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Cumulative Effects Assessment for West Coast Transient Population of Bigg's Killer Whale in the Northeast Pacific

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Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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ABSTRACT

The West Coast Transient (WCT) population of Bigg's Killer Whales (BKW) was listed as Threatened under the *Species at Risk Act* in 2003 and is vulnerable to the cumulative effects of human threats. The current understanding of threats (reduced prey availability, acoustic and physical disturbance, and contaminants), threat interactions, and potential impacts to fecundity and mortality were summarized in a Pathways of Effects (PoE) model that includes 16 pathway linkages from threats and threat interactions to effects on WCT fecundity and mortality. Sufficient knowledge and data were available to quantify eight of the pathway linkages in the PoE model and include them in a population viability analysis (PVA) model.

Using the most recent available data, the PVA model quantified impacts of threats on the population parameters of the Canadian Salish Sea subset of the WCT. The impacts of individual and cumulative threat scenarios on the modelled population were compared to the observed population demographics to define a model that best captured real-world dynamics. The cumulative model incorporating all threats simulated a population abundance trend that included the observed abundance within the 90% distribution of model simulations. The final model included impacts of prey availability on the population carrying capacity, vessel noise masking of prey sounds, vessel strikes, and PCB contamination. Sensitivity analysis demonstrated the importance of prey availability for the model results, as a result of the direct effect on carrying capacity and interactions with noise and contamination. The cumulative effects PVA model projected continued increase in the modelled population to the carrying capacity, with the rate depending on the trend in the prey populations, indicating that the current threat levels are unlikely to limit population growth.

There are knowledge gaps in our understanding of the WCT population of Bigg's killer whales compared to the better studied fish-eating resident killer whale populations. Building a quantitative population model for the WCT population required higher effort in scoping, population data analysis, and threat quantification than for resident killer whales (RKW). In particular, the impact of acoustic disturbance remains a significant knowledge gap for WCT. The PoE model and the quantitative sub-model explicitly identified knowledge and data gaps and needs for future research that can inform the next iteration and reduce uncertainty. Uncertainties and assumptions inherent in the conceptual PoE model and quantitative PVA model have been described in the document. The cumulative effects model for WCT advances the field and provides a valuable tool that can be used to examine scenarios of mitigation and management for the continued recovery of the population.

1 INTRODUCTION

1.1 BACKGROUND

Coastal waters of the Northeastern Pacific are home to three distinct ecotypes of killer whale: Bigg's (also referred to as transient), resident and offshore. These populations do not interbreed and have clear differences in morphology, social structure, diet, acoustic behaviour, and genetics (Deecke et al. 2005; Ford and Ellis 1999; Ford et al. 2013; Ford et al. 1998; Morin et al. 2024). Several distinct populations of Bigg's killer whales have been documented in the Northeastern Pacific, including the West Coast Transients, AT1 Transients, and Gulf of Alaska Transients (Ford et al. 2013; Matkin et al. 2012; Parsons et al. 2013). The West Coast Transients (WCT) have been photo-identified since the 1970s (Bigg 1982) and are the focus of this work as they are the only population of the Bigg's killer whale (BKW) ecotype to regularly occur in waters off the Pacific coast of Canada (Ford et al. 2013). A few individuals from the Gulf of Alaska population have been sighted in northern British Columbia (Ford et al. 2013; Matkin et al. 2012; Parsons et al. 2013), but too infrequently to be included in this assessment. In this document, we refer to BKW when discussing the ecotype, WCT when referring to the focal population, and the Canadian Salish Sea subset when discussing WCT observed in the Salish Sea. The Canadian Salish Sea subset is the focus of this assessment.

1.1.1 Range and population structure

WCT killer whales have a large range along the Pacific coast of North America, extending from at least southern California to southeastern Alaska (Figure 1). Within this region, they are primarily found inshore of the continental shelf edge (Ford et al. 2013; Towers et al. 2019). WCT can be found throughout coastal waters of western Canada year-round and have been documented with increasing regularity in the Salish Sea from the 1970s to present (Baird and Dill 1995; Houghton et al. 2015).

