A FRAMEWORK TO ESTIMATE THE RISK OF NOISE EXPOSURE FROM VESSELS FOR ENDANGERED CETACEAN SPECIES

by © Simone Cominelli

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Abstract

This thesis proposes a framework for assessing and visualizing exposure of the Southern Resident Killer Whale (SRKW) population to vessels' noise. First, SRKW distribution was mapped and the risk for this population to be exposed to vessel-noise was estimated. The study identified six vessel classes as being the main contributors to noise exposure for SRKW. Building on this result, the second study presents an analytical framework focused on exposure hotspot mapping, the computation of probabilistic levels of exposure, and the identification of shipping routes minimizing exposure for SRKW. The framework was tested in the Salish Sea, leading to the identification of four hotspots of exposure for SRKW. Small spatial changes in the current shipping lanes could lead to a large reduction of the overlap between vessel traffic and sensitive areas for SRKW. The results highlight how effectively addressing vessel noise requires the implementation of adaptive management strategies.

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List of Abbreviations and Symbols

AIS	Automatic Identification System
AOI	Area of Interest
BCCSN	British Columbia Cetacean Sighting Network
c.d.f.	Cumulative Distribution Function
COSEWIC	Committee on the Status of Endangered Wildlife in Canada
DFO	Fisheries and Oceans Canada
ECHO	Enhancing Cetacean Habitat and Observation program
EHM Tool	Noise Exposure Hotspots Mapping Tool
ESA	Endangered Species Act
GIS	Geographic Information Science
IMO	International Maritime Organization
KDE	Kernel Density Estimation
LCP	Least Cost Path
Leq	Equivalent Continuous Sound Pressure Level or Equivalent Time-averaged Sound Pressure Level
Leq-50th	Median Leq or Level of Exposure
LOE Tool	Level of Exposure Tool
MARS	Maritime mobile Access and Retrieval System
MONM	Marine Operations Noise Model
MPA	Marine Protected Area
MSFD	Marine Strategy Framework Directive
NARW	North Atlantic Right Whale
NEMES	Noise Exposure to the Marine Environment from Ships
NOAA	National Oceanic and Atmospheric Administration
NRKW	Northern Resident Killer Whale

OLS	Ordinary Least-Squares
PCB	Polychlorinated Biphenyl
PTS	Permanent Threshold Shift
PVC	Percentage Volume Contour
RMS	Root-Mean-Square
RG Tool	Route Generator Tool
S-AIS	Satellite Automatic Identification System
SARA	Species at Risk Act
SEL	Sound Exposure Level
SL	Source Level
SPL	Sound Pressure Level
SRKW	Southern Resident Killer Whale
TL	Transmission L
TTS	Temporary Threshold Shift
UNCLOS	United Nations Convention on the Law of the Sea
UNCTAD	United Nations Conference on Trade and Development
VFPA	Vancouver Fraser Port Authority
VMS	Vessel Monitoring System

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CHAPTER 1 INTRODUCTION AND OVERVIEW

1.1 SOUND, NOISE, AND WILDLIFE

Exploring the complex interactions taking place among humans, wildlife, and the environment requires a deep understanding of the space in which these interactions occur (Wiens et al., 1993). Space, however, is not solely defined by its tangible and visible elements, but also by its sonic characteristics (Schafer, 1994). Introduced by Michael Southworth in 1969, the term soundscape (Southworth, 1969) defines all the acoustic energy produced by the biotic, abiotic and anthropogenic components of the landscape. A soundscape emerges from the overlap of three distinct sources of sonic energy: geophonies (e.g. wind, waves, ice), biophonies (e.g. vocalizations, alarm calls, songs) and anthrophonies (e.g. cars, airplanes, vessels) (Farina, 2014; Pijanowski et al., 2011). Geophonies are generally undisturbed by anthropogenic activities. Biophonies and anthrophonies, on the other hand, display opposite trends. Along a gradient of growing anthropogenic presence (i.e. moving from pristine to urbanized environments), biophonies tend to decrease while anthrophonies tend to increase. Advancements in the fields of acoustics and soundscape ecology allowed us to recognize that anthrophonies are not confined to human-dominated environments only, but also extend beyond inhabited areas on land (e.g. noise from roads, railways and flyways) (Potvin, 2017; Ware et al., 2015) and permeate our oceans (Hildebrand, 2009; P. L. Tyack, 2008).

Within the conceptual framework of soundscape ecology, noise is defined as an unintentional sound, usually containing a low level of information and masking or interfering with an acoustic signal (Farina, 2014). Due to its subjective nature, noise can originate from any of the three components of a soundscape (i.e. geophonies, biophonies, anthrophonies) (Farina, 2014). However, it is only the noise produced by anthropogenic activities that can assume the connotation of a pollutant. Since the beginning of the industrial age, anthropogenic activities have been altering existing natural soundscapes by generating increasing levels of noise pollution (Potvin, 2017). The alteration of natural soundscapes is a source of habitat degradation (Ware et al., 2015), has an influence on species interactions (Arévalo and Newhard, 2011; Francis et al., 2009), alters communities (Proppe et al., 2013), and interferes with foraging (Senzaki et al., 2016) and reproduction (Schmidt et al., 2014; Tennessen et al., 2014) for a multitude of terrestrial species. Similar consequences have also been documented in marine taxa (Carroll et al., 2017). However, since sound travels approximately five times faster in water (average of 1531 m/s in sea water at 20-25°C) (Engineering ToolBox, 2004) than in air (average 343 m/s in dry air at 20°C) (Engineering ToolBox, 2003), the magnitude of noise pollution impacts may be greater in marine than in terrestrial environments. For example, there are no records of terrestrial wildlife mortality events associated with the emission of anthropogenic noise, whereas the first documented evidence of noise-related mortality in marine species dates back to the early 2000s (i.e. the Bahamas stranding) (Balcomb and Claridge, 2001). The Bahamas stranding was not an isolated episode, and in the following years many cetacean mass mortality events occurred worldwide were linked to the use of military sonar (Parsons, 2017).

The exposure to acute sources of noise (e.g. navy sonar, high-power echosounders, air-guns, pile driving) has also been linked to several non-lethal effects such as hearing loss and increases in stress (Bailey et al., 2010; Erbe, 2012). Animals exposed to air-gun noise can experience, depending on their proximity to the source and on the duration of the exposure, temporary (TTS) and permanent (PTS) auditory threshold shifts. Estimating the level of sound required for the onset of TTS in marine mammal species provided the basis for the development of regulations relative to the mitigation of acute noise pollution in Germany and the US (Erbe, 2012; NOAA, 2016). Currently, TTS are used to quantify, predict and mitigate the potential consequences of several noisy activities such as oil and gas exploration (Mikhail, 2016) and offshore constructions (Bailey et al., 2010). The exposure to chronic sources of oceanic noise (e.g. ships) has been related to other non-lethal effects, such as behavioral disruption (Wisniewska et al., 2016), communication masking (Holt et al., 2011; Holt et al., 2009), and echolocation masking (Veirs et al., 2016). Yet, understanding the population consequences and evaluating the effects of chronic noise exposure still represents a challenge for both researchers and marine managers. Oceans are currently pervaded by shipping noise, making comparative studies (i.e. before-after the introduction of shipping noise) particularly rare. For example, thanks to an unexpected reduction in the number of vessels navigating the Bay of Fundy immediately after the tragic events of 11 September 2001, Rolland et al. (2012) were able to provide the first evidence that vessel-noise may be related to chronic stress in baleen whales. The drop in vessel traffic resulted in a 6 dB reduction of the low-frequency underwater noise recorded in the region. This reduction in underwater noise levels was then identified as the cause of a drop in the concentration of stress-related hormones (i.e. glucocorticoids) for the North Atlantic Right

Whale population (Rolland et al., 2012). Another possible approach for the assessment of the effects of chronic noise exposure is the estimation of the energetic cost of disturbance (Noren et al., 2016). Disturbance induces ephemeral behavioural responses that have been estimated to increase daily energy consumption by approximately 4% for several odontocete species (Noren et al., 2016). Furthermore, by affecting feeding behaviour, disturbance has the effect of reducing prey acquisition, a response now considered ubiquitous in cetacean species (Senigaglia et al., 2016).

Marine mammals are not the only marine taxa affected by noise pollution. Responses to the increasing levels of oceanic noise have been observed in fish (Holt and Johnston, 2014; Simpson et al., 2016), crustaceans (Wale et al., 2013), cephalopods (McCauley and Fewtrell, 2008), and planktonic species (McCauley et al., 2017), indicating that there is a growing need for methods, mitigation measures, and regulatory frameworks to assess the levels of noise exposure for marine species and address the threats noise pollution poses to their conservation.

1.2 Vessel Noise

Approximately 90% of the global trade relies on commercial vessels and in less than 50 years the world's cargo fleet showed a six-fold increase in its capacity (UNCTAD, 2017). In 2016, the world's commercial fleet included 90,917 vessels (UNCTAD, 2017). Although underwater explosions (e.g. SL > 300 dB re 1 μPa), airgun arrays for seismic exploration (e.g. SL ~ 260 dB re 1 μPa), pile-driving hammers (e.g. SL ~ 237 dB re 1 μPa for a 1000 kJ hammer) and Navy sonar (e.g. 235 dB re 1 μPa for a US Navy 53C ASW sonar) are known to be the loudest sources of anthropogenic noise in our oceans (Hildebrand, 2009), vessels are the most widespread, generating sound over longer periods of time and larger areas (P. L. Tyack, 2008). Nonetheless, only limited knowledge relative to the sources of radiated noise from modern vessels is currently available (Arveson and Vendittis, 2000; McKenna et al., 2012; Wittekind, 2014).

Noise radiates asymmetrically from a vessel, with recorded noise levels 5-10 dB re 1μ Pa higher on the stern than on the bow, as demonstrated for six different vessel classes transiting in the Santa Barbara Channel (US) by McKenna et al. (2012). Vessels' noise emissions vary depending on ship design and operational conditions (McKenna et al., 2013a), however, not all the parameters influencing vessel's radiated noise have been identified and quantified (Wittekind, 2014). Propellers and the associated machinery (i.e. diesel generators) are responsible for the majority of the noise produced by modern vessels. More specifically, propeller noise is mainly attributed to cavitation, the formation of steam-filled bubbles as a consequence of the negative pressures generated by the propeller's blades (Wittekind, 2014). Noise from propeller cavitation peaks at frequencies of 50-150 Hz but also extends to frequencies higher than 10,000 Hz (McKenna et al., 2013a). Machinery noise is related to the vessel's diesel generator as well as to its hull design. These two factors are highly variable from one ship to another, making their detection and quantification particularly challenging (Wittekind, 2014). Although ship size, speed and time of the year have been identified as good predictors of vessel noise, the heterogeneity of the fleet introduces high variability in noise emissions from one vessel to another, even within the same class (McKenna et al., 2013a). The age of the fleet is another relevant factor to consider. For example, it has been suggested that removing 15% of the oldest and noisiest vessels could lead to a 3 dB reduction in the overall noise radiated by the fleet navigating the Salish Sea, British Columbia, Canada (Holt et al., 2009; Veirs et al., 2018).

When assessing cumulative vessel noise pollution, understanding how noise is generated at the source represents only the first step. When noise leaves the vessel and propagates into marine environments, several factors can affect its propagation, including bathymetry, geoacoustic properties of the seafloor, and sound speed profiles (MacGillivray et al., 2017). Furthermore, an assessment of vessel noise requires information relative to fleet composition as well as knowledge of vessel traffic spatial and temporal trends. For these reasons, in the past, estimating noise pollution for a multitude of vessels navigating over large regions has been incredibly challenging. The introduction and diffusion of vessel tracking devices, such as the Automatic Identification System (AIS) and the Vessel Monitoring System (VMS), allowed for the collection of large volumes of vessel movement data including not only geographical positions, but also vessel speeds, draughts, headings and navigation times. Although not perfect, AIS and VMS records allowed researchers to apply acoustic propagation models over large scales and for multiple moving sources.

Currently, noise from vessels, but also other sources of anthropogenic noise in the ocean (Mikhail, 2016), are assessed through modeling studies (Chion et al., 2017; Erbe et al., 2014, 2012; Keyel et al., 2017). Modeled noise levels are commonly reported using three metrics: Sound Pressure Level (SPL), Sound Exposure Level (SEL) and Time-averaged Sound Pressure Level (L_{eq}).

These three metrics are usually expressed in decibels (dB), a logarithmic unit defined as:

$$L = 10 \ \log_{10} \frac{I}{I_{ref}} ,$$

where L defines a generic "level", and I_{ref} represents a reference value established by convention (Caruthers, 1977). For the propagation of sounds in air the reference sound pressure is 20 µPa, whereas in water the reference pressure is 1 µPa (IEEE, 1996). Although SPL, SEL and L_{eq} are all commonly reported in decibels, these three metrics express very different acoustic quantities. Representing an instantaneous measure of the time-varying sound pressure, the peak SPL is used to evaluate exposure for sounds characterized by transient components, and is recommended by NOAA as the metric to measure exposure to impulsive sounds, such as the high-intensity pulses produced by airguns arrays used for seismic exploration (NOAA, 2016). SEL is defined as the constant sound pressure corresponding to the same amount of energy attributed to the time-varying SPL of the original acoustic event, for a reference duration of 1 second. SEL is a metric used to assess cumulative noise exposure and provides an indication of the total sound energy received by an organism (Soloway and Dahl, 2014). However, the recommended application of cumulative SEL is the estimation of exposure levels from a single source and may not represent the ideal metric for the estimation of exposure from multiple sources (NOAA, 2016). L_{eq} represents the integration of the time-varying SPL over a specific time interval. Commonly, L_{eq} is either integrated over a 24h period or over specific portions of the day (e.g. day vs. night) (Brink et al., 2017). Since L_{eq} allows to estimate noise exposure from multiple sources, to account for the duration of the exposure, and to analyze large areas, it is frequently used in vessel noise modelling studies (Erbe et al., 2014; MacGillivray et al., 2017; O'Neill et al., 2017).

1.3 Vessel Noise Management

Noise management practices began to be adopted since 1995 when both the US and the UK introduced the first guidelines and thresholds aiming at the prevention of injury and behavioural harassment for marine mammals (Dolman and Jasny, 2015). Initial concerns were focused on the mitigation of acute events such as the use of Mid Frequency Navy Sonar (MFNS) and the use of airguns for seismic exploration. Another connotation of the early noise management practices was the mitigation of near-field effects from single sources rather than the adoption of broad-scale noise management plans (Dolman and Jasny, 2015). Mounting evidence that the effects of marine noise pollution extended beyond the near-field and that chronic exposure could lead to long-term population effects on cetacean as well as other species led to a shift in the approach taken to manage noise in the ocean. In response to concerns raised by the scientific community as well as well as by numerous conservation-oriented NGOs, different resolutions and statements of concern directed toward the mitigation or reduction of vessel noise have been issued by both international and national regulatory bodies. For example, starting from 2011 (Commission Decision 2010/477/EU) the Marine Strategy Framework Directive (MSFD, 2008/56/EC) in the EU included an indicator for the determination of Good Environmental Status (GES) for low-frequency ambient noise (11.2.1), which is largely attributed to vessels. In 2008 the International Workshop on Shipping Noise and Marine Mammals asked for a "reduction in the contributions of shipping to ambient noise energy in the 10-300 Hz band by 3 dB in 10 years and by 10 dB in 30 years relative to current levels" (Wright, 2008). This target was subsequently recognized and adopted by the Scientific Committee of the International Whaling Commission (IWC, 2009). In April 2014 the International Maritime Organization released voluntary guidelines for the reduction of underwater noise from commercial ships (IMO, 2014).

Despite the growing effort, vessel noise remains largely unregulated worldwide. Yet, noise pollution in the ocean is a global environmental issue with a possible solution. Vessel-noise is mainly caused by propeller cavitation, and, as recognized by the scientific community and by the International Maritime Organization, the adoption of source-quieting technologies could greatly improve the acoustic quality of marine environments (Weilgart, 2010; IMO, 2014; Dolman and Jasny, 2015). However, the development of effective quieting technologies would require: i) a deeper understanding of the relationship between noise and propeller cavitation; ii) The correlation of noise measurements with datasets containing tracks and other vessels' operational conditions (e.g. AIS); iii) the implementation and testing of quieting measures on individual vessels; iv) the quantification of the relationship between ship noise reduction and ambient noise levels (Southall et al. 2017). Another factor that could limit the implementation of quieting technologies is represented by the time and cost required to retrofit the existent fleet. Since the average lifetime of a large commercial vessel is 25 years (Dinu and Ile, 2015), and in the absence of mandatory regulations, retrofitting the existing fleet could take decades. Although beneficial in the long-term for many species, endangered cetacean species (e.g.

the North Atlantic Right Whale (NARW)) and populations (e.g. the Southern Resident Killer Whale (SRKW)) could experience a severe decline during the transition towards quieter technologies. For example, a population viability study by Lacy et al. (2015) showed how the cumulative effects of oil spills, pollution, decline in prey availability, and vessel disturbance could lead SRKW towards complete extinction (8.6% probability) or to quasi-extinction (i.e. < 30 individuals surviving in the wild) (53.5% probability) within the next 100 years. To prevent the further decline of these two species while waiting for new quieting technologies, short-term solutions based on the introduction of guidelines and on the implementation of spatial management practices are needed.

With respect to vessel-noise, there are two main approaches that are currently being explored in North America: the implementation of mandatory regulations (USA) and the introduction of voluntary rules (Canada - CA). The adopted spatial measures include: speed limitations (i.e. Salish Sea - CA, Saguenay St-Lawrence Marine Park - CA, Glacier Bay National Park - USA), safety distances (i.e. Salish Sea - USA), no-go areas (i.e. Saguenay St-Lawrence Marine Park), routing restrictions (i.e. Saguenay St-Lawrence Marine Park, Glacier Bay National Park), and vessel quotas (Glacier Bay National Park). An example of mandatory rule introducing restrictions to the distance between animals and vessels is represented by the federal vessel regulation adopted by the USA, in 2011, for the protection of SRKW from disturbance and noise (Holt et al., 2017). The regulation forbids vessels to approach killer whales along their path within 366 m or within 183 m if approaching from any other direction (NOAA, 2011). In this case, no specific zone is defined, and the regulation applies to all killer whales found within the inland waters of Washington State.

An alternative approach is the one followed by the Vancouver Fraser Port Authority (VFPA). In 2017, the VFPA was the first port authority to explore the introduction of voluntary speed limitations as well as the introduction of incentives for vessels equipped with "quieter" technology (VFPA, 2018) aimed at reducing acoustic disturbance for SRKW in the Salish Sea. In this case, the voluntary slow-down was implemented within a portion of the Salish Sea located in Haro Strait, a key foraging area for SRKW (Scott-Hayward, 2015; Hauser et al., 2007). Another example is represented by the voluntary measures adopted in the Saguenay-St. Lawrence Marine Park for the protection of the cetacean species commonly found in the area (Chion et al., 2017). The protection measures include a no-go area, a speed reduction area and a recommendation to navigate in the northern portion of the St. Lawrence estuary. The measures were introduced with the objective of reducing the occurrence of lethal collisions between commercial ships and baleen whales (i.e. *Balaenoptera musculus*, *Balaenoptera physalus*) without increasing the level of noise exposure for the St. Lawrence beluga population (*Delphinapterus leucas*). In Glacier Bay National Park, the park regulations allow transit to a limited number of vessels per day (e.g. two cruise ships, three tour vessels, and 31 charter and private vessels per day), and speed limits are implemented in areas characterized by a high probability of cetacean presence (McKenna et al., 2017).

The introduction of spatial management practices in the planning of shipping operations could help reduce the degree of overlap between these anthropogenic sources of noise and marine mammal populations (Dolman and Jasny, 2015). However, the benefits deriving from the application of these measures, in terms of reduction of noise exposure for cetacean species, need to be carefully evaluated. For example, the speed restriction adopted in the Saguenay St-Lawrence Marine Park resulted in a 40% drop in the risk of lethal collisions for baleen whales, but also in a 2.4% increase in the cumulative noise received by beluga whales (Chion et al., 2017). In the US portion of the Salish Sea, the introduced vessel regulations were found not to be significant predictors of the noise levels experienced by SRKW, whereas the number of vessels and their speed resulted to be significant predictors (Holt et al., 2017). In conclusion, although the adoption of spatial management practices for the reduction of noise exposure has the potential to improve the current conditions, adequate tools and frameworks for their design and evaluation are needed (Chion et al. 2017; Stelzenmüller et al. 2013; Dolman and Jasny, 2015).

1.4 THE SOUTHERN RESIDENT KILLER WHALE POPULATION

As discussed in Section 1.1, chronic noise pollution is suspected of having negative impacts on the survival and health of Cetacean species. Such negative impacts could, in the case of endangered species (i.e. the North Atlantic Right Whale (NARW)) and populations (i.e. the resident killer whales) hinder the conservation efforts aimed at their recovery. Followed by researchers since the early 1970s, resident killer whales are one of three *Orcinus orca* ecotypes inhabiting the north-eastern Pacific (Ford, 1994) and, most likely, one of the most studied ecotypes across the globe. In contrast with other ecotypes which mainly feed on marine mammals, resident killer whales are fish-eaters, and their population dynamics is strictly connected to the population dynamics of their elective prey (Ford et al., 2010). Resident killer whale populations are also characterized by a highly organized social structure. Matrilines, groups constituted by all the surviving members of a female lineage, constitute the fundamental unit of resident killer whale societies, and individuals belonging to the same matriline use specific vocalizations to communicate with each other (Miller and Bain, 2000). Closely related matrilines are then associated into pods, groups of orcas that invest a significant portion of their time socializing, feeding, traveling and resting together (Bigg et al., 1990). Initially believed to be as stable as matrilines (Bigg et al., 1990), pods are now considered to be dynamic associations that evolve through time (Parsons et al., 2009). Each pod of resident killer whales has distinctive vocalizations; however, some calls are shared across pods. Pods that share similar vocalizations are considered to be part of a high-order social unit, the acoustic clan (Ford et al., 2000). Resident killer whales in British Columbia are organized in two distinct communities: the Northern Resident Killer Whale (NRKW) and the Southern Resident Killer Whale (SRKW). NRKW is constituted of three clans and 34 matrilines while SRKW consists of a single clan, the J clan (Ford et al., 2000). The J clan includes three pods, namely J, K, and L, which comprise 20 matrilines in total (Ford et al., 2000).

Currently, all the individuals constituting the Northern and Southern Resident Killer Whale populations have been identified and cataloged through photo identification studies. Deaths, births and other life-history events have been documented for this population since the early 70s. When first censused in 1974, NRKW was strong of approximately 120 individuals, while a total of 70 individuals constituted the SRKW population. Due to their particularly low population numbers and to the increasing anthropogenic pressures threatening their survival, SRKW has been designated as endangered and protected under the Species at Risk Act (SARA) as early as 2002 (COSEWIC, 2008). Considered to be growing in numbers in the first half of the 1960s, SRKW population size dropped dramatically between the late 1960s and the early 1970s. This first decline was attributed to the development of a unique type of fishery: the live capture of killer whales. It is estimated that, over the period 1964-1973 a total of 47 individuals belonging to SRKW, half of which did not survive in captivity for more than a few years, were taken from their natural environment and transferred into aquaria and amusement parks (Bigg, 1975). Live captures of killer whales were banned in 1973, and, over the years following the ban, SRKW started to rebound. With a 19% increase over the period 1973-1980, SRKW went from 70 to 83 individuals. From 1981 to 1984, as a consequence of diminished birth rates and increased mortality rates of reproductive females and juveniles, SRKW experienced another phase of decline that brought the population to 74 individuals. This second phase of decline was the result of selective captures of mature individuals occurring during 1964-1973 (Olesiuk et al., 1990). During the following 10 years (1985-1995), SRKW experienced a peak in the number of mature individuals, which was accompanied by reduced mortality rates and increased birth rates. In 1995, SRKW reached 99 individuals, a number that has not been surpassed, or even approached, over the past 22 years. As of June 16, 2018, SRKW accounts for 75 free-living orcas and for one individual living in captivity, at the Miami Seaquarium. More specifically, the L pod currently includes 34 individuals, while the J and K pods account for 23 and 18 individuals, respectively.

Decline in chinook salmon populations (*Oncorhynchus tshawytscha*), physical disturbance caused by sources of anthropogenic noise (e.g. vessels, aircrafts), and high levels of contaminants are thought to be the main threats preventing SRKW from reaching the 2.3% yearly growth rate that would allow the full recovery of the population (DFO, 2017a; Lacy et al., 2017). Although vessel noise is generally considered to be associated with chronic, sub-lethal effects, the combined effects of noise disturbance, food deprivation and high concentration of toxins are currently hindering SRKW's recovery. Even though chinook's abundance is the main driver of SRKW population growth, achieving the 2.3% recovery goal by solely increasing prey availability would require reaching abundances that have not been observed in British Columbia's salmon populations since 1979 (Lacy et al., 2017). Completely removing chemical pollution, leaving the other threats unaltered, may lead to a 0.3 % growth rate, which would not be enough to achieve SRKW's recovery target (Lacy et al., 2017). Similarly, eliminating acoustic disturbance for SRKW, leaving the other threats unaltered, may lead to a 1.7% growth rate. Consequently, the achievement of full recovery for this endangered population requires the adoption of synergic management strategies. For example, a 15% increase in chinook abundance and a 50% reduction of noise disturbance may result in a 2.3% growth rate for SRKW (Lacy et al., 2017). However, determining the effects of chronic exposure to noise for large free-ranging marine animals is particularly challenging (Holt et al., 2009) and our knowledge relative to the impacts of vessel noise on SRKW is still limited. Recent studies started to shed some light on this issue. In 2009, Holt and colleagues (Holt et al., 2009) provided the first evidence that members of SRKW respond to increased background noise levels by increasing the loudness of their acoustic signals. This phenomenon, known as Lombard effect, is a common response of many mammals to increased levels of noise, including humans (Lombard, 1911; Tiesler et al., 2015). For example, noise levels in a classroom are worsened by the Lombard effect, leading to increased stress, fatigue, and loss of concentration in both students and teachers (Tiesler et al., 2015). Similarly, bats alter the frequency and amplitude of their echolocation signals as a response to increased ambient noise (Hage et al., 2013). Avoidance is another noise-induced response that has been documented for several species (Hastie et al., 2017; Luo et al., 2015; Potvin, 2017). For example, harbour porpoises avoid areas where tidal turbines noise is particularly loud (Hastie et al., 2017).; although, avoidance is not always possible and some species may be forced to cope with a increasingly noisy environment (Potvin, 2017). This may be the case of SRKW, for which key foraging areas overlap with areas of intense vessel traffic. SRKW's feeding behaviour is disrupted by the presence of vessels (Lusseau et al., 2009) and orcas increase source levels of calls to cope with growing noise levels (Holt et al., 2011). SRKW's feeding behaviour may also be affected by high-frequency vessel noise, which overlaps with orca echolocation sounds (Veirs et al., 2016). Even though high-frequency noise is usually attenuated over short distances, members of SRKW are often seen in proximity to multiple vessels (Lusseau et al., 2009; Veirs et al., 2016). Consequently, members of SRKW may lose foraging opportunities due to disturbance from high-frequency vessel noise. Another recognized consequence of prolonged exposures to chronic noise pollution is the increase in the concentration of stress-related hormones (Blickley et al., 2012; Kaiser et al., 2015; Rolland et al., 2012). Although evidence of the physiological consequences of noise exposure for cetacean species are still limited (Rolland et al., 2012), recent studies showed that the combined effects of food deprivation and vessel disturbance increase SRKW physiological stress and may be at the root of the low reproductive success of this population (Ayres et al., 2012; Wasser et al., 2017). In conclusion, even though the quantification of vessel noise impacts is still challenging, and further research is needed, it is clear that noise pollution contributes to the degradation of SRKW's habitat. For this reason, SRKW is the first Canadian endangered marine mammal species for which specific vessel noise management strategies are currently being explored (DFO, 2017b).

1.5 RESEARCH QUESTIONS

Ocean shipping is currently the most energy efficient type of long-distance transport for the mobilization of large amounts of goods (Rodrigue et al., 2006). Shipping routes across the globe form a complex network in which traffic flows following unidirectional, circular routes. While advantageous for the global exchange of goods, this highly clustered network is also a vehicle for the distribution of pollutants and the spreading of invasive species. Projections on the growth of the global commercial fleet predict a doubling in the number of ships over the next 15 years. This would correspond to an 87% increase in ocean noise emitted at the source (Kaplan and Solomon, 2016). At the same time, although recognized as a ubiquitous threat to the conservation of marine megafauna (e.g. exposure to chemicals, noise, risk of strike) (Jarvela Rosenberger et al., 2017; Rolland et al., 2012; Williams and O'Hara, 2010), vessels are rarely actively managed. Of the 1076 Marine Protected Areas (MPAs) that have been established for the protection of marine mammals across the globe (Hoyt, 2011), only 33 employed active vessel management tools (McWhinnie et al., 2018). And even smaller is the number of studies assessing the effects of vessel management strategies on the acoustic environment (Chion et al., 2017; Holt et al., 2017; McKenna et al., 2017). The development and implementation of successful management strategies require two key achievements: the production of sound scientific knowledge (McShea, 2014) and the establishment of a steady flow of information from the researchers to the decision-makers and back (Cvitanovic et al., 2016). Marine planning inherently requires integrating a spatial component in data processing and analysis. Such integration, as well as the establishment of a flow of information between researchers and decision-makers, could both be achieved through the development of Geographic Information Science (GIS) based planning and decision-making tools (Stelzenmüller et al., 2013).

Noise pollution management can play a central role in the successful recovery of SRKW (Fisheries and Oceans Canada, 2011; Lacy et al., 2017). However, the implementation of effective noise management strategies would require:

1. spatially explicit information relative to vessel traffic and vessel-noise emissions;
2. the combination of vessel-noise emissions with the estimated distribution of cetacean populations;
3. the development of analytical frameworks and practical tools to support the management of vessel-noise.

The present thesis combines information relative to the spatial distribution of SRKW with information on the heterogeneous distribution of commercial vessels and their relative contribution to noise pollution in the Salish Sea. Using these approaches, it aims at answering the following research questions relative to the impacts of vessel noise on SRKW:

1. What is the modeled level of noise exposure from commercial vessels experienced by SRKW within their critical habitat?
2. Which classes of commercial vessels are driving the level of noise exposure for SRKW?
3. What is the spatial distribution of noise exposure for specific vessel classes?

Furthermore, ten distinct populations of cetacean commonly found in the Salish Sea (i.e. Dall's porpoise; Pacific white-sided dolphin; minke whale; humpback whale; transient killer whale, offshore killer whale; northern and southern resident killer whale) and a total of 22 different vessel classes, defined by AIS, navigate and emit noise within the same environment. This complex reality makes the identification of areas, species, and vessel classes toward which management efforts should be focused on a challenging task for planners and decision-makers. Consequently, a fourth research question arises:

4. How can an analytical framework and a set of GIS tools help explore and analyze data relative to noise pollution from shipping and cetacean species distribution?

1.6 RESEARCH GOAL

This project is part of the MEOPAR funded Noise Exposure to the Marine Environment (NEMES) project. It aims at bridging the gap between acoustic modeling and spatial decision making for the adoption of effective noise mitigation measures to promote the recovery of SRKW. More specifically, the objectives of this research are twofold. First, we aim to identify which classes of commercial vessels should be targeted by specific management strategies and where, within the Salish Sea, noise mitigation measures could be put in place to support the recovery of SRKW. Second, we aim to develop an analytical framework, applicable to other cetacean species, which can help identify noise exposure hotspots, assess vessels' relative contribution to the modeled cumulative noise, estimate noise exposure levels within the identified hotspots, and explore and compare of possible management solutions.

1.7 CASE STUDY

SRKW was listed as Endangered under the Canadian Species at Risk Act in 2001 (COSEWIC, 2008) and under the United States' Endangered Species Act (ESA) in 2005 (Federal Register / Vol. 70, No. 222). As discussed in Section 1.3, the SRKW population has not been showing signs of recovery over the past two decades, and, among other anthropogenic causes, disturbance from vessel presence and noise are deemed to be one of the drivers of the observed decline (Lacy et al. 2015, DFO 2016). During the summer and fall SRKW is commonly sighted in the Salish Sea (Fig. 2.1 B), with the highest number of observations located in Haro Strait, Boundary Pass, along the eastern portion of the Juan de Fuca Strait, and in the southern portion of the Strait of Georgia (Fig. 2.1 C) (DFO, 2016). At the same time, the Salish Sea hosts one of the largest coastal human populations in Canada. It includes major shipping lanes and ports such as Vancouver in British Columbia, and Seattle in Washington State, and is a renowned touristic destination. This results in a large and heterogeneous volume of vessel traffic navigating in the Salish Sea, ranging from large merchant vessels (e.g. Containers, Bulkers, Tankers) to recreational, passenger, and whale watching vessels (e.g. ferries, Roll-on/roll-off Vehicle Carriers).

Especially during the summer, there is a high degree of overlap between vessel traffic and SRKW within the waters of the Salish Sea. An average of 20 vessels per day are found in the proximity of members of SRKW, and during the peak of the summer season, this number can rise up to 50 vessels per day (Holt and Noren, 2009). The large volume of traffic and the dependency of SRKW on the Salish Sea make this an ideal case-study for the development of spatial management practices (i.e. methods, analysis frameworks, and tools) aimed at minimizing the degree of overlap between anthropogenic sources of noise and an endangered marine mammal population.

