wide range of congeners in resident killer whale habitat and biota; however, the model includes only those congeners with adequate data for the study area (see **Appendix A: Table A-1**). After the model calculates concentrations of all included congeners, a total PCB (ΣPCB) concentration is calculated consisting of the sum of the concentrations of all congeners included in the model.

2.3.2 Sediment-Associated PCBs in the Roberts Bank Candidate DAS Area Following Disposal at Sea

The characteristics of the dredge materials that will be disposed at sea during the construction of the Project are described in detail in Hemmera 2014a. The marine environmental fate of sediment in the proposed marine discharge is discussed in detail in Tetra Tech EBA 2014.

Table 2-1 shows the predicted concentrations of individual PCB congeners in surface sediment within the candidate DAS area following the marine discharge of poorly settleable fines, for those PCB congeners that were detected using high resolution analytical techniques (modification of EPA method 1668a; USEPA 2008) in sediment samples from the dredge basin, tug basin, intermediate transfer pit (ITP), and lower Fraser River navigational channel. The sediment concentrations for each PCB congener were derived based on the following assumptions:

- **Congeners of Interest**: Those sediment sample results for which fewer than ten congeners were detected were excluded from further consideration. For the remaining samples, non-detected results were assumed to have negligible contribution to the ΣPCB concentration of individual sediment samples, given the very low sample detection limits achieved for individual PCB congeners (detection limits for individual samples were in the range of 0.6 to 3 pg/g dw), and were assigned a value of zero. The concentrations of detected congeners for a sample were then normalised to the ΣPCB concentration of that sample (unit-normalised). Congeners were retained for inclusion in the model if they comprised 3% or more on average of the ΣPCB concentration across all non-excluded samples. The average estimated concentrations in sediment of an additional 29 congeners were retained for modelling purposes since these were included in the original model used to predict sediment thresholds for killer whale protection in critical habitat in general.

The one exception to the above-described congener selection scheme is that PCB congener 11 (3,3'-DiCB) was not included in the site-specific model, in spite of the fact that it comprises an estimated 6.5% on average of the ΣPCB concentrations in sediments that will be discharged to the Roberts Bank DAS site. This congener was not included because it was not among the suite of congeners considered in the original Alava et al. (2012) model, and the relevant congener-specific physico-chemical properties were not immediately available. Furthermore, PCB 11 is not expected to be biaccumulated appreciably by marine mammals or other mammalian and avian species since it is highly amenable to metabolic modification and excretion.

For congeners that overlap in the chromatographic separation, the reported concentration was assigned to the single congener that is routinely reported to exhibit the highest concentration in PCB technical fluids such as Aroclors, and in contemporary abiotic environmental samples (e.g., soil, sediment). In a few instances where more than one co-eluting congener is considered to credibly occur in environmental mixtures, the concentration was split according to their expected relative concentration.

The PCB congeners that contribute disproportionately to the ΣPCB concentrations in the study sediment samples include especially PCB 138 (6.5% of ΣPCB), PCB 118 (5.8% of ΣPCB), PCB 110 (5.2% of ΣPCB) as well as lower chlorinated (dichlorinated to tetrachlorinated) PCB congeners 11, 28, 31, 66, and 70/74 (6.5%, 9.2%, 6.9%, 5.1% and 10.4% of ΣPCB, respectively).

- To arrive at a specific dry weight concentration in sediments within the Roberts Bank candidate DAS area following marine discharge, it was further assumed that the particulars of dredgeate loading to the dyked containment cells of the terminal footprint will result in retention of the coarser sandy fraction (>74 µm effective particle diameter) and the marine discharge of suspended sediments exclusively in the silt and clay size range (100% less than 74 µm, or 100% fines).

  Expectations regarding the relationship between ΣPCB concentrations in contemporary era sediments, expressed on a dry weight (dw) basis, and the percent fines content are discussed in Hemmera 2014a. The observed statistically significant relationship (least-squares best fit) between ΣPCB concentrations in candidate dredgeate is as follows:

$$\mathbf{Log_{10}[\Sigma PCBs(pg/g\ dw)] = 0.99 \times Log_{10}[\%fines] + 0.729} \qquad \textbf{[1]}$$

  Based on this relationship, a dredgeate sample comprised of 100% fines (silt plus clay fraction) is expected to have a ΣPCB concentration of 510 pg/g dw.

  Based on the 95% upper confidence limit around the PCB – fines relationship, sediments discharged to the Roberts Bank candidate DAS site were assumed to exhibit a ΣPCB concentration of 685 pg/g dw. The concentrations of 39 congeners already captured in the existing food-web transfer model for the SRKW critical habitat in Canada (Alava et al. 2012), which are a reasonable match for the dominant congeners in sediment samples representative of discharged dredgeate, were estimated by multiplying the ΣPCB concentration in sediment by the average proportional contribution of each congener to the total. Based on this approach, the sum of the sediment concentrations for congeners listed in **Table 2-1** was 593 pg/g dw. This equates to 87% of the sediment, in good agreement with the 95% upper confidence limit estimate of 685 pg/g dw ΣPCB when including all reported congeners, for a sediment with 100% fines. The summed concentration for the 39 modelled congeners (**Table 2-1**) was also higher than the central tendency estimate for a sediment composition of 100% fines (**Equation 1**).

  Based on observed relationships between percent fines and total organic carbon content (TOC) in surficial samples from the proposed dredge basin, tug basin, and ITP, as well as the lower Fraser River navigational channel, the TOC of the discharged dredgeate (100% fines) was estimated to be 2.88%.

Figure 2-1 Contour Plots of Total PCBs (ΣPCB) and PCB Congeners 118 and 138 in Surficial Sediments in the Strait of Georgia (from Grant et al. 2011)

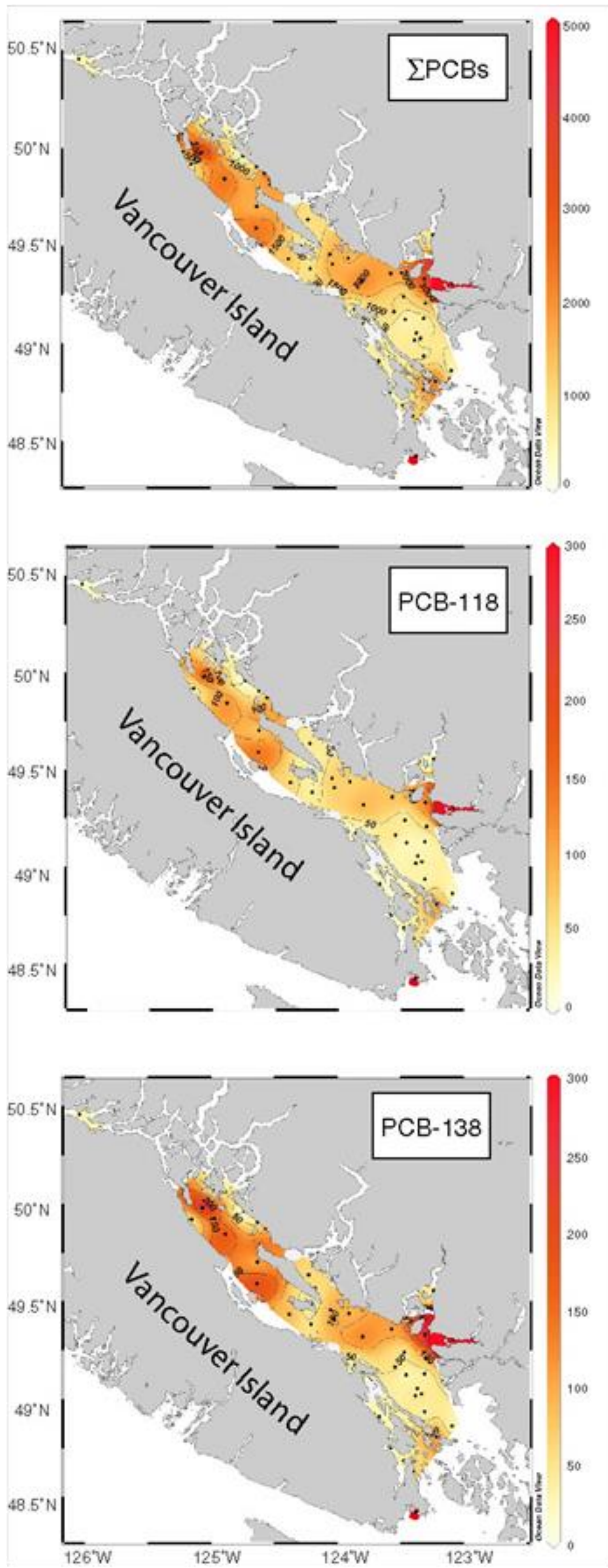


Table 2-1 Predicated Concentrations of PCB Congeners for Sediment and Water in the RBT2 DAS Site Following Discharge of Dredgeate

PCB Congeners/CAS #	Sediment Concentration (ng/kg dw)	Total Water Concentration (ng/L)
PCB 8	1.73	8.75E-05
PCB 18	6.17	2.45E-04
PCB 28	63.0	1.60E-03
PCB 31	47.2	1.04E-03
PCB 33	9.14	2.35E-04
PCB 44	29.5	6.10E-04
PCB 49	8.61	1.51E-04
PCB 52	19.1	3.50E-04
PCB 56	9.96	1.58E-04
PCB 60	3.07	4.31E-05
PCB 66	35.0	5.71E-04
PCB 70	71.4	1.04E-03
PCB 74	6.74	9.73E-05
PCB 87	14.6	1.56E-04
PCB 95	15.1	2.28E-04
PCB 99	12.1	1.26E-04
PCB 101	25.3	2.82E-04
PCB 105	11.5	6.93E-05
PCB 110	35.3	3.93E-04
PCB 118	39.4	3.11E-04
PCB 128	1.32	8.13E-06
PCB 132	3.08	2.63E-05
PCB 138	44.4	1.67E-04
PCB 141	1.29	8.33E-06
PCB 149	19.9	1.53E-04
PCB 151	4.37	3.46E-05
PCB 153	31.7	1.73E-04
PCB 156	0.99	4.85E-06
PCB 158	0.61	3.53E-06
PCB 170	2.36	9.12E-06
PCB 174	1.81	8.21E-06
PCB 177	1.12	5.20E-06
PCB 180	9.47	3.64E-05
PCB 183	0.97	3.95E-06
PCB 187	4.68	1.97E-05
PCB 194	0.68	1.27E-06
PCB 195	0.07	1.97E-07
PCB 201	0.01	1.74E-08
PCB 203	0.35	8.46E-07

Finally, the area of seabed within critical habitat affected by construction-related marine discharge was estimated from predictions provided through dispersion modelling, as discussed in Tetra Tech EBA 2014. In particular, two scenarios were modelled based on the predicted areal extent of re-deposition of sediments with assumed PCB concentration and TOC concentrations as discussed above:

1. The seabed area predicted to exhibit accumulations of ≥1.0 mm was estimated to be 21.6 km^2. A Project-related accumulation of 1.0 mm is estimated to be approximately 10% of the natural annual sedimentation rate in the candidate DAS site, (estimated ~10 mm/y).
2. The seabed area predicted to exhibit accumulations of ≥0.1 mm was estimated to be 196.7 km^2. A Project-related accumulation of 0.1 mm is estimated to be approximately 1% of the natural annual sedimentation rate in the candidate DAS site (estimated ~10 mm/y).