Two putative sub-populations of WCT killer whales have been based on habitat preferences (Ford et al. 2013). They are referred to as "inner coast" and "outer coast" sub-populations and individuals within them were demarcated based solely on number of encounters over time, with inner-coast individuals being documented relatively frequently compared to outer coast individuals (Ford et al. 2013; Smith 2017). Both inner and outer coast individuals regularly travel together where their preferred ranges overlap so a revised approach to defining the "coastal subset" of the WCT population in Canada was developed that relied on three criteria: number of encounters with adult whales over time, number of years each was identified, and number of years since they were last encountered (Towers et al. 2019). The approaches in Ford et al. (2013) and Towers et al. (2019) result in different arbitrary definitions of putative sub-populations or population subsets. The multiple criteria of the latest approach helps to eliminate animals with unknown statuses from the data as well as individuals that may belong to populations of Bigg's killer whales other than WCT, considering that other genetically distinct populations have occasionally been documented in Pacific Canada (Towers et al. 2019).

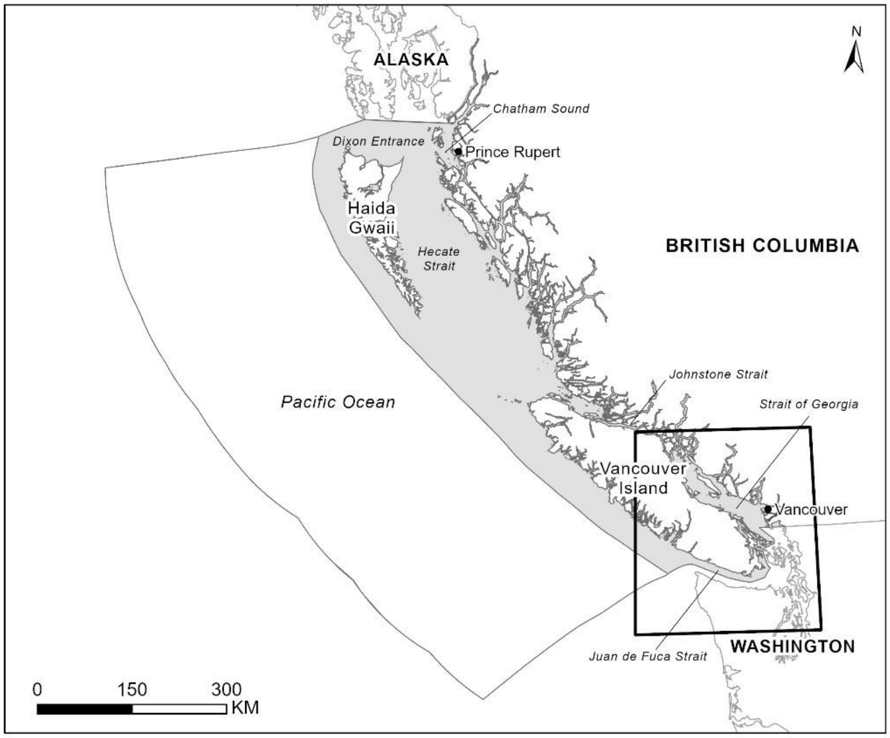


Figure 1. The range of the West Coast Transient (WCT) population of Bigg's Killer Whale in Canadian Pacific waters (DFO 2021b). Black rectangle delineates the focal area for the vessel noise model.

1.1.2 Abundance and Growth

Effort to photo-identify WCT killer whales has increased spatially and in regularity each decade from the 1970s to present. The first attempt to assess abundance of this population used photo-identification data from 1974 to 2006 and resulted in an estimate of 243 individuals (Ford et al. 2007). By 2012, over 500 individuals of the Bigg's ecotype had been documented in Pacific Canada, with 304 assigned to the putative inner coast sub-population and 217 assigned to the outer-coast sub-population (Ford et al. 2013). In 2018, 766 unique individuals had been documented, At that time, the coastal subset of this population was re-assessed by Towers et al. (2019) and found to include 349 individuals, most adults of which were included in the assessment by Ford et al. (2007) and assigned to the putative inner-coast sub-population by Ford et al. (2013).

Ford et al. (2007) reported that the WCT population grew at an average rate of 6% per year from 1974 to 2006. Between then and 2011, the inner coast sub-population grew at an average rate of 2.7% per year (Ford et al. 2013) and between 2012 and 2018 the coastal subset of the population had an average annual growth rate of 4.1% (Towers et al. 2019).