1.8 RESEARCH METHODS

To answer the research questions presented in Section 1.4, the information contained in a fine scale Kernel Density Estimation (KDE) (Worton, 1989) describing SRKW summer core areas was combined with the output of a cumulative vessel noise model (O'Neill et al., 2017) (i.e. Research question 1). Cumulative distribution functions (CDF) (Nicholson, 2014) were used to evaluate SRKW's noise exposure levels from 15 vessel categories over three zones located within SRKW summer core areas and the resulting median cumulative noise values were used to group categories based on their associated exposure levels (i.e. Research question 2). Exposure maps were then used to assess the spatial distribution of noise pollution for the different vessel classes (i.e. Research question 3). A framework for the analysis of species' exposure to noise from shipping was then developed and integrated into a set of geovisualization tools (i.e. Research question 4). The proposed framework was built around the concepts of exposure mapping (Lahr and Kooistra, 2010), the use of CDF for the computation of probabilistic levels of a pollutant's exposure (Jin et al., 2015; Uddh-Söderberg et al., 2015; Zandbergen and Chakraborty, 2006), and on the application of a least-cost path (LCP) analysis for the identification of shipping routes minimizing the overlap between vessels and cetaceans within the study area. The framework was applied to estimate noise exposure for the SRKW population, and for the exploration of possible ship traffic displacement scenarios in the Salish Sea. The geovisualization tools where developed using Python 3.0 programming language and ArcGIS Pro and allow potential users to apply the framework to species-vessel combinations that were not explored in the present thesis.

1.9 THESIS OUTLINE

Chapter 2 reports on a study that aimed at assessing noise exposure levels from shipping for the endangered SRKW population. The study is organized in two parts. The first portion focuses on the estimation of SRKW's summer distribution in the Salish Sea. The second portion focuses on the results of a cumulative noise modeling study and their integration with SRKW summer distribution for the assessment of the population's levels of vessel-noise exposure. Building on the results described in Chapter 2, Chapter 3 presents a framework and a set of geovisualization tools developed to support marine spatial planners in the decision-making process relative to the issue of anthropogenic noise pollution from shipping. The study first describes the three analytical components of the framework: Exposure Mapping; Probabilistic Level of Exposure; Generation of Ship Traffic Scenarios. Then a description of the corresponding geovisualization tools and examples of possible applications are provided. The studies presented in Chapter 2 and 3 are brought together in Chapter 4, and the main results, methodological challenges, and limitations are critically discussed.

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1.11 Co-authorship Statement

This thesis was written in manuscript format with the goal of submitting the two core chapters (Chapter II and Chapter III) for peer-reviewed publication. The candidate led the writing of all the chapters, and feedback provided by the candidate's supervisor and co-authors contributed to the final version of the manuscript.

Authors' roles:

Chapter 1 was written by the candidate and reviewed by Dr. Rodolphe Devillers and Dr. Rosaline Canessa.

Chapter 2 was written as a paper that was submitted on October 7th, 2017, for publication to the Marine Pollution Bulletin peer-reviewed journal. The candidate was the primary author, led the study, ran the analysis, and prepared the manuscript for publication. Dr. Harald Yurk provided input on the methodological approach adopted to estimate the Southern Resident Killer Whale (SRKW) population distribution maps, helped with the interpretation of the acoustic modeling study results, and provided feedback throughout the preparation of the manuscript. Alexander MacGillivray provided input for the methodological approach adopted in the estimation of SRKW's spatial risk of noise exposure, support in the interpretation of the acoustic modeling study results, and feedback throughout the preparation of the manuscript. Dr. Rodolphe Devillers supervised the candidate, provided input to the study design and contributed to the preparation and review of the manuscript. Dr. Lauren McWhinnie and Dr. Rosaline Canessa contributed to the preparation and review of the manuscript.

Chapter 3 was written as a paper that was submitted on April 25th, 2018, for publication to the Ocean and Coastal Management peer-reviewed journal. The candidate was the primary author, designed the proposed analytical framework, and developed the geovisualization tools presented in the manuscript, run the analysis for the application examples, and prepared the manuscript for publication. Dr. Brent Hall and Dr. Michael Leahy provided input in the design of the analytical framework and in the scripting of the tools, guidance to the Candidate by sharing their expertise in the use of Python programing language and GIS software, and provided feedback throughout the preparation and review of the manuscript. Dr. Rodolphe Devillers supervised the candidate, provided input to the study design and contributed to the preparation and review of the manuscript.

Chapter 4 was written by the candidate and reviewed by Dr. Rodolphe Devillers and Dr. Rosaline Canessa.

Chapter 2 Noise Exposure from Commercial Shipping for the Southern Resident Killer Whale population

2.1 Introduction

It is thought that millions of species inhabit our oceans and seas (Appeltans et al., 2012; Mora et al., 2011), an environment where sound is often the most effective means to transmit and receive information (Simmonds et al., 2014). Sound can be used, depending on the species, for the perception of features in the environment, such as underwater topography and prey or predator detection (Simpson et al., 2016, 2015), or to help support complex social interactions, such as mating, competition and cooperation (Bruintjes and Radford, 2013; Eskelinen et al., 2016). Similarly to many acoustically active terrestrial species (Brumm and Todt, 2002; Penna et al., 2005; Schaub and Schnitzler, 2007), signaling and audition of marine species have evolved in environments with sometimes high levels of natural background noise (Baumann-Pickering et al., 2015; Foote and Nystuen, 2008; Holt et al., 2011). For example, many fish species show preferences for specific soundscapes and respond to changes in the natural background noise levels by increasing the loudness of their signals, a phenomenon called the Lombard effect (Filiciotto et al., 2013; Holt and Johnston, 2014; Lugli, 2014). Furthermore, evidence of the role of sound and background noise for crustacean species are also starting to be documented. Marine tidal turbine noise was shown to affect the length of estuarine crabs' time to metamorphosis (Pine et al., 2016) and anthropogenic noise has been linked to behavioral, physical and physiological effects in several invertebrate species (Carroll et al., 2017). The effects of anthropogenic noise can also extend to the lower levels of the trophic chain. A study conducted off the southern coast of Tasmania, Australia recently showed how air-gun noise might cause high mortality in plankton species as far away as 1.2 kilometers from the source (McCauley et al., 2017).

During the past 50 years, the increase in human activities in our oceans has caused a progressive increase in background noise levels in various marine ecosystems (Chapman and Price, 2011; McDonald et al., 2008). Sounds produced by Seismic explorations, navy sonar exercises, pile driving for offshore construction, ice-breaking, and commercial or recreational vessels have all been recognized as sources of anthropogenic noise that occur in addition to natural ambient sounds (Hildebrand, 2009; Merchant et al., 2012; Cosens and Dueck, 1993).

The past two decades of research on the impacts of anthropogenic noise have shown that noise caused by human activities can affect several aspects of a species' life cycle. Responses from exposure to anthropogenic noise range from the alteration of animal's physiology (Habib et al., 2007; Nichols et al., 2015), to modifications and disruption of its anti-predatory, reproductive and feeding behaviors (Meillère et al., 2015; Schmidt et al., 2014; Voellmy et al., 2014). Amid the known anthropogenic sources of noise in the oceans, commercial shipping is the most ubiquitous. According to the United Nations Conference on Trade and Development (UNCTAD 2017), commercial shipping represents approximately 90% of the global trade occurring worldwide, a number that is expected to grow in the future. In less than 50 years, the world's cargo fleet showed a six-fold increase in capacity, from the 262,070 thousand of deadweight tonnage reported in 1968 (UNCTAD, 1969), to the 1.8 billion thousand reported on January 1st, 2016 (UNCTAD, 2017). Currently, the world's commercial fleet accounts for 90,917 vessels (UNCTAD, 2017).

Amongst all marine species, marine mammals and more specifically, cetaceans, are considered to be highly susceptible to sound and impacted by noise. While the complexity and intensity of acoustic activity may vary amongst individuals, groups, populations, and species (Au et al., 2000; Perrin et al., 2009), the production and perception of sound permeate every aspect of their life-cycles. Odontocetes (i.e. toothed whales) use echolocation to perceive the surrounding environment and to identify and pursue prey (Geisler et al., 2014; Gutstein et al., 2014), while several species of Mysticetes (i.e. baleen whales) are known to produce elaborate mating calls and songs (Payne and McVay, 1971; Delarue et al., 2009; Garland et al., 2013; Paniagua-Mendoza et al., 2017). Evidence of the use of sound in complex social interactions exist for both groups, such as feeding calls during foraging bouts called bubble-net feeding produced by humpback whales (Megaptera novaeangliae) (Friedlaender et al., 2011) and group hunting in killer whales (*Orcinus orca*) (Van Opzeeland et al., 2005). Furthermore, acoustic communication plays a role in mother-calf interactions (Vergara and Barrett-Lennard, 2008; Videsen et al., 2017) and in the transmission of social behavior from one generation to the next through vocal learning (Janik, 2014; Reiss and McCowan, 1993). As a consequence, changes to the soundscape experienced by these animals could potentially have a negative effect on their survival (Harwood et al., 2016; Videsen et al., 2017).

Anthropogenic noise has the potential to cause adverse impacts when its frequencies overlap with the frequencies of a species' audiogram, the spectrum of acoustic frequencies that can be perceived by the animals' auditory system (i.e. hearing range). Large commercial vessels generate noise with most energy being emitted at frequencies below 1 kHz. Mysticetes, hear and produce sounds in a similar range of frequencies, and are considered vulnerable to noise from shipping (Southall et al., 2007). Odontocetes, signaling using higher frequencies and having lower sensitivity to low-frequency sounds, are generally considered less impacted than mysticetes by low-frequency noise (Southall et al., 2007). Nonetheless, recent findings suggest that odontocetes' sensitivity to noise from shipping might have been underestimated (Dyndo et al., 2015; Aguilar Soto et al., 2006). In particular, a study undertaken in the Haro Strait (Fig. 2.1C), located along the Canada-US border, documented how ship noise within the core habitat (Fig. 2.1B) of the endangered Southern Resident Killer Whale (SRKW) population raised background noise levels (91±4 dB re 1 μPa) not only in the low-frequency domain, but also for high frequencies, with an increase of 5-13 dB re 1 μPa in the 10 kHz to 40 kHz band (Veirs et al., 2016). As argued by Veirs et al. (2016), sound from shipping may not only mask killer whale communications but can also interfere with their echolocation signals within a range of several km around the noise source. Such interference has the potential to lower survival rates and lower reproductive success of individuals, and, in the long term, may affect the survival and dynamics of the entire population (Harwood et al., 2016). High extinction risk was the reason SRKW were listed as endangered and protected under Canada's Species at Risk Act (SARA) and the United States' Endangered Species Act (ESA). Furthermore, the majority of the Salish Sea has been recognized by both Canada and the US as a critical habitat for SRKW (NOAA, 2006; DFO, 2011). Yet, the designated areas only delineate the limits of SRKW's critical habitat at the time of the designation and extensions to the protected habitat are currently under consideration (DFO, 2017a). Like all resident killer whales, the members of SRKW are socially organized into clans, pods, and matrilines. Matrilines, consisting of a mother and all her offspring, travel and forage in close proximity to each other throughout their lives, while pods are temporally stable social groups that consist of related matrilines and share most of their vocal repertoires. Clans comprise pods that share calls and are therefore considered to be acoustically related (Ford 1991). SRKW consist of one clan (J-clan) and three pods (J, K, and L) (Bigg et al., 1990; Parsons et al., 2009). The complex social organization of this population is thought to influence SRKW spatial distribution within their critical habitat (Hauser et al., 2007). Hauser et al. (2007) investigated the spatial distribution of SRKW, identifying shared areas among all SRKW as well as pod-specific core areas for this population.

Only 76 SRKW individuals survive in the wild (www.whaleresearch.com/orca-population) (Center for Whale Research, 2017) and several anthropogenic activities undertaken within the Salish Sea are threatening the persistence of this population. Both the survival and the reproductive success of SRKW's individuals have been linked to prey availability (Baird 2001, Krahn et al. 2002, Ward et al. 2009, Ford et al. 2009). SRKW's diet is largely composed of salmonid species, Chinook mainly, but also Steelhead, and occasionally Sockeye, Chum, and Coho salmon (Hanson 2010, Ford and Ellis, 2006). As concluded by Williams et al. (2011), the current decline of both SRKW and their elective prey, and the transboundary nature of these two species represent a challenge for their successful conservation. Being framed around the concept of production optimization for the benefit of both the Us and Canada Salmon fisheries, the objectives of the Pacific Salmon

Treaty are in contrast with the current conservation goals for SRKW in both countries (Williams et al., 2011). SRKW was estimated to consume 12-23% of Fraser River Chinook in the summer and a fully recovered population could consume up to 20-40% of the available Chinook (Williams et al., 2011).

Other examples of current threats are the high levels of contaminants observed in members of SRKW (Krahn et al., 2009, 2007; Ross et al., 2000) and the physical and acoustic disturbance caused by vessel traffic (Holt et al., 2009; Houghton et al., 2015; Lusseau et al., 2009; Veirs et al., 2016).

With Canada's Policy for Conservation of Wild Pacific Salmon far from being fully implemented (Price et al., 2017) and the limited actions that can be undertaken to lower SRKW's level of contaminants, disturbance from vessel traffic arguably represents the only major environmental stressor for SRKW that could be addressed in the short-term. Both the US Recovery Plan for Southern Resident Killer Whales (National Marine Fisheries Service, 2008) and the Canadian Recovery Strategy for the Northern and Southern Resident Killer Whales (DFO, 2011) recognize the potential impact that noise could have on the recovery of SRKW. In particular, the new action plan to achieve recovery of the threatened and endangered Northern and Southern resident killer whales (DFO, 2016) explicitly introduces noise as a threat to the recovery of British Columbia's killer whale populations and specifies a list of action measures that should be put in place to reduce disturbance from anthropogenic noise to the acoustic habitat of killer whales and the marine environment.

As part of the MEOPAR (Marine Environmental Observation Prediction and Response Network) funded Noise Exposure to the Marine Environment (NEMES) project, this study investigated the predicted levels of noise exposure modeled from commercial vessel traffic within SRKW's summer core areas and aimed to inform managers and decision-makers on the spatial distribution of noise and whales in specific locations of the Salish Sea. Our goal was to identify areas in the Salish Sea where high levels of noise from shipping and high probability of SRKW presence co-occur. This was done by combining fine scale Kernel Density Estimation (KDE) of the SRKW population's core habitat with the output of a cumulative vessel noise model (O'Neill et al., 2017) that was informed about vessel density and distribution by Satellite Automatic Identification System (S-AIS) records.

2.2 METHODS

2.2.1 STUDY AREA

In 2013, British Columbia (BC) waters accounted for more than 50% of the ship traffic density occurring nationally (Simard et al., 2014). With major ports like Vancouver, Prince Rupert in BC and Seattle in Washington State (WA) serving major Canadian and USA economic centers, the distribution of shipping along the southern BC coast is mostly concentrated within the Salish Sea, an inland sea encompassing Canadian and USA national waters (Fig. 2.1 and 2.2). The Salish Sea extends from Olympia (WA, USA) in the South to Campbell River (BC, Canada) in the North (Barrie et al., 2014). It covers an area of 16,925 km^2 and includes 7,470 km of coastline (Gaydos et al., 2008). The complexity of the Salish Sea ecosystem is also reflected in the geomorphology of the region. The Salish

Sea's landscape was formed during a succession of geological events that shaped the Southern coast of BC into an intricate network of waterways. The Salish Sea region hosts the largest coastal population in Canada, with consequent high levels of coastal development. The Salish Sea is not only an area characterized by intense human activity but also a hotspot for marine biodiversity. Previous studies identified 172 species of birds and 37 species of mammals in this region (Gaydos and Pearson, 2011), as well as 253 species of marine fishes (Pietsch and Orr, 2015) that are highly dependent on this ecosystem for the full expression of their biological functions. The increase in human pressure on the Salish Sea ecosystem is threatening its biodiversity. The consequences of anthropogenic activities are reflected by the growing number of species, sub-species and ecologically-significant units of populations that are mentioned in provincial and federal lists of threatened species, both in Canada and the USA (Zier and Gaydos, 2016).

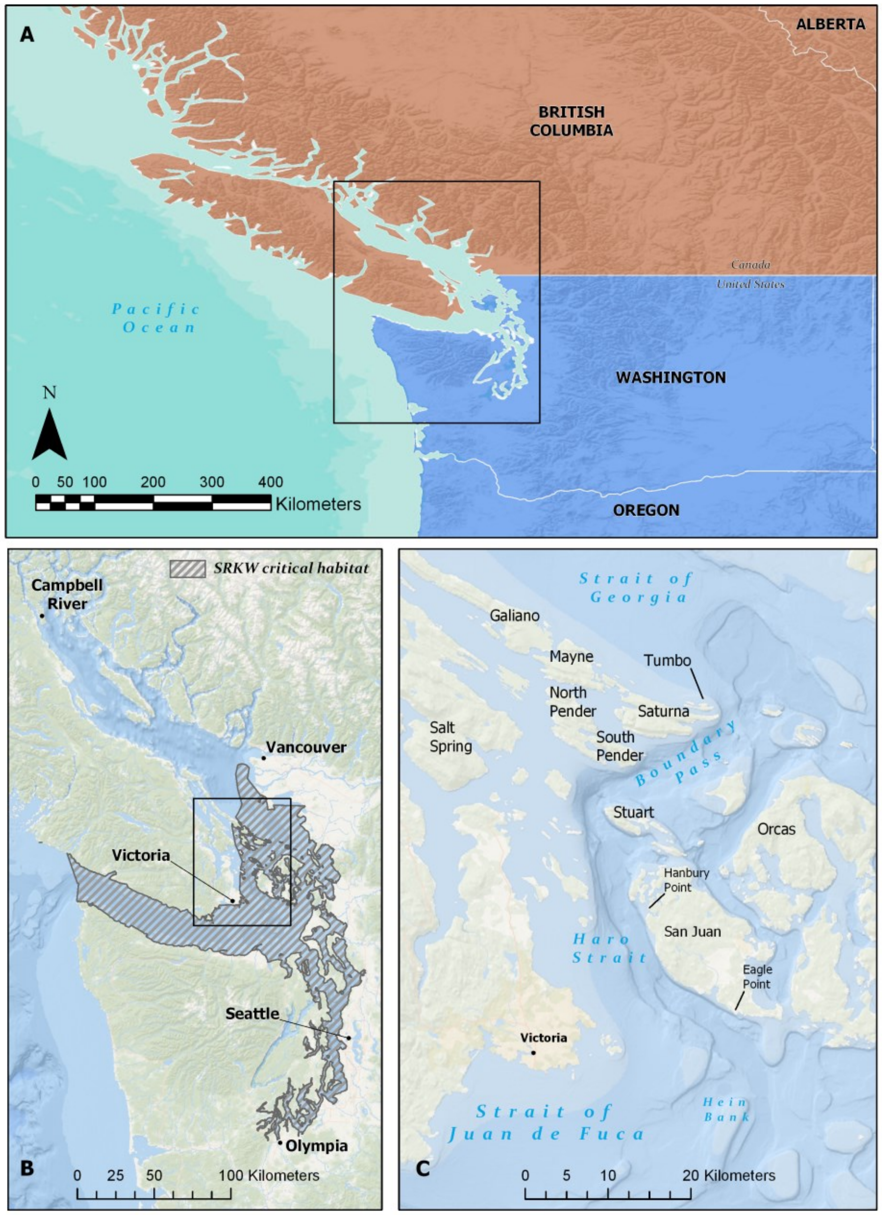

Figure 2.1. Canada's west coast (A), the Salish Sea and SRKW critical habitat (B) and the study area considered for the analysis of SRKW's levels of noise exposure (C).

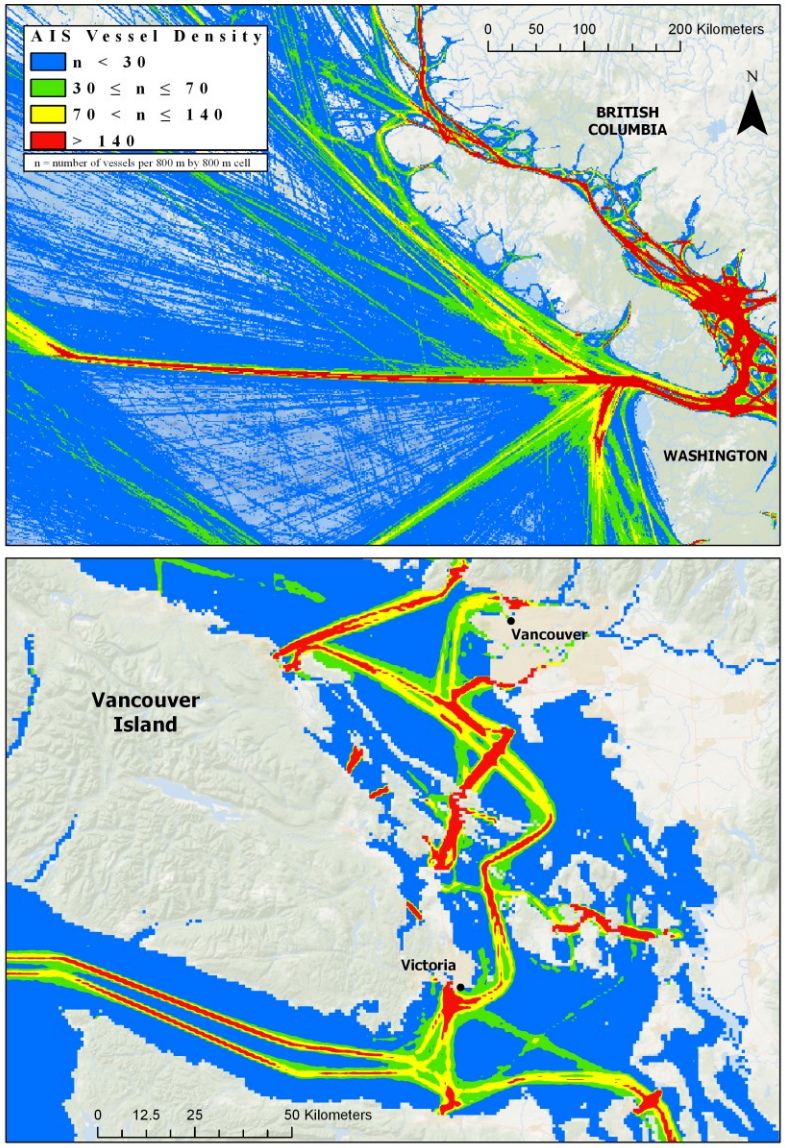

Figure 2.2. Map of aggregated AIS vessel density for the year 2015 (A). Map of aggregated AIS vessel density for the month of July 2015 (B). Both maps were derived from the same AIS dataset used by Jasco Applied Science to estimate levels of cumulative noise from shipping in the Salish Sea. The legend refers to both maps.

2.2.2 CETACEAN SIGHTINGS

We used SRKW sighting data collected by the Soundwatch Boaters Education Program (SBEP) between May 2011 and September 2014. These observations are part of a larger compilation of sightings, the "Southern Resident Killer Whale Sighting Compilation", produced by The Whale Museum (Friday Harbor – Washington, US). Sightings are collected from opportunistic platforms such as commercial whale-watching boats and private vessels as well as dedicated research vessels. This dataset contains a total of 83,474 records (1948 to 2014) describing the date, time and location of the sightings, the observed pod or pod combination and includes notes about the observed behavior of killer whales. Since the compilation collects data from observers with different levels of experience, the accuracy of pod designation may vary from one observer to another. Since 2009, each pod designation is accompanied by a "likely pod" designation determined by staff members of the Whale Museum, increasing the accuracy of the dataset. However, pod misidentifications cannot be completely removed and some of the reported SRKW sightings might be of individuals belonging to one of the other orca ecotypes (i.e. offshore, transient) present in the area. Sightings are summarized using a grid (Fig. 2.7E), with regular cells in open water and irregular ones within channels and in proximity of the coast Cell size is approximately 5 km by 5 km (i.e. quadrants) and the observations are not corrected for search effort. SBEP provides detailed geo-referenced sightings which are not aggregated in quadrants.

Another benefit of this dataset is the possibility to estimate the effort per unit area invested by the SBEP's volunteers from their yearly reports (Eisenhardt et al. 2012; Eisenhardt and

Koski 2011, 2013 and 2014). For each season of operation, SBEP reports the number of vessels contacted in each one of the 444 quadrants in which the Salish Sea is subdivided. Where a "contact" consists of SBEP's volunteers approaching boaters to inform them on the best practices for the operation of vessels in the proximity of marine mammals. The relationship between the number of sightings and the number of contacts recorded per hour was tested using Spearman's correlation's test. The area of each quadrant, as well as the number of vessels contacted within it, were computed using Esri ArcMap 10.3.1. The number of contacts divided by the total area of a quadrant provided an estimation of the effort per unit area invested by the volunteers in each quadrant (Figs. 2.7, A to D). Cetacean sightings were recorded at 30-minute intervals throughout the summer, applying the same protocol followed for SRKW's sightings compilation. Hence, the number of vessel contacts per unit area can be considered as a proxy for the search effort invested by SBEP in collecting cetacean presence data.

The SBEP dataset consists of 13,179 sightings collected during 16 years of activity in the Salish Sea. The purpose of this study was to estimate the probability for SRKW to be exposed to certain levels of noise by area, under the current intensity of ship traffic. Since the noise model outputs hereby considered are representative of the summer season, only relatively recent sightings, collected for the period May-September 2011 to 2014 were used for the identification of SRKW's summer core areas, totaling 3150 sightings. The derived effort ($E_{z,y}$) was computed for each year as follows:

$$E_{z,y} = \frac{N_{z,y}}{A_{z,y}}, \qquad [1]$$

where $N_{z,y}$ is the average number of contacts occurring in quadrant z for the y season and $A_{z,y}$ is the total area of quadrant z for the y season. As a consequence, before proceeding with the creation of the summer core area maps, each sighting occurring in a quadrant was divided by the corresponding $E_{z,y}$ value computed for the zone. Sightings recorded within a quadrant with $E_{z,y} = 0$ were considered as being "off effort" observations and excluded from the analysis. Assuming that quadrants with no contacts are "off effort" introduces a limitation: some quadrants might have no contacts but still, be highly frequented by SRKW.

To test whether or not the resulting KDE relative to the entire population was descriptive of SRKW summer distribution, the final results were compared to the British Columbia Cetacean Sightings Network (BCCSN) (http://wildwhales.org) dataset. Established in 2000 and hosted by the Vancouver Aquarium, the BCCSN is a network of more than 6000 volunteer observers distributed across British Columbia. Contributors include whale watching naturalists, lighthouse keepers, commercial mariners and recreational boat operators, as well as researchers. The information collected through the network is shared with government agencies, universities and ENGOs for the conduction of conservation research. For example, Williams and O'Hara (2010) used the information collected by the BCCSN to compile a list of the known ship strikes events involving BC cetacean species from 1999 to 2007. The sightings collected by BCCSN's volunteers were instrumental to the delineation of SRKW's critical habitat within Canadian waters (DFO, 2011) and were used by the Port Metro Vancouver to inform the environmental assessment for the Robert's Bank Terminal 2 project (Wood et al., 2014).In the BCCSN dataset, effort-weighted summer sightings of orcas (Rechsteiner et al., 2013) (i.e. resident and transient together) collected since the early 1980s are reported for the same 444 quadrants used by SBEP. The correlation between the KDE produced in this study and the number of sightings per-unit effort reported by BCCSN was tested using the ArcGIS software ordinary least squares (OLS) geoprocessing tool.

Each SRKW sighting reported by SBEP is accompanied by a pod designation. A member of the SRKW population can belong to one of three socially distinct units (i.e. J, K or L pods). Furthermore, individuals often form mixed groups, where one or more members of a pod are typically observed within a larger group of individuals belonging to another pod. As a consequence, there is a total of seven pod combinations recorded in the SBEP dataset: J, K, L, JK, KL, JL, JKL. Considering each combination as a separate social entity within SRKW would have greatly reduced the number of sightings available for the estimation of SRKW summer areas. In order to produce an area estimation representative of the entire population, as well as of its three main social groups, each sighting was assigned to one of three clusters: J-group, K-group, and L-group. Where the J-group included the J pod as well as the JK and JL pod combinations, the K-group included the K pod and the KL pod combination, and where the L group included the L pod and the JKL combination. Such group compositions were first described and used by Hauser et al. (2007).

2.2.3 SRKW SUMMER CORE AREA ASSESSMENT

Kernel Density Estimation (KDE) Kernel Density Estimation (KDE) is a well-established approach for assessing habitat use (Worton, 1989). KDEs have been used to derive home ranges for several terrestrial and aquatic species from a variety of data sources: radio-tracked animals (Tumenta et al., 2013), indirect signs of presence (Sawyer, 2012), photo-identification records (Rayment et al., 2009) and visual surveys (Hauser et al., 2007). KDEs were computed using ArcGIS 10 Kernel Interpolation With Barriers geoprocessing tool, a variant of a first-order local polynomial interpolation which allows to obtain accurate estimates close to irregularly shaped shores (i.e. barriers). KDEs were computed for each group and for the entire population, applying a 5th-degree polynomial function:

$$1-\left(\frac{r}{h}\right)^3\left(10-\left(\frac{r}{h}\right)\left(15-6\left(\frac{r}{h}\right)\right)\right), for\ \frac{r}{h}<1. \qquad [2]$$

Where r is the radius, centered on a point within the study area, and h is the bandwidth or smoothing parameter. In order to allow a comparison of animals' home ranges and sound from shipping, the KDEs were computed at the same spatial resolution as the cumulative noise model (i.e. 800 m) (see Section 2.2.4). The boundaries of SRKW summer core areas were then identified using the 95% and 50% Percentage Volume Contours (PVCs) of each KDE. PVCs are a common measure of the extent of animal's home ranges (Garitano-Zavala et al., 2013; Sprogis et al., 2016; Tumenta et al., 2013). Since the outcomes of a KDE are highly dependent on the selection of the appropriate bandwidth (Worton, 1989), appropriate h values were selected following the method described in Kie et al. (2013). Starting from a reference bandwidth, h_{ref}, a set of bandwidth values ranging from 1.4 x h_{ref} to 0.1 x h_{ref}

was used to derive 95% percentage volume contours (PVC) for the entire population and each pod-group. The optimal bandwidth was then selected as the minimum value of h generating the least fragmented 95% PVC. Fragmentation of the 95% PVC was evaluated considering the number of polygons and the perimeter-area ratio (Fig. 2.3). Once an h value was selected for a pod-group, the corresponding KDE was designated as representative of the pod-group summer habitat-use while the other alternatives were discarded. In order to allow comparisons between the various KDEs, raster values were re-scaled between 0 and 1.

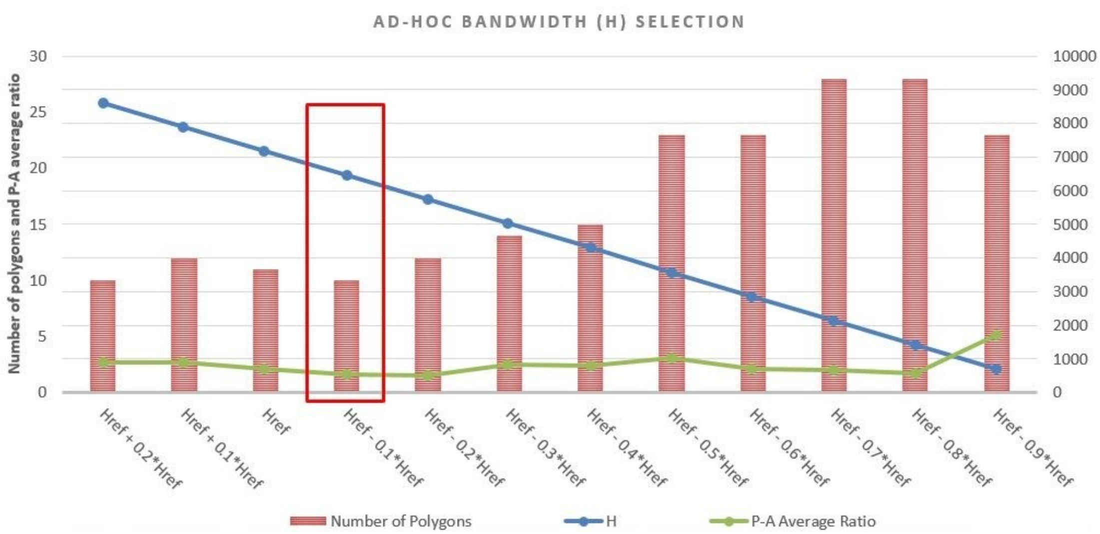

Figure 2.3. Bandwidth selection procedure. Bars represent the number of polygons constituting each iteration of a KDE. Bandwidth (H) values are represented by the blue line and the average perimeter-area ratio of the polygons constituting the 95% PVC of the KDEs are shown in green. The red box highlights the H value that led to the generation of the least fragmented 95% PVC, $H_{ref} - 0.1*H_{ref}$, in this case.

Representing a non-parametric approach to the evaluation of animals' home-ranges, KDEs do not require a priori identification of the sample's distribution (Anderson, 1982; Worton, 1989). However, the quantification of uncertainty for non-parametric methods is often problematic and not immediate. Bootstrapping (Briggs et al., 1997) was used to overcome this limitation, allowing to re-sample animal locations for the creation of confidence intervals. Applying an ArcGIS geoprocessing custom model developed for this study and starting from a randomized sample of the original animal's location, a total of 200 iterations of the KDE analysis were performed for the four summer core area maps. This allowed for the identification of a minimum and a maximum possible extent for each one of the 95% PVCs describing SRKW summer spatial distribution.