Table 2-1 also shows PCB congener water concentrations, estimated using the partitioning relationship between sediment and water concentrations (i.e., based on the sediment-water partition coefficient, K_{SW} = Cs/Cw; where Cs is PCB concentrations in sediment and Cw is PCB concentrations in water), and assuming equilibrium partitioning. This probably represents a conservative over-prediction of waterborne PCB concentrations, especially based on partitioning from older stores of PCBs in deeper sediments into recent and transient water masses.

Killer whale respiration provides a route for uptake and elimination of PCBs, so airborne PCB congener concentrations were incorporated in the food web models. This is likely a minor route of exposure to killers whales relative to uptake in food. Air concentrations of ΣPCBs were obtained from the Saturna Island station to represent air concentration (mean = 9.3×10^{-6} $ng{\cdot}L^{-1}$) in critical habitats within the Strait of Georgia, and the remote Ucluelet station was selected to represent air concentrations (mean = 8.9×10^{-6} $ng{\cdot}L^{-1}$) in habitat on the west coast of Vancouver Island (Noël et al. 2009). These ΣPCB air concentrations are very low, and therefore, are unlikely to be a significant source to the killer whale body burden of PCBs.

2.4 KILLER WHALE FOOD WEBS

The structure of the SRKW food web in their B.C. critical habitat, including foraging preferences and prey items for various marine animals including killer whales, is described in detail in Lachmuth et al. (2010) and Alava et al. (2012). Briefly, the following criteria were applied to develop the food web structure for modelling PCB bioaccumulation:

1. Species of primary interest included: northern and southern resident killer whales, chinook salmon, chum salmon (*Oncorhynchus keta*), coho salmon (*Oncorhynchus kisutch*), Pacific halibut (*Hippoglossus stenolepis*), sablefish (*Anoplopoma fimbria*), lingcod (*Ophiodon elongates*), Dover sole (*Microstomus pacificus*), and Pacific herring (*Clupea pallasi*).

2. Species considered local (i.e., those species that forage primarily in the two areas considered-critical habitat in general and the RBT2 DAS site) were included in the model. For example, resident killer whales have been observed foraging principally on salmonids, for up to 12 months per year in the coastal waters of B.C., (Ford et al*.* 1998).
3. Species from different trophic guilds relevant to the transfer and bioaccumulation of PCBs in the food web were included. Relevant trophic guilds include phytoplankton, zooplankton (i.e., copepods), filter feeding invertebrates (i.e., mussels and oysters), benthic detritivores (i.e., amphipods, crabs, shrimp, and polychaetes), juvenile and adult forage fish as well as fish that are higher predators, and resident killer whales.
4. Species for which empirical PCB concentration data exist were included to allow evaluation of the accuracy of the model predictions. Those species included Harrison chinook salmon, NRKW and SRKW. Published tissue data for NRKW were used in the development of the existing model (Alava et al. 2012).

The number of species included in the model was limited to reduce complexity and make model calculations more transparent. Similar approaches have been used and validated by others for evaluations of food webs that are sediment-driven (von Stackelberg et al. 2002). In order to reflect present day feeding ecology, only the most abundant prey for each species was included. This approach produced a food web bioaccumulation model that included one category for phytoplankton, one category for zooplankton, eight invertebrate species (including detritivores and filter feeders), 12 fish species, and resident killer whales (male, female, juvenile, and newborn).

Most of the data on ecology, foraging habits/diet composition and trophic position for fish were retrieved from www.fishbase.org (Froese and Pauly 2010) and for other aquatic biota from www.sealifebase.org (Palomares and Pauly 2010). In addition, various peer-reviewed papers were consulted when information on life history parameters, prey items, and diet composition were unavailable in the web link sources. Weight and lipid content for chinook salmon within killer whale critical habitats (i.e., Strait of Georgia), were obtained from Cullon et al. (2009).

An updated and revised resident killer whale diet based on field observations by Ford et al. (2010) was established as follows (Lachmuth et al. 2010, Alava et al. 2012): 70% chinook salmon, 15% other salmonids (i.e., 10% chum, 5% coho), and 15% groundfish (i.e., 3% halibut, 3% sablefish, 3% lingcod, 3% dover sole, 3% gonatid squid), and is provided (**Appendix A: Table A-2**).

2.5 HABITAT DISTRIBUTIONS FOR SRKW AND CHINOOK SALMON

Southern resident killer whales are composed of three pods: J, K, and L. These pods range from Monterey Bay, California to Langara Island, B.C., over a distance of approximately 2,000 km along the Pacific coast (Ford 2006). From early summer to late fall, they frequent the coast of southeastern Vancouver Island and Puget Sound (Ford 2006) and in July and August, 90% of their time is spent in their critical habitat in Canada and the U.S. (Ford et al. 2010). In winter and spring, SRKW travel extensively in outer coastal waters (Ford et al. 2000, Nichol and Shackleton 1996, Osborne 1999, Wiles 2004). However, J pod is often sighted in inshore waters all months of the year. Typically, pods K and L return to the Strait of Georgia in May or June then leave in October or November. From about December-May, all three pods travel to outer coastal areas for several days at a time (Ford 2006). Based on this information and data reported by Lachmuth et al. (2010), the annual distribution of SRKWs in the areas and critical habitats is shown in **Table 2-2**. These area-specific distributions reflect the time spent foraging by SKRWs in each area.

Table 2-2 SRKW Distribution on the Pacific Coast Based on Field Observations by Others (see Figure 2-2) (Lachmuth et al. 2010, Alava et al. 2012)

Southern Resident Killer Whales (SRKW)	Outer Coast	SRKW Critical Habitat in Canada	Strait of Georgia	SRKW Critical Habitat in USA (Puget Sound)	SRKW Critical Habitat in USA (summer core & Juan de Fuca Strait)	Total
% Time spent per area	37.0	18.0	3.0	6.0	36.0	100

Efforts have been made to assess the degree to which SRKW frequent the area around Roberts Bank (Hemmera 2014b and c; SMRU 2014)**.** Data exist for the year 2013 for acoustic detections of SRKW vocalisations by hydrophone and acoustic identification in the vicinity of the RBT2 Project area (SMRU 2014). SRKW spend a small percentage of time in the vicinity of the RBT2 Project area (**Table 2-3**). SRKW were detected acoustically in proximity to an underwater hydrophone deployed at Roberts Bank for a total duration of 13 hours through 125 observational days (SMRU 2014; **Table 2-3**). Based on this, the estimated percent of time for which there was a recorded occurrence of SKRW in the Project area, from August to December 2013, was 0.4%. For the first five months of 2014, SRKWs had not been detected acoustically in the vicinity of the Project (Jason Wood, SMRU, personal communication, 11 June 2014).

Approximately 58% of chinook salmon consumed by resident killer whales in all areas of the B.C. coast are from Fraser River stocks primarily from the South Thompson River and Lower Fraser River (Ford et al. 2010). Resident killer whales consume approximately 75% ocean-type chinook salmon, which spend a significant amount of their life history in coastal waters (Ford et al*.* 2010). To simplify the modelling process, resident killer whales were assumed to only consume South Thompson and Fraser River stocks of chinook salmon. This may be an oversimplification, since SRKW also forage on chinook salmon in Puget Sound which exhibit higher PCB concentrations on average than South Thompson and Fraser River chinook.

Figure 2-2 SRKW Distribution Regions on the Pacific (Alava et al. 2012)

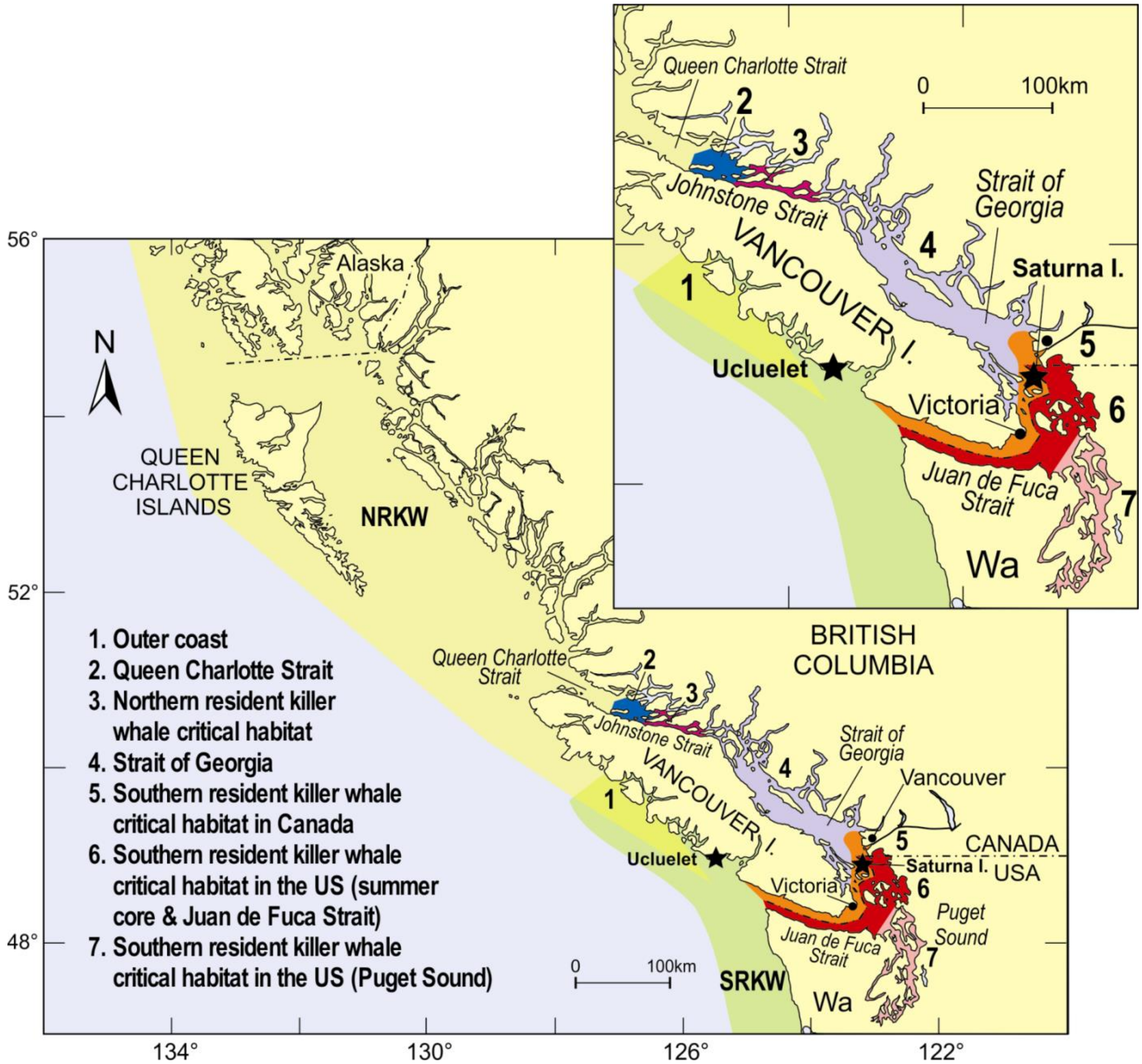


Table 2-3 Acoustic Detection Data for SRKW in the Roberts Bank Area (SMRU 2014)