Increasing population abundance has resulted in higher density of WCT killer whales within their range. This is most apparent in their core habitat around Vancouver Island, specifically the Salish Sea, where the observed mean group size has increased from roughly four to six individuals between the 1970s and the 2010s (Baird and Dill 1995; Ford et al. 2013; Houghton et al. 2015; Shields et al. 2018). The frequency with which WCT have been documented in this region has also increased significantly over time from only a few encounters documented each year through the 1970s and 1980s (Bigg 1982; Ford and Ellis 1999) to hundreds of encounters each year throughout the 2010s and 2020s (Ford et al. 2013; Houghton et al. 2015; Shields et al. 2018). Increased WCT presence in the Salish Sea is also due to movement of whales into the area, including several matrilines that were not previously known to use the Salish Sea (Towers et al. 2019), likely in response to increased prey abundance (Shields et al. 2018). The dataset of WCT encounters, including dates, locations, individuals, their reproductive status, documented births and deaths, and familial relationships, is maintained by J. Towers, and is used throughout the paper, cited as “DFO-CRP and Bay Cetology unpublished data”.

1.1.3 Prey and Social Structure

The Bigg’s ecotype of killer whales specialize in hunting marine mammals. The primary prey species of the WCT population in Pacific Canada are harbour seal (Phoca vitulina), followed by harbour porpoise (Phocoena phocoena) and Steller sea lion (Eumetopias jubatus) (Ford et al. 2013). In Pacific Canada, they also hunt and consume Dall’s porpoise (Phocoenoides dalli), Pacific white-sided dolphin (Lagenorhynchus obliquidens), California sea lion (Zalophus californianus), Northern elephant seal (Mirounga angustirostris), common minke whale (Balaenoptera acutorostrata), and grey whale (Eschrichtius robustus) (Baird and Dill 1995; Ford and Ellis 1999; Ford et al. 2013; Ford et al. 1998). Increasing abundance of primary prey species, especially pinnipeds, are probably responsible for the observed growth of the coastal subset of the WCT population (Shields et al. 2018), especially around Vancouver Island where harbour seal, Steller sea lion and California sea lion have exhibited major increases of abundance in recent decades (DFO 2021b, 2022, 2023). WCT killer whales have also been documented chasing and sometimes killing sea and river otters as well as several species of bird and fish, but consumption of these animals has not been confirmed (Ford and Ellis 1999; Ford et al. 1998). In total, various species are preyed on by all WCT and although there are regional and temporal differences in focal prey species, to date there are few known individual or family level prey specializations within the coastal subset of this population.

WCT killer whales, like other populations of this species, live in a matrifocal society where the basic social unit consists of an adult female and her offspring (Ford and Ellis 1999). These groups remain cohesive over long periods of time, but daughters often permanently disperse from the natal group when they mature and begin to have their own offspring (Nielsen et al. 2023; Towers et al. 2019). Adult males and juveniles of both sexes occasionally will also temporarily or permanently disperse from their natal groups (Baird and Whitehead 2000; Ford and Ellis 1999; Nielsen et al. 2023; Towers et al. 2019). Dispersal serves to achieve low optimal group size for WCT killer whales, which is important when social activities such as vocal and surface-active behaviours can result in avoidance by prey and reduce hunting success (Deecke et al. 2005; Ford and Ellis 1999). Nevertheless, vocal communications between individuals does result in regular but temporary associations with other related and non-related social units within the population. These social groupings may have temporal and spatial trends based on prey abundance and affinity between sex and age classes.

1.1.4 Life history

Analysis of the WCT encounters database (DFO-CRP and Bay Cetology unpublished data) provided life history values (Table 1) for female first reproductive age (mean 15.7 years, minimum 10 years) and calving interval (mean 4.7 years) (Towers et al. 2019). Analysis of this dataset by Nielsen et al. (2023) estimated the mean life expectancy for males as 29 years (±0.7 years standard deviation (SD)) and 43 years (±11 SD) for females. The maximum age was estimated from the 90% life span as 69 year for females and 50 years for males (Nielsen et al. 2021). The proportion of males in the population was 0.36 (Nielsen et al. 2021). Using morphometrics from drone photogrammetry, male WCT were estimated to reach physical maturity at 18.4 years while females reached physical maturity at 14.2 years (Kotik et al. 2023), which was consistent with estimates for Southern Resident Killer Whales (SRKW; females = 15 years and males = 18 years) (Fearnbach et al. 2011). Other life history metrics for WCT killer whales are likely similar to those for the sympatric resident populations (Smith 2017).

Table 1. Life history traits and values for WCT killer whales and the source of the information.