2.2.4 Cumulative Noise Assessment

A cumulative noise model generated by JASCO Applied Science for the NEMES project (O'Neill et al., 2017) was used to determine areas of high levels of noise exposure for SRKW. Vessel broadband source levels for the modeling study were compiled in 1/3 octave bands from 10 Hz to 63.1 Hz (Appendix A.1, Fig. 12 in O'Neill et al., 2017). Broadband vessel Source Levels (SLs) are shown in Table 1. One of the data inputs for the noise model was S-AIS ship movement data provided by exactEarth (http://www.exactearth.com). Originally developed as a navigation safety measure, the Automatic Identification System (AIS) allows for the tracking and modeling of large vessels' movements. According to Canadian's regulations every ship of 500 tons or more, fishing vessels excluded, needs to carry an AIS device. Raw S-AIS data were cleaned and processed by the Institute for Big Data Analytics (Dalhousie University, Canada), resulting in traffic density gridded maps for 22 different vessel categories. Vessels categories were defined using AIS types, when applicable, and using the International Telecommunication Union's Maritime mobile Access and Retrieval System (MARS) as well as Marine Traffic (www.marinetraffic.com) when the AIS records failed in reporting the class of a vessel. Grids (800 m resolution) recorded vessel counts, the total number of hours with vessels, and the average vessel speed for each cell. Traffic density data (i.e. vessel density, speed, source levels) were used by JASCO to determine ship contributions to ambient sound levels (i.e. background noise level). Vessel's sound source levels by category (e.g. tankers, container ships, bulk carriers, fishing vessels) together with the vessel density grids were entered into a sound propagation loss model which then together with other variables (e.g. bathymetry, sea surface pressure, geoacoustics and ambient noise), generated monthly cumulative noise distribution maps of the Salish Sea at an 800 m by 800 m grid scale (O'Neill et al., 2017). Source levels for each vessel class (in dB re 1 μPa @ 1 m) were specified in 1/3-octave frequency bands from 10 Hz to 63.1 kHz. Acoustic transmission loss (TL) for each 1/3-octave band was calculated using a parabolic-equation-based sound propagation model (JASCO's Marine Operations Noise Model, MONM), based on the computationally-efficient split-step Padé algorithm (Collins, 1993). TL was averaged over five frequencies inside each 1/3 octave band and the TL versus range curves were smoothed inside a 200 m window to remove fine-scale interference effects. At high frequencies, mean TL computed by MONM is expected to converge to a high frequency (i.e., ray-theoretical) limit; therefore, TL values for bands above 5 kHz were approximated by adjusting TL at 5 kHz to account for frequency-dependent absorption at higher frequencies (François and Garrison, 1982a, 1982b). MONM was used to pre-calculate curves of TL versus range for twenty different geographic zones, covering the study area, representing four different seabed types (i.e. sand, silt, clayey-silt, and sand-silt-clay) and five different depth ranges (i.e. < 50m, 50-100m, 100-150m, 150-200m, > 200m). For each geographic zone, TL was modeled using two different sound speed profiles, representing July and January conditions, and for two source depths, representing the nominal acoustic emission centers of small (2m) and large (6 m) draft vessels. The 1/3-octave band received SEL in each grid cell was computed as the total time-integrated squared sound pressure originating from all adjacent grid cells not blocked by land within a 75 km radius. For the range-dependent case, where the ray between a source and a receiver traversed more than one zone, the total TL was computed as the range-weighted average of the zone-dependent TL. The monthly L_{eq} in each grid cell was calculated from the SEL and the number of seconds in a single month, T_{mon}, as follows:

$$L_{eq} = SEL - 10\ x \log_{10}(T_{mon}). \qquad [3]$$

Monthly L_{eq} was calculated separately for each vessel category. The relative differences between category-specific L_{eq} at each geographic location provided a measure of the relative exposure risk from the different types of shipping, based on the overall noise budget.

In order to validate the results of the cumulative noise model, modeled received levels were compared to measured sound levels for several ships of opportunity on a hydrophone station located within the study area (Fig. 2.4). The validation results showed good agreement between the model predictions and received sound pressure levels (i.e. RMS model-data mismatch of 3.53 dB). However, due to the opportunistic nature of the validation process, not all the vessel categories could be assessed.

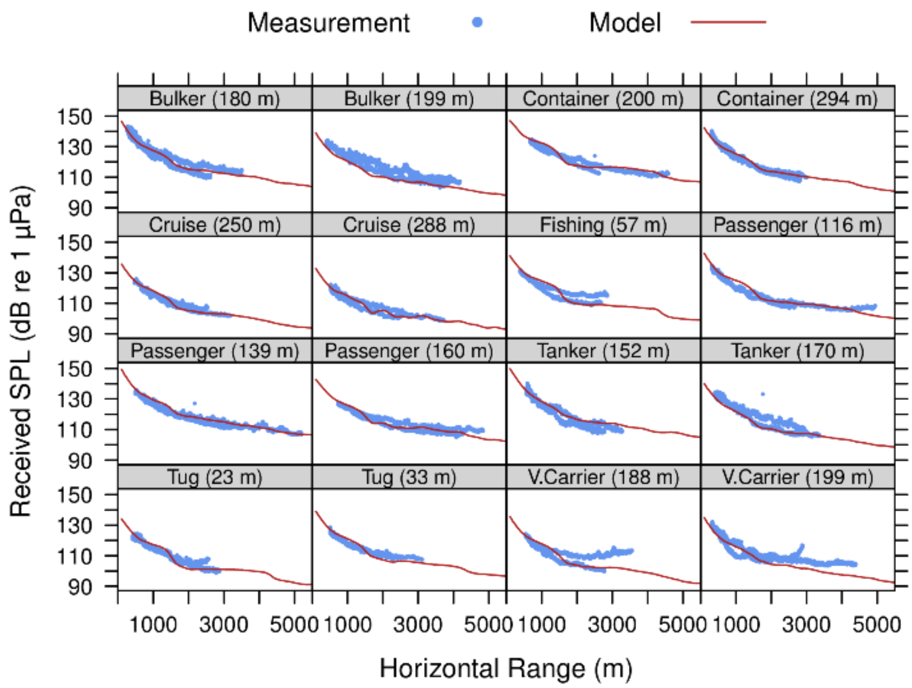

Figure 2.4. Comparison of measured and modeled SPL (broadband, 0.01-63 kHz) versus range, from 16 ships of opportunity measured on a cabled hydrophone station located in Georgia Strait. The type and length of each vessel are shown in the strip above the plots. Ships were tracked on AIS as they passed the hydrophone, and SPL was calculated in 1-second intervals from the acoustic data. The measurements show both the approach and departure of each vessel past the hydrophone, with higher levels generally measured during departure (i.e., in the aft direction).

The noise model results for the month of July 2015 were used here for a comparison with SRKW summer distribution. Noise models outputs (Fig. 2.5) are estimations of cumulative noise expressed in terms of Equivalent Continuous Sound Pressure Level (L_{eq}).

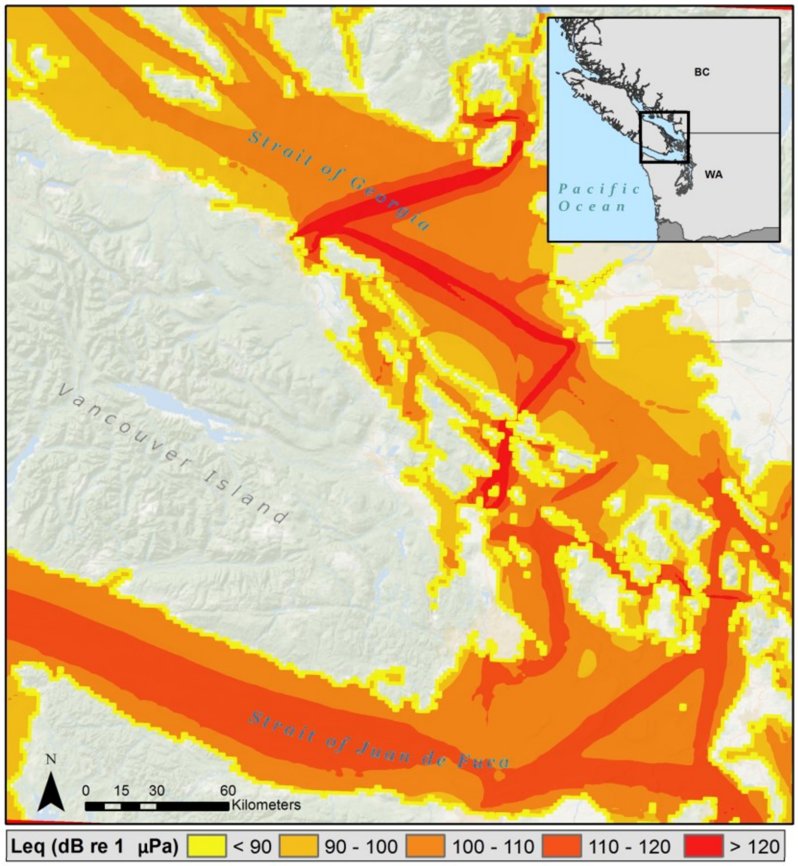

Figure 2.5. The output of the cumulative noise model produced by Jasco Applied Science. The map shows the cumulative L_{eq} values relative to all the ship categories combined.

Cumulative noise was mapped at 800 m resolution, providing a L_{eq} value for each vessel category and for all the categories combined. For the purpose of this study, the noise contributions of the initial 22 vessel categories identified in the cumulative noise model were reduced to 15 distinct groups (Tab.2.1) using the dB summation formula:

$$L_{eq_{(A+B+C+\cdots+n)}} = 10\ x\ log_{10}(10^{\frac{L_{eq_A}}{10}} + 10^{\frac{L_{eq_B}}{10}} + 10^{\frac{L_{eq_C}}{10}} + \ldots + 10^{\frac{L_{eq_n}}{10}}) \qquad [4]$$

One of the vessel categories, Dredgers, was excluded from the analysis because of its small AIS aggregated density and its localized contribution to the cumulative noise, resulting in a total of 14 pooled categories included in the noise exposure risk assessment.

Table 2.1. List of vessel categories included in the cumulative noise mode and the corresponding pooled categories used to evaluate SRKW' levels of noise exposure. Dredgers, marked with an *, were not included in the analysis.

Model Categories	Broadband SL (db re 1 µPa)*	Pooled Categories
Bulk Carriers <200m	167.1	Bulkers
Bulk Carriers >200m	170.9	
Container Ships <200m	178.6	Containers
Container Ships >200m	178.6	
Crude Oil Tankers <200m	161.2	Crude Oil Tankers
Crude Oil Tankers >200m	161.2	
Dredgers#	167.5	-
Ferries <50m	173.3	Ferries
Ferries >50m	173.3	
High-Speed Ferry	166.3	
Fishing Vessels	146.2	Fishing Vessels

Model Categories	Broadband SL (db re 1 µPa)*	Pooled Categories
Government/Research	146.7	Government/ Research
Naval Vessels	146.7	Naval Vessels
Passenger <100m	152.3	Passenger
Passenger >100m	166.3	
Recreational Vessels	144.3	Recreational Vessels
Reefers	170.9	Reefers
Tankers	161.2	Tankers
Tug <50m	167.5	Tugboats
Tug >50m	167.5	
Vehicle Carriers	170.9	Vehicle Carriers
Other	145.8	Other

* Representative source levels for the different categories of vessels used in the model were compiled from a number of publicly available papers and reports (Austin et al., 2013; Warner et al., 2013; Cybulski, 1977; Arveson and Vendittis, 2000; MCR International, 2011; McKenna et al., 2012; Kipple and Gabriele, 2004; Zykov et al., 2008; Mouy et al., 2012; Breeding et al., 1994; Veirs et al., 2016).

2.2.5 Spatial Noise Exposure Risk by Vessel Categories

The levels of exposure reported in this study represent the spatial distribution of SRKW's risk to be exposed to a certain cumulative noise level, from one of the 14 pooled vessel categories, within its summer core areas. The spatial noise exposure for SRKW pod-groups within their summer core areas was estimated by computing the cumulative distribution function (c.d.f.) of the L_{eq} values modeled for each vessel category over the KDE relative to the entire population.

Ship traffic within the study area is heterogeneous and varies from one region of the Salish Sea to the other (MacGillivray et al., 2017). Exposure levels were evaluated over three sub-areas (Fig. 2.6) capturing the different components of vessel traffic transiting through the study area. Located on the southern Gulf Island and outside of the commercial shipping lanes, Zone 1 (Fig. 2.6) is characterized by the presence of several ferry routes and frequently used by recreational as well as fishing vessels. Zone 2 (Fig. 2.6), located in Haro Strait, is an area characterized by high intensity of large commercial traffic. Zone 3 (Fig. 2.6), located in Boundary Pass and extending in the Strait of Georgia, is also characterized by high intensity of large commercial traffic. Zone 1 included the entire L-group's unique core area (Fig. 2.11B). Zone 2 included the core area common to all the three groups (Fig. 2.8B) as well as J-group's unique core area (Fig. 2.9B). Zone 3 included the entire K-group's unique core area (Fig. 2.10B).

Cumulative probabilities were computed as follows. First, the probability, P_i, of having an animal (or group of animals) in one of the cells constituting the KDE was computed, for each cell, as:

$$P_i = KDE_i / \sum_{i=1}^{n} KDE_i, \qquad [5]$$

where KDE_i was the value of the density estimation stored in cell i, and where n was the number of cells. Using Esri ArcMap 10.3 software, the sum of all the KDE cell values was obtained by multiplying the average value by the total number of cells. For a real random variable, *X*, the corresponding c.d.f. is given by:

$$F_X(x) = P(X \leq x) \qquad [6]$$

Where $F_X(x)$ represents the probability that the considered random variable, X, will assume a value equal or less than x (Nicholson, 2014). By substituting X and x with the modelled L_{eq} values (Equation 3) and *P* values with the $P_{x,y}$ values computed from Equation 5, Equation 6 was re-written as:

$$F_{L_{eq}} = \sum P_j \quad \forall\, j \text{ where } L_j \leq L_{eq} \qquad [7]$$

Where $F_{L_{eq}}$ is the cumulative probability of having an animal (or group of animals) exposed to a noise value equal or less than L_{eq}. To create the c.d.f. for each vessel category, v, a Python script was used to iteratively compute cumulative probability values using Equation 7 starting from $Leq = \min(L_{eq_v})$ and proceeding by 1 dB increases until $Leq = \max(L_{eq_v})$. This process allowed for the creation, for each vessel category as well as for all the categories combined together, of a set of points representing cumulative probabilities and corresponding L_{eq} values. From each distribution, L_{eq} values corresponding to the 5th, the 50th and the 95th percentiles were computed and used to compare vessel categories in terms of levels of exposure.

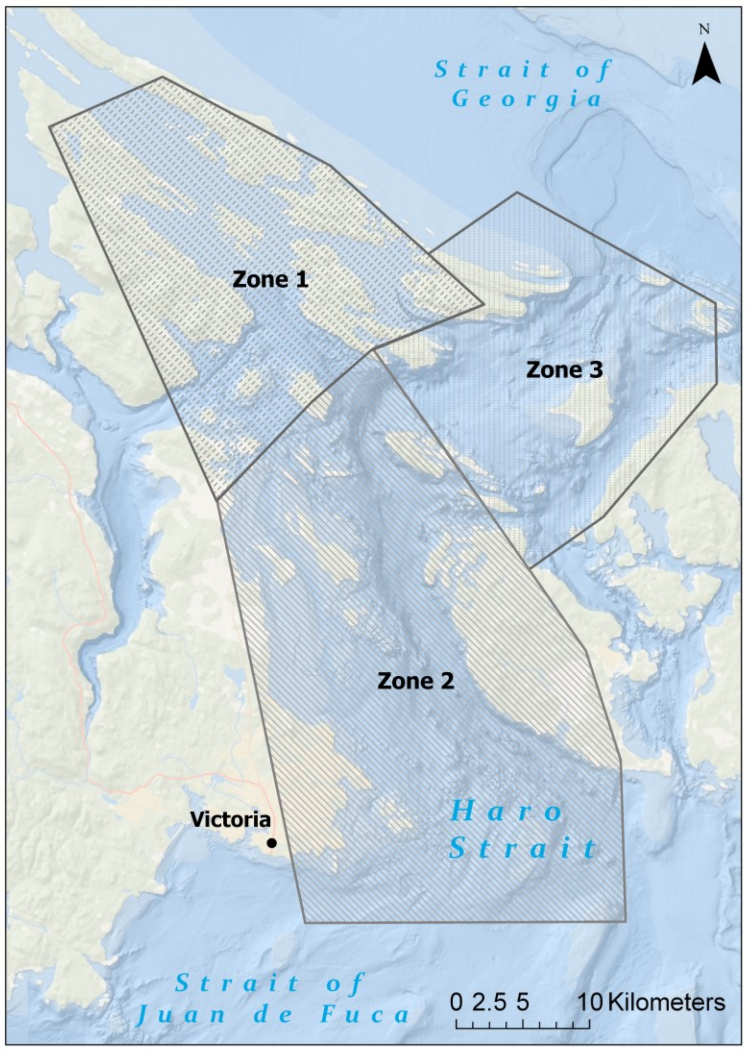

Figure 2.6. Zones over which the cumulative distribution functions (c.d.f.) of L_{eq} values over the KDE describing the entire SRKW population summer core areas were computed.

This approach allowed us to identify, within a specific area, the vessel categories associated with the highest median levels of exposure, $L_{eq-50^{th}}$, corresponding to the 50th percentile of the relative c.d.f.. The 5th and 95th percentiles were included to give an indication of the range of variation of the L_{eq} values attributed to a vessel category within a zone. Since there are no regulations or thresholds relative to the amount of noise from shipping considered harmful to SRKW, the L_{eq} values produced by each category were divided in three exposure level groups: $L_{eq-50^{th}} < 60\ dB\ re\ 1\ \mu Pa$; $60 < L_{eq-50^{th}} < 90\ dB\ re\ 1\ \mu Pa$; $L_{eq-50^{th}} > 90\ dB\ re\ 1\ \mu Pa$. The three exposure levels were used to reclassify the noise maps for the categories belonging to the $L_{eq-50^{th}} > 90\ dB\ re\ 1\ \mu Pa$ group.

2.3 RESULTS

2.3.1 SRKW SIGHTINGS

Of the initial 3150 sightings recorded by Soundwatch volunteers during the summer seasons from 2011 to 2014, only 2994 were retained for the creation of the KDEs (Table 2). A total of 156 observations, corresponding to 5% of the dataset, were removed because they were either incomplete or falling in quadrants with $E_z = 0$, thus considered as "off-effort" observations.

Table 2.2. SRKW sightings summarized by pod, pod-combination, and pod-group. The number of sightings and percentages relative to the total are reported. The J-group represents 60% of the sightings, with J-pod, JK and JL pod-combinations accounting for 47%, 36% and 17% of the J-group sightings, respectively. The K-group represents 5.5% of the sightings, with K-pod and the KL pod-combination accounting for 62% and 38% of the K-group sightings, respectively. The L-group represents 44% of the sightings, with L-pod and the JKL pod-combination accounting for 44% and 56% of the L-group sightings, respectively. Pod-groups were first defined and used by Hauser et al. (2007).

Pod-group	**Pod**			**Pod combination**				Total
	J	K	L	JK	JL	KL	JKL	
J-group	848 (47%)	-	-	660 (36%)	295 (17%)	-	-	1803 (60.23 %)
K-group	-	102 (62%)	-	-	-	62 (38%)	-	164 (5.47%)
L-group	-	-	457 (44%)	-	-	-	570 (56%)	1027 (34.30%)
Total (%)	28%	4%	15%	22%	10%	3%	18%	2994

SRKW sightings are not evenly distributed within the study area. Out of 444 quadrants, four quadrants (i.e. 175, 180, 183 and 185), located along the west and south-west coasts of San Juan Island, contain approximately 55% of the sightings. The remaining 45% is spread over 131 quadrants, with the majority of the quadrants (n=310) containing no sightings. The three distinct pods; J, K, and L, together totaled 47% of the sightings, while the pod combinations; JK, JL, KL, and JKL, accounted for 53%. Among the three pods, J (28%) is the most represented, followed by L (15%) and K (4%). Among the pod combinations, JK (22%), is the most represented, followed by JKL (19%), JL (10%) and KL (3%). Spearman's test results showed that the number of contacts and the number of sightings per hour were significantly correlated ($rho = 0.88$, $p < 0.05$). Quadrants adjacent to the south-west and to the south coast of St. Juan Island delineate the area where the number of contacted vessels reached the highest values (Fig. 2.7, A to D). Quadrants surrounding this area were characterized by an intermediate number of contacts, while the remaining quadrants were characterized by low numbers of vessel contacts (Fig. 2.7, A to D). The effort per unit area derived from Soundwatch boat contacts is variable among years as well as among zones. 2011 is the year with the highest average effort per unit area ($E_{z-2011} = 0.433$, $SD_{Ez-2011} = 0.893$), followed by 2012 ($E_{z-2012} = 0.381$, $SD_{Ez-2012} = 0.589$), 2013 ($E_{z-2013} = 0.306$, $SD_{Ez-2011} = 0.685$) and 2014 ($E_{z-2011} = 0.196$, $SD_{Ez-2011} = 0.167$). The high standard deviations reflect the wide variability in the effort per unit area among the different zones, ranging from the maximum value of 4.507 contacts per unit area recorded in quadrant 184, to the minimum value of 0.018 recorded in quadrant 122.

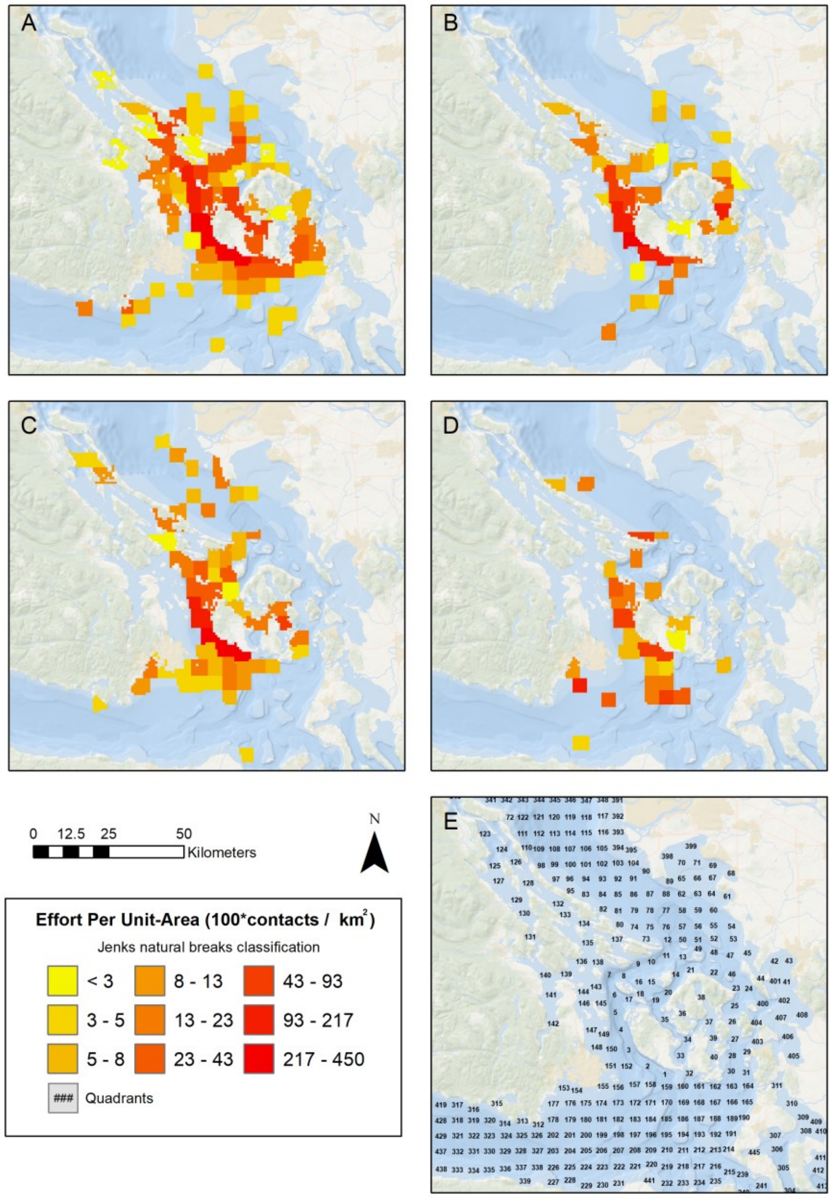

Figure 2.7. Distribution of Soundwatch effort across the study area for the years 2011 (A), 2012 (B), 2013 (C) and 2014 (D). Quadrants used to compile the Southern Resident Sighting Compilation (E).

2.3.2 SRKW SUMMER CORE HABITAT

Using a bandwidth selection method allowed for the identification of optimal h values for all four KDEs. The 95% PVC for the K-group was the least fragmented, followed by the entire population, and the J and L groups. The selected 95% PVCs were then used to estimate the full extent of SRKW summer core habitat. The 95% PVC for the entire population (Figs. 2.8 and 2.12A) showed the largest extent (i.e., 1805 km2). The corresponding minimum and maximum extents obtained from the bootstrap analysis resulted in 864 km2 and 3333 km2, respectively. The J-group 95% PVC (Figs. 2.9 and 2.12B) covered an area of approximately 1372 km2, with a minimum and maximum extent of 812 km2 and 2814 km2, respectively. The L-group 95% PVC (Figs. 2.11 and 2.12C) covered an area of approximately 1142 km2, with an estimated minimum and maximum extent of 446 km2 and 1541 km2, respectively. The K-group presented the smallest 95% PVC (Figs. 2.10 and 2.12D), covering an area of approximately 1218 km2. Minimum and maximum extent for this KDE captured a range of variation comprised between 180 km2 and 1007 km2, indicating that the estimated KDE might not be representative of the K-group core summer area (Fig. 2.12D).

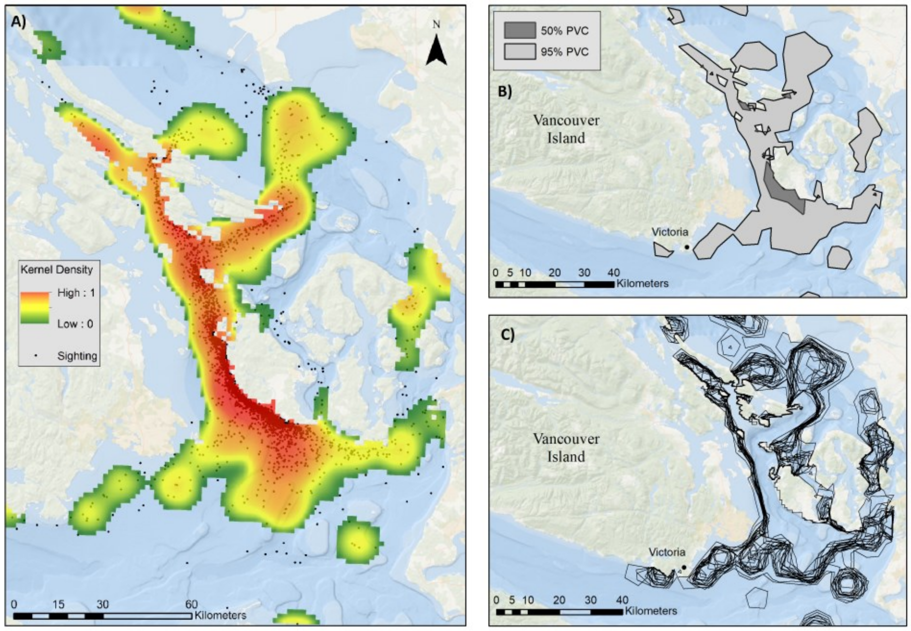

Figure 2.8. Results of the kernel density analysis for the entire SRKW population. A) KDE values within the 95% PVC and SRKW sightings. B) The extent of the 95% (light gray) and 50% (dark gray) PVCs. C) Results of the bootstrap procedure, for visualization purposes only the first 20 iterations are displayed.

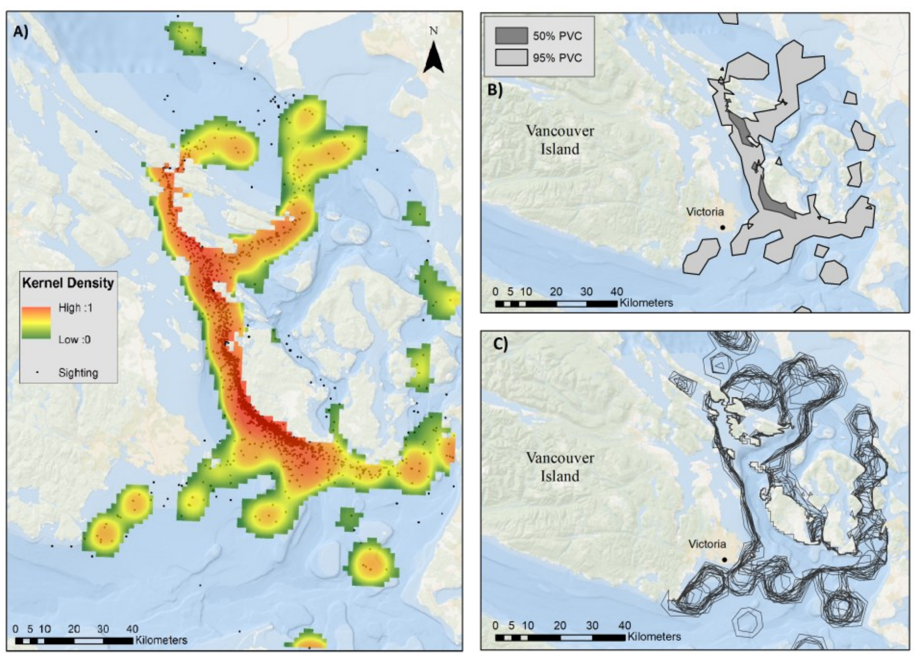

Figure 2.9. Results of the kernel density analysis for the J-pod. A) KDE values within the 95% PVC and J-pod sightings. B) The extent of the 95% (light gray) and 50% (dark gray) PVCs. C) Results of the bootstrap procedure, for visualization purposes only the first 20 iterations are displayed.

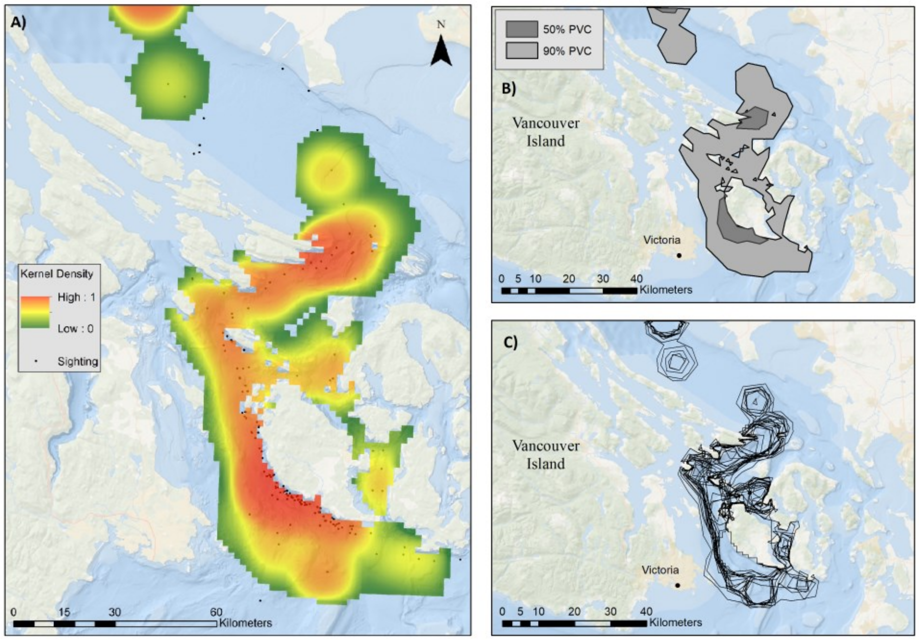

Figure 2.10. Results of the kernel density analysis for the K-pod. A) KDE values within the 95% PVC and K-pod sightings. B) The extent of the 95% (light gray) and 50% (dark gray) PVCs. C) Results of the bootstrap procedure, for visualization purposes only the first 20 iterations are displayed.

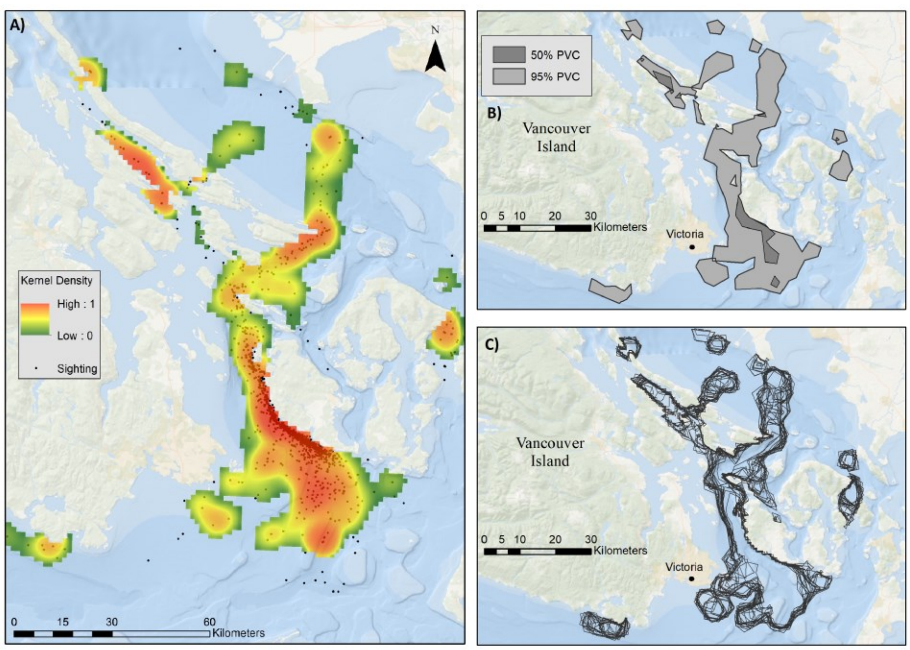

Figure 2.11. Results of the kernel density analysis for the L-pod. A) KDE values within the 95% PVC and L-pod sightings. B) The extent of the 95% (light gray) and 50% (dark gray) PVCs. C) Results of the bootstrap procedure, for visualization purposes only the first 20 iterations are displayed.