Acoustic Data	Roberts Bank[2]
Number of recording days[1]	125
Number of recording minutes[1]	180,070
Number of detection events	16
Total duration of detection events (h:m)	**13:11**
Total duration of detection events (minutes)	791
Average duration of detection events ± SD (h:m)	0:49 ± 0:39
Median duration of detection events (h:m)	0:41
Min-Max duration of detection events (h:m)	0:01 - 2:00
No. of detection events/recording day	0.1
Probability of detection	0.4

[1] No. of recording days and no. of recording minutes take into account the data gaps, i.e. those periods of time ranging from a few hours to a few days, when the system was down and no data were collected.
[2] Only the August to June 30 2014 data are considered in this table, since the 2012 data from the JASCO study are based on an analysis of the first 72 seconds of every 30 minute recording and are thus not directly comparable (July to September 2012).

Resident killer whales spend a considerable portion of their time within their critical habitat areas. During July and August, SRKWs spend approximately 90% of their time in critical habitat, likely consuming almost 100% chinook (Ford et al. 2010). During winter months when chinook salmon abundance is low, SRKWs consume other prey such as groundfish and spend more time feeding on salmon in Puget Sound (Ford et al. 2010). Both NRKWs and SRKWs leave critical habitat and travel to outer coastal areas ranging from central Vancouver Island to Monterrey Bay, California through winter and spring (Ford et al. 2010).

Fishing mortality distribution tables (from 1985 to 2007) for chinook salmon in different fishery regions (based on catch data) were used as a proxy for the annual percentage of time chinook spend in the model areas (**Appendix A: Table A-3**). Data for ocean-type chinook stocks were provided by Gayle Brown (Fisheries & Oceans Canada, personal communications), as originally reported in Lachmuth et al. (2010). South Thompson River chinook are represented by the lower Shuswap hatchery indicator stock, and Fraser River stocks by Chilliwack River hatchery stock.

2.6 SPATIAL RESOLUTION OF THE FOOD WEB MODEL

The Alava et al. (2012) model was designed to focus on SRKW critical habitat in Canada (**Figure 1-2**), an area of 2,495.52 km^2. The disposal at sea of dredgeate from the Project is estimated to result in sediment accumulations along and adjacent to the Roberts Bank foreslope over an area of 21.6 km^2 based on an accumulated thickness of ≥ 1.0 mm, and an area of 196.7 km^2 based on an accumulated thickness ≥ 0.1 mm. Thus, the predicted area of influence associated with the RBT2 DAS site comprises 0.9% to

7.9% of the critical habitat. The predicted average bulk sediment (dry weight concentration) for ΣPCBs in the area of seabed affected on the delta foreslope at Roberts Bank, resulting from the re-deposition following sediment disturbance, is 685 pg/g dw (0.685 ug/kg dw) as discussed above.

2.7 STEADY STATE PCB CONCENTRATIONS VERSUS TIME DEPENDENCY

Steady state models (i.e., no net change between environmental compartments) assume that contaminant concentrations have enough time to partition between the water column, the sediments, and biota in the food web such that contaminant concentrations no longer change over time, and reach a dynamic "equilibrium" (Gobas and Arnot 2010). Seasonal changes and the effect of killer whale or prey item age on PCB concentrations can still be captured with a steady state approach by using appropriate parameters. Therefore, a steady state, rather than time-dependent approach, was adopted for the resident killer whale food web bioaccumulation model because the time response of sediment PCB concentrations to changes in loadings and external conditions is relatively slow compared to that in biota.

The environmental half-life for PCBs has been estimated to range from a few years to 100 years (Jonsson et al*.* 2003, Sinkkonen and Paasivirta 2000), while the half-life of PCB 126 in rainbow trout (a salmonid) ranges from 82 to 180 days (Brown et al. 2002). This assumption of steady state is valid for small aquatic organisms (e.g., plankton) as equilibrium between uptake and elimination is quickly reached; however, this process can be much longer for larger organisms (e.g., seals and killer whales), as their body burden often lags behind changing environmental conditions (Hickie et al. 2007). Thus, steady-state models often overestimate concentrations in larger organisms because those concentrations are unlikely to be reached in the short time-span that the model considers (Natale 2007). To reduce model complexity, a steady state approach was used, and included different age classes for certain organisms in the food web to account for age-specific differences in PCB concentrations. The temporal response of PCB concentrations in the sediments is ultimately the "rate controlling" step in the model.

PCB concentrations in the food web were predicted by inputting estimated sediment and water concentrations into the model. Separate model estimates are produced for the seabed area potentially influenced by the Project as opposed to critical habitat areas in general. The model is designed to predict the steady state concentrations in biota as a result of exposure to PCBs in air, water, and sediments, although not how quickly this equilibrium will be achieved. A time-dependent model may be more appropriate to predict the time frame for this equilibrium to be achieved.

3.0 MODEL DESCRIPTION

The following section describes the model used to predict Project-related changes of PCB exposures of SRKW in critical habitat.

3.1 GENERAL MODEL DESCRIPTION

The PCB bioaccumulation model used to quantify potential RBT2 Project-related effects was based on a toxicokinetic/trophodynamic food web bioaccumulation model for PCBs developed for killer whale critical habitat in the marine region of B.C. (Lachmuth et al. 2010, Alava 2011, Alava et al. 2012). Such models attempt to quantify toxicity by simulating the processes that lead to toxicity in organisms over time (Alava et al. 2012). The aim of the PCB model is to characterise the relationship between the concentrations of PCBs in sediments and key SRKW prey (i.e., chinook salmon) in SRKW critical habitat, and their role as a vector for PCB exposure of SRKWs and resulting eco-toxicological risk (Cullon et al. 2009). The theoretical approach is based on the calculation of the Biota Sediment Accumulation Factors (BSAFs) for PCB congeners and ΣPCBs (see **Table 2-1**). Thus, the main output of the model is the BSAF, which characterises the relationship between PCB concentrations in biota (C_B; g PCB·kg^{-1}, wet weight organism) and concentrations in sediments (C_S; g PCB·kg^{-1}, dry weight sediment):

$$\mathbf{BSAF = C_B / C_S} \qquad [1]$$

The model calculates BSAF values (kg dry sediment/kg wet weight organism) for individual PCB congeners in each species included in the model. The BSAF values are calculated as statistical distributions rather than a single point estimate, to allow for seasonal variation. In the management module, BSAF values are used to forward calculate and backward calculate PCB concentrations (**Figure 3-1**).

<u>Forward calculations</u> use BSAFs to predict PCB concentrations in biota (C_B) based on measured/anticipated PCB concentrations in sediments (C_S):

$$\mathbf{C_B = BSAF \cdot C_S} \qquad [2]$$

<u>Backward calculations</u> use PCB concentrations in biota (C_B) to predict PCB concentrations in sediments (C_S):

$$\mathbf{C_S = C_B / BSAF} \qquad [3]$$

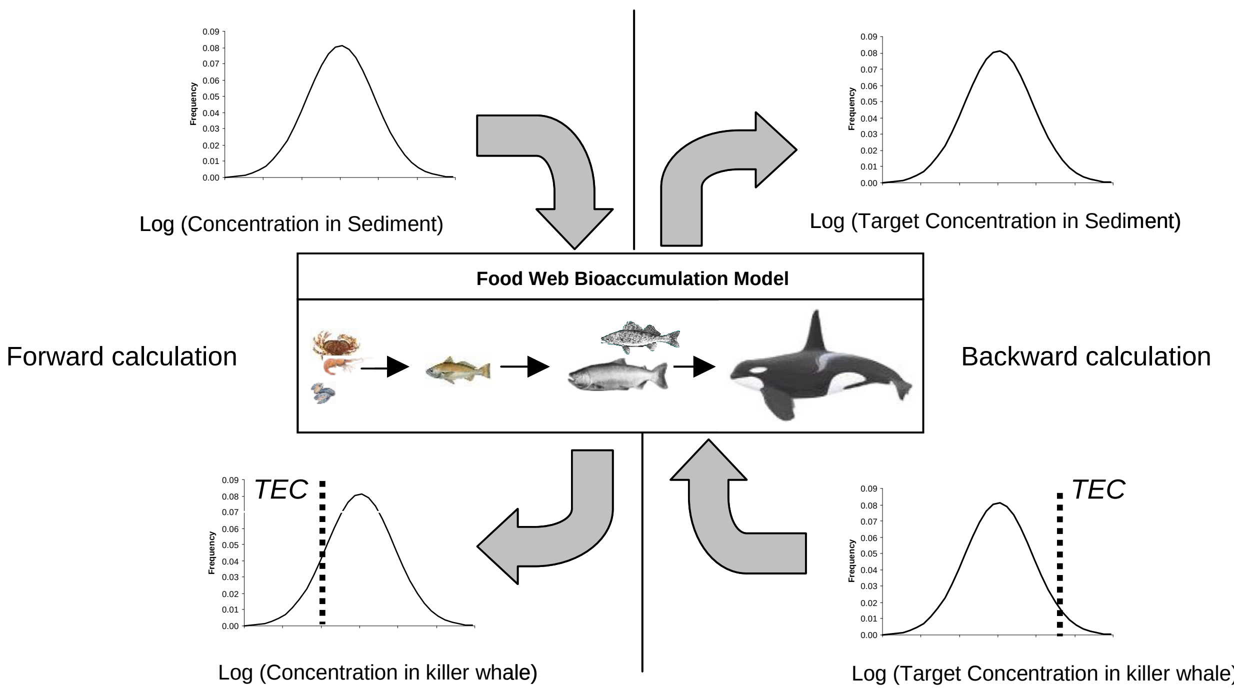
Backward calculations are used to predict concentrations in sediments that are below thresholds for adverse health effects in biota. For the purpose of this work, however, the backward calculation was not applied, as the development of a targeted PCB sediment benchmark for the disposal site was not required.