Life history trait	**Value**	**Source**
Age of sexual maturity	Male – 18.4 (SE = 2.3)	Morphometrics of WCT from drone photogrammetry, data range 2014-2019 (Kotik et al. 2023)
	Female - 14.2 (SE = 2.8)	Morphometrics of WCT from drone photogrammetry, data range 2014-2019 (Kotik et al. 2023)
Age of first reproduction	Male - 15 years (range 10-17.4)	From SRKW population dynamics: Male sexual maturity, defined as when the dorsal fin shape changes sufficiently to distinguish males from females (Olesiuk et al. 1990).
	Female - Mean = 15.74 years, min = 10 years.	Analysis of WCT encounters dataset (DFO-CRP and Bay Cetology unpublished data; Towers et al. 2019)
Maximum age of reproduction	Male - 50	Assigned value same as maximum credible interval of 90% life span (50 ±1.3 years) (Nielsen et al. 2021)
	Female = 69 years	Maximum credible interval or 90% life span (69 ±2.9 years) (Nielsen et al. 2021)
Maximum age	Male - 70	Nielsen et al. (2021) estimated a credible interval for BKW 90% life span as males = median 44 ±0.7 SD years (confidence interval (CI) 37-52 years)
	Female - 90	Nielsen et al. (2021) estimated a credible interval for BKW 90% life span as females = median 59 ±1.5 SD years (CI 48-73 years).
Calving interval	Mean 4.71 years	Analysis of WCT dataset (DFO-CRP and Bay Cetology unpublished data; Towers et al. 2019)
Gestation period	16-17 months	From RKW population dynamics (DFO 2021b)
Adult sex ratio	0.36 (male:total individuals)	Fraction of all adult BKW individuals that are male (Nielsen et al. 2021)

1.1.5 Natural sources of mortality

Sources of natural mortality in WCT include infanticide, accidental stranding, entrapment in small water bodies, disease, parasitism, biotoxins, and starvation (DFO 2007, 2021b).

1.1.5.1 Sexual Conflict

Infanticide is among the most extreme examples of sexual conflict and was documented once in WCT killer whales in 2016 (Towers et al. 2018). The neonate killed was very young, defended by its mother, dispatched quickly by an unrelated adult male and the male's post-reproductive mother. The neonate was not consumed suggesting that, like in most other infanticidal mammals, this behaviour is sexually selected to increase inclusive fitness benefits of the perpetrators (McEntee et al. 2023; Towers et al. 2018). It is unclear how common infanticidal behaviour is in WCT killer whales but, considering how quickly it occurred after the whale was born, may be something that occurs more frequently than documented.

Sexual conflict is also likely to be an underlying potential cause of death in both adult male and female WCT killer whales found dead with blunt force trauma and/or scarring suggestive of deliberate aggression from conspecifics (Lee et al. 2023; Raverty et al. 2020). These pathological findings are consistent with potential causes of death of individuals, including neonates, in some other populations of killer whale (Lee et al. 2023). Signs of calf-directed aggression are prevalent in WCT and other killer whale populations with younger individuals, particularly males, having higher scarring rates from the teeth of conspecifics on their bodies than any other age or sex class (van Weelden et al. 2025). Such aggression is likely rooted in sexual conflict and the injuries resulting from it could provide a pathway for infection and subsequent mortality (van Weelden et al. 2025).

1.1.5.2 Live-strandings

Live-stranding events for killer whales can occur as a result of chasing prey into shallow waters and incidentally becoming beached out of water (as opposed to carcasses washed ashore). These incidental strandings are uncommon along the west coast of North America and most historical accounts of such incidents do not provide information on which population was implicated (Towers et al. 2020). However, since 2002, all live-stranding events of killer whales within the range of the WCT killer whale population along the west coast of North America have been members of this population. Live-stranding risk is likely much higher for WCT than other killer whale populations in this region because they often hunt harbour seals in very close proximity to shore and may become accidentally live-stranded in the process (Towers et al. 2020).

Live-stranding events of WCT killer whales include four events involving five whales between 2002 and 2015 (Towers et al. 2020) and two cases involving two whales between 2016 and 2024 (DFO-CRP and Bay Cetology unpublished data). Adults and juveniles of both sexes were documented stranded on sandy shores and rocky outcroppings. Most individuals survived the events by swimming away when the tide came in. However, an adult female drowned on the incoming tide in 2024 (DFO-CRP and Bay Cetology unpublished data) and an adult male died, perhaps due to internal injuries associated with the stranding, in 1976 (Ford and Ellis 1999). Also, one of the juveniles reported in Towers et al. (2020) may have drowned on the incoming tide if it weren't for responders who were able to apply leverage to her body to move her off the rocks (Towers et al. 2020).