The south-western coast of San Juan was identified as part of the 50% PVC in each one of the KDEs. The 50% PVCs also included pod specific areas, one for each pod-group. The J-group 50% PVC also included an area of approximately 32 km2 extending from the northern shore of Stuart Island to the southern shore of Pender Islands. The K group 50% PVC also included an area of approximately 40 km2 located on the eastern outskirts of Tumbo and Saturna islands. The L-group 50% PVC also included an area of approximately 14 km2 located between the islands of Salt Spring in the south and Galiano in the North. The KDE of the entire population did not identify these pod-specific areas as high use areas and only identified the South-western coast of San-Juan as SRKW summer core area. The OLS analysis showed a positive correlation between the KDE relative to the entire population and the distribution of the BCCSN sightings-per-unit-effort ($R2 = 0.58$, $p < 0.05$). Since no pod designation was included in the BCCSN sightings-per-unit-effort map, the test could not be performed for the remaining three KDEs. For this reason, the evaluation of noise exposure levels was limited to the KDE relative to the entire population (Fig. 2.8A).

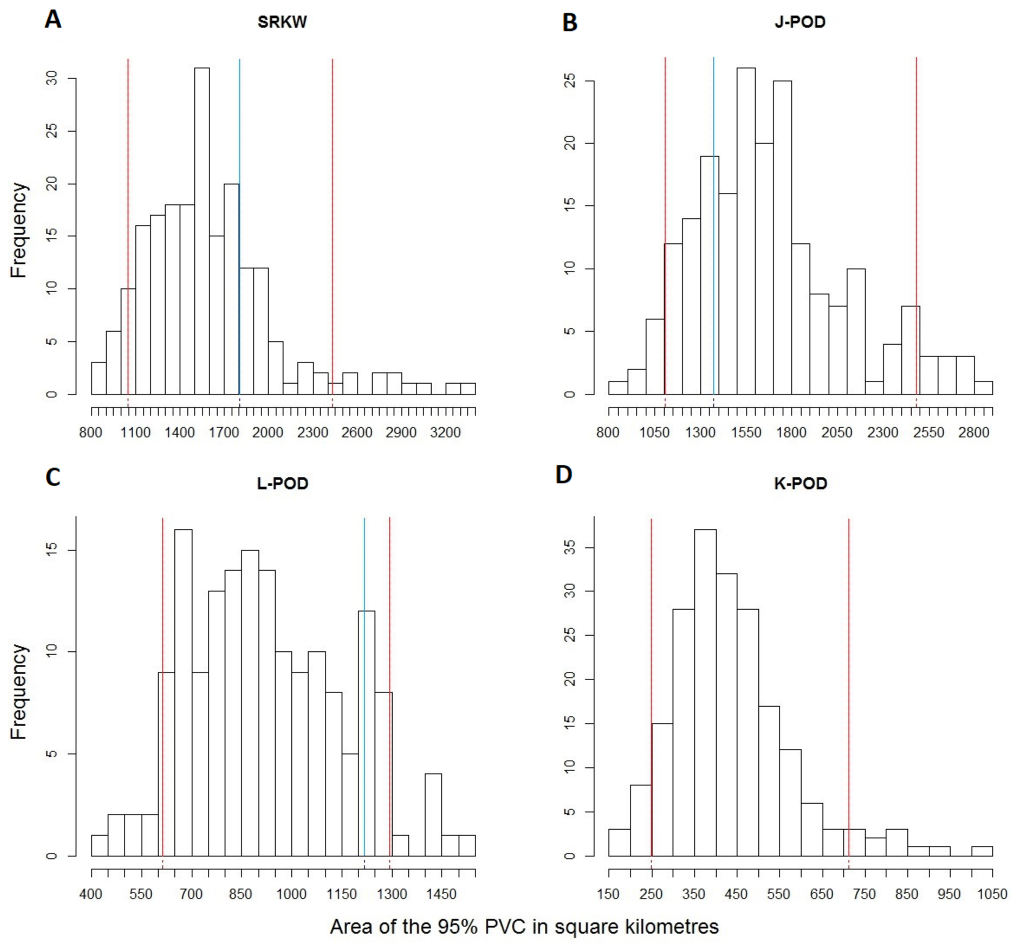

Figure 2.12. Frequency distribution of the 95% PVCs extent obtained from 200 iterations of the KDE bootstrap procedure for the entire population (A), the J-pod (B), the L-pod (C) and the K-pod (D). Red lines indicate the 5th and 95th percentiles of the distribution, blue lines represent the area of the 95% PVCs obtained from the full dataset for the entire population as well as for each pod. The K-pod (D) 95% PVC area resulted to be greater than the upper limit of the distribution, indicating a possible overestimation of the extent.

2.3.3 Spatial Noise Exposure Risk by Vessel Categories

The c.d.f. relative to the total traffic showed median values of 110 dB re 1 μPa ($L_{eq-5^{th}}$ = 95 dB re 1 μPa, $L_{eq-95^{th}}$ = 126 dB re 1 μPa), 107 dB re 1 μPa ($L_{eq-5^{th}}$ = 97 dB re 1 μPa, $L_{eq-95^{th}}$ = 114 dB re 1 μPa) and 105 dB re 1 μPa ($L_{eq-5^{th}}$ = 95 dB re 1 μPa, $L_{eq-95^{th}}$ = 112 dB re 1 μPa) for Zone 1, Zone 2 and Zone 3, respectively. By analyzing the single c.d.f. curves (Figs. 2.13, 2.14 and 2.15), vessel categories were divided into three groups. Vessels having more than 50% of their c.d.f. falling under L_{eq} values < 60 dB re 1 μPa were considered as belonging to a "Low Level" exposure group. Vessels having more than 50% of their c.d.f. values falling between 60 dB re 1 μPa and 90 dB re 1 μPa were considered as belonging to a "Medium Level" exposure group. Vessels having more than 50% of their c.d.f. values falling above 90 dB re 1 μPa were considered as belonging to a "High Level" exposure group. For example, since 50% of the Crude Oil Tankers' c.d.f. within Zone 1 fell below 60 dB re 1 μPa (Fig. 2.13), this vessel category was assigned to the low-level exposure group in this location. However, since, in Zone 2 and 3, more than 50% of the Crude Oil Tankers' c.d.f. was comprised within 60 re 1 μPa and 90 re 1 μPa (Figs. 13 and 14), the category belongs to the medium level exposure group in these two locations. Classification of the modeled vessel categories and corresponding L_{eq} values are reported in the following paragraphs as well as in Table 3.

Within Zone 1, four vessel categories were identified as having an $L_{eq-50^{th}}$ > 90 dB re 1 μPa: Ferries ($L_{eq-50^{th}}$= 110 dB re 1 μPa); Tugboats < 50 m ($L_{eq-50^{th}}$= 101 dB re 1 μPa), Vehicle Carriers ($L_{eq-50^{th}}$= 94 dB re 1 μPa) and Recreational Vessels ($L_{eq-50^{th}}$= 90 dB re 1 μPa). Of the remaining 10 categories, 7 (i.e. Fishing Vessels, Naval Vessels, Containers, Bulkers, Government/Research, Tankers, Other) were identified as having $L_{eq-50^{th}}$ values comprised between 60 and 90 dB re 1 μPa, while 3 (i.e. Passenger, Crude Oil -tankers, Reefers) showed $L_{eq-50^{th}} < 60$ dB re 1 μPa. Within Zone 2, four vessel categories were identified as having an $L_{eq-50^{th}} > 90$ dB re 1 μPa: Tugboats ($L_{eq-50^{th}}$= 101 dB re 1 μPa), Containers ($L_{eq-50^{th}}$= 98 dB re 1 μPa), Bulkers ($L_{eq-50^{th}}$= 97 dB re 1 μPa), Vehicle Carriers ($L_{eq-50^{th}}$ = 93 dB re 1 μPa). Of the remaining 10 categories, 9 (i.e. Tankers, Ferries, Naval Vessels, Recreational Vessels, Fishing Vessels, Passenger, Government/Research, Crude Oil Tankers, Other) were identified as having $L_{eq-50^{th}}$ values comprised between 60 and 90 dB re 1 μPa, while only Reefers showed $L_{eq-50^{th}} < 60$ dB re 1 μPa . Within Zone 3, four vessel categories were identified as having an $L_{eq-50^{th}} > 90$ dB re 1 μPa: Tugboats ($L_{eq-50^{th}}$= 99 dB re 1 μPa), Containers ($L_{eq-50^{th}}$= 98 dB re 1 μPa), Bulkers ($L_{eq-50^{th}}$= 97 dB re 1 μPa), Vehicle Carriers ($L_{eq-50^{th}}$= 94 dB re 1 μPa). Of the remaining 10 categories, 9 (i.e. Tankers, Naval Vessels, Ferries, Government/Research, Passenger, Recreational Vessels, Fishing Vessels, Crude Oil Tankers, Other) were classified as medium exposure categories, while only Reefers were identified as a low exposure category.

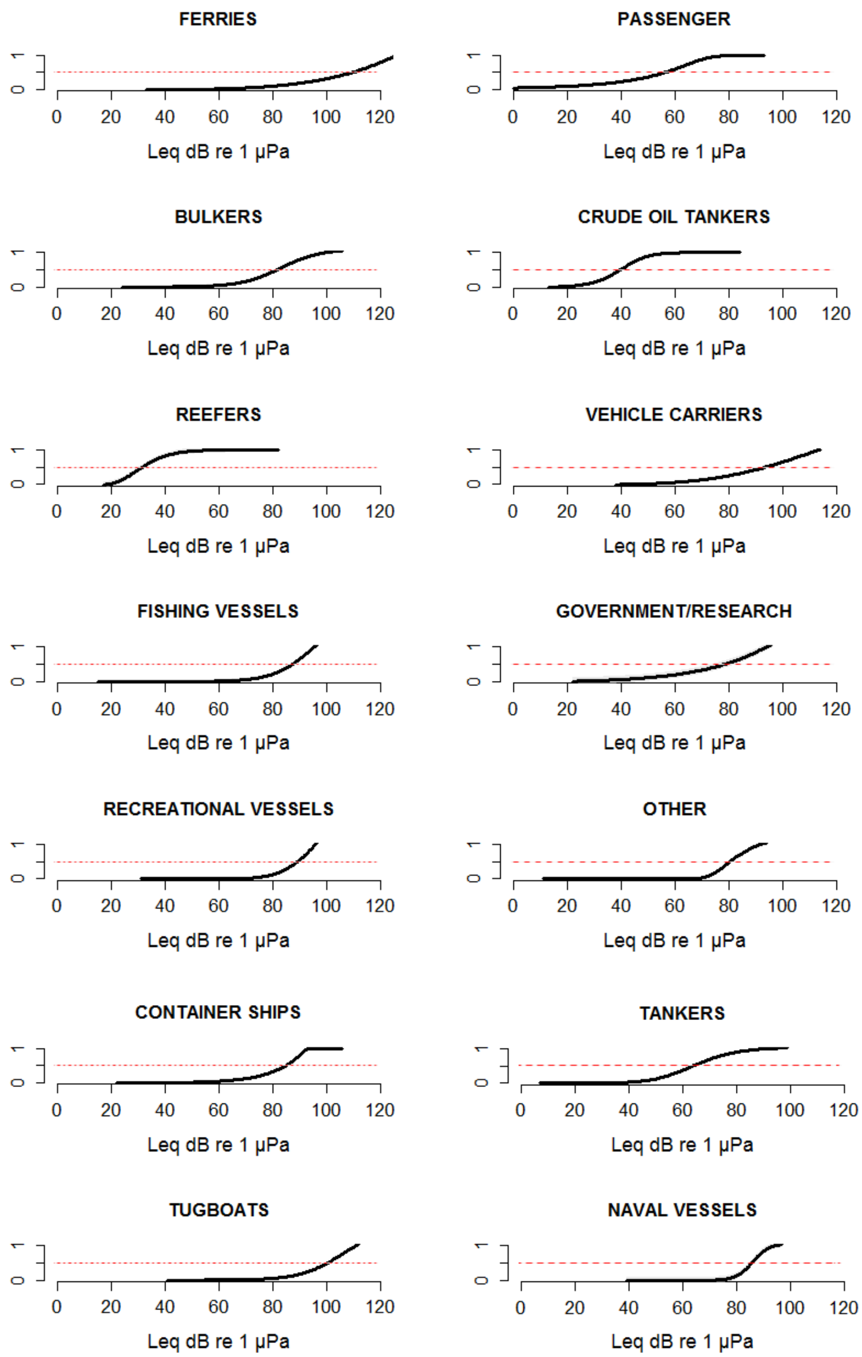

Figure 2.13. The c.d.f. curves of the L_{eq} values modelled within Zone 1 (Fig. 2.6). A curve was computed for each vessel category producing noise emissions within Zone 1. Cumulative probabilities are on the y axes while the corresponding L_{eq} values are on the x axis. The red dashed line marks the $L_{eq-50^{th}}$ for each class (i.e. $F_{L_{eq}} = 0.5$) (Equation 7).

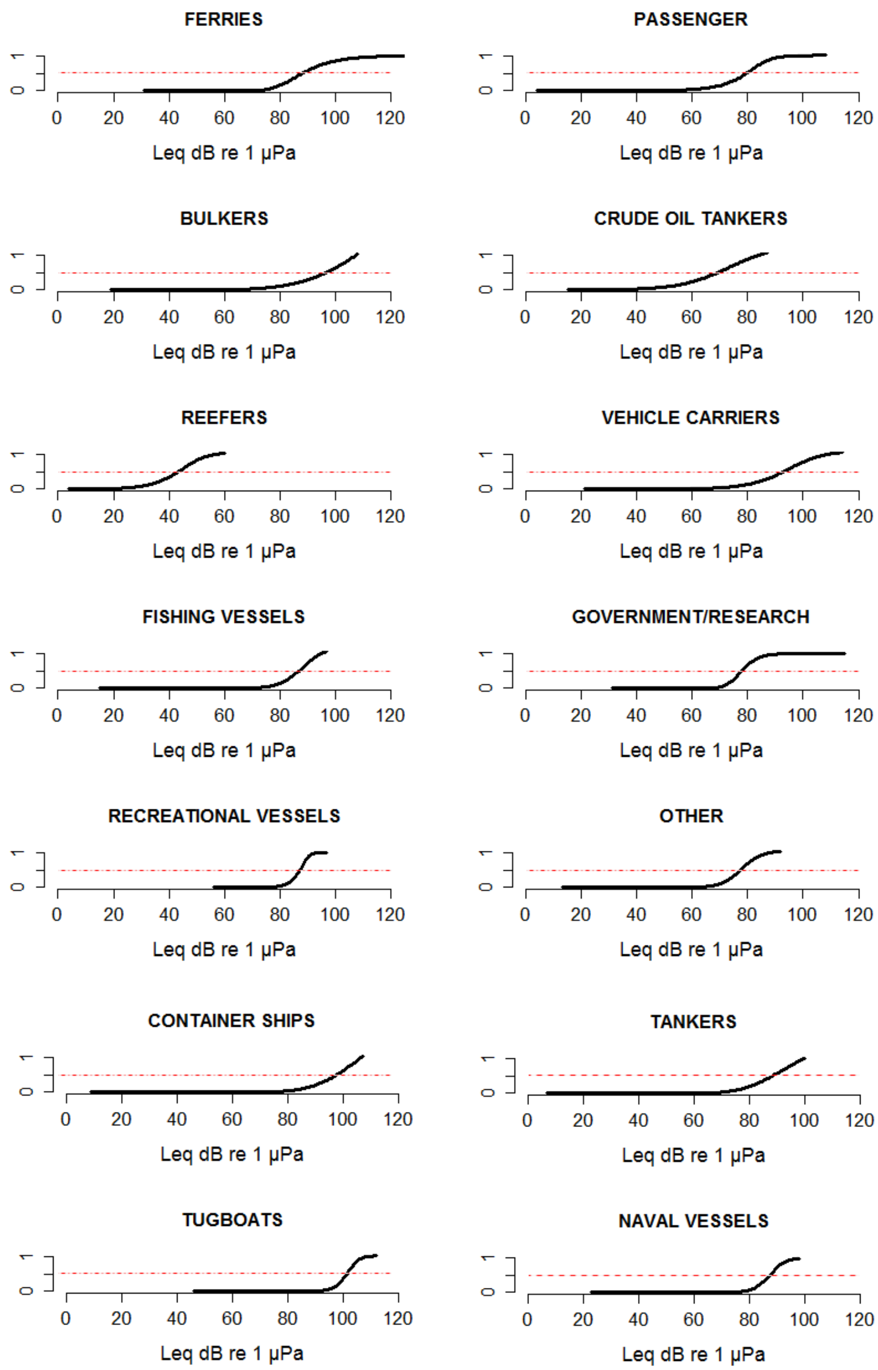

Figure 2.14. The c.d.f. curves of the L_{eq} values modelled within Zone 2 (Fig. 2.6). A curve was computed for each vessel category producing noise emissions within Zone 2. Cumulative probabilities are on the y axes while the corresponding L_{eq} values are on the x axis. The red dashed line marks the $L_{eq-50^{th}}$ for each class (i.e. $F_{L_{eq}} = 0.5$) (Equation 7).

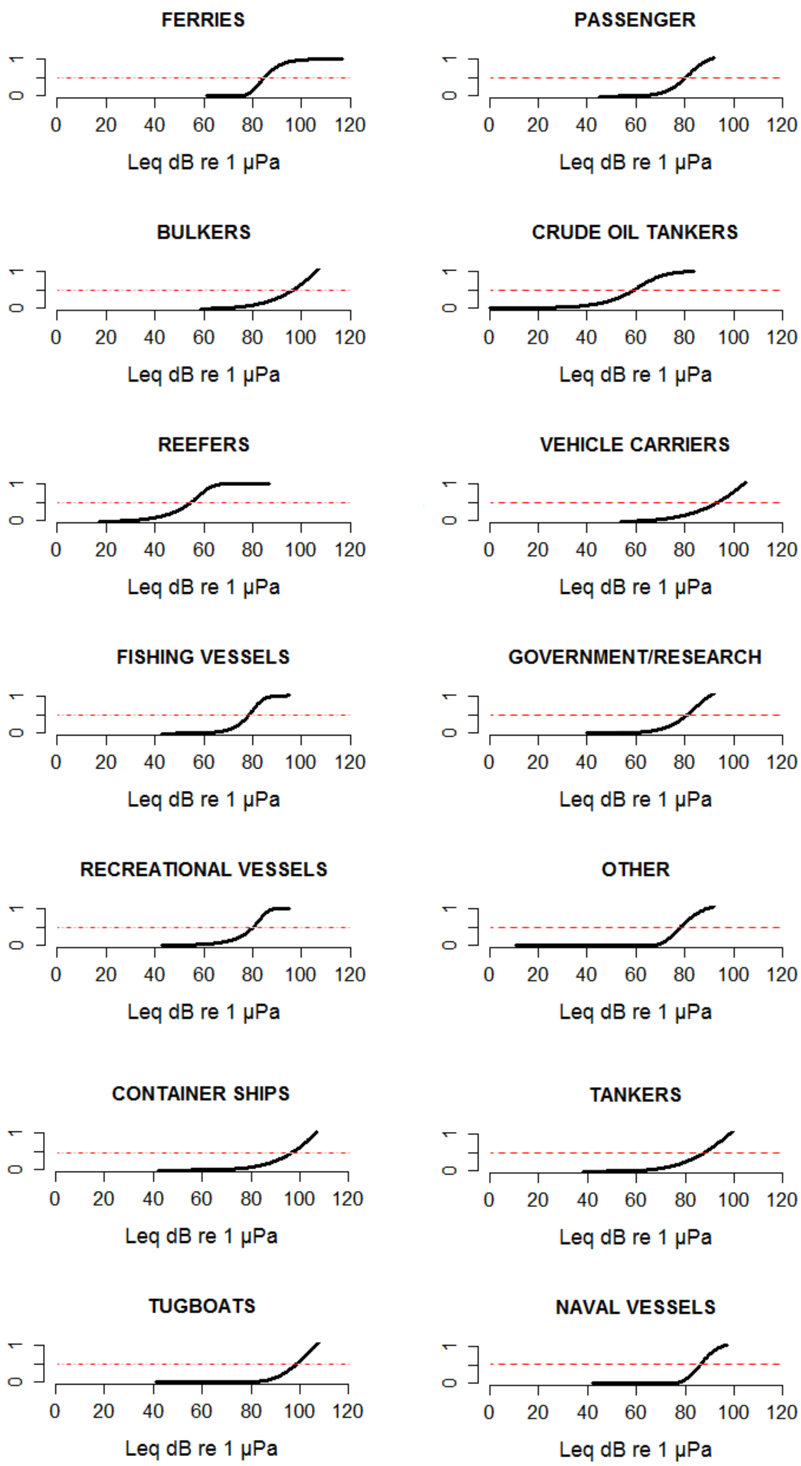

Figure 2.15. The c.d.f. curves of the L_{eq} values modelled within Zone 3 (Fig. 2.6). A curve was computed for each vessel category producing noise emissions within Zone 3. Cumulative probabilities are on the y axes while the corresponding L_{eq} values are on the x axis. The red dashed line marks the $L_{eq-50^{th}}$ for each class (i.e. $F_{L_{eq}} = 0.5$) (Equation 7).

Table 2.3. L_{eq} values corresponding to the 5th (0.05), 50th (0.5) and 95th (0.95) percentiles of the c.d.f. of each vessel category over Zone 1, 2 and 3 (Figs. 2.12, 2.13 and 2.14). For each zone, the assigned exposure levels ($L_{eq-50^{th}}$< 60; 60 < $L_{eq-50^{th}}$ < 90; $L_{eq-50^{th}}$ > 90) are reported. Pooled categories are in bold.

Zone 1					**Zone 2**					**Zone 3**				
Category	L_{eq}			**Exposure Level $L_{eq-50^{th}}$ (dB re 1 µPa)**	**Category**	L_{eq}			**Exposure Level $L_{eq-50^{th}}$ (dB re 1 µPa)**	**Category**	L_{eq}			**Exposure Level $L_{eq-50^{th}}$ (dB re 1 µPa)**
	0.05	***0.5***	***0.95***			***0.05***	***0.5***	***0.95***			***0.05***	***0.5***	***0.95***	
Ferries	70	110	125	**> 90**	**Tugboats**	94	101	107	**> 90**	**Tugboats**	86	99	106	**> 90**
Tugboats	80	101	110		**Containers**	81	98	106		**Containers**	77	98	106	
Vehicle Carriers	59	94	112		**Bulkers**	73	97	107		**Bulkers**	76	97	105	
Recreational Vessels	75	90	95		Vehicle Carriers	72	93	107		Vehicle Carriers	71	94	104	
Fishing Vessels	69	88	95	**60 - 90**	Tankers	72	88	99	**60 - 90**	Tankers	67	88	98	**60 - 90**
Naval Vessels	77	86	93		**Ferries**	76	88	109		Naval Vessels	78	86	94	
Containers	61	85	92		Naval Vessels	79	87	96		**Ferries**	78	85	98	
Bulkers	61	82	97		Recreational Vessels	80	87	92		Government/ Research	65	81	89	
Other	71	81	90		Fishing Vessels	76	87	94		**Passenger**	68	81	89	
Government/ Research	38	79	93		**Passenger**	62	80	91		Recreational Vessels	62	80	87	
Tankers	44	64	85		Government/ Research	71	78	88		Fishing Vessels	67	79	86	
Passenger	2	56	74	**< 60**	Other	67	77	86		Other	70	79	88	
Crude Oil Tankers	24	40	54		**Crude Oil Tankers**	45	69	83		**Crude Oil Tankers**	34	60	76	
Reefers	22	32	49		Reefers	26	44	55	**< 60**	Reefers	36	55	65	**< 60**

2.3.4 Exposure Maps

Exposure maps were produced for the six vessel categories belonging to the "High Level" exposure group (Fig. 2.16). Some of these categories showed analogous L_{eq} distribution patterns, while others displayed a unique pattern. Containers (Fig. 2.16E) and Bulkers (Fig. 2.16F), were characterized by high exposure levels ($L_{eq-50^{th}} > 90$ dB re 1 µPa) covering approximately 50% of both Zone 2 and 3 and by medium exposure levels ($60 < L_{eq-50^{th}} < 90$ dB re 1 µPa) within Zone 1.

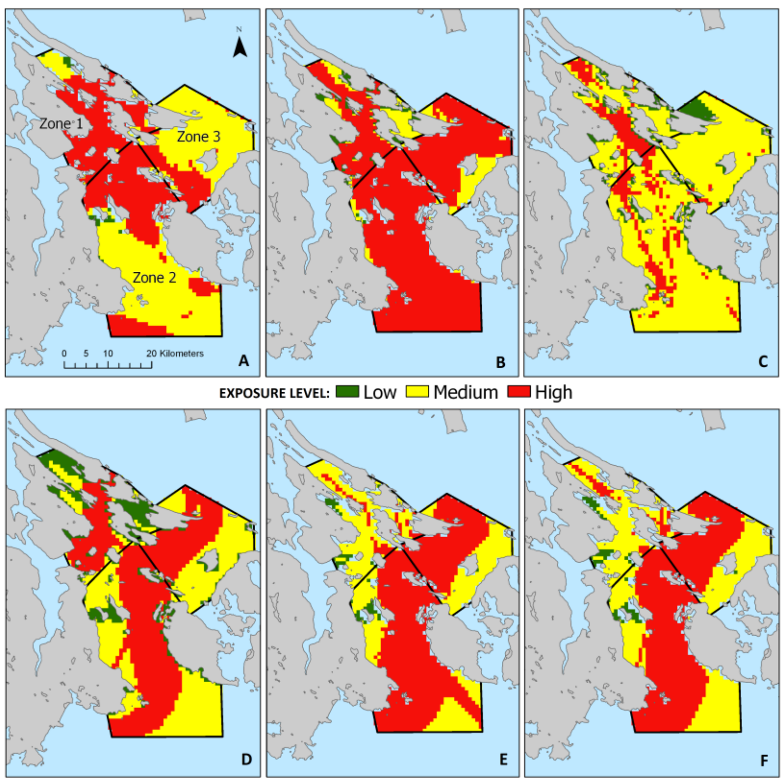

Figure 2.16 Maps showing exposure levels for ferries (A), tugboats (B), recreational vessel (C), vehicle carriers (D), containers (E) and bulkers (F). Low exposure levels (green) correspond to $L_{eq} \leq 60$ dB re 1 µPa, medium exposure levels (yellow) correspond to $60 < L_{eq} \leq 90$ dB re 1 µPa while high exposure levels (red) correspond to $L_{eq} > 90$ dB re 1 µPa.

Ferries (Fig. 2.16A) were characterized by high exposure levels concentrated in Zone 1 and in the northern portion of Zone 2, while the central portion of Zone 2 and the majority of Zone 3 displayed medium exposure levels. Recreational Vessels (Fig. 2.16C) displayed high exposure levels concentrated in the center of Zone 1 and along the western portion of Zone 2. When compared to other classes, areas with high exposure values from Recreational Vessels showed the smallest extent. High exposure levels from tugs (Fig. 2.16B) covered the majority of all the three zones, with only small portions characterized by a medium or low exposure level. Showing a similar pattern, vehicle carriers (Fig. 2.16D) displayed high exposure levels over approximately half of each one of the three zones.

2.4 Discussion

Findings in this study complement and update findings from previous studies that identified the same SRKW summer core areas (Hauser et al. 2007, DFO 2011). However, the high spatial resolution of the KDEs produced in this study and the large number of observations considered for their computation allowed us to describe current SRKW summer areas at a finer spatial scale. By combining these distributions with vessel noise maps, we also provide a first insight into the locations of areas characterized by high levels of noise exposure for part of the SRKWs critical habitat within the central Salish Sea. Such information can help support the management of this endangered population.

2.4.1 SRKW Summer Core Areas

A large core area, commonly used by all three pods as evinced by the 50% PVCs, was identified along the south-western shore of San Juan Island (Fig. 2.1C). Foraging on Chinook salmon is the main activity undertaken by members of SRKW within the boundaries of this core area (Hanson et al., 2010; Scott-Hayward et al., 2015) and vessel traffic off the coast of San Juan has been associated with the disruption of SRKW feeding behavior (Lusseau et al., 2009). This area in the Haro Strait borders international shipping lanes, making it likely that individuals belonging to all three pods, when feeding here, will at times be exposed to high levels of noise from vessel sources. Moreover, potential disturbance from noise may vary among different groups. For example, the current San Juan core area for the L-group extends southwards into the Strait of Juan de Fuca, reaching the northern end of Hein Bank, while the K and J groups current core areas are between Eagle Point and Hanbury Point, representing the southern and northern ends, respectively. Looking at the pod-specific areas identified by the 50% PVCs, the J and K specific core areas (Figs. 9B and 10B) overlap with the international shipping lane, while the L-group specific core area (Fig. 2.11B) is located in a portion of the Salish Sea characterized by relatively low levels of traffic from large vessels. Pod-groups include multiple pods formed by SRKW. For example, the J-group includes observations of individuals all belonging to the J-pod but also mixed groups such as JK and JL. These mixed groups are usually short-term association between members of different pods (Hauser et al., 2007). Our approach did not allow to draw pod-specific conclusions. However, as noted by Hauser et al. (2007), the long-lasting social associations (i.e. J, K, and L) may be driving the movement and space use of the less frequent mixed groups. For example, the J-pod appeared to be driving the spatial distribution of the JK and JL pod combinations, and similar observations were made for the K and L groups.

Estimating data uncertainty is thought to be fundamental for incorporating species distribution studies into conservation planning (De Ornellas et al., 2011; McShea, 2014; Scott-Hayward et al., 2015). In this study, the bootstrap iteration allowed for an estimation of uncertainty related to the KDEs, providing upper and lower boundaries of the summer core areas for which there has been Soundwatch activity. According to the results of the bootstrap iteration (Fig. 2.12D), the KDE representing the K-group (Fig. 2.10C) overestimated the extent of the corresponding core area. This is probably associated with the relatively low number of K-group sightings available for this study (Tab. 2.2). However, this overestimate is not only influenced by biases in the methodology but also a result of SRKW's peculiar population structure. K-pod is the least numerous of the three pods, made up of only 4 matrilines, comprising of only 18 individuals, consequently, further data needs to be collected to improve the quality of this pod's core area estimation. Since the Strait of Juan the Fuca, the Strait of Georgia and the Northern Gulf Islands are rarely frequented by Soundwatch, the KDEs are probably unreliable across these areas. For this reason, none of the three analyzed zones included portions of the KDEs extending over these three areas. Biases related to the uneven spatial and temporal distribution of sightings effort could also have influenced the reported location of the pod-specific core areas. Soundwatch mainly operates where private boaters are more likely to encounter members of SRKW, and, in some years the activity is limited to the South-west coast of San Juan Island (Fig. 2.7). Similarly, most Soundwatch activities are undertaken within US waters, a bias that might have caused underestimations of the extent of SRKW summer areas within Canadian waters. Data from Straitwatch, the Canadian counterpart of Soundwatch, could not be accessed and included in our study but could be used in the future to refine our analyses. The positive correlation observed between the KDE relative to the entire population and the BCCSN sightings-per-unit-effort map indicates that the KDE effectively depicts SRKW summer distribution within the study area. The low R2 value obtained from the OLS analysis might be due to the different spatial resolution of the two maps: 800 m and approximately 5 km for the KDE and the BCCSN sightings-per-unit-effort map, respectively. Another factor affecting the level of correlation between the two estimates could be related to the inclusion, in the BCCSN dataset, of both the resident (i.e. northern and southern) and transient (Bigg's) killer whale ecotypes occurring in the Salish Sea. Due to these limitations, noise exposure levels were evaluated only for the KDE describing the entire population summer core area. Nonetheless, the identification of pod-specific areas suggests that the three pods constituting SRKW could be exposed to different levels of noise from shipping.