The model uses various state variables (e.g., octanol water partition coefficient (K_{ow}), lipid content, temperature, weight) to derive BSAFs (Alava et al., 2012). Several mathematical equations are used in the model to describe PCB uptake and elimination in biota, as described in Alava et al. (2012). Equations for mammals (killer whales) are different than those for invertebrates and fish.

3.2 ENVIRONMENTAL CONDITIONS IN SRKW CRITICAL HABITAT IN CANADA

Environmental condition input variables used for the SRKW critical habitat are listed in **Appendix A: Table A-4**. In water, PCBs can be freely dissolved or absorbed to particulate organic matter (POM) and dissolved organic carbon (DOC). Values for POM and DOC were obtained from the literature, or were estimated based on the relationship that most organic carbon (~80%) in water is in the form of DOC (Lachmuth et al. 2010).

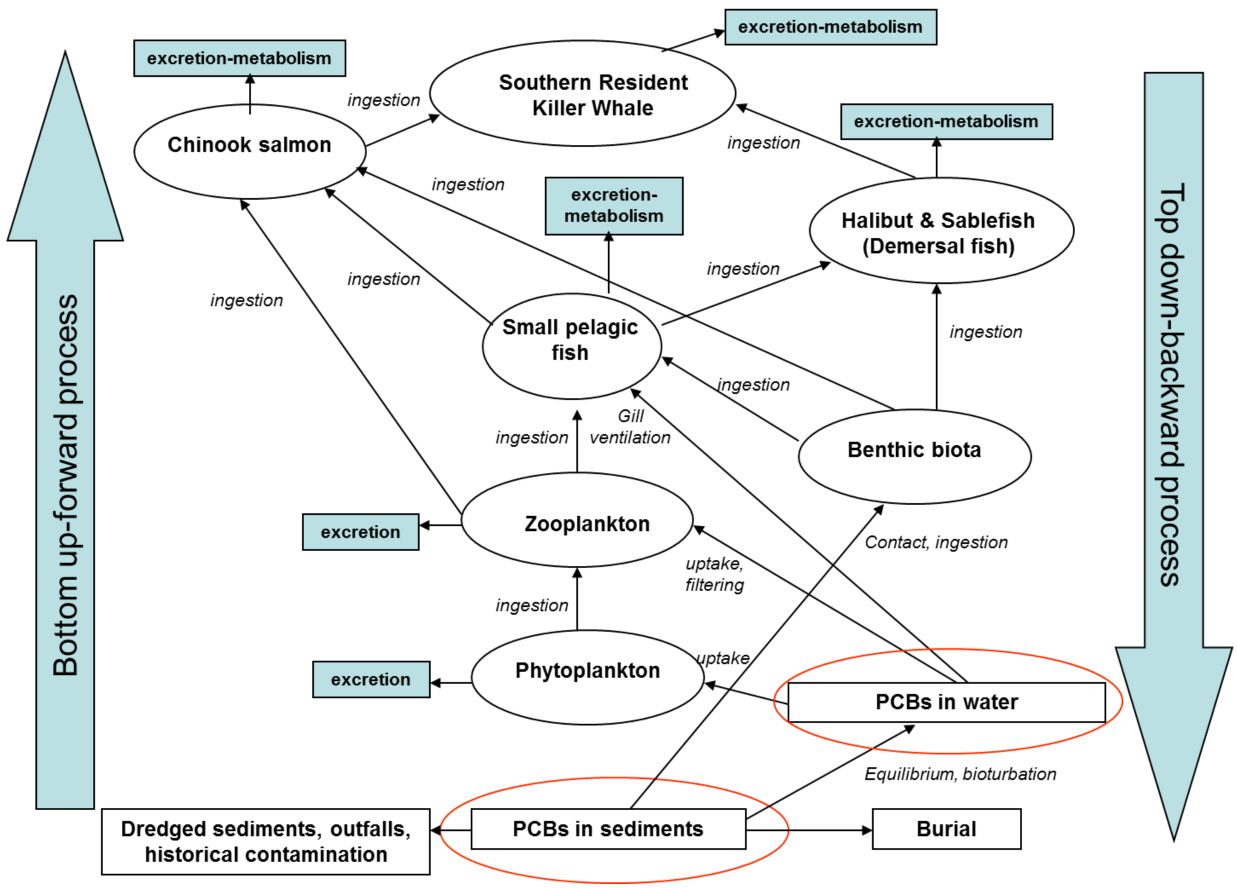
Figure 3-1 Illustration of Forward and Backward Calculation in the Food Web Bioaccumulation Model for PCBs (adapted from Gobas and Arnot 2010)



3.3 BIOLOGICAL VARIABLES IN THE MODEL

A schematic diagram of organisms included in the coastal food web and the representative trophic interactions considered is shown in **Figure 3-2**. Airborne concentrations of PCBs were also included but are not depicted in **Figure 3-2**. The biological and physiological parameters used in the model are listed in **Appendix A: Table A-5**. A detailed description of foraging preferences for the species included in the model in coastal and oceanic food webs respectively, are provided in **Appendix A: Table A-2**.

Figure 3-2 Schematic for Coastal Food Webs in SRKW Critical Habitat Showing Biota Included in the Model, their Trophic Level (TL), and Assumed Food web Linkages (adapted from Lachmuth et al. 2010)



3.4 Forward Calculation: ΣPCB Concentration Estimates in Fish and Wildlife

As noted above, forward calculations determine PCB concentrations in fish and wildlife (C_B) based on measured or predicted PCB concentrations in the sediment (C_S). For this site-specific ecological risk assessment, sediment concentrations are a major model input. Sediment PCB concentrations are in logarithmic format (log C_S) so that the lognormal distributions of sediment concentrations are able to be depicted as normal distributions of log C_S. Similarly, the BSAF (model output) is also depicted in logarithmic format (log BSAF) based on the same reasoning. The calculation is:

$$\log C_B = \log C_S + \log BSAF \quad [4]$$

Mathematically, this is equivalent to:

$$C_B = BSAF \cdot C_S \quad [5]$$

C_B is calculated for each PCB congener and ΣPCBs in the forward calculations, and its uncertainty is based on uncertainty in sediment ΣPCB concentrations and BSAFs. The predicted species PCB concentration can be used to determine whether target threshold PCB concentrations are exceeded.

3.5 Potential for Ecotoxicological Effects to Fish and Wildlife

The model can predict the frequency of likelihood of toxicological effects in modelled species by comparing a species' predicted ΣPCB concentrations to a tissue-based toxic effect concentration (TEC) associated with toxicological effects. **Table 3-1** provides a summary of published estimates of tissue PCB concentration thresholds for various adverse effects.

Table 3-1 Marine Mammal Health Effects Thresholds for ΣPCBs

Toxic Effect Concentrations (TEC)	Health Endpoint Affected	TEC ($\mu g \cdot kg^{-1}$ lipid)	Log TEC ($\mu g \cdot kg^{-1}$ lipid)
Harbour seal PCB toxicity (Ross et al. 1996)	Immune function • Natural killer cell activity • T-cell function • Antibody responses Vitamin A and thyroid hormones	17,000	4.23
Bottlenose dolphin PCB toxicity (Hall et al. 2006)	Mortality	10,000	4.00
Revised harbour seal PCB toxicity (Mos et al. 2010)	EC_5; Immune function, Vitamin A; and, thyroid hormones; Thyroid hormone receptors	1,300	3.11

Note: All thresholds were derived based on studies involving free-ranging or captive fed marine mammals, wherein PCBs represented the dominant contaminant of concern and the contaminant which best correlated with observed effects.

The lowest TEC in **Table 3-1** is 1,300 µg/kg lipid of ΣPCBs, which in turn is based on biochemical responses indicative of PCB exposures and which might be considered as sensitive indicators of further effects on reproduction at the whole animal level of biological organisation. None of the available studies have provided direct observations of changes in reproductive capacity through one or more generations of marine mammals exposed to PCBs.

3.6 Assessment of SRKW PCB Exposures With and Without RBT2 DAS

Pre- and post-disposal scenarios were conducted using an area-adjusted PCB value for the RBT2 DAS site situated within the SRKW critical habitat in Canada. This approach integrates the outcomes generated from the model and existing data (PCB sediment data and BSAFs) for all areas and critical habitats used for SRKWs in B.C. (Canada) and Puget Sound (U.S.A.), as reported in Alava et al. (2012) and Lachmuth et al. (2010) (**Table 3-2**). Considering the extensive range of SRKW, it is reasonable to assume that sources from other areas have contributed to their PCB burdens.

The pre-disposal scenario included the PCB concentrations in marine sediments and BSAFs for the outer coast, Strait of Georgia, SRKW critical habitat in Canada (in this case, the ΣPCB concentrations were updated using the sum of PCB congeners reported in **Table 2-1**), and SRKW critical habitat in U.S.A. (Juan de Fuca Strait and Puget Sound).

The post-disposal scenario included all the PCB concentrations for sediments and BSAFs from all these areas and critical habitats. However, the post-disposal scenario was adjusted by multiplying the proportion of the RBT2 seabed affected by RBT2 DAS within the SRKW critical habitat area, which was calculated as the ratio of the disposal area (i.e., 21.6 or 196.7 km^2, as shown below) to the SRKW critical habitat area in Canada (2,495.52 km^2), by the predicted bulk sediment ΣPCB concentration (i.e., 685 pg/g dw) in the area of disposal affected seabed on the delta foreslope at Roberts Bank. Based on the accumulation of sediment (Tetra Tech EBA 2014), the following assumptions were included:

i. Area of seabed affected by ≥ 0.1 mm of accumulated sediment (i.e., 196.7 km^2) or 7.9% of the SRKW critical habitat; and

ii. Area of seabed affected by ≥ 1.0 mm of accumulated sediment (i.e., 21.6 km^2) or 0.9% of the SRKW critical habitat.

The post-disposal scenario also includes the fraction of the total SRKW critical habitat (92.1% or 99.1%) times the ΣPCB concentration for sediment in the SRKW critical habitat. In addition, both the pre- and post- disposal scenarios include the proportion of time or distribution % that the SRKWs spent in each area or critical habitat, as reported elsewhere (Lachmuth et al. 2010, Alava et al. 2012) and the fraction of time SRKW spend (i.e., 0.4%) in the vicinity of the RBT2 site (**Section 2.5**).