1.1.5.3 Entrapment

Natural entrapment is defined by Jourdain et al. (2021) as a situation where one or more killer whales remain in an unusual, restricted habitat where they were unwilling or unable to leave for a prolonged period. In such situations killer whales can become resource limited and die. Several killer whale populations have been documented entrapped, including in sea ice (Westdal et al. 2017), and with four cases up to 2019 involving WCT killer whales (Jourdain et al. 2021). Since then, two more cases have been documented (DFO-CRP and Bay Cetology unpublished data).

In total, three of these cases involved lone individuals, two of which were young juveniles and one of which was an adult male. All lone whales were successfully coaxed out of the entrapped area by responders and survived, even though the two juveniles were showing signs of malnutrition at the time of the rescue operation (Jourdain et al. 2021; DFO-CRP and Bay Cetology unpublished data). The other three cases involved groups of WCT killer whales, two of which left the entrapment on their own (Jourdain et al. 2021) and the other case included a pair of adult males being encouraged out of the entrapment by responders (DFO-CRP and Bay Cetology unpublished data).

1.1.5.4 Disease

A review of available literature on bacterial, viral, and fungal pathogens in killer whales found three pathogens reported from wild killer whales, including *Brucella* spp., *Edwardsiella tarda*, and cetacean pox virus (Gaydos et al. 2004). *E. tarda* is highly virulent across age groups and was the confirmed cause of death for a SRKW in 2000 (Gaydos et al. 2004); cetacean pox virus is only highly virulent in neonates; and *Brucella* spp. can reduce fecundity in infected individuals, although it is not regarded as being highly virulent (Gaydos et al. 2004). Gaydos et al. (2004) also found 13 pathogens reported from captive killer whales, nine of which were known to be highly virulent. Since that study was completed, two whales were confirmed through necropsy to have died due to bacterial infections; an offshore killer whale died due to *Salmonella* in 2005 in California, and a BKW individual died due to an infection with an unknown bacterial agent in 2007 in BC (Raverty et al. 2020).

Disease may also affect recovery, causing injuries and trauma from which whales may have otherwise recovered to become fatal. Ingestion of foreign bodies, such as fishhooks, may open a pathway for infection. For example, an Alaskan killer whale succumbed to septicemia in 2005 after swallowing a fishing hook and, while the cause of trauma was unknown, the BKW (N018) died due to secondary infection with *Taxoplasma gondii* and *Sarcocystis neurona* after suffering a vertebral fracture (Raverty et al. 2020).

1.2 GOAL OF THE ASSESSMENT

The West Coast Transient population of the Bigg's Killer Whale (BKW) ecotype was listed as Threatened under the *Species at Risk Act* (SARA) in 2003. Under SARA, the federal government has a commitment to prevent wildlife species from being extirpated or becoming extinct, to provide for the recovery of wildlife species that are extirpated, endangered, or threatened as a result of human activity, and to manage species of special concern to prevent them from becoming endangered or threatened. The Minister of Fisheries and Oceans Canada and the Minister responsible for the Parks Canada Agency are the competent ministers for the recovery of aquatic species at risk.

A cumulative effects assessment (CEA) is required in order to address the high priority recovery measure (RM7) in the draft Action Plan: “Assess cumulative effects of potential anthropogenic impacts on Transient Killer Whales using an appropriate impact assessment framework for aquatic species”. Cumulative effects are the combined, incremental impacts that threats from multiple human activities can have on individuals, populations, communities, and ecosystems through space and time. The most pressing threats to WCT have been identified as: environmental contaminants (both legacy and emerging) and physical and acoustic disturbance (DFO 2021b), while also vulnerable to changes in prey availability and/or quality, disease, and toxic spills.