2.4.2 Spatial Noise Exposure Risk by Vessel Categories

This study considered cumulative noise expressed as unweighted equivalent time-averaged sound pressure level (L_{eq}), which results from the long-term integration of time-varying sound exposure. More specifically, in this study L_{eq} represents the average rate of accumulation of sound exposure over a period of a month. When computed over prolonged periods of time, L_{eq} will tend toward an asymptotic value, and, assuming that the daily vessel traffic is broadly similar throughout the month, the monthly accumulation rate will be comparable to the daily accumulation rate of sound exposure experienced by SRKW within the study area. L_{eq} is a commonly used metric for the assessment of Human exposure to continuous, non-physically damaging noises (Maling 2007). In an analogous way to the measurement of human noise exposure, the L_{eq} maps from this study may be understood as a measure of typical daily noise exposure for whales at different geographic locations within the study area. An animal (or group of animals), occupying a cell of the model may be exposed to higher or lower sound levels at any particular instant, but the long-term exposure will tend toward the average value (i.e. L_{eq}). Since animals are known to move within the study area, having a member of SRKW continuously occupy a single cell for a day would be very unlikely. The results presented, therefore, should be seen as the maximum exposure an animal would receive if it were to stay within the same general area. The L_{eq} at a given percentile level (e.g. $L_{eq-50^{th}}$) therefore expresses the probability for a pod (or group) to accumulate a certain amount of daily noise exposure within such an area. Thus, while the modeled L_{eq} values used in the present study were not intended to provide a cause-effect relationship between noise exposure and its impacts on the population, it is nonetheless a useful proxy to identify areas characterized by a higher risk of exposure for SRKW on the basis of the spatial distribution of vessels as sources of noise and killer whales as receivers. Furthermore, a study measuring variation in stress hormones in North Atlantic Right Whales showed that after the events of September 11th, 2001, resulting in fewer commercial vessels travelling through Right Whale habitat causing a 6 dB drop in sound levels, stress hormones in whale fecal samples were reduced (Rolland et al., 2012). The authors, however, did not differentiate between the potential effects of ship presence and noise presence in their study.

The computation of the c.d.f. allowed taking into account the probability of observing SRKW within a specific cell of the KDE during the summer months. Furthermore, the use of c.d.f. suggests that both the spatial and temporal components of commercial shipping should be considered when introducing management solution aimed at the reduction of chronic noise pollution.

The various ship categories considered in this study were characterized by different cumulative distribution functions which could be grouped based on their 50th percentiles. Ferries, Tugboats, Vehicle Carriers and large commercial ships (i.e. Container Ships, Bulkers) produced the highest levels of sound exposure for SRKW within their summer core areas. Ferries, Tugboats, and large commercial ships are also responsible for the vast majority of the sound energy input by commercial vessels in the Salish Sea (MacGillivray et al., 2017).

The modeled L_{eq} values are driven by the source levels (SL) (Tab. 2.1), the SPL scaled to nominal distance of one meter from the source, estimated for each vessel category. Container ships are the category of commercial ships that produces the highest SLs. SLs reaching 178 dB re 1 μPa have been estimated from container ships transiting through Haro Strait (Veirs et al., 2016). Source levels of 183 dB re 1 μPa and of 185 dB re 1 μPa have been estimated from container ships navigating the waters of Puget Sound (Bassett et al., 2012), both in the Salish Sea, and in Santa Barbara Channel (McKenna et al., 2012), along the coast of California. Tugboats show lower estimated source levels: 170 dB re 1 μPa (Bassett et al., 2012; Veirs et al., 2016). Tugboats navigate at relatively constant low speeds, one of the main factors influencing the amount of noise produced by a vessel (McKenna et al., 2013), but their cargo, can be highly variable. Therefore, source levels estimated from a small sample of Tugboats or limited to a small area, might not capture the full extent of Tugboats' noise emissions. One of the factors making ferries one of the main contributors to the cumulative noise within SRKW summer core areas is that ferries travel the same route several times a day while other vessel categories are less frequent.

The use of estimated SL also introduces an element of uncertainty in the modeled L_{eq} values. A recent study (Veirs et al., 2017) demonstrated how approximately half of all noise energy released in Haro Strait is produced by approximately 15% of the total commercial fleet. These "large" noise polluters are characterized by SLs > 179 dB re 1 μPa, indicating that a small population of particularly loud vessels might be affecting the average SL attributed to a category. The SL of a ship is highly variable depending on speed, draught, maintenance as well as several other factors, and actual cumulative noise levels could only be established through the use of models validated for particular ships navigating in a specific environment. In order to validate the results of the cumulative model, the modelled received levels were compared with the available vessel noise measurements in the Salish Sea (Fig. 2.4).

Another limitation of the cumulative noise model used in this study is that AIS data inevitably underestimates the actual density of ships in the Salish Sea as not all the categories considered in this study are equipped with mandatory AIS devices. This is particularly true for recreational vessels which resulted to be a category associated with high levels of exposure within Zone 1 (Fig. 2.16C and Tab. 2.3). This result might be underestimating the actual contribution of recreational traffic to the cumulative noise experienced by SRKW because only a small fraction of the private pleasure crafts, fishing vessels and whale watching boats are equipped with AIS transponders. Consequently, an analysis of these specific sources of noise is highly recommended.

2.4.3 Management Implications

Our results can help inform decisions relative to ship traffic within the study area and help design scenarios that could reduce noise from shipping within SRKW summer core areas. From August 7 until October 6th, 2017, the Vancouver Fraser Port Authority introduced a voluntary speed limit of 11 kn for all the traffic transiting within SRKW summer core area

(VFPA ECHO Program). Since most of the large commercial ships transiting through Haro Strait move at speeds of approximately 8 m/s (i.e. 15.5 kn) and since the traffic is concentrated within the international shipping lane, this management solution aims to reduce the noise produced by these vessel categories. However, within the study area, Tugboats move at speeds below 6 m/s (i.e. 11.5 kn) and showed, for the month of July 2015, a volume of traffic approximately 4 times larger than the traffic volume of large commercial ships (MacGillivray et al., 2017). The imposition of an 11 kn speed limit to this category might not reduce its contribution to the cumulative noise within SRKW core areas. Even for those categories which are affected by the slowdown protocol, reducing vessel speed increases the duration of noise exposure (albeit, at a lower sound level). Thus, it remains uncertain to what extent slowdown mitigations reduce acoustic impacts on SRKW. Future work should investigate how the introduction of slow-downs affects the duration of noise exposure. A possible approach could be the estimation of noise exposure from an SEL perspective, as reported by McKenna et al. (2013) in the Santa Barbara Channel.

A possible application of these results could be the implementation of speed and density limits for the six vessel categories identified as causing high levels of exposure for SRKW. Vessel density could be controlled by re-routing part of the traffic toward other areas as well as by imposing a limit on the number of vessels allowed to navigate through an area at the same time. Re-routing vessels navigating through a complex system of narrow seaways and islands such as the Salish Sea could be challenging. Another possible approach could be the adoption of what DFO defined as “lateral displacement”, the introduction of small changes in the routes typically followed by vessels to avoid ecologically vulnerable areas (DFO, 2017b). Although lateral displacement would most likely not be an efficient solution for the abatement of low-frequency noise, it could lead to a reduction of the amount of high-frequency noise released within SRKW’s core areas. Furthermore, re-routing and lateral displacement options could be feasible in ports where shipping lanes are not too geographically constrained. Speed limits and avoidance areas have already been implemented to address lethal ship strikes for the North Atlantic Right Whale in both the US (Laist et al., 2014) and Canada (Daoust et al., 2017), aiming to achieve a reduction in the number of vessel-caused deaths for this endangered population. Along with the risk of ship strike and entanglement, chronic noise pollution is thought to be a limiting factor for the recovery of the NARW population (Petruny et al., 2014). Ports which are not as geographically constrained, such as Boston’s Harbor (US), could more easily adopt re-routing and lateral displacement as strategies to reduce the risk of exposing endangered cetacean species to vessel noise pollution. However, in other areas re-routing would not be challenging, but rather impossible. Representing the main point of access for the Gulf of St. Laurence and the St. Lawrence Seaway, the Cabot Strait is characterized by a considerable amount of vessel traffic. In this context, other solutions such as real-time notifications of whale presence, convoying, and the creation of “quiet” periods where navigation is forbidden could be adopted (DFO, 2017b). Nonetheless, as recognized by the International Maritime Organization (IMO 2014), the ideal long-term solution for the reduction of shipping noise is the adoption of quiet design practices for the construction of commercial vessels.

2.5 Conclusions

In the narrow seaways of the Salish Sea, killer whales are frequently in close proximity to ships, therefore exposed to both noise in low- and high-frequencies generated by propeller cavitation (Veirs et al., 2016). For these endangered odontocetes, this might have the dual effect of masking communication as well as the reception of echolocation signals, thus affecting the feeding success and the social interactions of SRKW. It is important, however, to consider noise pollution as only one of the many anthropogenic impacts affecting this marine species. Impact at the population level is likely the result of cumulative impacts from several different stressors interacting with each other. For example, the SRKW population was considered to have 71 individuals in 1973 (Olesiuk et al. 1990) while 76 were reported by the Center for Whale Research in September 2017. In between the population rose to over 90 individuals and declined again and repeated this cycle a few times. The population numbers in one year appeared to be directly connected to the availability of their main prey, the Chinook salmon (*Oncorhynchus tshawytscha*) during the year before (Ford and Ellis 2006; Hanson et al. 2010) and periods of decline in the abundance of chinook have been associated with periods of increased mortality rates for SRKW (Ford et al. 2010). The interdependency of these two species is considered so strong that, without the implementation of adequate conservation measures, a full recovery of SRKW might compromise the recovery of Chinook salmon populations (Williams et al., 2011). All killer whales in the North-eastern Pacific also show high levels of contaminants, such as polychlorinated biphenyl (PCB) as well as other pollutants, which have been associated with reduced survival and reproduction rates (Buckman et al., 2011; Lachmuth et al., 2011; Alava et al., 2016). Disturbance from small vessel traffic near the whales may represent another relevant threat to the recovery of SRKW, with positive correlations between increases in small vessel presence around the animals and reduced foraging rates in this population (Lusseau et al., 2009). Moreover, over the period 2011-2016, the Soundwatch Boaters Education program recorded more than 13,300 negative interactions between boats and killer whales that had the potential to damage the animals or interfere with their behavior (Eisenhardt et al., 2012; Eisenhardt and Koski, 2011, 2013 and 2014; Seely, 2015 and 2016). During 2016 each one of the 77 members of SRKW (Center for Whale Research, 2016) experienced on average approximately 30 negative interactions with boats. Furthermore, at least one of the six deaths that occurred in 2016, taking SRKW back to population sizes recorded in the late 1980s, was suspected to be the consequence of a ship strike, an unprecedented threat to the survival of this population (Lopes, 2016).

These concurring threats highlight the need for comprehensive adaptive management strategies, in order to ensure the survival of the SRKW population and to improve the habitat for other marine life. Adaptive management goes beyond the trial and error approach and requires the exploration of alternative strategies including modeling simulations of effects as well as the systematic evaluation and modification of those strategies through continuous monitoring of effects and outcomes (Aldridge et al., 2004; Allen and Garmestani, 2015). For these reasons, appropriate adaptive management measures for the reduction of cumulative noise from shipping in the Salish Sea should be adopted. For example, although speed is generally correlated with the noise emitted by commercial ships, the relationship between speed and noise varies among vessel types and propulsion systems (Wales et al., 2002; McKenna et al., 2012), suggesting that the effectiveness of improving SRKW habitat via vessel slowdown needs to be tested and compared with other methods. It may turn out that, in addition to slowdowns, other strategies are needed to address this complex issue. For example, modifying existing shipping routes, as suggested by IMO's guidelines (IMO, 2014), represents another possible strategy for the reduction of vessel noise.

However, in the absence of a regulatory framework addressing the issue of oceanic anthropogenic noise and its impacts, the successful application of quieting measures is dependent on voluntary compliance by noise producers. Although noise has been included in SRKW recovery strategy as a source of disturbance (DFO, 2016), currently no law limiting chronic anthropogenic noise output in the ocean exists in Canada. Yet, a regulatory infrastructure that recognizes noise as a marine pollutant already exists. The United Nations Convention on the Law of the Sea (UNCLOS) is at the core of many national and international regulations for the protection of marine environments (Boyes and Elliott, 2014; Firestone and Jarvis, 2007). UNCLOS defines pollution as the: "*introduction by man,* *directly or indirectly of substances or energy into the marine environment, including estuaries, which results or is likely to result in such deleterious effects as harm to living sources and marine life, hazards to human health, hindrance to marine activities, including fishing and other legitimate uses of the sea, impairment of quality for use of sea water and reduction of amenities*" (UNCLOS, supra note 21, at article 1(4)). Considering that high amplitude sound as a by-product of anthropogenic activities is recognized to be potentially harmful to humans and other terrestrial species (Fritschi et al., 2011; Luo et al., 2015; Ware et al., 2015), all countries that ratified UNCLOS should adopt measures to regulate the emission of underwater sound in order to reduce its impact. Furthermore, other jurisdictions have already introduced legislative frameworks aimed at reducing the output of underwater sound energy. The EU's Marine Strategy Framework Directive (MSFD) (2008/56/EC) identifies annual thresholds for low-frequency continuous sounds and level thresholds for impulsive sounds introduced into the waters around its member states (Erbe et al., 2012). The MSFD explicitly refers to underwater noise as a form of pollution and required member states to implement ambient noise monitoring programs by 2015 (Dekeling et al 2014).

In conclusion, the absence of national regulations and the slow implementation of international regulations might jeopardize the conservation efforts for SRKW as well as for many other species inhabiting our oceans.

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Chapter 3 Geovisualization tools to Inform the Management of Vessel Noise in Support of Species' Conservation

3.1 Introduction

Environmental noise produced by anthropogenic activities is gaining increasing attention worldwide. According to the European Union (EU) and the World Health Organization (WHO), after atmospheric pollution, noise is the second largest environmental cause of health problems in Europe (Gouveia et al., 2004). As a consequence, EU member states are now required to implement noise management actions for sensitive areas (i.e., large urban agglomerations, major roads, railways, and airports) (Environmental Noise Directive, 2002/49/EC). Noise pollution is also harmful to wildlife. Road traffic noise has been linked to habitat degradation, and effects on avian and mammal species are being widely documented in the wild (Luo et al., 2015; Pepper et al., 2003; Senzaki et al., 2016; Ware et al., 2015).

Of growing concern is the impact that noise has on marine ecosystems, as sound travels approximately 4.5 times faster in seawater than in air. Early research efforts focused attention on marine mammal species considered to be particularly sensitive to acoustic disturbance (Southall et al., 2007; L. Tyack, 2008). However, evidence suggests that the effects of anthropogenic noise in the oceans may be broader than expected, affecting several levels of the trophic chain. Both marine mammal and fish species, when exposed to growing levels of background noise, show reactions similar to humans, increasing the loudness of their communications, a phenomenon called the Lombard effect (Filiciotto et al., 2013; Holt and Johnston, 2014; Marla M Holt et al., 2009; Lugli, 2014). Recently, McCauley et al. (2017) reported high mortality of plankton species and altered fish behaviour after the use of a single air-gun, providing one of the first pieces of evidence that anthropogenic noise affects the lower levels of the marine trophic chain over wide areas. Marine species evolved in the absence of anthropogenic noise. Consequently, changes in the acoustic conditions of the marine environment caused by expanding anthropogenic activities could affect a large number of species and may be particularly negative for those that depend on sound for foraging, detecting predators, and mating (Halliday et al., 2017).

The global increase in oceanic noise pollution recorded over the past decades (Merchant et al., 2012b), and the growing number of studies reporting on the effects of noise on commercial and non-commercial species (Carroll et al., 2017), require the attention of marine managers. In the Canadian context, oceanic noise pollution is gaining importance, especially for endangered cetacean species. Noise pollution is currently suspected to be a relevant contributor to the decline of at least two endangered populations: the North Atlantic Right Whale (Petruny et al., 2014), and the Southern Resident Killer Whale (SRKW) (DFO, 2017a; Veirs et al., 2016). The newly proposed action plan for the recovery of SRKW recognizes noise pollution as a source of disturbance and as a potential threat to the conservation of this population (DFO, 2017a). The Department of Fisheries and Oceans (DFO) also concluded that adopting a combination of mitigation measures, rather than a single solution may be the most effective way to achieve a reduction in noise exposure for SRKW (DFO, 2017b).

Noise exposure from shipping for endangered species can be reduced following different management solutions. Since propeller cavitation and machinery have been identified as the main sources of noise radiating from commercial ships, an ideal management solution would be a reduction of noise at its source (IMO, 2014). However, since the average lifecycle (i.e. design, construction, operation and maintenance, and disposal) of a modern commercial ship is 25-30 years (Dinu and Ilie, 2015), upgrading the existing commercial fleet to meet the required noise standards is difficult. For these reasons, short-term management solutions are needed to assure the protection of endangered species during the transition from the current conditions toward quieter shipping technologies.

However, the design and implementation of ship-noise mitigation measures are far from being straight-forward. For example, although speed is generally correlated to the noise emitted by large commercial ships (McKenna et al., 2012), the relationship between speed and noise varies from one vessel type to another (Wales and Heitmeyer, 2002). A modeling study conducted by Chion et al. (2017), showed how the adoption of speed limits may actually result in an increase of the total amount of acoustic energy released into the environment due to the resulting longer duration of travel. Modifying the existing routes, as suggested by the International Maritime Organization (IMO, 2014), represents a possible mitigation measure for the reduction of the negative effects of ship noise (DFO, 2017b). Two of the 12 mitigation measures explored by DFO in 2017 are related to the modification of existing routes: “relocation of shipping traffic lanes” and “redirecting a portion of vessel traffic” (DFO, 2017b).

A possible approach to the design of an effective noise management solution is to achieve a better understanding of the spatial relationship between anthropogenic noise and the distribution of endangered marine species. Such understanding can be gained using spatial analysis and geovisualization tools within geographic information systems (GIS) software.

Visual analytics, combining automated analysis with interactive data visualizations to achieve a deep understanding of complex phenomena (Keim et al., 2008), was shown to be effective for communicating scientific knowledge (Schroth et al., 2014) and as a tool to support decision-making (Al-Kassab et al., 2014). Examples can be found in the literature dedicated to the role of 3D-geovisualization in climate change communication (Schroth et al., 2015, 2014; Shaw et al., 2009). Also, in their work, Schroth et al. (2015, 2014) demonstrated how the use of scenarios representing possible local consequences of global climate change in a 3D environment could help communicate adaptation policies and create a sense of local responsibility. Furthermore, studies from the field of landscape genetics highlighted how visualization techniques and tools can convey relevant information from researchers to decision-makers and stakeholders without GIS or genetics expertise (Aoidh et al., 2013). Examples relative to the application of GIS in the field of noise propagation modeling are the Sound Mapping Tools (SMT) for terrestrial environments (Keyel et al., 2017) and the Effects of Sounds in the Marine Environment (ESME) workbench (Mountain et al., 2013) for marine environments. SMT allows for the spatially-explicit estimation of sound pressure levels emitted by existing, as well as future terrestrial noise sources (e.g., road traffic, construction sites). The ESME workbench was designed to allow government, industries, and researchers to explore the effects of impulsive noise on marine species by combining acoustic propagation and marine mammal movement simulations.

The present study aims to conceptualize and test GIS-based geovisualization tools to support marine planners and managers in the decision-making process relative to the issue of vessel noise. Even though trials are unarguably a necessary step in the development of new management strategies, the analysis of alternative scenarios allows decision-makers to explore solutions, and possibly anticipate negative outcomes and weak points before entering the testing phase of a policy (Al-Kassab et al. 2014). In particular, the present work focuses on the creation of alternative scenarios relative to ship traffic displacement practices (i.e., "relocation of shipping traffic lanes" and "redirecting a portion of vessel traffic") as a short-term solution for the reduction of shipping noise for the endangered SRKW population.

Three geoprocessing tools are presented that allow spatial planners and managers to explore and analyze data relative to noise pollution from shipping and cetacean species distribution. More specifically, the proposed noise exposure analysis framework is centred on the concepts of exposure mapping (Lahr and Kooistra, 2010), on the use of cumulative distribution functions (CDF) (Nicholson, 2014) for the computation of probabilistic levels of a pollutant's exposure (Jin et al., 2015; Uddh-Söderberg et al., 2015; Zandbergen and

Chakraborty, 2006), and on the use of a least-cost path (LCP) analysis for the identification of shipping routes that minimize the overlap between vessels and cetaceans within the study area.

3.2 METHODS

3.2.1 DATA SOURCES

3.2.1.1 CUMULATIVE NOISE FROM SHIPPING

Cumulative ocean noise, generated by 22 different classes of commercial ships, was modeled in the study area by Jasco Applied Sciences (O'Neill et al., 2017) for the Canadian project NEMES (Noise Exposure to the Marine Environment from Ships). The modeling approach described in O'Neill et al. (2017). For a more detailed description of the modeling approach see Section 2.2.4. Ship movement data, collected through the Satellite Automatic Identification System (S-AIS), and provided by exactEarth (http://www.exactearth.com), were used to model vessels' noise in the Salish Sea. S-AIS records, combined with vessel noise source levels and oceanographic as well as geoacoustic data were used to estimate cumulative noise in terms of Equivalent Continuous Sound Pressure Level (Leq), expressed in Decibels with a reference pressure of 1 μPa. Leq values were computed over three distinct periods (i.e., January 2015, July 2015 and January 2016) and for simulated receivers placed at two different depths (i.e., 10 m and 50 m). The output of the model was made available as gridded data at 800 m resolution covering the entire Salish Sea. The data include an estimation of Leq values for each vessel class, as well as for all the classes together (Fig. 3.1 A). Furthermore, the model includes Leq values weighted according to the SRKW's hearing sensitivity (Fig. 3.1 B), only expressing noise within the auditory frequency threshold of the species (Owen et al., 2016).

For this study, the gridded data were converted into raster datasets for each combination of vessel class, period, and receiver depth, for both unweighted and weighted Leq values.

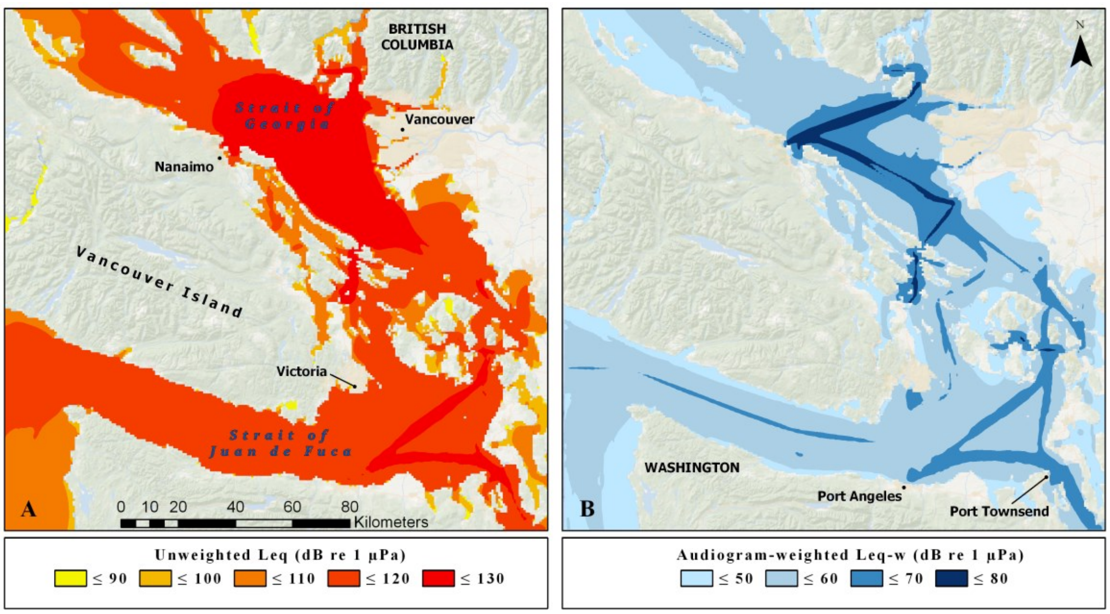

Figure 3.1. Two examples of the cumulative noise model output. Unweighted cumulative noise values (L_{eq}) for all 22 vessel classes (A). Audiogram-weighted cumulative noise values ($L_{eq\text{-}w}$) for all 22 vessel classes (B). Both maps are based on AIS records transmitted during January 2015.

3.2.1.2 CETACEAN DATA

Information relative to the summer distribution of seven cetacean species was provided by the British Columbia (BC) Cetacean Sighting Network (BCCSN) (https://www.vanaqua.org/act/direct-action/bc-cetaceans-sighting-network). The dataset contains more than 80,000 cetacean observations summarized in 5x5 km cells that cover the entire Salish Sea (Fig. 3.2). According to the Canadian Species at Risk Act (SARA), three of the cetacean populations found in BC waters are listed as "not at risk" (i.e., Dall's porpoise; Pacific white-sided dolphin; minke whale), two are listed as "special concern" (i.e., harbor porpoise; grey whale), four are listed as "threatened" (i.e., humpback whale, transient killer whale, offshore killer whale, and northern resident killer whale), and one is listed as "endangered" (i.e., SRKW). The gridded cetacean dataset contains effort weighted sightings for each one of the species. However, the BCCSN dataset does not distinguish between resident, transient, and offshore killer whales. Since SRKW is the only cetacean population listed as "endangered" in the Salish Sea, four maps created as part of an earlier study (Chapter 2) were included to describe SRKW's summer core areas in more detail. These maps (Fig. 3.3, A to D) provide a fine scale estimation of the entire population's summer distribution, as well as an estimation of the distribution of the three social groups constituting the SRKW population: the J, K and L pods. Similarly to the cumulative noise data, all the maps describing cetacean presence were converted into raster datasets with a spatial resolution of 800 m.

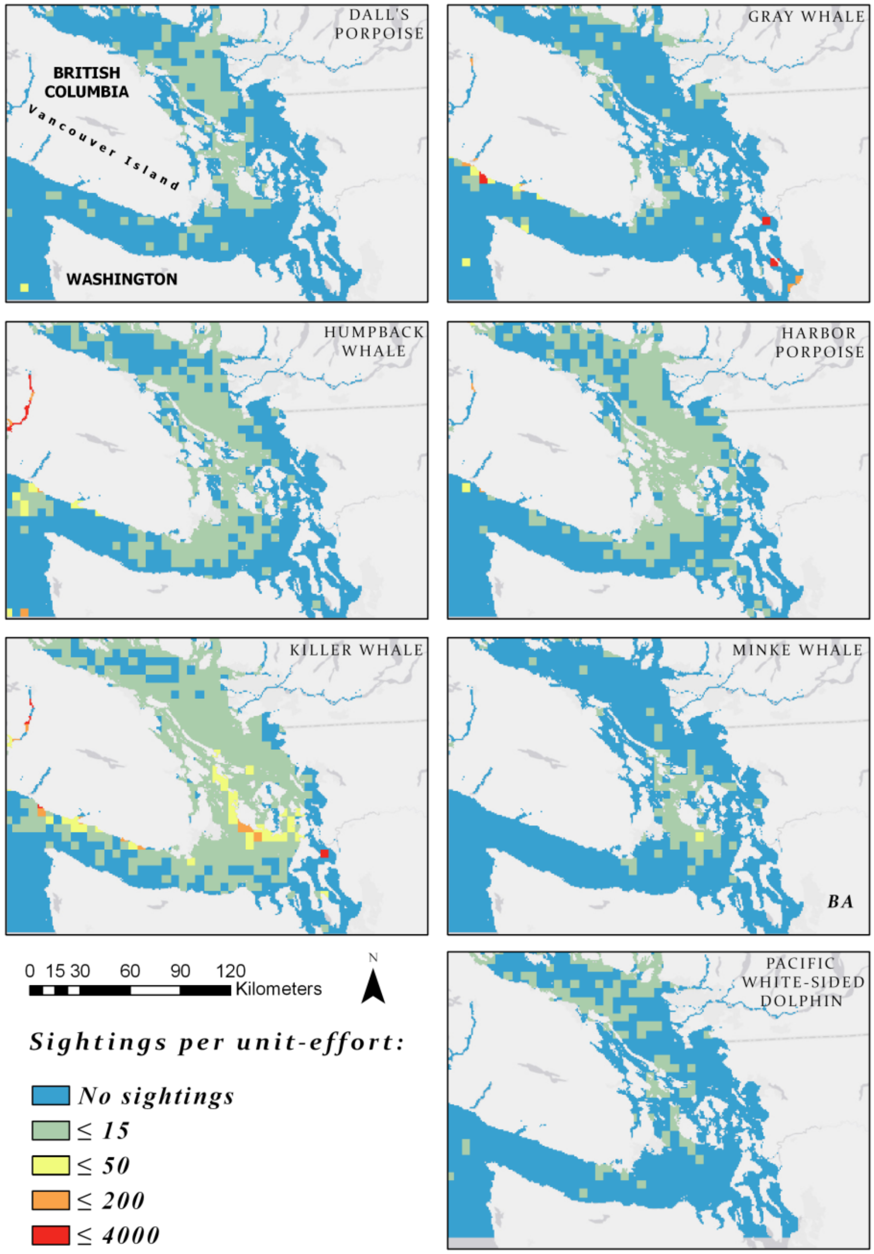

Figure 3.2. Sightings per unit-effort from the BCCSN. Dall's porpoise (*Phocoenoides dalli*); grey whale (*Eschrichtius robustus*); humpback whale (*Megaptera novaeangliae*); harbor porpoise (*Phocoena phocoena*); killer whale (*Orcinus orca*); minke whale (*Balaenoptera acutorostrata*); Pacific white-sided dolphin (*Lagenorhynchus obliquidens*).

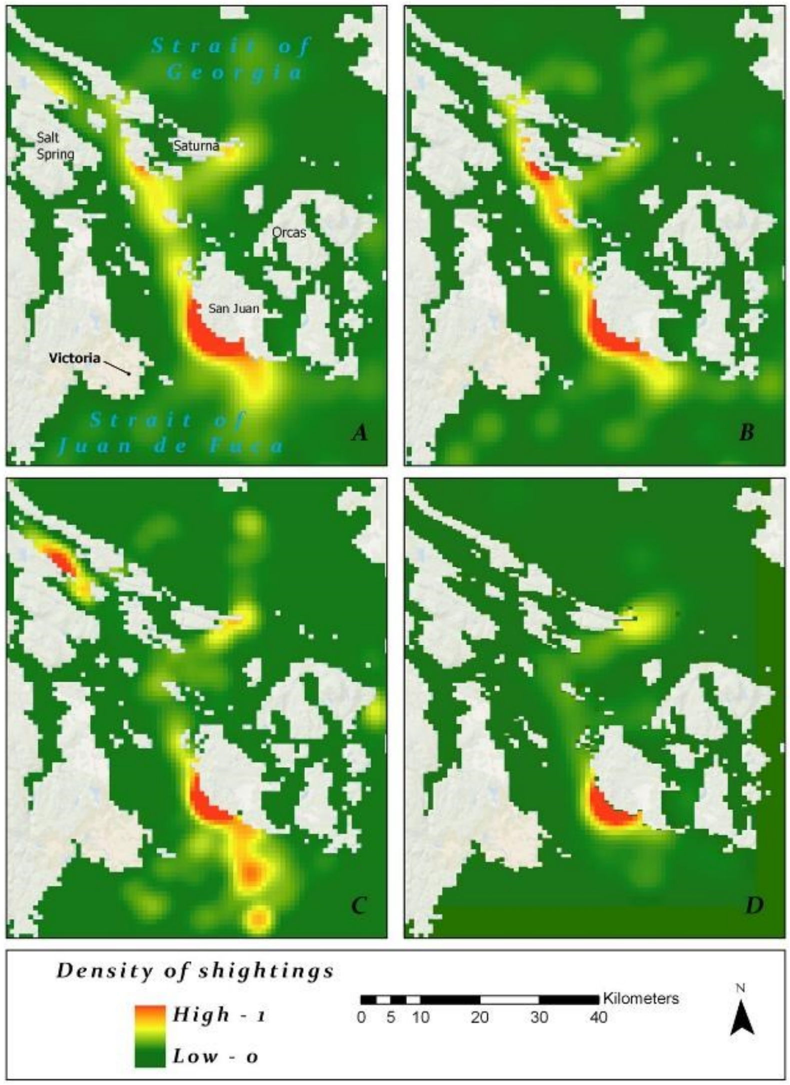

Figure 3.3. Kernel Density Estimations (KDEs) describing SRKW summer core areas for the entire population (A) and for the J (B), the L (C) and K (D) pods.

3.2.2 Exposure Mapping

Exposure maps combine measured or predicted levels of pollutants relative to an area of interest (AOI) with the geographical distribution of ecological or human receptors (Lahr and Kooistra, 2010). In the present study, exposure mapping was applied to the issue of ship-related noise exposure for cetacean populations following three main steps. First, for each of the selected vessel classes, the receiver depth and the season values were converted into a unique alphanumeric code used to extract the corresponding cumulative noise maps. The code consisted of two digits for the receiver depth, six for the modeling period (i.e., four for the year and two for the month) and either 2 or 3 digits unique to each single vessel class.

In the second step, the noise values, expressed as Leq dB re 1 µPa, were summed using the dB summation formula:

4 $$L_{eq_i} = 10 * \log_{10}\left(\sum_{v=1}^{n} \sum_{i=1}^{k} 10^{\frac{Leq_{v,i}}{10}}\right), \qquad [1]$$

5

where L_{eq_i} is the total Equivalent Continuous Sound Pressure Level stored in a cell, *i*, of the resulting noise map; $10^{\frac{Leq_{v,i}}{10}}$ is the exponential value corresponding to the cumulative noise expressed in dB, $L_{eq_{v,i}}$, for a vessel class, *v*; *n* is the number of vessel classes being considered; and k is the total number of cells.