Table 3-2 Summary of Data for Area Adjusted PCB Uptake

SRKW Critical Habitat (CH)/areas	SRKW Habitat Distribution[1] (%)	PCB sediment concentration[2] (ug/kg dw)	BSAF[2] male SRKW (Lipid normalised)	BSAF[2] females SRKW (lipid normalised)
Outer coast	37.0	0.70	89139	13808
SRKW CH-Canada	18.0	0.60	35137	4650
Strait of Georgia	3.0	1.05	85334	13170
SRKW CH-USA (Puget Sound)	6.0	74.4	69777	10757
SRKW CH-USA (summer core & Juan de Fuca Strait)	36.0	6.10	125599	19355

1 % time spent by SKRW in each critical habitat (CH) and area taken from **Table 2-2**.
2 PCB data for sediments and BSAFs for each critical habitat and areas, except for the SRKW critical habitat in Canada (updated here with the data for the Roberts Bank area), were retrieved from Lachmuth et al. (2010).

4.0 MODEL TESTING AND PERFORMANCE ANALYSIS

The following sections discuss model bias assessment and sensitivity analysis.

4.1 MODEL BIAS

Model performance analysis compares each PCB congener's (i) model predicted concentration ($C_{pred,i}$), to the observed concentration ($C_{obs,i}$). Measured sediment and estimated water PCB congener concentrations were input parameters for calculation of PCB concentrations in biota. This measure of model performance was described quantitatively by the model bias (MB), which is species-specific:

$$MB_i = 10^{\sum_{i=1}^{n} \frac{[\log(Cpred,i / Cobs,i)]}{n}} \qquad [6]$$

Assuming a log-normal distribution of the ratio $C_{pred,\ i}$ / $C_{obs,\ i}$, the MB_i is the geometric mean of the ratio of predicted and observed concentrations for all <u>individual PCB congeners</u> in a particular species (*i)*. MB indicates the model's systematic over- (MB>1) or under-prediction (MB<1). For example, MB = 2 means that the model over-predicted the species empirical PCB congener concentrations by a factor of two on average. Over- and under-estimations of observed PCB congeners tend to cancel out while calculating MB, which causes MB to track the central tendency of the model's ability to predict PCB congener concentrations. The standard deviation of MB represents the variability of the over- and under-estimation of measured values.

To quantitatively express model performance for ΣPCBs, we used the model bias MB*, which is derived for each species as:

$$MB^*_i = 10^{\sum_{i=1}^{n} \frac{[\log(Cpred,i / Cobs,i)]}{n}} \qquad [7]$$

Assuming a log-normal distribution of the ratio $C_{pred,\ \Sigma PCB}$ / $C_{obs,\ \Sigma PCB}$, MB_i* is the geometric mean of the ratio of predicted and observed concentrations for <u>ΣPCB</u> in species *i*. MB* indicates the model's systematic over- (MB*>1) or under-prediction (MB*<1) for ΣPCB. The variability of over- and under-estimation of measured values is represented by the standard deviation of MB*, and is an indication of the variability and uncertainty of model predictions. The error of MB* can be described as a factor (rather than a term) of the geometric mean because of the log-normal distribution of the ratio of predicted and observed concentrations.

4.2 SENSITIVITY ANALYSIS

The template PCB model (Lachmuth et al. 2010, Alava et al. 2012), has undergone extensive use and testing to determine which parameters the model is most affected by. A general overview of relative sensitivity of the various parameters, including those having a high impact on the model outcomes, is provided in **Appendix A: Table A-6**.

Specific model parameters, including water and sediment concentrations and organic carbon content in sediments, were tested in Lachmuth et al. (2010) and Alava et al. (2012). For instance, the sensitivity of PCB concentrations in killer whales to PCB concentrations in water was greater than that to PCB concentrations in sediments, indicating that PCBs in the water column are the main source of PCBs for killer whales, with PCBs partitioning predominantly from water to phytoplankton and zooplankton through the pelagic food web into killer whale prey (Alava et al. 2012). Since the relationship between PCB concentrations in sediment and the killer whale food web is affected by the sediment-water concentration ratio for PCBs (Burkhard et al., 2005), there may be potential risks to killer whales associated with sediment disposal and disturbance activities (Alava et al., 2012).

5.0 RESULTS AND DISCUSSION

There are inherent challenges and logistical constraints associated with studying free-ranging populations of resident killer whales, as well as their principal prey species (adult chinook salmon) which also exhibit a very large distributional range and potentially complex foraging patterns across stocks and individuals. This modelling work is intended to provide an alternative approach for predicting the food web mediated uptake of PCBs from alterations in seabed chemistry, associated with at-sea disposal of dredgeate from RBT2 construction at Roberts Bank.

5.1 MODEL APPLICATION TO CHINOOK SALMON AND SOUTHERN RESIDENT KILLER WHALES

Empirical sediment PCB values were used to predict PCB concentrations in chinook salmon and SRKWs within SRKW critical habitat. Summary tabulations are presented here for chinook salmon and killer whales (**Tables 5.1** and **5.2**). Predicted PCB concentrations for several other diet items of resident killer whales (chum and coho salmon, sablefish and halibut) are provided in **Appendix A: Table A-7**.

Model predicted ΣPCB concentrations for male SRKWs PCBs (17.4 mg/kg lipid) exceeded PCB toxicity health effect thresholds for marine mammals, [i.e., 17 mg/kg lipid (Ross et al. 1996); 10 mg/kg lipid (Hall et al. 2006); and 1.3 mg/kg lipid, (Mos et al. 2010)], under the assumption that they are confined and spend 100% of their time in the SRKW critical habitat. In contrast, model predicted ΣPCB concentrations for female SRKWs (2.4 mg/kg lipid) assumed to be confined in SRKW critical habitat are below these thresholds, with the exception of the revised Toxic Reference Value (TRV) for harbour seals (i.e., 1.3 mg/kg lipid; Mos et al. 2010)(**Table 5.2**).

Table 5-1 Predicted and Observed PCB Congener Concentrations in Chinook Salmon

PCB Congener	PCB Concentration (µg/kg lipid)	
	Predicted	Observed*
PCB18	0.74	1.87
PCB28	15.3	5.00
PCB31	14.0	3.70
PCB33/20	2.17	1.26
PCB44	9.55	7.82
PCB49	3.45	7.20
PCB73/52	7.24	17.0
PCB56/60	4.53	2.67
PCB66	15.4	6.04
PCB70/76	18.0	7.30
PCB61/74	18.1	4.34
PCB115/87	10.1	12.0
PCB95	7.30	19.9
PCB99	8.47	23.6

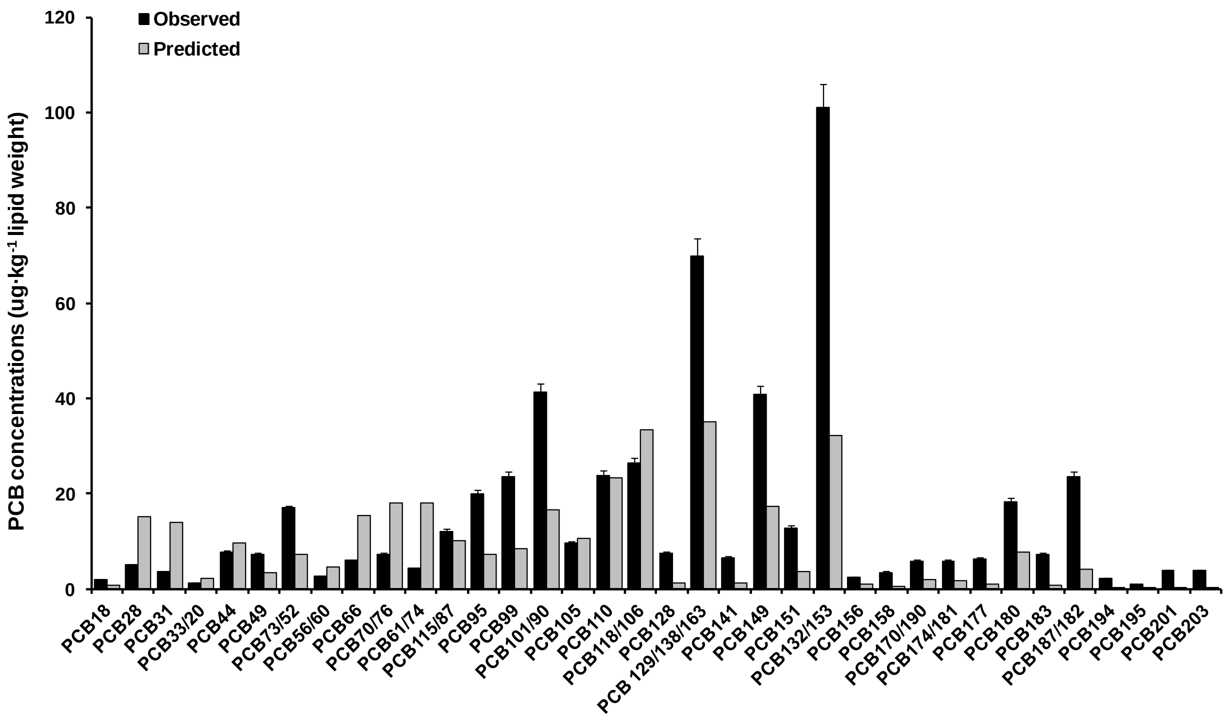
PCB Congener	PCB Concentration (µg/kg lipid)	
	Predicted	Observed*
PCB101/90	16.7	41.3
PCB105	10.7	9.60
PCB110	23.4	23.8
PCB118/106	33.5	26.4
PCB128	1.24	7.44
PCB 129/138/163	35.0	70.0
PCB141	1.19	6.47
PCB149	17.3	40.8
PCB151	3.74	12.8
PCB132/153	32.2	101
PCB156	0.91	2.37
PCB158	0.57	3.50
PCB170/190	1.95	5.87
PCB174/181	1.65	5.81
PCB177	1.03	6.21
PCB180	7.80	18.2
PCB183	0.83	7.14
PCB187/182	4.11	23.5
PCB194	0.18	2.30
PCB195	0.04	0.94
PCB201	0.00	3.81
PCB203	0.16	3.92
Total PCBs (∑PCB)	328	543

*Observed PCB congener data for chinook salmon (Lower Fraser River and Puget Sound) were obtained from Cullon et al. (2009).

Table 5-2 Predicted PCB Congener Concentrations in Adult Male and Female SRKW based on the set of PCB congeners detected in sediment (Section 2.3.2; Table 2-1).