For example, for the two vessel classes *Reefers* and *Tankers*, the sum of their contributions to the cumulative noise within a single 800 m cell, *i*, is equal to:

$$L_{eq_{tot,i}} = 10 * \log_{10}(10^{\frac{L_{eq_{Reefers,i}}}{10}} + 10^{\frac{L_{eq_{Tankers,i}}}{10}}) \ . \qquad [2]$$

This was calculated for each of the 800 m cells of the cumulative noise model. During this phase, a set of diagnostic maps was generated, representing the relative (i.e., percentage) contribution of each selected ship class to the cumulative noise. The relative contribution of a vessel class, *v*, within a cell, *i*, was computed as follows:

$$RC_{v,i} = 100 * \ 10^{\frac{L_{eq_{v,i}}}{10}} / \sum_{v=1}^{n} \sum_{i=1}^{k} 10^{\frac{L_{eq_{v,i}}}{10}}, \qquad [3]$$

where $RC_{v,i}$ is the percentage contribution of a vessel class, *v*, relative to the overall noise attributed to the selected vessel classes, $\sum_{v=1}^{n} \sum_{i=1}^{k} 10^{\frac{L_{eq_{v,i}}}{10}}$. Similar to the summation of the contributions, this was calculated for each of the 800m cells of the cumulative noise model.

In the third phase, the resulting noise map and the species distribution map were combined to create a noise exposure hotspot map. First, the species distribution map was standardized to range from 0 to 1 as follows:

$$Raster_{STD} \ = \ \frac{(Raster - Raster.minimum)}{(Raster.maximum - Raster.minimum)}, \qquad [4]$$

where *Raster* is a species distribution map; *Raster.minimum* and *Raster.maximum* are respectively the minimum and maximum values stored in the raster dataset, and $Raster_{STD}$ is the resulting standardized raster. Values below 0.1 were removed from the species distribution map to exclude all the areas characterized by very low or null presence of the selected species. Similarly, in the case of unweighted noise maps, all values below 60 dB re 1 μPa were removed and the map was standardized by applying Eq. (4). This step was skipped for the audiogram-weighted noise maps. The noise exposure hotspot map was computed as:

$$Noise\ Exposure\ Hotspots\ Map = \ Noise\ Map_{STD} * Species\ Distribution\ raster_{STD}\ . \qquad [5]$$

The resulting exposure map was then rescaled to range from 0 to 1 by applying Eq. (4). This process can be applied to query combinations of vessels types, receiver depth, and seasonal values and to calculate exposure maps for any of the species distribution maps described above.

3.2.3 PROBABILISTIC LEVEL OF EXPOSURE

Cumulative distribution functions (CDF) (Nicholson, 2014) have been used for the probabilistic estimation of median levels of exposure to arsenic (Uddh-Söderberg et al., 2015) and copper (Jin et al., 2015), and have been proposed as a valid approach to map oceanic sound exposure levels (Gervaise et al., 2015). CDFs were used in the present study to estimate median sound exposure levels for SRKW following three distinct steps. First, probability values were computed as:

$$P_i = SDmap_i / \sum_{i=1}^{n} SDmap_i\ , \qquad [6]$$

where $SDmap_i$ is the value stored in cell i of the species distribution map, $\sum_{i=1}^{n} SDmap_i$ is the sum of all the values contained in the species distribution map.

Second, the accumulated probability for every single value of Leq stored in the noise map were iteratively computed within the selected AOI as follows:

$$F_{L_{eq}} = \sum P_i \quad \forall\ i \text{ where } L_i \leq L_{eq}\ , \qquad [7]$$

where $F_{L_{eq}}$ is the cumulative probability of a species to be exposed to a noise value equal to or less than L_{eq}. Starting from the minimum L_{eq} value, and proceeding by 1 dB increments, this step was repeated until L' was equal to the maximum L_{eq} value stored in the noise map. In the third step, the analysis results were saved in the form of a table and as a two-dimensional plot showing the cumulative probability values along the y axis and the corresponding L_{eq} values along the x axis (Fig. 3.7). These two outputs were then used to identify $L_{eq-50th}$, which is the CDF value corresponding to the median level of noise exposure computed over the selected AOI.

3.2.4 Generation of Ship Traffic Scenarios

The LCP model combines a set of spatially explicit variables, quantifying the level of friction associated with a surface, and identifies a path (or set of paths) characterized by the lowest possible cumulative cost (Adriaensen et al., 2003; Douglas, 1994). In the past, LCP analyses have been applied in different research contexts, such as identifying ecological corridors for wildlife populations (Alexander et al., 2016), routing power lines to minimize environmental impacts (Bagli et al., 2011), and identifying archaeological sites in remote areas (Gustas and Supernant, 2017).

The generation of the ship traffic scenarios presented in this study required the creation of a cost surface and the generation of a new shipping route through an LCP analysis. A set of raster datasets representing different costs (i.e., "species" and "vessel" costs) were combined into a single cost surface. Each cost raster was weighted (i.e. multiplied) by a constant value ranging from 0 to 1 with the sum of all values being equal to 1. Portions of the cost surface that represent hazards to navigation (e.g., shallow areas) were excluded from the analysis. Of the two scenarios described in Section 4, the re-routing scenario (i.e., Scenario A) was computed using a single cost surface: SRKW summer core area (Fig. 3.3 A). The lateral displacement scenario (i.e., Scenario B) was computed combining two distinct cost surfaces. The first cost, a species cost, was represented by the KDE values falling within the limit of the 50% Percentage Volume Contours (PVCs) identified for SRKW (Fig. 3.8 A). The 50% PVC represents the smallest polygon encompassing 50% of the density of a kernel (Anderson, 1982). The second cost, a vessel cost, was represented by a raster dataset storing distances computed from the cell containing the origin of the route to each one of the cells of the raster. Since the purpose of the analysis was to identify possible solutions to reduce noise exposure for SRKWs, species costs were considered more relevant than the vessel costs. Tests conducted using weights of 0.5, 0.2, and 0 for the species cost, and weights of 0.5, 0.8, and 1 for the vessel cost all resulted in similar solutions (Appendix C, Fig C.1), which minimized the travelled distance. The resulting least-cost paths did avoid the 50% PVC along the coast of San Juan Island (Appendix C, Fig C.1), but no changes were observed in the remaining two areas. Assigning a weight of 1 to the species cost and 0 to the vessel cost generated the same result obtained in Scenario A. Assigning a weight of 0.8 to the species cost and 0.2 to the vessel cost resulted in an additional reduction of overlap with the 50% PVC located along the east coast of Saturna Island. Consequently, a weight of 0.8 was assigned to the species cost, while a weight of 0.2 was assigned to the vessel cost.

The resulting cost surface was then used as input in the LCP analysis to generate optimal routes.

The methods described above were tested in a GIS environment and the results were visualized using Esri's ArcGIS Pro software. Scripts were developed for the tools using the Python 3.5 programming language and presented in a custom programmed user interface that can be added to the ArcGIS Pro ribbon as an add-in.

3.3 Noise Exposure Geovisualization Tools

This section provides an overview of the geoprocessing and geovisualization tools, their required inputs and their outputs. A high-level representation of the framework for the analysis of noise exposure is shown in Fig. 3.4.

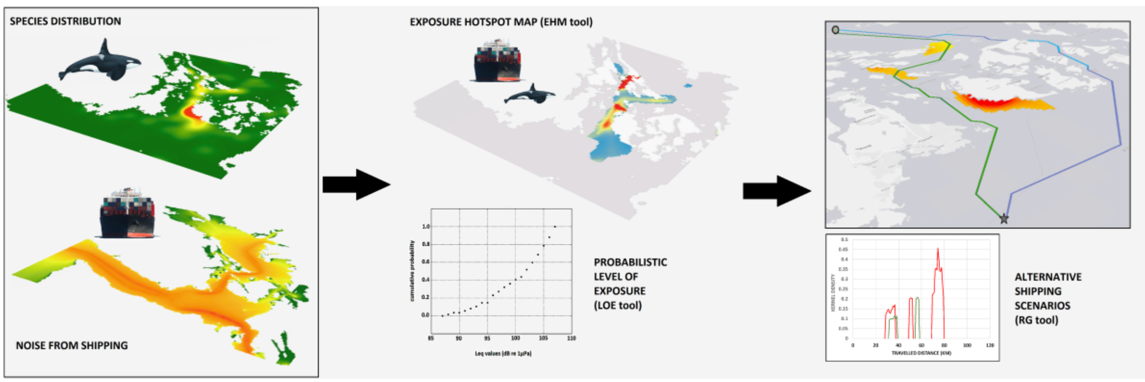

Figure 3.4. Noise exposure analysis framework. Left: initial raster datasets (i.e., species distribution and noise from shipping). Center: outputs of the Exposure Hotspots Maps (EHM) and Level of Exposure (LOE) tools, derived from the combination of the initial raster datasets. Right: The output of the EHM and LOE tools are used to guide the creation of alternative shipping scenarios when running the Route Generator (RG) tool.

The exposure hotspot maps (EHM) tool allows for the creation of noise exposure maps representing the co-occurrence of high levels of cumulative vessel noise and cetacean species (i.e., hotspots). To be run, the tool requires five inputs: a *cetacean species or population of interest*, a *species distribution map*, *vessel classes* of interest, a *receiver depth*, and a *season*.

The level of exposure (LOE) tool allows computing the level of exposure attributed to one or several vessel classes, depending on the selected noise map. The tool generates a cumulative distribution function (CDF) over a user-defined AOI. The median L_{eq} value, corresponding to an accumulated probability, $P = 0.5$, is returned as the level of exposure characterizing the user-defined area. The tool requires three inputs: *species distribution map*, *noise map,* and *area to analyze*. The noise map can contain values from a single class as well as from a combination of classes, such as the noise map obtained from the EHM tool.

The route generator (RG) tool is based upon an LCP analysis. While the EHM tool helps identify critical areas within the Salish Sea characterized by both high levels of modeled cumulative noise and high probability of cetacean presence, the RG tool can help identify potential routes that reduce the level of exposure for the selected species. A prerequisite for running the RG tool is the creation of a cost surface (Section 2.4). The creation of a cost surface is handled through a separate tool and requires two inputs: a list of raster datasets representing different costs that can be used to compute an overall cost, and a list of weights representing the relative importance of each cost. The tool multiplies each user-selected raster dataset by its corresponding weight and returns the sum of the weighted raster datasets. The use of optional parameters (e.g., *polygon feature to exclude*) allows specific areas to be excluded from the cost surface (e.g., navigation hazards, forbidden areas). The *depth limit for safe navigation* parameter, another optional parameter, allows the user to remove from the cost surface all those areas that are not compatible with the selected ship classes' maximum navigable depths.

The RG tool accepts two parameters: *cost surface* and *sequence of points*. The user can insert a sequence of point features representing the origin, destination, and waypoints of a shipping route. These points can either be extracted from an existing track or represent a hypothetical route. Once all the parameters are set, the tool processes the selected input following two steps. First, the RG tool generates two LCP (i.e., the shortest and optimized paths) for each consecutive pair of points, starting from the origin (i.e., the first point specified by the user) and progressively analyzing each point-to-point segment until the destination point is reached. Multiple runs of the RG tool, performed using different cost surfaces, can be used to generate alternative scenarios. The resulting routing options can then be compared in terms of traveled distance and degree of overlap with a species' core areas.

3.4 APPLICATION EXAMPLES

In order to test the prototype tools, the analytical framework (Fig. 3.4) and models were applied using real data (Figs. 3.1 and 3.3) in the context of the Salish Sea to answer the following analysis and management questions:

- Which portions of SRKW summer core areas can be identified as noise exposure hotspots for this population?
- Which vessel classes are driving the estimated level of noise exposure within the hotspots?
- Which management solution (i.e., re-routing vs. lateral displacement) could be adopted to reduce noise exposure for SRKW?

The EHM tool was first used to compute the total contribution to the cumulative noise attributed to the six classes identified as main contributors to SRKW's noise exposure levels (Ferries, Tugboats, Recreational Vessels, Vehicle Carriers, Containers, and Bulkers) (Fig. 3.5 A) (Chapter 2). The contribution of these classes was then combined with the KDE describing SRKW summer core areas to identify hotspots of exposure (Fig. 3.5 B). Finally, the different classes were compared based on their percentage contribution to the cumulative noise (Fig. 3.6).

The EHM tool allowed a visual identification of four hotspots within SRKW summer core areas: HS1, HS2, HS3 and HS4 (Fig. 3.5 B). The classes driving noise exposure in each one of these four locations were identified using the percentage contribution maps (Fig. 3.6, A to F). The contribution of Ferries crossing the Strait of Georgia (Fig. 3.6 A) was mainly located in the northern portion of the study area, mostly along a few fixed routes. Noise exposure in HS3, where Ferries showed 100% contribution to the cumulative noise, was most likely driven by this class. Similarly, noise exposure in HS2 could almost entirely be attributed to Ferries. Tugboats contributed 100% in all those areas where large commercial vessels (i.e., Containers, Vehicle Carriers, and Bulkers) and Ferries were absent (Fig. 3.6 B). However, this vessel class dropped to 0% noise contribution along ferry routes while its contribution ranged from 10% to 50% within the boundaries of the international shipping lane. HS1 and HS4 were both localized in areas where tugboats reached > 50%, suggesting that this class may be responsible for a large portion of the noise produced by commercial vessels in the area. Recreational vessels contributed 10% of the noise over the majority of the study area, with the exception of a few locations where the percentage contribution reached 100%. The high-contribution areas for Recreational vessels were located within the narrow seaways of the San Juan Archipelago and of the Southern Gulf Islands, in Indian Arm (i.e. north of Vancouver), along the northern and southern tips of Denman Island, and along the south coast of Camano Island, in Washington State (Fig. 3.6 C).

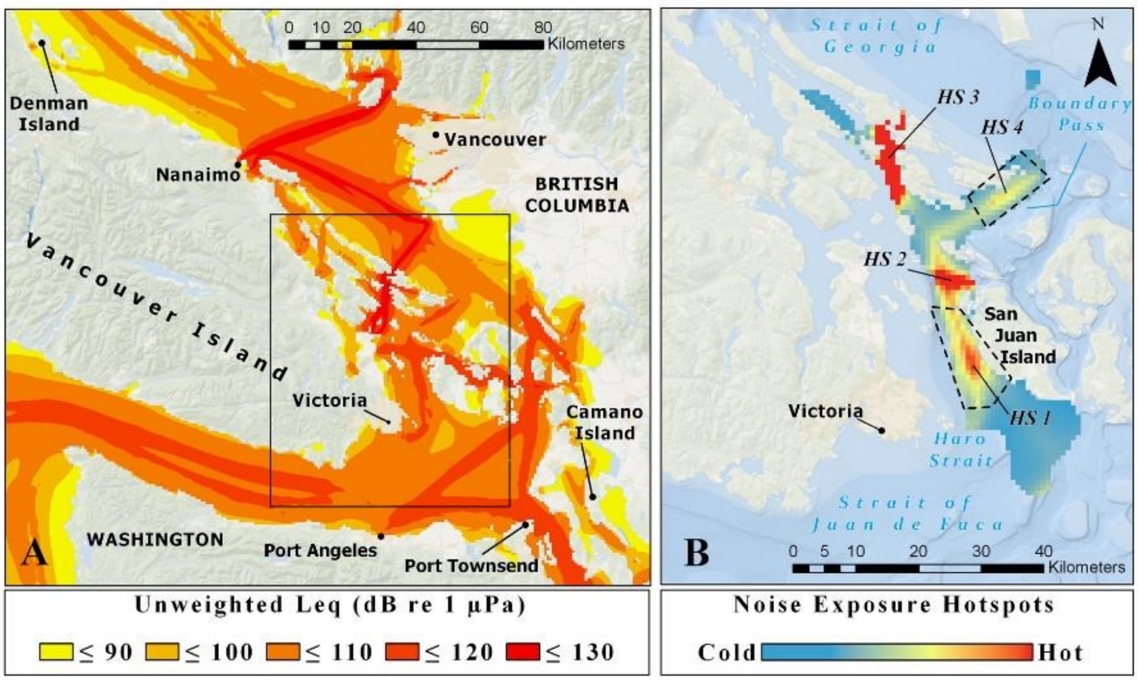

Figure 3.5. (A) Total cumulative noise attributed to Ferries, Tugboats, Recreational Vessels, Vehicle Carriers, Containers, and Bulkers together. (B) Noise Exposure Hotspot Map for the aforementioned classes. Areas in red are characterized by a high degree of overlap between vessel noise and SRKW whereas areas in blue are indicating a low degree of overlap. HS1-4 indicate noise exposure hotspots. Dashed lines display the areas used to compute noise exposure levels for hotspots HS1 and HS4.

Vehicle Carriers (Fig. 3.6 D) showed contributions ranging from 20% to 30% along the international shipping lane, with peaks of > 50% located in the proximity of major ports such as Vancouver (BC), and Port Townsend (WA). Similarly, Containers (Fig. 3.6 E) and Bulkers (Fig. 3.6 F) showed 10-40% contribution to recorded noise along the international shipping lanes and peaks in the proximity of major ports. Containers also displayed a peak at the western end of the Strait of Juan de Fuca. Ferries and Tugboats seem to be driving the cumulative noise and were attributed 100% contribution over large portions of the study area. The contribution to cumulative noise from Recreational Vessels was characterized by highly localized peaks. Vehicle Carriers, Containers, and Bulkers displayed similar patterns in their percentage contribution, with Containers showing the most relevant contribution amongst these three classes.

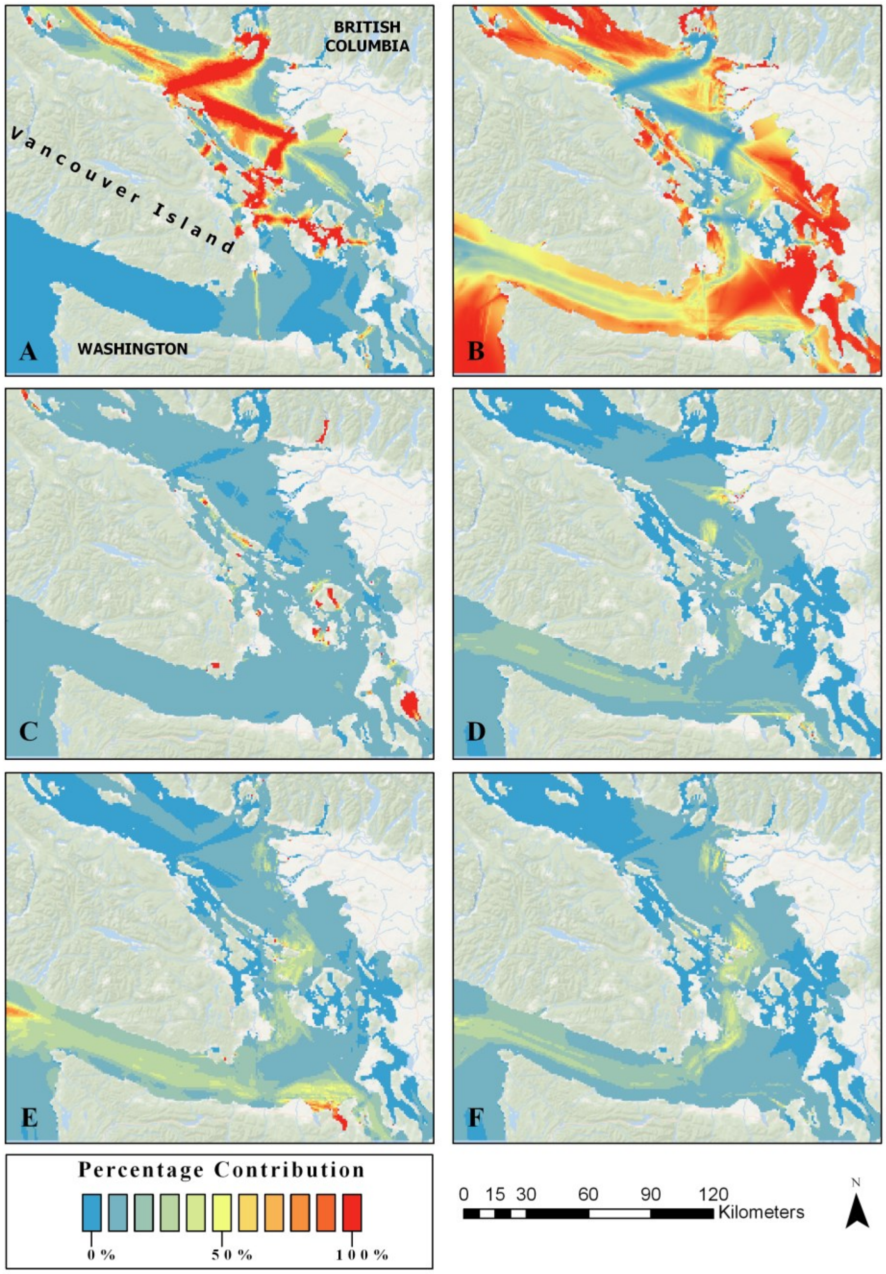

Figure 3.6. Percentage contribution raster produced by the EHM tool for Ferries (A), Tugboats (B), recreational Vessels (C), Vehicle Carriers (D), Containers (E), and Bulkers (F).

Although the EHS tool helped identify Ferries as the class driving noise exposure over HS3 and HS2, it was not possible to define which classes were contributing the most to noise exposure over HS1 and HS4. A deeper understanding of vessel noise over these hotspots could be obtained by running the LOE tool. One run of the LOE tool was completed for each one of the six vessel categories over the two hotspots, HS1 and HS4, for a total of 12 runs. The resulting CDF curves are reported in Fig. 3.7 while the corresponding exposure levels are summarized in Table 1.

Table 3.1. Exposure levels for Ferries, Tugboats, Recreational Vessels, Vehicle Carriers, Containers, and Bulkers computed over the hotspots HS1 and HS4.

Vessel Classes	**$L_{eq\text{-}50th}$ (dB re 1 µPA)**	
	HS1	***HS4***
*Ferries**	86	83
*Tugboats**	103	104
Recreational Vessels	87	78
Vehicle Carriers	100	100
*Containers**	102	103
*Bulkers**	102	103

Note: The corresponding CDF curves are reported in Fig. 3.7. Classes marked with * are pooled (i.e., are representative of more than one class). Noise maps for pooled classes were created using the EHM tool. Noise for the Ferries category is the sum of noise from the Ferries > 50 m, Ferries < 50 m and High-Speed Ferries categories. Tugboats includes Tugs < 50m and Tugs > 50 m. Containers include container ships <200 m and container ships > 200 m. Bulkers include bulk carriers < 200 m and bulk carriers > 200m.

Ferries and recreational vessels showed relatively low levels of noise exposure within HS1 and HS2. Consequently, these two categories were omitted from further analysis. The remaining four classes were divided into two groups: Tugboats, and Large Commercial Vessels (i.e., Vehicle Carriers, Containers, and Bulkers). The $L_{eq\text{-}50th}$ for Tugboats in HS1 and HS4 was respectively 103 and 104 dB re 1 μPa. The combined total $L_{eq\text{-}50th}$ for Large Commercial Vessels in HS1 and HS4 was respectively 106 and 107 dB re 1 μPa. Since Tugboats showed a large noise footprint over the entire study area (Fig. 3.6 B), it was possible to conclude that re-routing might not be a viable solution for this category. Hence, small adjustments to the current routes (i.e., lateral displacement) might be a better option for this category. However, since Large Commercial Vessels showed localized noise footprints (Fig. 3.6 D to F), re-routing part of the traffic from this group, may significantly reduce SRKW noise exposure levels. To explore these two possible scenarios (i.e., re-routing and lateral-displacement) starting from an existing vessel route, multiple runs of the RG tool were completed and compared.

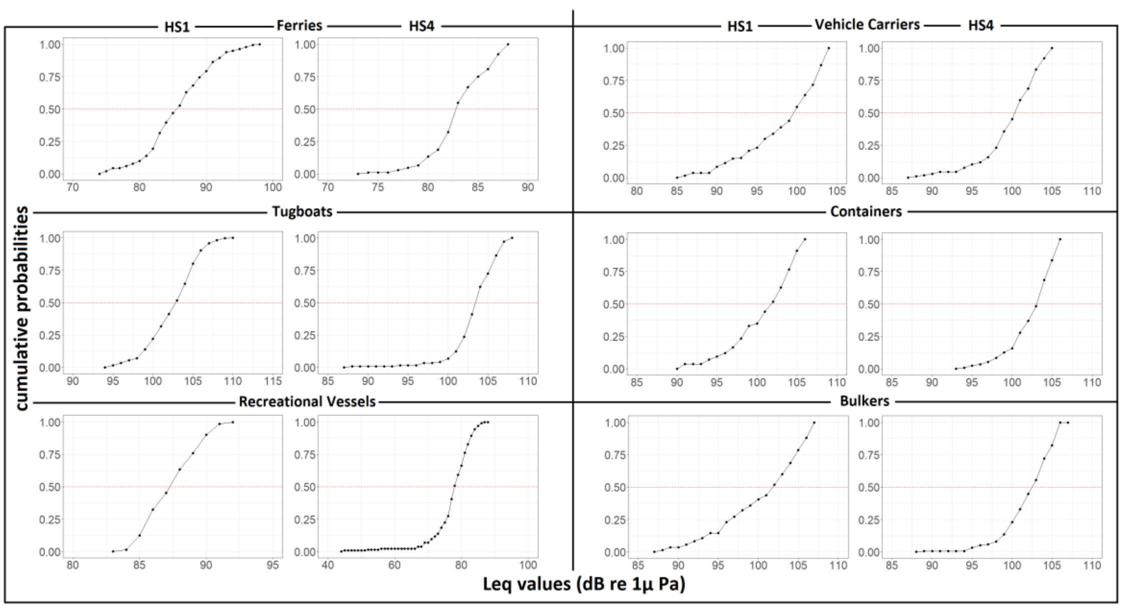

Figure 3.7. Results from 12 runs of the LOE tool. CDF for Ferries, Tugboats, Recreational Vessels, Vehicle Carriers, Containers and Bulkers over the hotspots areas HS1 and HS4 (Fig. 3.5 B). Cumulative probability values are reported along the y-axis while noise values (L_{eq}) are reported along the x-axis. CDFs are computed starting from the minimum L_{eq} value recorded within the AOI and for each 1 dB increase until the maximum L_{eq} value is reached. Consequently, the x-axis has different ranges from one vessel category to the other. Levels of exposure ($L_{eq\text{-}50th}$) are reported in Table 3.1.

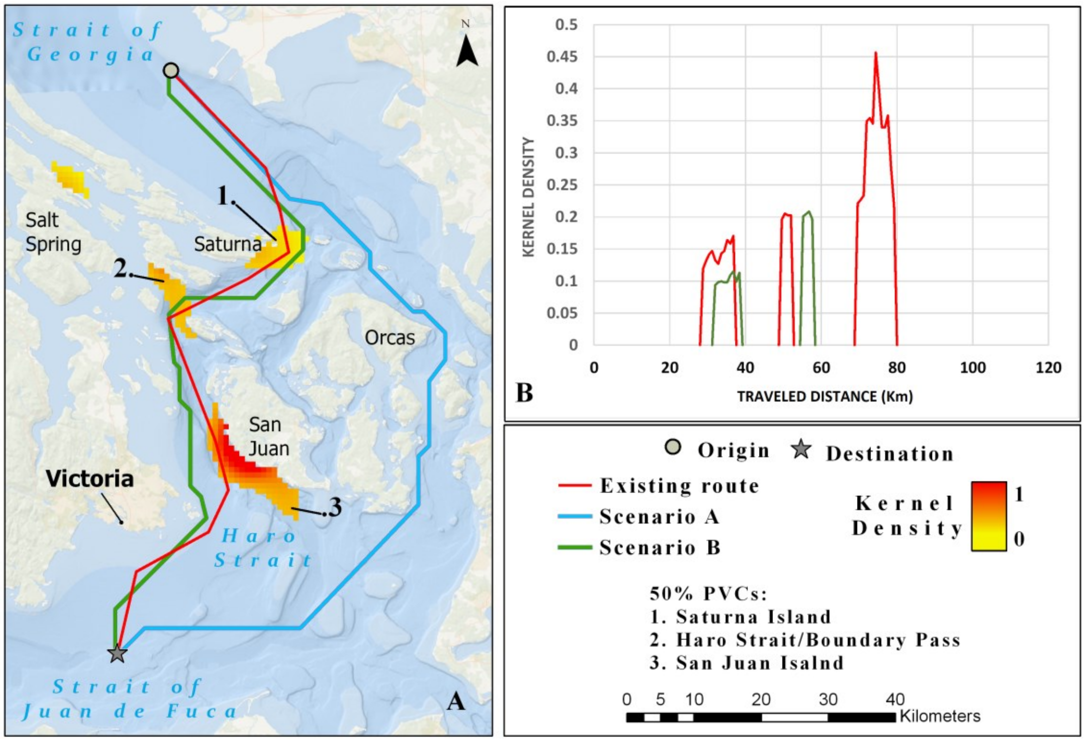

Figure 3.8. (A) The existing route (red) was used as the starting point to run the RG tool and the two least-cost path scenarios: A (blue) and B (green). SRKW 50% PVCs and kernel density values within the PVCs are included to show the degree of overlap between the three routes and SRKW summer core areas. (B) Plot showing the KDE values along the original route as well as along Scenario B. Scenario A is not displayed in the graph because this solution does not overlap with SRKW 50% PVCs.

Comparisons between scenarios used two metrics: route length and overlap with the 50% PVCs extracted from SRKW summer core areas map. The original route was 112.6 km long and took approximately 5:30:00 and 4:00 to be completed at constant speeds of 11 kn and 15 kn, respectively.

Table 3.2. Comparison of the three routes: the original route, scenario A, and scenario B.

Route	Length (km)	Increase in length (km)*	Increase in length (%)*	50% PVC†	50% PVC overlap	
					km$^{\Delta}$	Reduction‡
Original	112.6	-	-	1	8	-
				2	2.4	-
				3	9.6	-
				Total	**20.0**	**-**
Scenario A	140.1	27.4	24.0%	1	0	-100%
				2	0	-100%
				3	0	-100%
				Total	**0**	**-100%**
Scenario B	116.5	3.8	3.4%	1	6.4	-20%
				2	2.4	0%
				3	0	-100%
				Total	**8.8**	**-56%**

*Absolute (km) and relative (%) increase in traveled distance for scenario A and B in comparison to the original route.
†SRKW 50% PVCs as shown in Fig. 8A. $^{\Delta}$Absolute (km) distance traveled within the 50% PVCs for each route.
‡ Relative (%) reduction in the traveled distance within the 50% PVCs in comparison to the original route.

The original route (Figs. 3.8 A and 3.9 A, Table 3.2) had 20.0 km of navigation overlapping with SRKW 50% PVC, which corresponded to a p = 0.60 (i.e. the area under the curve in Fig. 3.8 B). Of these 20 km, approximately 10 km overlapped with areas characterized by a density of SRKW sightings (i.e. kernel density) between 0.1 and 0.2 (i.e. PVC 1, PVC 2) (Fig. 3.8 A and B, Table 3.2), and the remaining 10 km reached values of 0.45 (Fig 3.8 B and Table 3.2), indicating overlap with areas characterized by high probability of encountering SRKW over the summer. Scenario A resulted in a 100% reduction in the distance traveled within SRKW 50% PVCs (Fig. 3.8 A and B, Table 3.2) but also in a 24% increase in the total traveled distance from 112.6 km to 140.1 km. Scenario B resulted in a 56% reduction (i.e. from 20.0 km to 8.8 km) in the distance traveled within SRKW 50%

PVC and in a corresponding 50% reduction in the probability of encountering a member of SRKW, which dropped from the initial p = 0.60 to p = 0.27. When compared to the original route, Scenario B resulted in the complete avoidance of PVC 3 (Fig. 3.8 A and B, Table 3.2). In Scenario B, only areas with a KDE value < =0.1 were crossed within PVC 1 (Fig. 3.8 A and B, Table 3.2), whereas the degree of overlap with PVC 2 remained substantially unchanged. The overall 56% reduction in overlap with SRKW PVCs was achieved with a 3.4% increase in the total traveled distance, from 112.6 km to 116.5 km.

The RG tool allows performing similar analysis starting from different user-defined routes and cost surfaces, depending on the objective of the user. The results can then be compared using simple metrics to understand which scenario meets the required goals. In this specific example, the goal was to compare re-routing and lateral displacement as noise management practices.

3.5 Discussion and Conclusion

Recent studies have highlighted how current vessel management solutions that aim to reduce anthropogenic impacts on marine mammals are often either not effective for the reduction of noise pollution (Holt et al. 2017) or can lead to unexpected results (Chion et al., 2017; McKenna et al., 2017). For example, Holt et al. (2017) demonstrated how noise levels received by SRKW vary, over different years, independently from the adoption of mitigation measures. Focusing on vessel noise in Glacier Bay National Park, McKenna et al. (2017) showed how vessel scheduling decisions, changes in commercial vessel services, routes and fleet composition are all factors contributing to the observed noise levels, independently of the current mitigation measures. Although limited by the availability of baseline recordings, these results suggest that the exploration and analysis of acoustic datasets, species distribution models, and possible scenarios, can lead to more effective noise management strategies. Building upon this research, the tools presented in this study allow marine managers to explore the results of acoustic models in a scenario-oriented GIS environment. The analysis of hotspot maps, percentage contribution raster, and probabilistic levels of exposure discussed above provide insight into the overlap between noise generated by vessel traffic and an endangered cetacean population in the Salish Sea. Moreover, the RG tool was used to explore two possible management solutions: re-routing and lateral displacement of vessels.