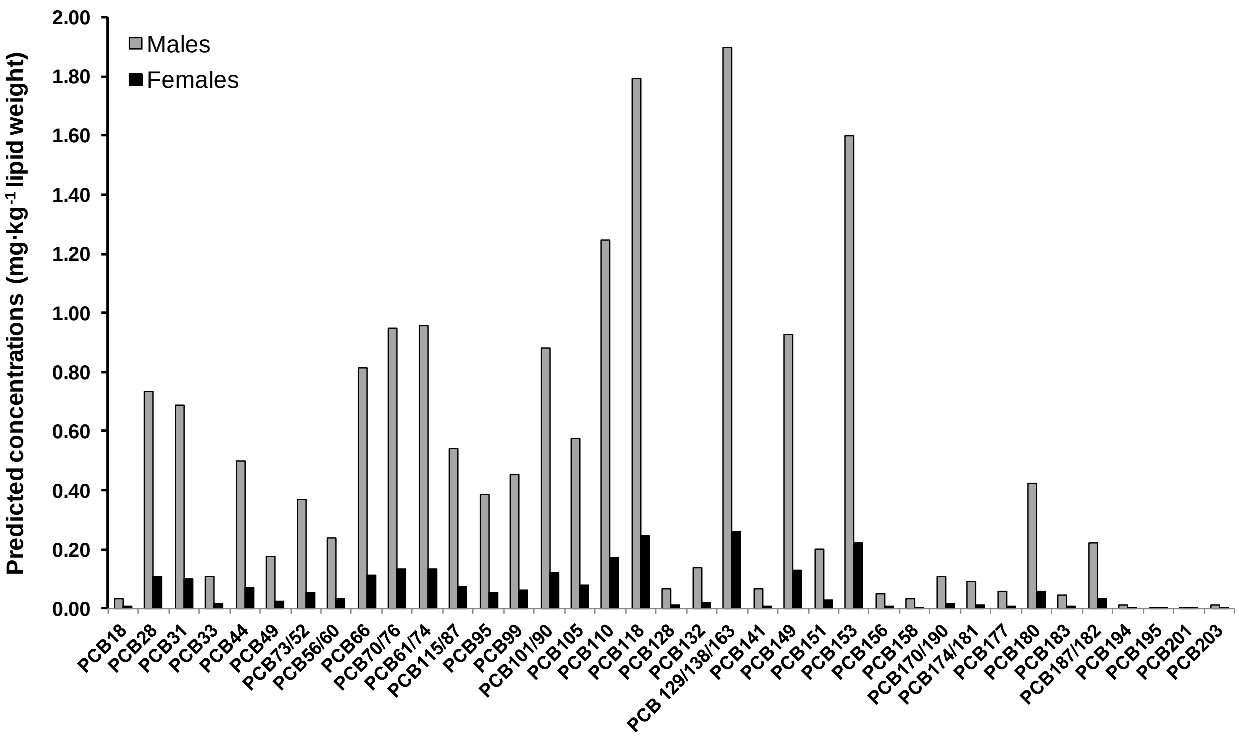
PCB Congener	Predicted Concentration (mg/kg lipid)	
	SRKW Critical Habitat	
	Male SRKW	Female SRKW
PCB18	0.03	0.01
PCB28	0.74	0.11
PCB31	0.69	0.10
PCB33	0.11	0.02
PCB44	0.50	0.07
PCB49	0.17	0.02
PCB73/52	0.37	0.05
PCB56/60	0.24	0.03
PCB66	0.82	0.11
PCB70/76	0.95	0.13
PCB61/74	0.96	0.13
PCB115/87	0.54	0.07
PCB95	0.39	0.05
PCB99	0.45	0.06
PCB101/90	0.88	0.12
PCB105	0.57	0.08
PCB110	1.24	0.17
PCB118	1.79	0.25
PCB128	0.07	0.01
PCB132	0.14	0.02
PCB 129/138/163	1.90	0.26
PCB141	0.06	0.01
PCB149	0.93	0.13
PCB151	0.20	0.03
PCB153	1.60	0.22
PCB156	0.05	0.01
PCB158	0.03	4E-03
PCB170/190	0.11	0.01
PCB174/181	0.09	0.01
PCB177	0.06	0.01
PCB180	0.42	0.06
PCB183	0.05	0.01
PCB187/182	0.22	0.03
PCB194	0.01	1E-03
PCB195	2E-03	3E-04
PCB201	2E-04	3E-05
PCB203	0.01	0.001
Total PCBs (ΣPCB)	17.4	2.41

Figure 5-1 Predicted and Observed Concentrations of Specific PCB Congeners (ug•kg-1 lipid weight) in Chinook Salmon in SRKW Critical Habitat



Predicted PCB congener data for male and female SRKW are illustrated in **Figure 5.2**. Because comparable empirical data for SRKW was not available, it was not possible to conduct comparisons with observed PCB congener concentrations. Using non-comparable data may result in over- or under-predicted values. Although only the predicted data are shown here as a reference, the PCB pattern reproduced by the model in **Figure 5.2** is similar to the pattern previously observed in SRKWs (Ross et al. 2000).

Figure 5-2 Predicted PCB Congener Concentrations (mg•kg^{-1} lipid weight) in Male and Female SRKW



5.2 MODEL BIAS AND PERFORMANCE

The ability of the model to estimate PCB congener concentrations in biota was tested by comparing predicted concentrations in individual adult chinook salmon from the Strait of Georgia to available empirical values (i.e., chinook salmon stocks from the Lower Fraser River, and Duwamish and Deschutes rivers in Puget Sound; Cullon et al. 2009). Model-predicted and empirical PCB congeners included are shown in **Figure 5.1** and **Table 5.1**. While the model bias (MB) geometric mean ± log MB (SD) was 0.40 ± 0.67 for PCB congeners in chinook salmon, underlining underprediction, the model bias for total PCBs (ΣPCBs MB*) indicates fairly good agreement between predicted versus empirical data, with a MB* geometric mean ± log MB (SD) of 0.75 ± 0.30, which is close to 1.0. The predicted concentrations of PCBs are fairly similar to, or within the range of, observed PCB concentrations in chinook salmon (**Figure 5.1**). This suggests that the congener patterns of PCBs in chinook salmon are plausibly reproduced by the model when compared against the empirical profiles for this species. This is further supported by the small uncertainty (i.e., error bias) of the model pointed out above (SDMB = 0.30).

5.3 Risk Assessment of Pre- and Post-Disposal Scenarios

In terms of increasing risk to both male and female SRKWs, the change in predicted concentrations of total PCBs is minimal or negligible when comparing the pre- and post- disposal data (**Table 5.3**). An increase of 0.00003% for males and 0.00002% for females is predicted if the affected seabed area accumulates sediment ≥ 0.1 mm in an area of 196.7 km^2 (**Table 5.3**). The increase in risk remains minimal, with a smaller increase predicted, if the affected seabed area accumulates sediment ≥ 1.0 mm in an area of 21.6 km^2 (0.000003% increase for both males and females; **Table 5.3**). To evaluate risk for SRKWs, the predicted concentrations were compared with existing guidelines for marine mammals. These predicted concentrations for PCBs in SRKWs exceed toxicity reference values (TRVs) or existing toxicity thresholds for marine mammals, i.e., 17 mg/kg lipid (Ross et al. 1996), 10 mg/kg lipid (Hall et al. 2006), and 1.3 mg/kg lipid, (Mos et al. 2010), either before or after disposal (**Figure 5.3**). The model results are consistent with previous food web modelling efforts in critical habitats and areas inhabited by SRKWs (Lachmuth et al. 2010; Alava et al. 2012).

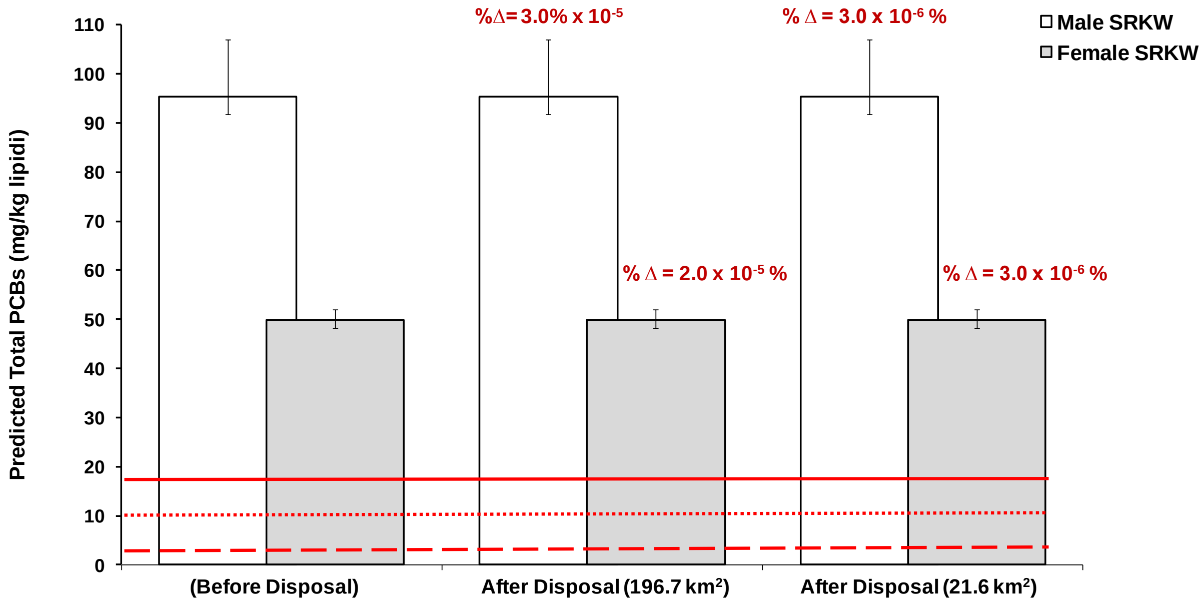
The predicted concentrations of PCBs in SRKW exceeded toxicity thresholds documented for marine mammals in both the RBT2 pre- and post-disposal scenarios. Natural siltation or smothering of sediments potentially buries DAS sites (especially near the mouth of the Fraser River), and may essentially reduce exposure of organisms in the food web to PCBs in disposal materials (i.e. Fraser River dredgeate). This is an important consideration for disposal practices, in terms of frequency of disposal and the site selection process.

Table 5-3 Pre- and Post-Disposal Scenarios Using an Area-Adjusted Sediment PCB Value for the Roberts Bank/ RBT2 DAS Area Scenarios

RBT2-DAS Area: 196.7 km^2 at ≥ 0.1 mm				
Male SRKW Concentration (mg/kg lipid)				
Robert Banks RBT2 Disposal Area	Before Disposal	After Disposal	Fraction Change	% Change
	mg/kg lipid	mg/kg lipid		
Uncorrected Values	611	611	2.9E-07	0.00003
Corrected Values*	95.3	95.3	2.9E-07	0.00003
Female SRKW Concentration (mg/kg lipid)				
Robert Banks RBT2 Disposal Area	Before Disposal	After Disposal	Fraction Change	% Change
	mg/kg lipid	mg/kg lipid		
Uncorrected Values	94	94	2.5E-07	0.00002
Corrected Values*	50	50	2.5E-07	0.00002
RBT2-DAS Area: 21.6 km^2 at ≥ 1.0 mm				
Male SRKW Concentration (mg/kg lipid)				
Robert Banks RBT2 Disposal Area	Before Disposal	After Disposal	Fraction Change	% Change
	mg/kg lipid	mg/kg lipid		
Uncorrected Values	611	611	3.2E-08	0.000003
Corrected Values*	95.3	95.3	3.2E-08	0.000003
Female SRKW Concentration (mg/kg lipid)				
Robert Banks RBT2 Disposal Area	Before Disposal	After Disposal	Fraction Change	% Change
	mg/kg lipid	mg/kg lipid		
Uncorrected Values	94.2	94.2	2.7E-08	0.000003
Corrected Values*	50	50	2.7E-08	0.000003

*The predicted (uncorrected) PCB concentrations in male and female SRKWs (PCB levels before disposal) were corrected by dividing it by 6.4 for males and 1.9 for females. The values 6.4 and 1.9 represent the antilog of the mean error bias, which were estimated after subtracting the empirical PCB mean concentration in males (i.e., 146.3 mg·kg^{-1} lipid) or in females (i.e., 55.4 mg·kg^{-1} lipid) reported by Ross et al. (2000), and the mean calculated in males (i.e. 62.1 mg·kg^{-1} lipid) or in females (i.e. 45.0 mg·kg^{-1} lipid) from the data reported by Krahn et al. (2007), from the predicted PCB concentrations in males or in females in a logarithm format [$10^{\Sigma(\log PCBpredicted - \log PCBobserved)/n}$].

Figure 5-3 **Predicted Concentrations for PCBs in SRKWs Using an Area-adjusted approach for PCB Values to assess Pre- and Post-disposal Scenarios at the Roberts Bank Candidate DAS Site (the dashed line represents the revised harbour seal PCB TRV (1.3 mg•kg^{-1} lipid; Mos et al. 2010); the dotted line represents the bottlenose dolphin PCB toxicity threshold (10 mg•kg^{-1} lipid; Hall et al. 2006); and the solid line represents the previous harbour seal PCB toxicity threshold (17 mg•kg^{-1} lipid; Ross et al. 1996).**



5.4 DATA ASSUMPTIONS

This model requires empirical sediment PCB concentrations and organic carbon content as critical inputs to calculate and subsequently predict PCB concentrations in marine biota. This data was collated from literature and the sediment characterisation technical report (Hemmera 2014a).

The accuracy of the model was tested by comparing the model predictions of PCB concentrations in biota (i.e., chinook salmon) to available empirical data. These empirical data were limited to chinook salmon stock from the Lower Fraser River (Strait of Georgia), Deschutes River, and Duwamish River (Puget Sound). Observed PCB data for the SRKW population was unavailable at the time of this modelling application. A more recent and comparable dataset for the SRKW population is required for comparison to avoid introducing more bias to the model predictions. Further inclusion of other food web species would improve model accuracy throughout trophic levels.

The model attempted to predict the consequences of disposal of dredgeate in critical habitat of SRKWs. However, actual PCB concentrations of the dredged material were based on a reliable estimated concentration for the bulk of sediments predicted to be accumulated at the Roberts Bank DAS site from dredgeate discharge (Hemmera 2014a, Tetra Tech EBA 2014).

Using models requires the use of assumptions for simplicity and transparency, as described throughout this report. A major assumption of this model is that contaminant-sediment partitioning is at steady-state, which may or may not be the case, particularly when the model includes long-lived species such as killer whales. Assumptions were made about the distribution of adult chinook salmon, as the principal prey item of SRKW, and the associated foraging time spent by SRKW in specific areas. In order to simplify the modelling process, it is assumed that SRKW consume only South Thompson and Fraser River chinook salmon stocks and that they spend 100% of their time within the SRKW critical habitat. These are conservative assumptions that would tend to over-predict PCB uptake from both the Canadian critical habitat areas and the Roberts Bank candidate DAS site relative to the true case. Despite these limitations, this study has completed its overall objective of ensuring that sufficient information is available to inform an effects assessment for the Project.

6.0 CONCLUSIONS

Based on the weighted-area approach for RBT2 pre- and post- disposal scenarios, the increase in risk in terms of predicted PCB concentrations for both male and female SRKW is sufficiently low that it is deemed to be negligible. The predicted increase in exposure is very small in comparison with both the known variability of tissue concentrations for killer whale individuals (male or female) and the precision with which changes in exposure can be predicted or measured.

7.0 CLOSURE

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9.0 STATEMENT OF LIMITATIONS

This report was prepared by Hemmera Envirochem Inc. (“Hemmera”), based on fieldwork conducted by Hemmera, for the sole benefit and exclusive use of Port Metro Vancouver. The material in it reflects Hemmera’s best judgment in light of the information available to it at the time of preparing this Report. Any use that a third party makes of this Report, or any reliance on or decision made based on it, is the responsibility of such third parties. Hemmera accepts no responsibility for damages, if any, suffered by any third party as a result of decisions made or actions taken based on this Report.

Hemmera has performed the work as described above and made the findings and conclusions set out in this Report in a manner consistent with the level of care and skill normally exercised by members of the environmental science profession practicing under similar conditions at the time the work was performed.

This Report represents a reasonable review of the information available to Hemmera within the established Scope, work schedule and budgetary constraints. The conclusions and recommendations contained in this Report are based upon applicable legislation existing at the time the Report was drafted. Any changes in the legislation may alter the conclusions and/or recommendations contained in the Report. Regulatory implications discussed in this Report were based on the applicable legislation existing at the time this Report was written.

In preparing this Report, Hemmera has relied in good faith on information provided by others as noted in this Report, and has assumed that the information provided by those individuals is both factual and accurate. Hemmera accepts no responsibility for any deficiency, misstatement or inaccuracy in this Report resulting from the information provided by those individuals.

APPENDIX A
Tables

Table A-1 Values for PCB Congeners and Physical-Chemical Properties Used in the Food Web Bioaccumulation Model for the SRKW Critical Habitat in Canada (adapted from Lachmuth et al. 2010)

Chemical Name	Congener CAS #	Molecular Weight (g/mol)	LeBas Molar Volume (cm^3/mol)	Log Kow Temperature 9.10°C and Salt Corrected	Log K_{OW} Temperature Corrected (37.5 °C)	Log K_{OA} Temperature Corrected (37.5 °C)
PCB	8	223.1	226.4	5.42	4.96	4.96
PCB	18	257.5	247.4	5.62	5.12	5.12
PCB	28	257.5	247.4	5.99	5.47	5.47
PCB	31	257.5	247.4	6.11	5.60	5.60
PCB	33	257.5	247.4	5.98	5.47	5.47
PCB	44	292.0	268.4	6.16	5.63	5.63
PCB	49	292.0	268.4	6.30	5.76	5.76
PCB	52	292.0	268.4	6.26	5.72	5.72
PCB	56	292.0	268.4	6.39	5.80	5.80
PCB	60	292.0	268.4	6.49	5.91	5.91
PCB	66	292.0	268.4	6.36	5.81	5.81
PCB	70	292.0	268.4	6.46	5.90	5.90
PCB	74	292.0	268.4	6.46	5.91	5.91
PCB	87	326.5	289.4	6.72	6.15	6.15
PCB	95	326.5	289.4	6.43	5.86	5.86
PCB	99	326.5	289.4	6.73	6.16	6.16
PCB	101	326.5	289.4	6.68	6.16	6.16
PCB	105	326.5	289.4	7.20	6.62	6.62
PCB	110	326.5	289.4	6.68	6.11	6.11
PCB	118	326.5	289.4	6.97	6.39	6.39
PCB	128	361.0	310.4	7.18	6.59	6.59
PCB	132	361.0	310.4	6.90	6.36	6.36
PCB	138	361.0	310.4	7.59	7.04	7.04
PCB	141	361.0	310.4	7.13	6.59	6.59
PCB	149	361.0	310.4	6.99	6.44	6.44
PCB	151	361.0	310.4	6.96	6.42	6.42
PCB	153	360.9	310.4	7.28	6.65	6.65
PCB	156	361.0	310.4	7.37	6.85	6.85
PCB	158	361.0	310.4	7.23	6.71	6.71
PCB	170	395.5	331.4	7.56	7.00	7.00
PCB	174	395.5	331.4	7.43	6.83	6.83
PCB	177	395.5	331.4	7.41	6.81	6.81

Chemical Name	Congener CAS #	Molecular Weight (g/mol)	LeBas Molar Volume (cm^3/mol)	Log Kow Temperature 9.10°C and Salt Corrected	Log K_{ow} Temperature Corrected (37.5 °C)	Log K_{OA} Temperature Corrected (37.5 °C)
PCB	180	395.5	331.4	7.57	6.95	6.95
PCB	183	395.5	331.4	7.52	6.92	6.92
PCB	187	395.5	331.4	7.49	6.89	6.89
PCB	194	429.8	352.4	8.18	7.56	7.56
PCB	195	430.0	352.4	7.87	7.25	7.25
PCB	201	430.0	352.4	7.92	7.31	7.31
PCB	203	430.0	352.4	7.95	7.33	7.33

Table A-2 Feeding Preferences Matrix – Dietary Composition and Trophic Levels (TL) of 22 Predator Species / Organisms for Southern Resident Killer Whales and Redistribution of Diet Composition for Some Fish Species

Offshore Food Web Species (Predators)	TL	Prey[1] (Diet %)																							Sum
		Det/Sed	Phy	Zoo	Pol-1	Pol-2	Mus	Oys	Amp	Mys	DCr	Shri	Sper	Herr	Wpol	Anch	Dsol	Chum	Sqd	Coho	Lcod	Sfish	Hal	Sal	
Zooplankton (Copepoda, *neocalanus*)	2.0		100																						100
Polychaete-1 *(Neanthes succinea)*	2.1	90	5	5																					100
Polychaete-2 *(Harmothoe imbricata)*	2.1	30	35	35																					100
Blue mussel (*Mytilus edulis*)	2.3	15	60	25																					100
Pacific oyster (*Crassostrea gigas*)	2.3	15	60	25																					100
Amphipods (*Themisto* sp.)	2.4	30	35	35																					100
Mysid shrimp (*Mysis* sp.)	2.5	10	45	45																					100
Dungeness crab (*Cancer magister*)	2.8	43	2	10	5	5	5	5	5	5		5	5	5											100
Crangon sp. (shrimp)*****	2.9	15		3.5		1.5			30	50															100
Shiner surfperch (*Cymatogaster aggregata*)	3.2	5	10	10	10	10			20	15		20													100
Pacific Herring (*Clupea pallasi*)	3.0			98	1				1																100
Walleye pollock (*Theragra chalcogramma*)	3.0			95	2.5				2.5																100
Northern anchovy (*Engraulis mordax*)	3.1		20	20					15	25		20													100
Dover Sole (*Microstomus pacificus*	3.3				27	27	7.25	7.25	1	10	10	10													100
Chum salmon (*Oncorhynchus keta*)	3.4	12		24	0.5	0.5			9		2			17.5		17.5			17						100
Gonatid squid (*Gonatius*)	3.5			50					3	5		5	9.3	9.3	9.3	9.3									100
Sablefish (*Anoplopoma fimbria*)	3.8			10	5				5		5	10	3	3	45	3	2.5		8						100
Coho salmon (*Oncorhynchus kisutch*)	4.2			26					34		4	4		16		8		8							100
Lingcod (*Ophiodon elongates*)	4.3								10	6.7	6.7	6.7			25		25		20						100
Halibut (*Hippoglossus stenolepis*)	4.0			1	1	1	1	1	1	10	14	14	5	5	38		1		5	1		1			100
Chinook salmon (*Oncorhynchus tshawytscha*)	4.0			5					1			4	10	25	25	10	10		10						100
Killer whale (*Orcinus orca*)	5.0																3	10	3	5	3	3	3	70	100