Noise from shipping overlaps with killer whale echolocation frequencies (Veirs et al., 2016), and a killer whale behavioural modeling study (Scott-Hayward et al., 2015b) predicted a high probability of observing SRKW entertaining feeding activities along the south-western coast of San Juan Island. Consequently, re-routing vessel traffic from the Haro Strait to the Rosario Strait could reduce the number of noise sources present within the SRKW summer core area. However, there are several limitations to this approach. Specifically, the outcome is variable depending on the selected quota of vessels to displace, there are geographic constraints to the navigation of certain classes of ships through Rosario Strait, and redirecting the noise in a narrower seaway may actually result in increased noise levels (DFO, 2017b). Furthermore, as displayed by Scenario A, traffic coming from Robert's Banks terminals (Delta, BC) and going toward the open Pacific Ocean through the Juan de Fuca Strait (and *vice versa*) would travel for almost 30 additional km within the Salish Sea ecosystem. Hence, this scenario may actually lead to increased rather than attenuated overall noise exposure.

According to the Canadian Department of Fisheries and Oceans (DFO), when a critical habitat is in close proximity to a shipping lane (e.g., 100 m), small adjustments in the order of 10s of meters (e.g., 20-100 m) could drastically reduce received noise levels (DFO, 2017b). Scenario B, which combines avoidance of SRKW core areas with total traveled distance to identify an optimal vessel route, provided a starting point to explore this solution. With a westward displacement when navigating the Haro Strait and an eastward displacement when navigating the Boundary Pass, two SRKW core areas may experience significant reductions in received noise levels. The achieved reduction in noise levels would, however, be variable depending on the actual frequencies. Since high-frequency sounds attenuate at shorter distances than low-frequency sounds, the lateral displacement of noise sources would have greater effects on the former rather than the latter. Nonetheless, high-frequency noise from shipping interferes with SRKW foraging activities (Marla M Holt et al., 2009; Veirs et al., 2016), making lateral displacement a viable strategy to reduce the risk of disrupting SRKW foraging behaviour.

Other than having a frequency-dependent effect on the reduction of noise levels, lateral displacement strategies have two main limitations. First, physical constraints to ship navigation, as for the re-routing strategy, may make this solution suitable only for specific vessel classes. Second, routes would need to be dynamically adjusted if the species of interest changes its geographic distribution. Although SRKW has been shown to exhibit high site fidelity for specific areas of the Salish Sea (Hauser et al. 2007, Chapter 2), recent information collected by DFO suggests that other areas along the coast of British Columbia (e.g., south-western Vancouver Island) are particularly relevant for this population (DFO, 2017c). Consequently, any management practice adopted to reduce SRKW received noise levels should be regularly updated to account for changes in the spatial and temporal distribution of the animals.

The RG tool was designed to accept any number of costs representing different constraints to the navigation of vessels in the Salish Sea. Defining new navigation routes and modifying existing ones requires a detailed understanding of navigation hazards and constrictions. Nevertheless, least-cost paths were found to be a valuable tool for the identification of (pre-)historic routes (Gustas and Supernant, 2017), the identification of safer alternatives to current routes (Choi et al., 2015), and for the prediction of future possibilities (Smith and Stephenson, 2013). The reported example only considered, as a proof of concept, SRKW distribution using Euclidean distance as the vessel cost layer. However, the tools developed support the use of more sophisticated cost layers as inputs to the analysis. For example, areas relevant to other activities, such as fish farms or eco-tourism, could be included as costs in the LCP analysis. More specific constraints on possible/alternative routes can hence be entered into the system to only permit options that would be considered as possible viable alternatives.

Although the examples reported here were all focused on the SRKW population, this paper only used the SRKW population in the Salish Sea as a proof-of-concept of the framework and the approach could be applied similarly to other species of interest and other locations. Also, while some tools were developed for the exploration of the cumulative noise models produced by Jasco for the NEMES project, their functionalities could be extended to assess other environmental stressors related to vessel traffic (e.g., chemical pollutants, or risk of ship strike). However, there are several limitations related to the data inputs of the framework presented in this study, and the results of each step should be carefully interpreted. First, data relative to both the environmental stressor and the species distribution of concern need to be spatially explicit. Second, the spatial resolution of the environmental stressor map (e.g., noise) and of the species distribution map should ideally be the same. Applying the analysis to low-resolution datasets (Fig. 3.2) might lead to scenarios that are not applicable to real life situations. Similarly, resampling fine-scale models at lower resolutions may reduce the predictive power of such models, thus leading to potentially unreliable estimations of exposure levels. Hence, caution should be taken when selecting input datasets for the tools described in this research.

Equally important is the selection of cost surfaces capturing species and vessel costs as well as the selection of their associated weights. For example, overexpressing a species cost could lead to optimal solutions for the species but be very costly from the vessel perspective. This is the case of Scenario A (Fig. 3.8 A and B, Table 3.2), where no vessel cost was considered. Similarly, overexpressing a vessel cost may result in solutions that are not suitable for the exploration of mitigation strategies. Consequently, it is important for possible users to have a clear understanding of which species and vessel costs will be selected as inputs for the analysis. Once the costs are defined, multiple iterations of the RG tool should be run applying different weights to the costs in order to find the correct balance between species and vessel costs. Iterating LCP analysis to identify converging solutions (Choi et al., 2015; Gustas and Supernant, 2017) or to evaluate alternative solutions (Bagli et al., 2011) is a common practice that should be followed when running the RG tool.

In general, these findings underline the need for a broader approach to vessel noise management, which should include various strategies to address noise pollution in its multiple dimensions. By enabling marine managers and planners to explore and analyze acoustic and biological data, the framework presented in this study facilitates the adoption and implementation of adaptive noise management strategies.

3.6 References

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CHAPTER 4 SUMMARY

4.1 FINDINGS

The studies presented in Chapters 2 and 3 add to the existing literature relevant to SRKW and their vulnerability to the increasing levels of noise pollution recorded in the Salish Sea (Hauser et al., 2007; Holt et al., 2011; Lacy et al., 2015; Veirs et al., 2016). More specifically, Chapter 2 provided an estimation of SRKW summer spatial distribution at a finer resolution than preceding studies (Hauser et al., 2007; Hemmera and SMRU Canada Ltd., 2014) based on sightings collected by an environmental non-governmental organization operating in the Salish Sea (i.e. Soundwatch), a method for the estimation of SRKW noise exposure levels from commercial vessels, and a first indication of which classes of vessels are responsible for the vast majority of the modeled cumulative noise produced in the Salish Sea. Such information could support the management of noise exposure for SRKW and guide decision-makers during the design and implementation of mitigation measures. Building on the results obtained in Chapter 2, Chapter 3 described a framework for the analysis of species' exposure to noise pollution from shipping and presented a set of geovisualization tools specifically designed to aid managers and decision-makers in the exploration of noise modeling studies and in the preliminary assessment of possible noise management solutions.

The analysis described in Chapter 2 helped achieve two key results. First, the use of KDEs to describe SRKW summer core areas led to the identification of a large area located along the south-western shore of San Juan Island as well as other pod-specific areas. The San-

Juan Island area is known to be a summer core area for SRKW (Hauser et al., 2007), and to be an important feeding ground for this population (Scott-Hayward et al., 2015), but is also bordered by international shipping lanes and ferry routes, suggesting that animals, when feeding there, may be exposed to high levels of vessel noise. Second, the use of CDFs to combined SRKW summer core areas with the results of JASCO's cumulative noise modeling study (O'Neill et al., 2017) allowed to quantify the spatial risk of noise exposure associated with each vessel category considered in this study. The results expressed as median L_{eq} (i.e. $L_{eq-50^{th}}$), suggest that of the 22 modeled vessel classes, six (i.e. Ferries, Tugs, Recreational Vessels, Vehicle Carriers, Containers, and Bulkers) are responsible for up to 98% of the cumulative noise produced in the central portion of the Salish Sea. Furthermore, as evinced from the exposure maps presented in Chapter 2, the levels of noise exposure for SRKW are variable depending on the selected vessel class and on the selected location within the study area, indicating that both the spatial and temporal traits of vessel traffic need to be considered when designing and implementing vessel-noise mitigation measures. $L_{eq-50^{th}}$ within Zone 2 (Table 2.3), which encompassed the San Juan Island core area, was the highest for Tugboats, Containers, Bulkers and Vehicle Carriers. $L_{eq-50^{th}}$ within Zone 1 are dominated by Ferries, with a median level of exposure 9 dB higher than the next class, Tugboats. Ferries are approximately twice as loud as Tugboats in Zone 1 (Table 2.3). These results are in line with other studies and suggest that different classes of vessels can be distinguished on the basis of their behavior (Kaluza et al., 2010), and that vessel scheduling and routing decisions play a role in determining the amount of acoustic energy released in the environment (McKenna et al., 2017).

In Chapter 3, the analyses presented in Chapter 2 were combined with an LCP analysis and integrated into a framework for the analysis of species' noise exposure from shipping. Applying the framework to the Salish Sea environment allowed answering fundamental questions relative to the distribution of noise pollution from shipping within SRKW summer core areas. The analysis led to the identification of four hotspots of vessel-noise located within SRKW's summer core areas (Fig. 3.5). The adoption of a probabilistic approach in the computation of noise exposure levels (i.e. the use of CDFs) allowed understanding which vessel classes, amongst the six major noise producers identified in Chapter 2, were driving noise exposure within the three hotspots. Moreover, through the application of an LCP analysis, two possible mitigation measures (i.e. lateral displacement, and re-routing) were compared. Although re-routing has been considered as a possible solution for the reduction of SRKW's levels of noise exposure (DFO, 2017b), re-directing traffic from Haro Strait toward Rosario Strait would lead to variable results depending on the quota of displaced traffic and, more generally, would result in vessels increasing their transit time within the Salish Sea. In other words, re-routing vessel traffic would successfully reduce the amount of noise received by SRKW within its summer core areas but may result in an increase of the overall amount of acoustic energy released by vessels in the Salish Sea ecosystem. According to the results of the analysis, an eastward displacement of the shipping lane in Haro Strait accompanied by a west-ward displacement in Boundary Pass may result in a significant reduction of noise exposure for two of the SRKW's summer core areas identified in Chapter 2. The effective reduction of noise emissions would still be highly dependent on the frequencies being considered. Lateral displacement may successfully decrease SRKW's exposure to high-frequency noise (i.e. from 10 to 40 KHz), suspected to interfere with *Orcinus orca*'s foraging behaviour by masking their echolocation signals (i.e. 20–70 KHz) (Holt et al., 2009; Veirs et al., 2016). On the other hand, the low-frequency component of ship noise, which dissipates further from the source than the high-frequency component, will most likely still reach SRKW's summer core areas. The adoption of lateral displacement would also require a regular reassessment of SRKW's summer core areas in order to adjust the shipping lanes and reflect changes in the animal's geographic and temporal distribution.

Another relevant finding of this research, supported by a growing geovisualization literature (Zhang and Gruenwald, 2008; Al-Kassab et al., 2014; Fuhrmann et al., 2015; Schroth et al., 2015; Kinkeldey et al., 2017), is that the use of GIS for the visualization of complex phenomena, such as noise pollution, offers support for decision makers in the exploration, design, and evaluation of possible mitigation measures. The scenarios described in Chapter 3 provided an example of the possibilities represented by the combination of acoustic modeling with the capabilities of GIS software. As long as a user is equipped with the technical skills required for the use of GIS software, the visual exploration of acoustic and biological datasets could potentially support non-acoustic experts in the development of spatial knowledge, enabling managers and decision-makers to draw conclusions and take informed decisions without necessarily relying on expert knowledge.

4.2 Discussion

Achieving the results described in Chapters 2 and 3 required us to overcome several issues related to the use of opportunistic cetacean observations for the delineation of SRKW summer core areas, and to the integration of different spatial datasets (i.e. SRKW summer core areas, cumulative noise models). The use of SBEP data allowed us to produce a fine-scale estimation of SRKW's summer core areas. However, this required the computation, through the analysis of SBEP activity on the water, of an effort index representing an approximation of the effective effort invested by SBEP volunteers during data collection. Furthermore, the Strait of Juan de Fuca, the Strait of Georgia, and the Northern Gulf Islands are rarely surveyed by SBEP, a factor that limits the reliability of the KDEs over these areas. Similarly, biases in the spatial and temporal distribution of SRKW observations may have had an influence on the identification of the pod-specific summer core areas. Although KDEs represent a valuable non-parametric approach for the estimation of an animal's habitat use, the methodology does not allow for the explicit estimation of confidence intervals and the results are highly dependent on the selection of an appropriate bandwidth (i.e. H). The adoption of an ad-hoc bandwidth selection method allowed us to identify the optimal H values for each one of the four KDEs presented in Chapter 2. Iterating the analysis using a random sub-sample of sightings for each iteration allowed us to define lower and upper confidence intervals for SRKW's KDEs. The results of this process provided us with a deeper understanding of the KDE results. For the entire population as well as for the J and L pods, the KDEs computed using the optimized H values showed 95% PVCs falling between the 5th and the 95th percentiles of their relative frequency distributions (Fig. 2.11 A-C). On the other hand, the KDE describing the K-pod summer core area (Fig. 2.11 D) was characterized by a 95% PVC falling outside of its relative frequency distribution. More specifically, the K-pod 95% PVC showed an extent larger than the 95th percentile of the frequency distribution, indicating a possible overestimation of this pod's summer core area.

Cumulative noise modeling heavily relies on the estimation of SLs, the SPL scaled to a nominal distance of 1 m from a noise-emitting source. The use of estimated SLs introduced uncertainty in the modeled L_{eq} values and, consequently, in the median L_{eq} values reported in Chapter 2. The modelled received levels used by JASCO for the computation of L_{eq} values were compared to actual vessel noise measurements collected in the Salish Sea (O'Neill et al., 2017), reducing the uncertainty linked to the use of estimated SLs. Using AIS data for the estimation of vessel density introduces another source of uncertainty in the modelled L_{eq} values. Small vessels (e.g. fishing, recreational, whale-watching) are not required by law to carry AIS transmitters, hence the modelled cumulative level of sound for these classes is most likely an underestimation of the actual level. Another limiting factor is represented by the biological relevance of expressing noise in terms of time-averaged sound pressure levels. Animals are exposed to fluctuating sound pressure levels, which are often higher or lower than the estimated L_{eq} value at any particular instant. However, it is expected that the level of exposure in the long term would tend towards the estimated time-averaged sound pressure. In this specific case, L_{eq} represents the average sound energy that an animal would receive every day if it were to occupy the same location over the course of a month, given the fluctuations of sound levels over that time. Cetaceans, including SRKW, are highly mobile species, and would most likely move within the study area and be exposed to different levels of noise depending on the time spent at specific locations. This issue was partially overcome through the adoption of a probabilistic approach in the computation of the median exposure values (i.e. $L_{eq-50^{th}}$). An $L_{eq-50^{th}}$ = 90 dB re 1 μPa indicates that, within the area defined for the computation of the metric (e.g. Zone 1, 2 and 3 in Chapter 2) an animal (or group of animals), has, over a period of a month, a 0.5 daily probability of being exposed to a cumulative level of noise ≤ 90 dB re 1 μPa.

The tools presented in Chapter 3 are still a prototype and a formal evaluation study should be performed in order to understand whether or not the proposed framework meets the needs of managers and decision-makers. A first step toward a user-based evaluation of the tools was the creation of a GitHub public repository containing a sample dataset and the encrypted prototype toolbox.

Besides usability testing, there are still several limitations to the application of the presented framework which are related to the nature of the original datasets. Two main limitations are relative to the structure of species and noise datasets. Both are required to be spatially explicit and their spatial resolution, if not matching, should be as close as possible. Executions of the Exposure Hotspot Maps tool (Section 3, Chapter 2) performed using low resolution (5 km) cetacean distribution maps resulted in very coarse noise exposure maps, which prevented their use as input for the Route Generator tool. The probabilistic levels of exposure show different results for the same vessel class when computed over different areas or when computed on a different selection of classes. Consequently, identifying AOIs as well as defining which vessel classes will be considered, are two prerequisites for the generation of exposure hotspot maps and relative contribution rasters as well as for the computation of noise exposure levels. In addition, a CDF, when computed from a small sample of L_{eq} values (i.e. small AOI), may lead to unreliable exposure levels. The LCP analysis, selected as our approach for the generation of ship-traffic management scenarios, can lead to very different results depending on the selected cost-surfaces and their relative weights. Species and vessel costs should be defined *prior* to running the RG tool and multiple iteration of the analysis may be required to generate reliable management scenarios.

4.3 Conclusions

The results presented in Chapter 2 (Paragraph 2.4.3) suggest that, at least in some portion of SRKW's core areas, recreational vessels are associated with high levels of noise exposure ($L_{eq-50^{th}}$> 90 db re 1 μPa). However, not all recreational vessels are equipped with AIS, and their contribution, as well as the contribution of other small vessels to the total cumulative noise, might have been underestimated. Future studies should be focusing on the exploration of alternative methods for tracking vessels and on the estimation of the level of noise produced by vessels that are not carrying AIS. Another possible future development could be the conduction of a usability test to assess both user performances and ease-of use of the tools described in Chapter 3 (Larusdottir, 2011) and guide the further development of both the proposed framework and toolbox.

The results presented in Chapters 2 and 3 are in agreement with the results of a growing body of literature relative to the effects of anthropogenic noise on marine species, and corroborate the idea that independently form the species, sources, and geographic locations of interest, the management of oceanic noise pollution could benefit from the implementation of adaptive management strategies. Adaptive management is a systematic and iterative approach applied for the improvement of natural resources and wildlife conservation management policies (Dreiss, 2017). Through each iteration, new knowledge is produced, reducing uncertainty and allowing to review and adjust policies in response to changes in the managed ecosystem (Dreiss, 2017). At the same time, this approach should not be interpreted as a trial and error process. To be an effective tool, adaptive management requires the exploration of alternative strategies, including the modeling and simulations of possible management scenarios, and heavily relies on monitoring (Aldridge et al., 2004; Allen and Garmestani, 2015).

As recognized by the latest publications relative to the effectiveness of vessel management strategies in tackling the issue of noise pollution (Chion et al., 2017; Holt et al., 2017; McKenna et al., 2017), both the spatial and temporal variability of vessel traffic and species distribution need to be taken into account when designing noise mitigation measures. This may be particularly true for the Salish Sea, where a large number of vessel classes with different structural (e.g. size, number and type of propellers) and operational (e.g. speed and draught) characteristics are often navigating within the critical habitat of an endangered population. Since propeller cavitation and machinery have been identified as the main sources of noise radiating from commercial ships (IMO, 2014), an ideal management solution would be the reduction of noise at the origin. The International Maritime Organization (IMO) guidelines for the reduction of anthropogenic noise from shipping highlights how: "*the largest opportunities for reduction of underwater noise will be during the initial design of the ship*" (IMO, 2014). However, the guidelines also recognize that upgrading the existing commercial fleet to meet the required noise standards would be impractical. In other words, since the average lifecycle of a modern commercial ship is 25 years (Dinu and Ilie, 2015), it would be unrealistic to expect, in the short term, a reduction of shipping noise due to the application of "quiet" design solutions. For these reasons, in order to prevent or mitigate population-level impacts on endangered species, management actions focused on the adoption of operational rather than structural solutions are needed. An adaptive management approach, in this case, would require the application of different strategies to different classes of vessels depending on their spatial distribution, their operational characteristics, and on their contribution to the cumulative level of noise. For example, and as discussed in Chapter 2 (Paragraph 2.4.3), imposing a speed limit might be an appropriate approach to reduce the noise emitted by large commercial vessels but not be as effective for tugboats, which could be managed following other strategies such as re-routing or the introduction of quiet-times. Similarly, since ferries follow regular routes, noise emissions attributed to this class could be more effectively reduced by the introduction of a class-specific rather than an area-specific speed limit.

Changes in operational decisions, such as speed reduction and the modification of existing routes are considered to reduce negative impacts on marine life (IMO, 2014). However, navigating at speeds lower than the design speed of a vessel (i.e., slow steaming, a widespread strategy used to reduce fuel consumption) may result in increased noise from propeller cavitation. Studies conducted by the European Union (Audoly et al., 2017; Badwin et al., 2013) highlighted how the effectiveness of speed reduction as a noise mitigation measure is highly dependent on the technology carried by the vessels: vessels with controllable pitch propellers might be affected by poor efficiency and excessive cavitation as a consequence of slow-steaming practices. In a few words, the links between noise, vessels design and life-cycles, operational practices, and the outcomes of noise management solutions are still largely unknown and challenging to quantify. This is equally true for the links between the emission of noise in the environment and the long-term population consequences deriving from the exposure to noise. In this case, adopting an adaptive vessel-noise management approach would require the evaluation of a set of possible mitigation measures, such as the ones listed by DFO for SRKW which include speed limits, time restrictions, convoying, rerouting, and lateral displacement (DFO, 2017b). For example, Glacier Bay National Park implemented four different measures (i.e.

vessel quotas, speed limits, course restrictions, and temporary no-go areas.) to reduce vessel disturbance and risk of strike for cetaceans (McKenna et al, 2017).

Furthermore, the full extent of a species habitat is often unknown, and species distribution tends to be variable through time. Such changes cannot be addressed by static forms of management. For example, Fisheries and Oceans Canada recently identified two habitats of special importance located outside of SRKW and NRKW recognized critical habitats (DFO, 2017a). At the same time, the Vancouver Frazer Port Authority (VFPA) Enhancing Cetacean Habitat and Observation (ECHO) program conducted a two months (i.e. from August 6th until October 7, 2017) voluntary vessel slow-down trial in Haro Strait, with the objective, among others, of assessing the potential benefit of the slowdown to the behaviour and foraging of killer whales (VFPA, 2018). However, between August and October 2017, SRKW individuals were present within the slow-down area for only nine days and a total of 72 hours (i.e. a 70 % reduction in presence when compared to 2016), limiting the evaluation of the potential benefits derived from the slowdown to modeling (VFPA, 2018). Another example is the apparent shift in the distribution of NARW occurred between June and September 2017. Opportunistic data collected over the period 1986-2012 showed regular, although low, presence of NARW in the Gulf of St. Lawrence (Daoust et al., 2017). These observations were supported by dedicated surveys carried out between Anticosti Island and the Gaspé Peninsula (Daoust et al., 2017). However, an increase in dedicated survey effort over the period 2015-2017 showed a remarkable change in the number of NARW individuals observed in the waters of the Gulf of St. Lawrence. During the 2015 and 2016 monitoring seasons, at least 74 individuals (i.e. approximately 15% of the population) were photographed and identified, and more than 650 sightings of NARW were collected over the period 2015-2017. At the same time, opportunistic effort remained essentially unchanged and led to the collection of approximately 30 observations over the same period (i.e. 2015-2017), a number in accordance with the previous 30 years of opportunistic data collection. This discrepancy supports the idea that the Gulf of St. Lawrence has always been a relevant habitat for NARW, but the lack of monitoring effort invested in the area prevented its recognition as a critical habitat for the species. These two examples underline the importance of regular monitoring for the successful implementation of environmental management strategies and its fundamental role in reducing uncertainty. Adaptive management recognizes monitoring as the elective process to reduce uncertainty and iteratively improve management practices (Wintle, 2007). Since the implementation of the slow-down area in 2017, the increased monitoring effort in the Gulf of St. Lawrence allowed to detect and follow the movements of NARW and to apply both dynamic speed limits and temporary restrictions to the fisheries active in areas where the animals are sighted.

Besides the need for regular monitoring of cetacean species, two main questions relative to vessel noise management arise that could be the focus of future works:

i) Are area-based mitigation measures appropriate to achieve a reduction in noise exposure for endangered cetacean species?

ii) Are voluntary measures an effective management tool to reduce noise pollution from vessels?

Spatial measures, such as the ones implemented in the Haro Strait and in the Gulf of St Lawrence, resulted in a reduction in both the risk of lethal ship strikes and in the levels of noise pollution over areas that are key habitats for the Southern Resident Killer Whale and the North Atlantic Right Whale. However, this may encourage vessels to navigate at speeds that are higher than their usual speed when outside from these areas in order to recover the time lost on their journeys. As a result, whales could be exposed to increased risk of strike and increased levels of noise when outside of the managed areas. Vessel-based measures may prove to be a better solution than area-based measures for the reduction of noise pollution at the scale required to protect large migratory species. Further research is needed to develop vessel-based measures and to evaluate their effectiveness in reducing noise pollution. Similarly, we still do not possess enough information to fully evaluate the outcomes of other vessel management measures, such as lateral displacements and rerouting. Another key issue is the adoption of voluntary measures when compared to the enforcement of mandatory measures. Measures such as the slow-down implemented in the Haro Strait rely on voluntary compliance as a metric to evaluate their success. This could lead to a relaxation of the measures aimed at increasing compliance, but without a clear indication of the deriving benefits to the acoustic environment. With 421 out of 951 vessels transiting in the slow-down area at speeds below 12 kn, the trial achieved a 1.2 dB reduction in sound intensity within one of SRKW key foraging areas (VFPA ECHO Program, 2018).

However, the new iteration of the trial raised the speed limits to 15 kn for vehicle carriers, cruise and container vessels and to 12.5 kn for bulkers, tankers, ferries and government vessels. With these new speed limits, a compliance of 80% of vessels transiting would be required to achieve the same reduction in noise observed in 2017.

Given the complexity of underwater acoustic pollution, the spatial and temporal heterogeneity of its sources and receivers, the uncertainty relative to the effectiveness of vessel management measures, and the uncertainty relative to how noise-related impacts will affect marine species, noise pollution is best addressed through the implementation of adaptive management strategies.

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5 APPENDICES

5.1 APPENDIX A - CUMULATIVE NOISE MODELING OUTPUT

The following maps show the output of the cumulative noise model (O'Neil et al. 2017) for all the AIS-defined vessel classes relative to July 2015. Cumulative noise values for each class were binned in 10 dB intervals ranging from L_{eq} < 90 dB re 1 µPa to $L_{eq} >$ 120 dB re 1µ Pa to allow comparison between classes.

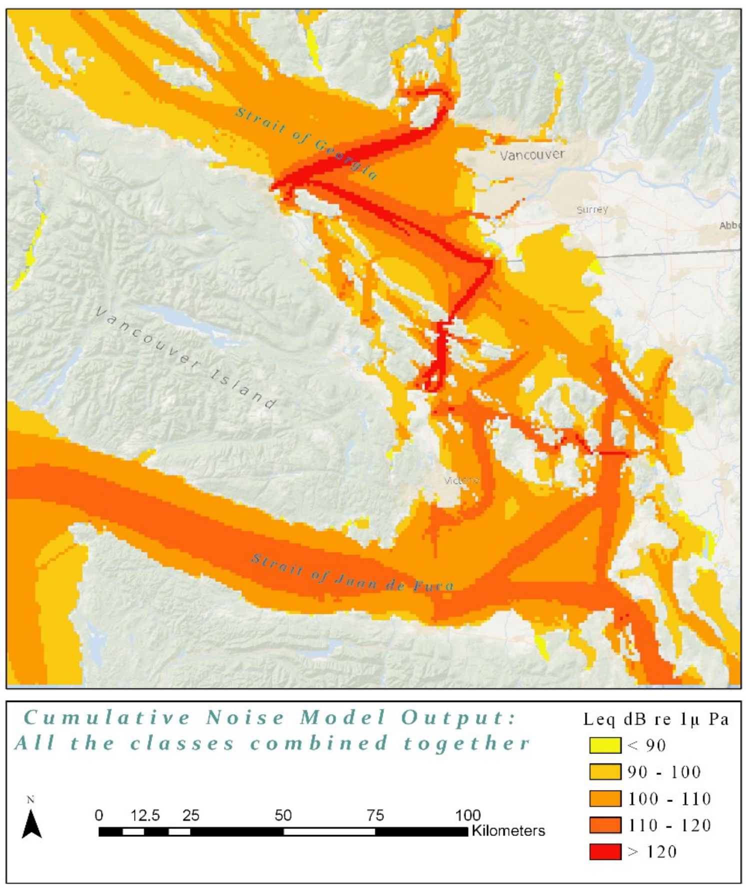

Figure A.1. Output of the cumulative noise modeling relative to all the vessel classes combined together.

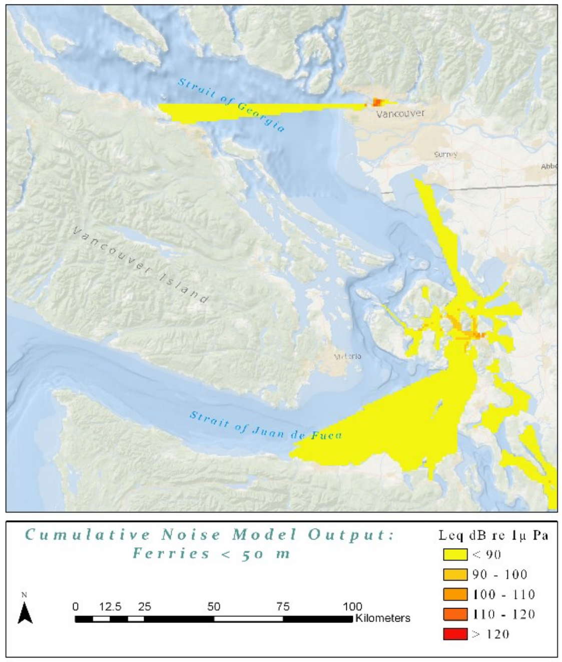

Figure A.2. Output of the cumulative noise modeling relative to the Ferries < 50 m class.

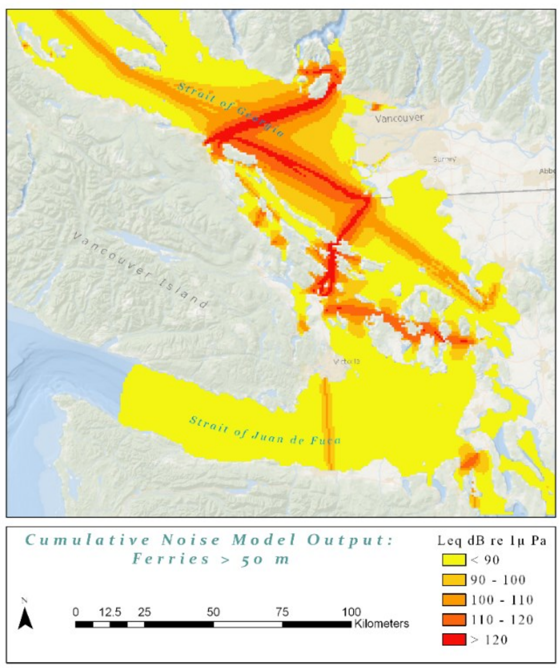

Figure A.3. Output of the cumulative noise modeling relative to the Ferries > 50 m class.

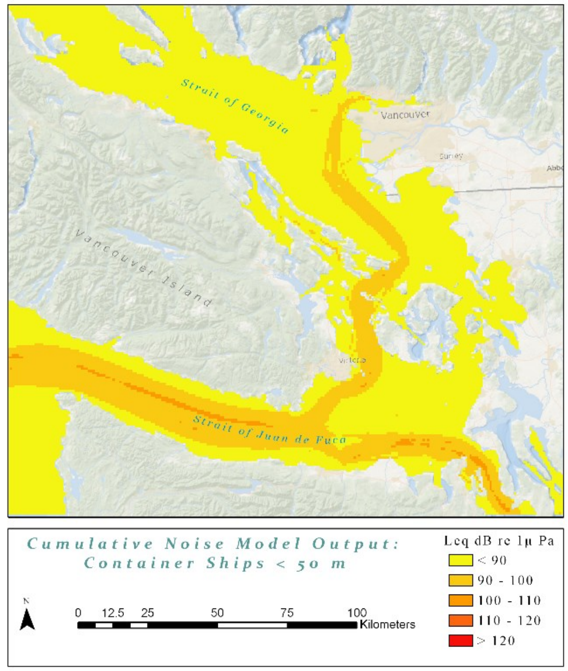

Figure A.4. Output of the cumulative noise modeling relative to the Container Ships < 50 m class.

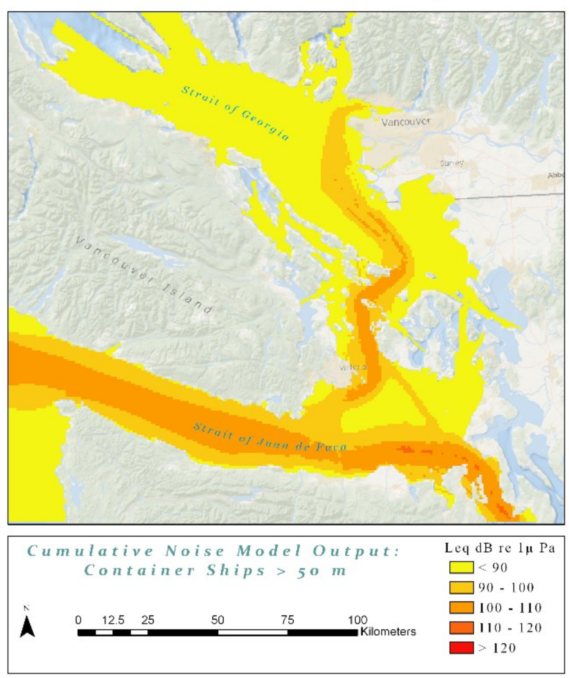

Figure A.5. Output of the cumulative noise modeling relative to the Container Ships > 50 m class.

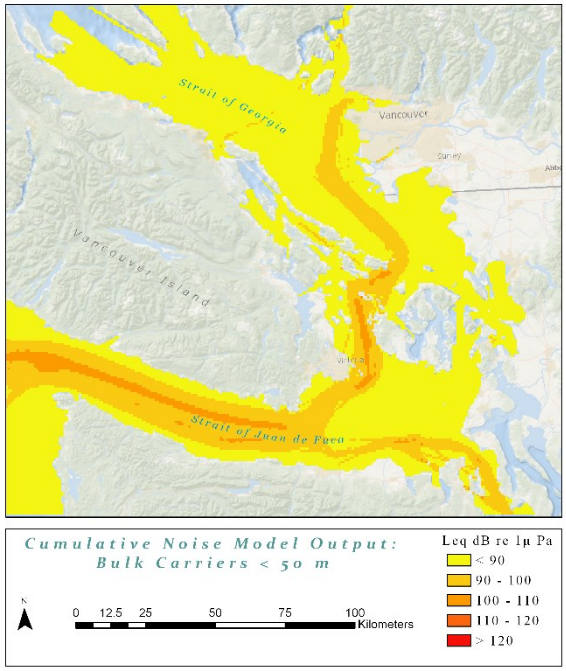

Figure A.6. Output of the cumulative noise modeling relative to the Bulk Carriers < 50 m class.

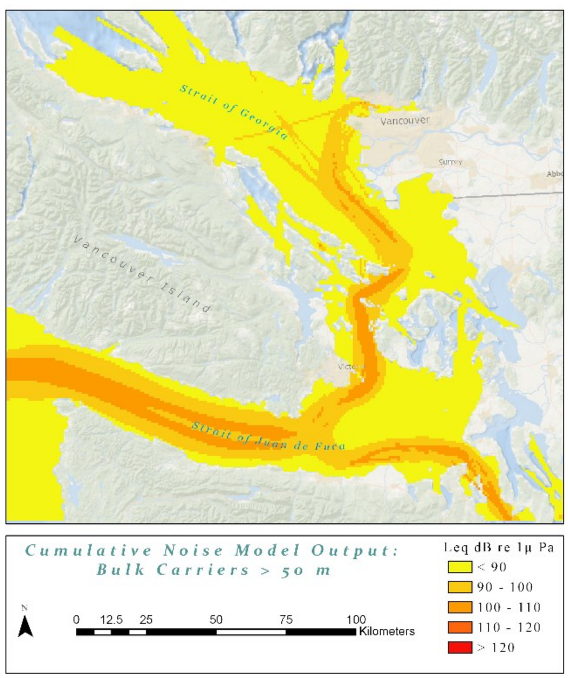

Figure A.7. Output of the cumulative noise modeling study relative to the Bulk Carriers > 50 m class.

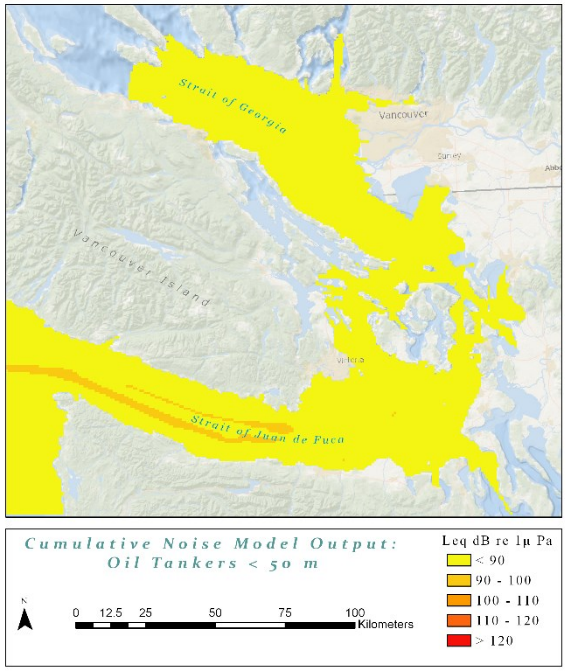

Figure A.8. Output of the cumulative noise modeling study relative to the Oil Tankers < 50 m class.

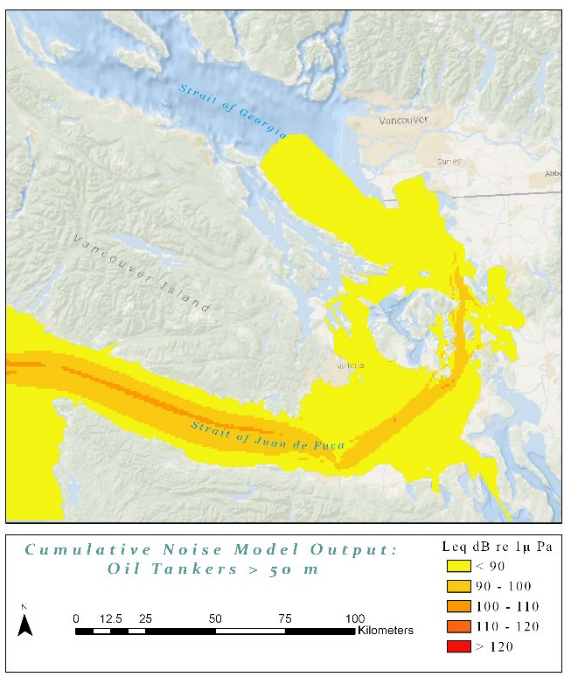

Figure A.9. Output of the cumulative noise modeling study relative to the Oil Tankers > 50 m class.

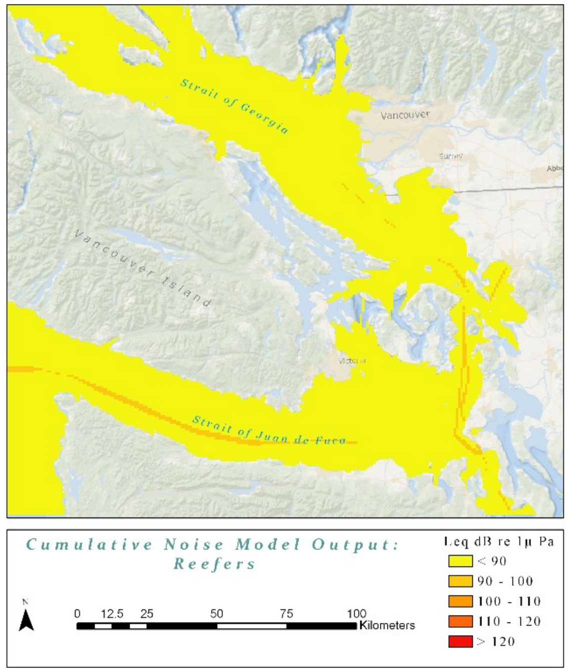

Figure A.10. Output of the cumulative noise modeling study relative to the Reefers class.

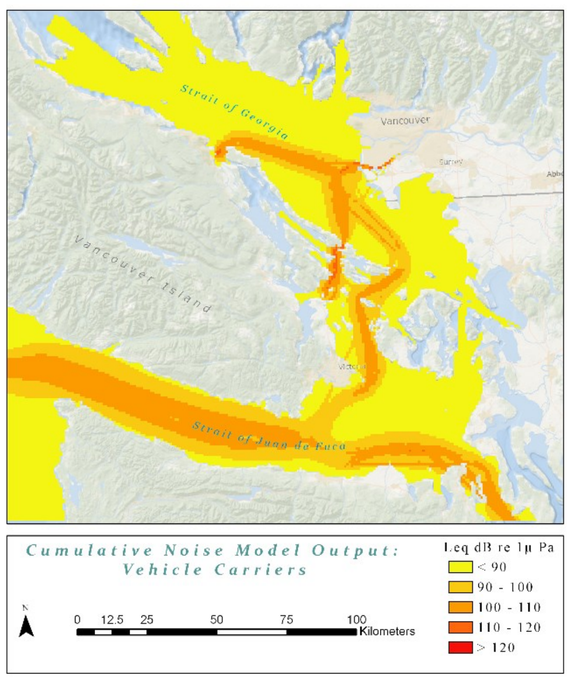

Figure A.11. Output of the cumulative noise modeling study relative to the Vehicle Carriers class.

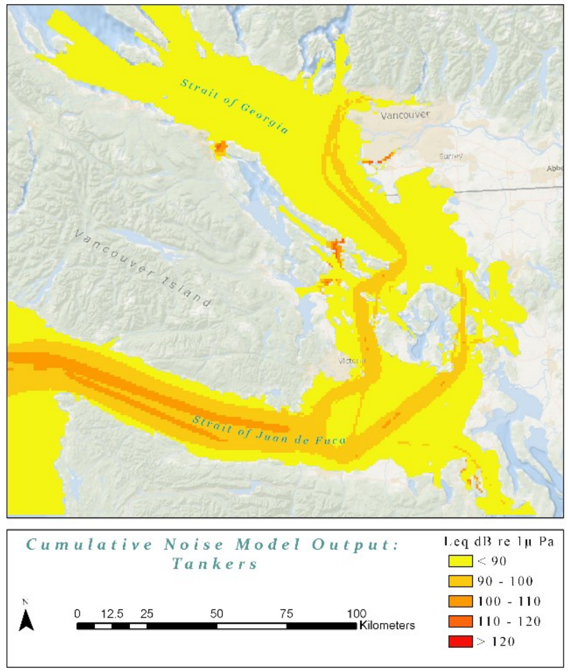

Figure A.12. Output of the cumulative noise modeling study relative to the Tankers class.

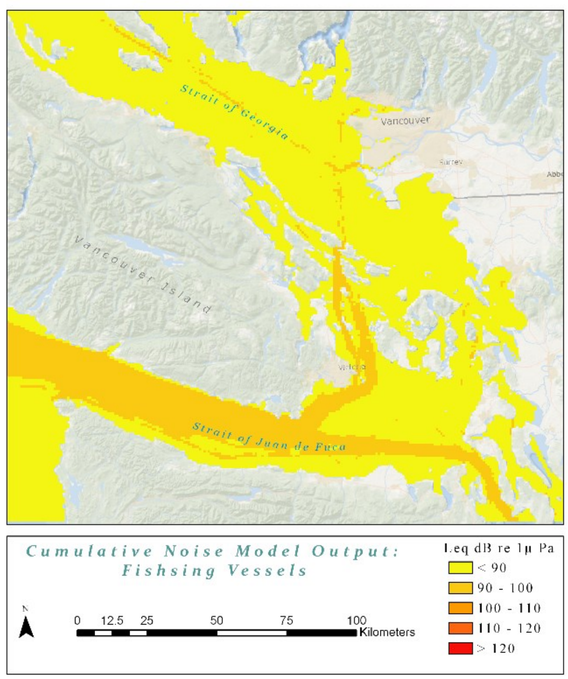

Figure A.13. Output of the cumulative noise modeling study relative to the Fishing Vessels class.

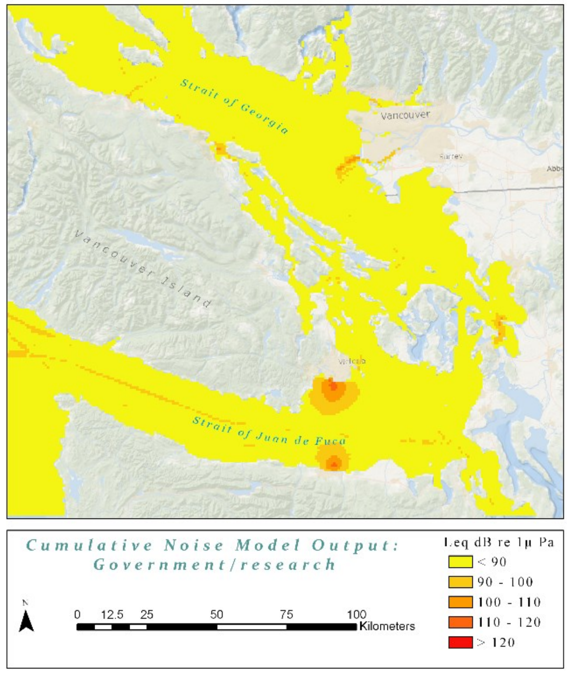

Figure A.14. Output of the cumulative noise modeling study relative to the Government/Research class.

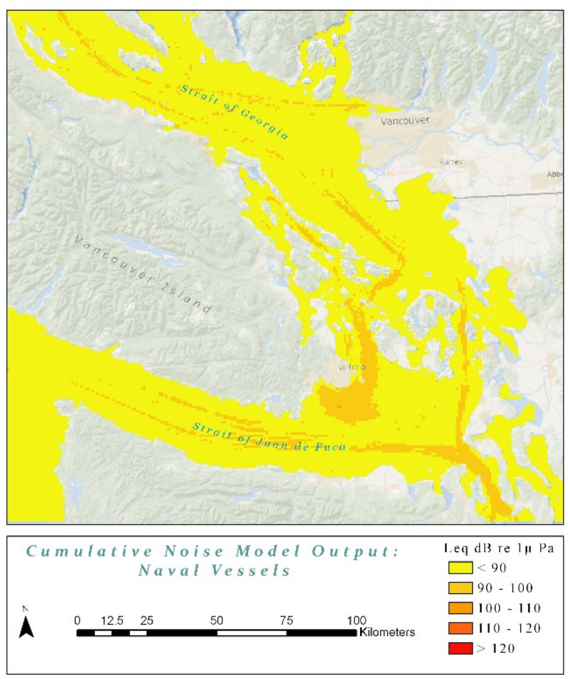

Figure A.15. Output of the cumulative noise modeling study relative to the Naval Vessels class.

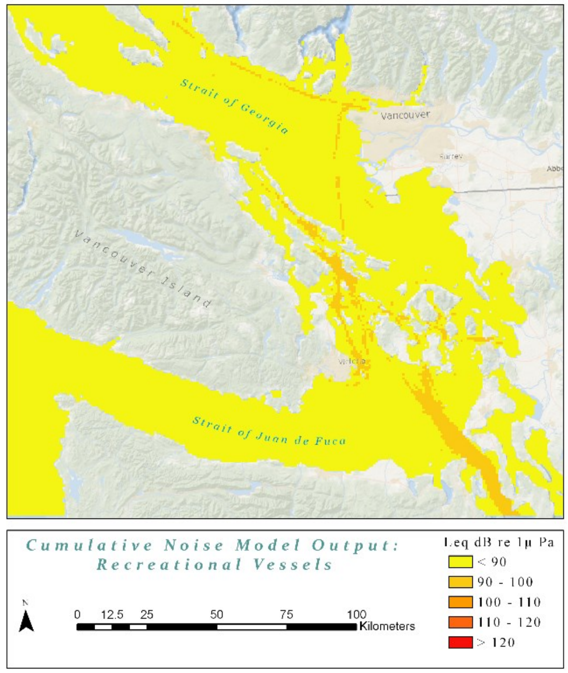

Figure A.16. Output of the cumulative noise modeling study relative to the Recreational Vessels class.

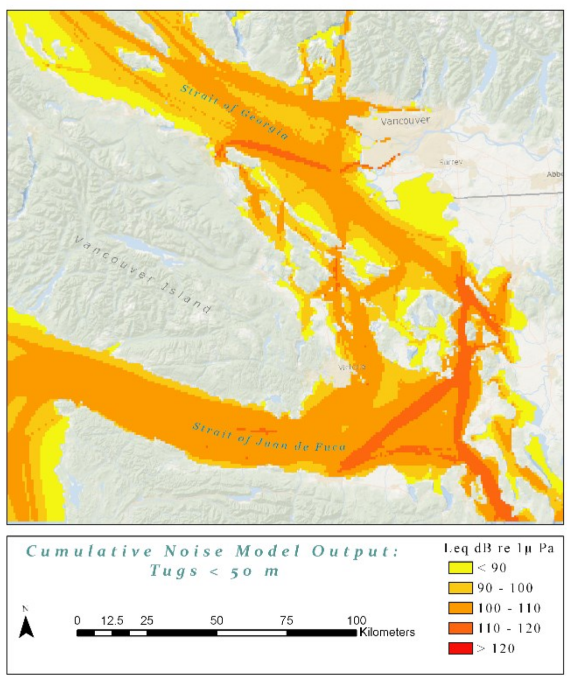

Figure A.17. Output of the cumulative noise modeling study relative to the Tugs < 50 m class.

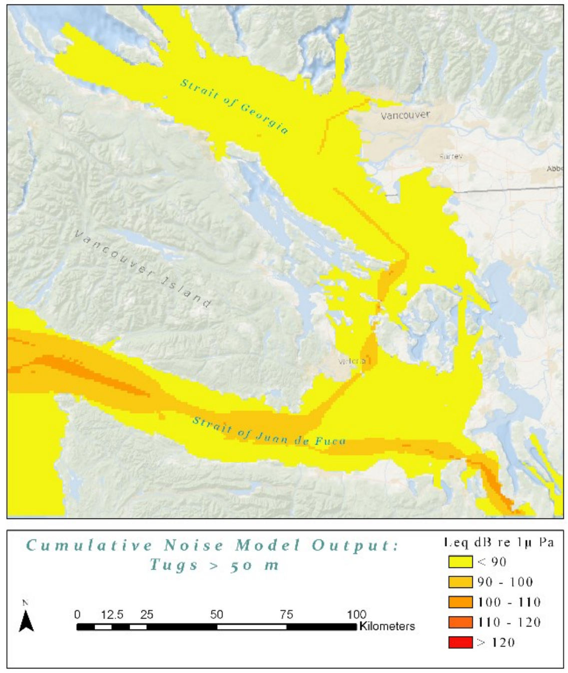

Figure A.18. Output of the cumulative noise modeling study relative to the Tugs > 50 m class.

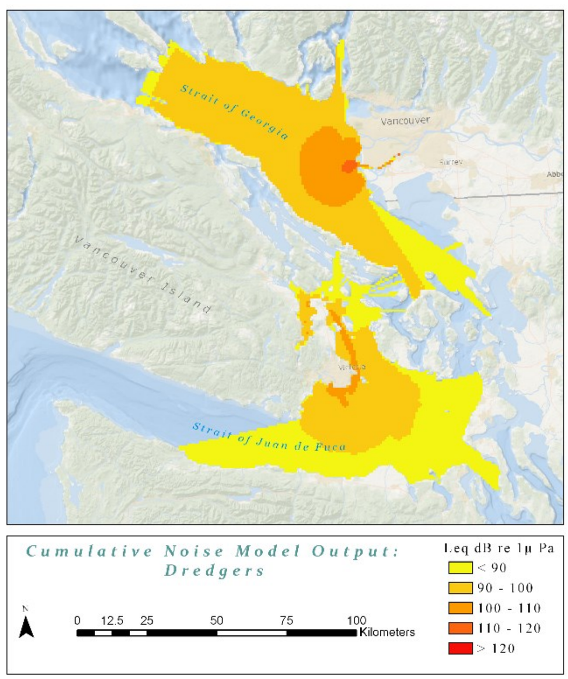

Figure A.19. Output of the cumulative noise modeling study relative to the Dredgers class.

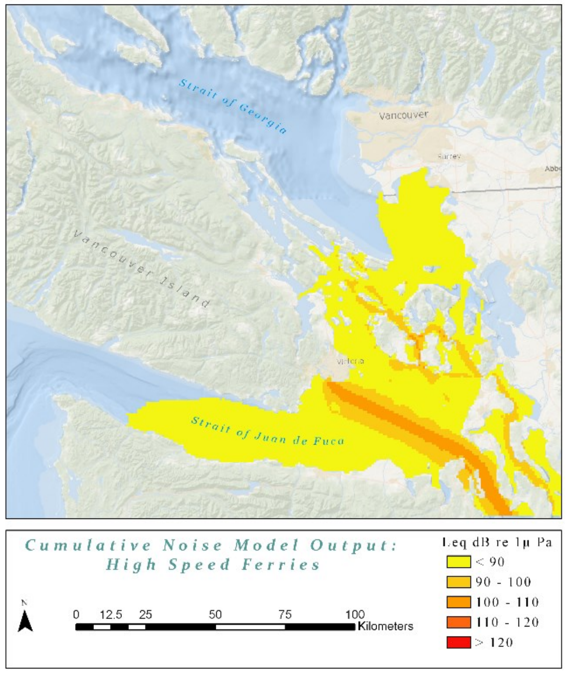

Figure A.20. Output of the cumulative noise modeling study relative to the High Speed Ferries class.

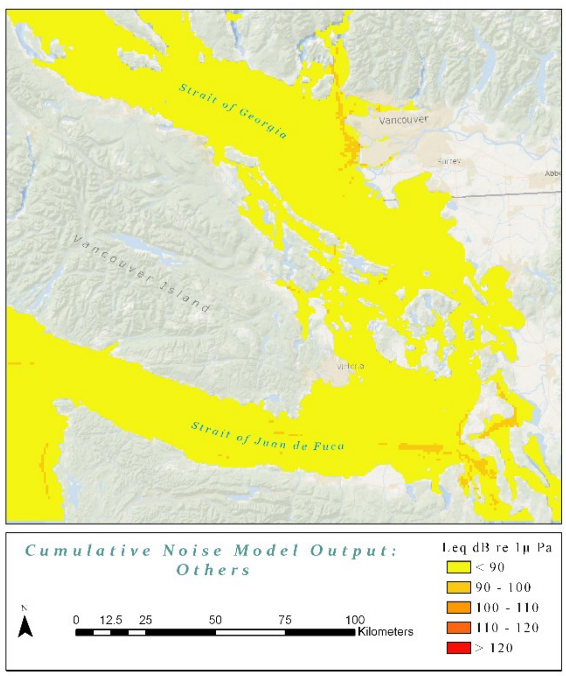

Figure A.21. Output of the cumulative noise modeling study relative to the Others class.

5.2 APPENDIX B – NOISE BUDGET

A Noise Budget represents an alternative approach to the methodology followed in Chapter 2 for understanding the contribution of each vessel class to the total amount of vessel-noise energy emitted within the Salish Sea. Although a probabilistic approach based on the modeled L_{eq} was deemed more suitable for the estimation of SRKW relative risk of noise exposure, a noise budget represents a valid approach for the classification of vessel classes based on their contribution to the total noise energy emitted into the marine environment. The average sound intensity is the preferred metric used for the computation of a noise budget. However, for long-term averages (e.g. a month), sound intensity can be assumed to be proportional to the mean squared sound pressure (Miller et al., 2008). Furthermore, for a large number of noise sources, distributed over a wide area, the mean squared sound pressure is an additive quantity, allowing to compute a noise budget for different portions of the Salih Sea (MacGillivray et al. 2016). The L_{eq}values were converted from the dB scale to the linear scale, allowing for the computation of total mean squared sound pressure values and for the computation of their relative contribution to the total mean squared sound pressure. The following tables report mean squared sound pressure values and percentage contribution to the total mean squared sound pressure for:

- The entire study area considered in Chapters 2 and 3;
- The 95% PVCs described in Chapter 2 for SRKW, J-group, K-group, and L-group

MacGillivray, A., M. Wood, Z. Li, A. Allen, and D. Hannay. 2016. Regional Ocean Noise Contributors Analysis: Enhancing Cetacean Habitat and Observation Program. Document 01195, Version 3.0. Technical report by JASCO Applied Sciences for Vancouver Fraser Port Authority.

Miller, J.H., Nystuen, J.A., Bradley, D.L., 2008. Ocean Noise Budgets. Bioacoustics 17, 133–136. doi:10.1080/09524622.2008.9753791

Table B.1. Noise Budget for the entire study area.

Average total mean squared sound pressure (kPa^2)	Entire Study Area	
	Regional Average	Contribution
	1.45E-01	100.0
TUGS	**2.91E-02**	**20.1**
tug< 50 m	2.81E-02	96.7
tug > 50 m	9.62E-04	3.3
FERRIES	**8.95E-02**	**61.9**
ferry < 50 m	5.91E-04	0.7
ferry > 50 m	8.73E-02	97.5
high-speed ferries	1.63E-03	1.8
PASSENGER	**4.90E-03**	**3.4**
passenger <100m	2.65E-06	0.1
passenger >100m	4.89E-03	99.9
CONTAINERS	**1.70E-02**	**11.7**
container ship < 200 m	1.10E-03	6.5
container ship > 200 m	6.27E-03	37.0
reefers	9.17E-05	0.5
bulk carriers < 200 m	1.48E-03	8.7
bulk carriers > 200 m	2.92E-03	17.2
vehicle carriers	5.09E-03	30.0
TANKERS	**2.81E-03**	**1.9**
oil tankers < 200 m	1.13E-04	4.0
oil tankers > 200 m	1.08E-03	38.5
tankers	1.61E-03	57.5
FISHING	**6.35E-04**	**0.4**
GOVERNMENT/RESEARCH/NAVY	**9.11E-04**	**0.6**
government/research	5.30E-04	58.1
navy	3.82E-04	41.9
RECREATIONAL	**2.22E-04**	0.2
DERDGERS	**1.13E-04**	0.1
OTHERS	**1.67E-04**	0.1

Table B.2. Noise Budget computed within the 95% PVC relative to the entire SRKW population.

Average total mean squared sound pressure (kPa²)	SRKW	
	95% PVC Average	**Contribution**
	1.76E-01	100.0
TUGS	**4.41E-02**	**25.1**
tug< 50 m	4.36E-02	98.8
tug > 50 m	5.11E-04	1.2
FERRIES	**1.07E-01**	**61.1**
ferry < 50 m	1.16E-04	0.1
ferry > 50 m	1.06E-01	98.7
high-speed ferries	1.33E-03	1.2
PASSENGER	**1.20E-03**	**0.7**
passenger <100m	4.05E-06	0.3
passenger >100m	1.20E-03	99.7
CONTAINERS	**1.86E-02**	**10.6**
container ship < 200 m	1.18E-03	6.3
container ship > 200 m	4.87E-03	26.2
reefers	7.89E-05	0.4
bulk carriers < 200 m	2.08E-03	11.2
bulk carriers > 200 m	4.55E-03	24.5
vehicle carriers	5.82E-03	31.3
TANKERS	**1.86E-03**	**1.1**
oil tankers < 200 m	3.23E-05	1.7
oil tankers > 200 m	4.69E-04	25.2
tankers	1.36E-03	73.1
FISHING	**5.74E-04**	**0.3**
GOVERNMENT/RESEARCH/NAVY	**1.47E-03**	**0.8**
government/research	8.07E-04	54.7
navy	6.68E-04	45.3
RECREATIONAL	**4.33E-04**	0.2
DERDGERS	**6.09E-05**	0.0
OTHERS	**1.07E-04**	0.1

Table B. 3. Noise Budget computed within the 95% PVC relative to the J-group.

Average total mean squared sound pressure (kPa^2)	**J-group**	
	95% PVC Average	**Contribution**
	1.80E-01	100.0
TUGS	**4.36E-02**	**24.2**
tug< 50 m	4.31E-02	98.9
tug > 50 m	4.77E-04	1.1
FERRIES	**1.09E-01**	**60.6**
ferry < 50 m	1.39E-04	0.1
ferry > 50 m	1.08E-01	98.9
high-speed ferries	1.03E-03	0.9
PASSENGER	**1.70E-03**	**0.9**
passenger <100m	3.75E-06	0.2
passenger >100m	1.70E-03	99.8
CONTAINERS	**2.11E-02**	**11.7**
container ship < 200 m	1.32E-03	6.3
container ship > 200 m	5.59E-03	26.5
reefers	9.36E-05	0.4
bulk carriers < 200 m	2.34E-03	11.1
bulk carriers > 200 m	5.15E-03	24.4
vehicle carriers	6.59E-03	31.3
TANKERS	**1.75E-03**	**1.0**
oil tankers < 200 m	2.28E-05	1.3
oil tankers > 200 m	3.68E-04	21.1
tankers	1.36E-03	77.6
FISHING	5.89E-04	**0.3**
GOVERNMENT/RESEARCH/NAVY	**1.62E-03**	**0.9**
government/research	9.56E-04	59.0
navy	6.63E-04	41.0
RECREATIONAL	4.33E-04	0.2
DERDGERS	5.17E-05	0.0
OTHERS	1.05E-04	0.1

Table B. 4. Noise Budget computed within the 95% PVC relative to the K-group.

Average total mean squared sound pressure (kPa^2)	K-group	
	95% PVC Average	**Contribution**
	2.71E-01	100.0
TUGS	**3.63E-02**	**13.4**
tug< 50 m	3.49E-02	96.2
tug > 50 m	1.38E-03	3.8
FERRIES	**1.93E-01**	**71.3**
ferry < 50 m	3.00E-04	0.2
ferry > 50 m	1.91E-01	98.9
high-speed ferries	1.87E-03	1.0
PASSENGER	**5.89E-03**	**2.2**
passenger <100m	3.40E-05	0.6
passenger >100m	5.86E-03	99.4
CONTAINERS	**3.06E-02**	**11.3**
container ship < 200 m	1.96E-03	6.4
container ship > 200 m	8.48E-03	27.7
reefers	1.16E-04	0.4
bulk carriers < 200 m	3.86E-03	12.6
bulk carriers > 200 m	8.14E-03	26.6
vehicle carriers	8.04E-03	26.3
TANKERS	**2.81E-03**	**1.0**
oil tankers < 200 m	7.12E-05	2.5
oil tankers > 200 m	7.61E-04	27.1
tankers	1.98E-03	70.4
FISHING	5.72E-04	**0.2**
GOVERNMENT/RESEARCH/NAVY	**9.82E-04**	**0.4**
government/research	2.55E-04	26.0
navy	7.27E-04	74.0
RECREATIONAL	4.63E-04	0.2
DERDGERS	1.02E-05	0.0
OTHERS	1.27E-04	0.0

Table B.5. Noise Budget computed within the 95% PVC relative to the L-group.

Average total mean squared sound pressure (kPa^2)	L-group	
	95% PVC Average	**Contribution**
	4.16E-01	100.0
TUGS	**4.87E-02**	**11.7**
tug < 50 m	4.73E-02	97.1
tug > 50 m	1.41E-03	2.9
FERRIES	**3.23E-01**	**77.6**
ferry < 50 m	0.00	0.0
ferry > 50 m	3.18E-01	98.5
high-speed ferries	4.67E-03	1.4
PASSENGER	**1.28E-03**	**0.3**
passenger <100m	1.16E-05	0.9
passenger >100m	1.27E-03	99.1
CONTAINERS	**3.56E-02**	**8.6**
container ship < 200 m	2.10E-03	5.9
container ship > 200 m	8.68E-03	24.4
reefers	1.61E-04	0.5
bulk carriers < 200 m	4.06E-03	11.4
bulk carriers > 200 m	8.90E-03	25.0
vehicle carriers	1.17E-02	32.9
TANKERS	**3.07E-03**	**0.7**
oil tankers < 200 m	0.00	2.9
oil tankers > 200 m	9.87E-04	32.1
tankers	0.00	64.9
FISHING	9.44E-04	**0.2**
GOVERNMENT/RESEARCH/NAVY	**2.35E-03**	**0.6**
government/research	1.54E-03	65.5
navy	8.12E-04	34.5
RECREATIONAL	5.11E-04	0.1
DERDGERS	4.48E-04	0.1
OTHERS	1.72E-04	0.0

5.3 APPENDIX C – SCENARIOS C-D-E

In order to understand the influence of different weights for vessels and species costs, in addition to Scenario A and B presented in Chapter 3 another 3 scenarios were explored. In Scenario C, we assigned equal weights (i.e. 0.5) to both costs. In Scenario D, we assigned weight 0.8 to the vessel cost and 0.2 to the species cost. In Scenario E, we assigned weight 1 to the vessel cost and 0 to the species cost. All these three scenarios resulted in the same least-cost path (Figure C.1), which had a total length of 111 km and corresponded to the shortest possible path in terms of travelled distance. These paths departed from the original route in the overlap with the 50% PVC located along the coast of San Juan Island, but did not avoid the 50% PVC located on the east coast of Saturna Island (Figure C.1).

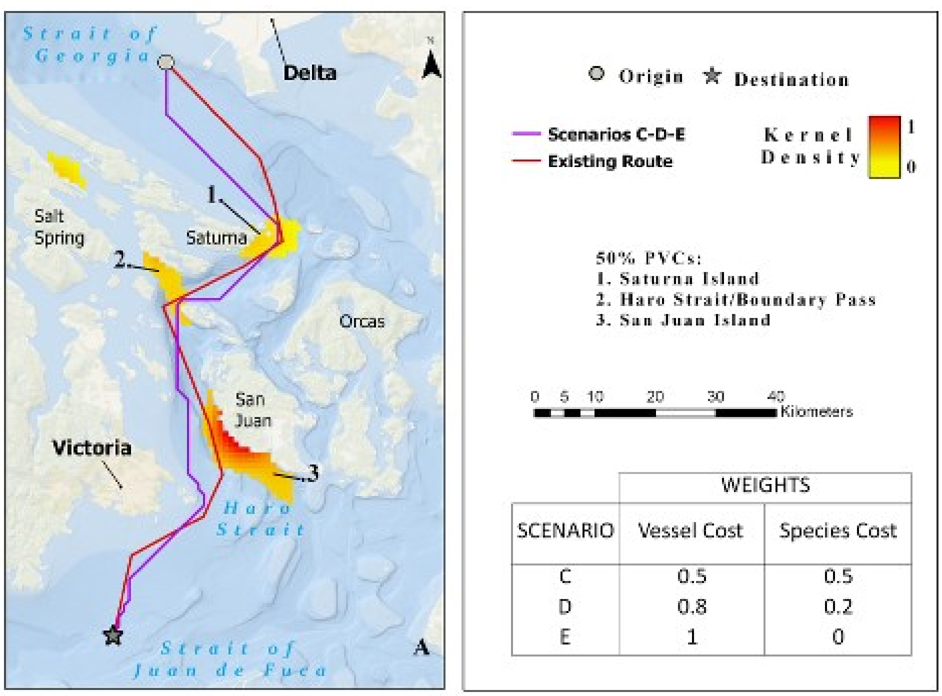

Figure C.1. Least-cost path resulting from scenarios C, D, and E.