[1]Legend prey species: Det/Sed = Detritus/Sediment; Phy = Phytoplankton; Zoo = Zooplankton; Pol-1 = Polychaete-1; Pol-2 = Polychaete-2; Mus = Blue Mussels; Oys = Oyster; Amp = Amphipods; Mys = *Mysis*; DCr = Dungeness crab; Shri =Shrimp (*Crangon*); Sper = Shiner Surfperch; Herr = Pacific herring; Wpol = Walleye pollock; Anch = Northern anchovy; Dsol = Dove sole; Coho = Coho salmon; Sqd= Gonatid squid; Sfish = Sablefish; Chum = Chum salmon; Lcod = Lingcod; Hal = Halibut; Sal = Chinook salmon. In the models trophic position values for detritus (TL = 1) and phytoplankton (TL = 1) were assigned according to Vander Zanden and Rasmussen (1996).*****Diet for *Crangon* shrimp was updated and modified from the original diet available in Alava *et al.* (2012)

Note: Prey species and their corresponding trophic levels are identified. Trophic position for fish and other aquatic biota were retrieved from www.fishbase.org (Froese and Pauly 2010) and www.sealifebase.org (Palomares and Pauly 2010), respectively.

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Table A-3 Average Annual Distribution (% Time) For Lower Fraser River and South Thompson Chinook Salmon Stocks Used in the Model Areas (Based on Lachmuth et al. 2010)

Lower Fraser River Chinook Areas	% Time Spent Per Area
Queen Charlotte Strait	1.71
Outer coast	55.0
NRKW Critical Habitat	14.47
SRKW Critical Habitat in Canada	7.68
Strait of Georgia	7.68
SRKW Critical Habitat in USA (summer core & Juan de Fuca Strait)	4.07
SRKW Critical Habitat in USA (Puget Sound)	9.41
Total	100

South Thompson Chinook Areas	
Queen Charlotte Strait	8.0
Outer coast	80.0
NRKW Critical Habitat	3.47
SRKW Critical Habitat in Canada	3.45
Strait of Georgia	3.45
SRKW Critical Habitat in USA (summer core & Juan de Fuca Strait)	1.63
SRKW Critical Habitat in USA (Puget Sound)	0.17
Total	100

Table A-4 Environmental Input Parameters for the SRKW Critical Habitat in the Bioaccumulation Food Web Model

Parameter	Input	Variability	Units	Reference
Mean Water Temperature	9.1	2.1	(^{O}C)	Masson 2006
Mean Air Temperature	9.3	8.1	(^{O}C)	Lachmuth et al. 2010
Mean Homeothermic Biota Temperature	37.5	1	(^{O}C)	Estimated
Mean Water Temperature	282.2	1.3	K	Masson 2006
Mean Air Temperature	282.4	6.0	K	Lachmuth et al. 2010
Mean Homeothermic Biota Temperature	310.65		K	Estimated
pH of Water	7.70	0.1	Unitless	Lachmuth et al. 2010
Practical Salinity Units (PSU)	30.4	3.0	(g/kg)	Masson 2006
Dissolved Oxygen Concentration @ 90% Saturation (DO)	4.11	2.03	(mg O_2/L)	Masson 2006
Dissolved Organic Carbon Content - Water (OCwater)	6.36E-07	1.19E-07	(kg/L)	Johannessen et al. 2008b
Particulate Organic Carbon Content - Water (POC)	9.20E-08	5.0E-08	(kg/L)	Johannessen et al. 2008b
Concentration of Suspended Solids (Vss)	2.62E-06	1.21E-06	(kg/L)	Komick et al. 2009
Percentage of Organic Carbon - Sediment (OCsed)	2.88%		(%)	Burd et al. 2008b; Dr. Doug Bright, (personal communication)
Density of Organic Carbon - Sediment (Docsed)	0.9		(kg/L)	Mackay 1991
Setschenow Proportionality Constant (SPC)	0.0018		(L/cm3)	Xie et al. 1997.
Molar Concentration of Seawater @ 35 ppt (MCS)	0.5		(mol/L)	Xie et al. 1997.
Absolute Temperature (K)	273.16		K	
Ideal Gas Law Constant (Rgaslaw)	8.314		(Pa.m^3/mol.K)	

Table A-5 General Biological and Physiological Parameter Definitions, Values, and References Used in the Food Web Bioaccumulation Model (ED = dietary chemical transfer efficiency)

General Aquatic Species Input Parameter	Mean		SD	Units	Source
Density of lipids	0.9	±		$kg \cdot L^{-1}$	1
Non-lipid organic matter content (NLOM)	20%	±	0.01	%	
NLOM proportionality constant (MAF)	0.05	±	5.0E-03	Unitless	2 (modified from)
Fish growth rate factor (FGR)	1.40E-03	±	7.0E-05	Unitless	3 (modified from)
Invertebrate growth rate factor (IGR)	3.50E-04	±	3.5E-05	Unitless	3
Dietary absorption efficiency of lipid in benthic invertebrate (ε_L)	75%	±	0.02	%	4
Dietary absorption efficiency of NLOM in benthic invertebrate (ε_N)	50%	±	0.02	%	4 (modified from)
Dietary absorption efficiency of lipid in fish (ε_L)	92%	±	0.02	%	4
Dietary absorption efficiency of NLOM in fish (ε_N)	60%	±	0.02	%	4, 5
Dietary absorption efficiency of lipid in mammals (ε_L)	100%	±	0.02	%	5, 7, 8
Dietary absorption efficiency of NLOM in mammals (ε_N)	98%	±	0.02	%	1 (modified from)
E_D - Constant A - All feeding species except marine mammals	8.5E-08	±	1.4E-08	Unitless	1
E_D - Constant B - All feeding species except marine mammals	2.00	±	0.600	Unitless	1
E_D - Constant A - Mammals	1E-09	±	1.7E-10	Unitless	1
E_D - Constant B - Mammals	1.025	±	1.2E-03	Unitless	1
E_W - Constant A - Water absorption efficiency in fish & invertebrates	1.85	±	0.13	Unitless	1
Water digestion efficiency in marine mammals (E_W)	85%	±		%	1
Lung uptake efficiency in marine mammals (E_L)	0.7	±		Unitless	1
Mean homoeothermic temperature (marine mammals)	37.5	±	1.00	°C	1
Metabolic transformation rate constant (k_M) - All species	0.00	±		$1 \cdot day^{-1}$	4
Particle scavenging efficiency (PSE)	100%			%	Default value

Table References:

1. Gobas and Arnot (2010)
2. Gobas et al. (1999)
3. Thomann et al. (1992)
4. Arnot and Gobas (2004)
5. Kelly et al. (2004)
6. Drouillard and Norstrom (2000)
7. Trumble et al. (2003)
8. Muelbert et al. (2003)

Table A-6 Food Web Bioaccumulation Model Sensitivity to Various Parameters

Parameter	Model Sensitivity
Dietary preference	High
Body weight	High
Lipid content	High
Gill ventilation rate	Low
Gill uptake efficiency	Low
Feeding rate	Low for chemicals with log $K_{OW} \leq 6.5$ High for PCBs with log $K_{OW} > 6.5$
PCB dietary uptake efficiency	Low
Growth rate	Low but increases in importance for larger organisms (fish & killer whales) and higher K_{OW} PCB congeners
Metabolism	Low – unless metabolic transformation rates are high compared to other elimination routes
K_{OW}	High
Food digestibility	High
Diet lipid content	High
Concentration in water	High
Concentration in sediments	High
Organic carbon content in sediments	High

Table A-7 Predicted Concentrations (ug·kg-1 wet weight) of PCB Congeners in other Fish Prey of the Southern Resident Killer Whales' Diet

PCB Congener	Predicted Concentration (ug/kg lipid)			
	Halibut	Sablefish	Coho Salmon	Chum Salmon
PCB18	0.04	0.04	0.02	0.01
PCB28	0.84	0.83	0.40	0.34
PCB31	0.75	0.74	0.38	0.32
PCB33/20	0.12	0.12	0.06	0.05
PCB44	0.50	0.50	0.26	0.22
PCB49	0.18	0.18	0.10	0.08
PCB73/52	0.37	0.37	0.20	0.18
PCB 56	0.23	0.23	0.13	0.11
PCB 60	0.08	0.08	0.05	0.04
PCB66	0.77	0.77	0.43	0.38
PCB70/76	0.89	0.89	0.51	0.46
PCB61/74	0.89	0.89	0.52	0.47
PCB115/87	0.47	0.48	0.30	0.28
PCB95	0.36	0.36	0.21	0.19
PCB99	0.40	0.40	0.25	0.23
PCB101/90	0.79	0.80	0.49	0.45
PCB105	0.48	0.48	0.34	0.32
PCB110	1.10	1.12	0.69	0.64
PCB118/106	1.52	1.55	1.03	0.97
PCB 128	0.06	0.06	0.04	0.04
PCB 132	0.12	0.12	0.08	0.07
PCB138	1.57	1.55	1.20	1.16
PCB141	0.05	0.05	0.04	0.04
PCB149	0.78	0.80	0.53	0.50
PCB151	0.17	0.17	0.11	0.11
PCB153	1.32	1.34	0.95	0.91
PCB156	0.04	0.04	0.03	0.03
PCB158	0.03	0.03	0.02	0.02
PCB170/190	0.09	0.09	0.07	0.06
PCB174/181	0.07	0.07	0.05	0.05
PCB177	0.05	0.05	0.03	0.03
PCB180	0.35	0.35	0.26	0.26
PCB183	0.04	0.04	0.03	0.03
PCB187/182	0.18	0.18	0.14	0.13
PCB194	0.01	0.01	0.01	0.01
PCB195	2.E-03	2.E-03	2.E-03	1.E-03
PCB201	2E-04	2E-04	1E-04	1E-04
PCB203	0.01	0.01	0.01	0.01