1.3 WHY A CUMULATIVE EFFECTS ASSESSMENT IS REQUIRED

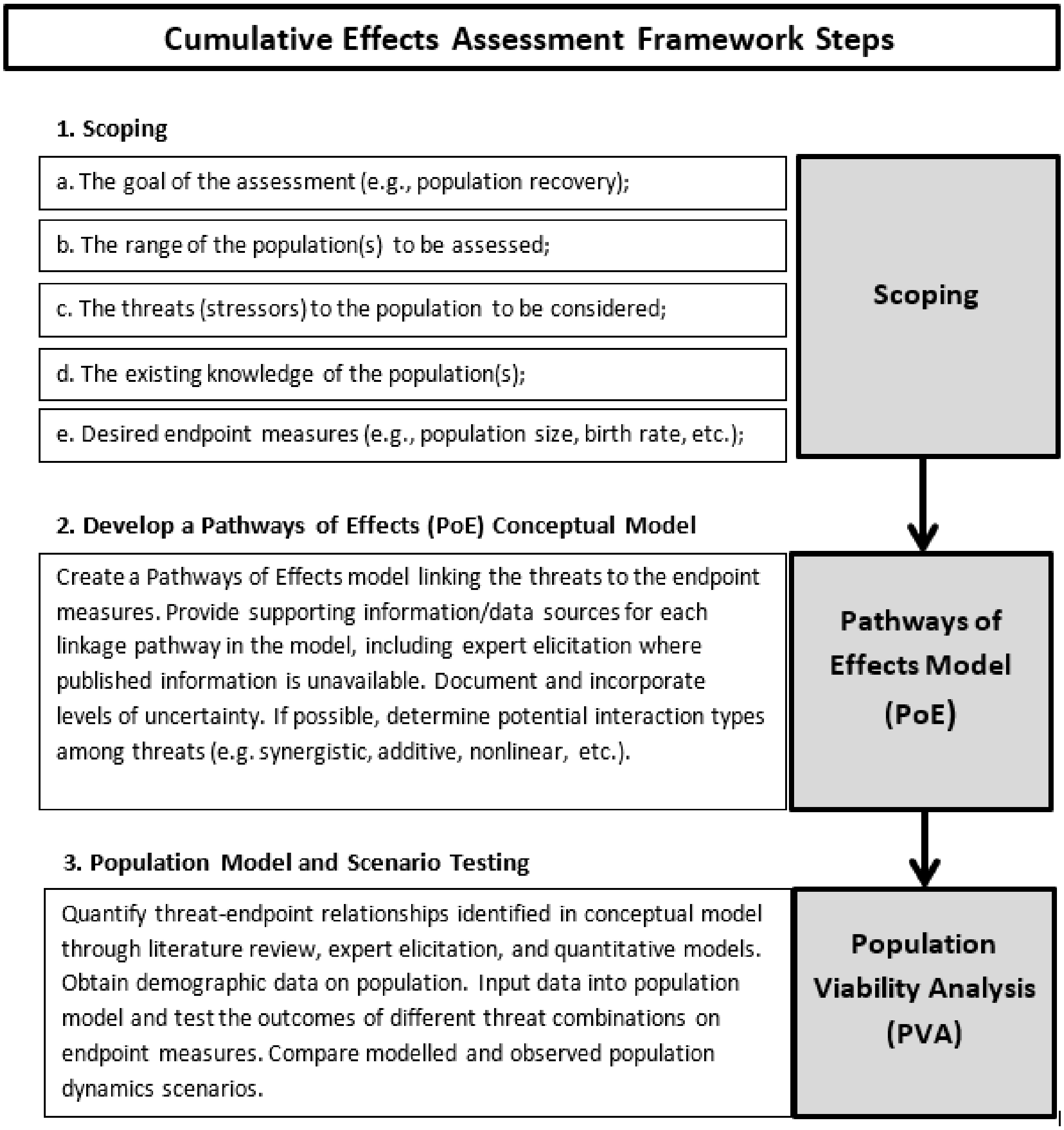
Fisheries and Oceans Canada (DFO) Species at Risk Program has requested that Science Branch provide an assessment of the cumulative effects of current and potential anthropogenic impacts on WCT population of Bigg's killer whales in Canadian waters. Cumulative effects assessments evaluate the effects of multiple threats by transforming impacts into a single currency or metric, thereby allowing for comparisons among threats and their combined impact on long-term population viability. A cumulative effects assessment was completed to address a similar recovery measure for Northern Resident and Southern Resident Killer Whale populations (Clarke Murray et al. 2019; DFO 2019). This study will advance previous methods developed for Resident Killer Whale (RKW) ecotypes (Clarke Murray et al. 2019; DFO 2019; Lacy et al. 2017; Murray et al. 2021) to assess cumulative effects to WCT.

1.4 CUMULATIVE EFFECTS ASSESSMENT FOR BIGG'S KILLER WHALES

An overview of the steps in a cumulative effects assessment for a population is outlined in

Figure 2. The first step of this process is scoping, a process that has already been completed for BKW by Fisheries and Oceans Canada (DFO) Species at Risk Program. The current cumulative effects assessment focuses on the second two steps of this process, referred to here as phase one and phase two. In phase one, a Pathways of Effects (PoE) conceptual model describes and visualises current knowledge on how priority threats can impact population parameters (fecundity and mortality) of the population directly, or through interactions between threats. As threats can interact over space and time, altering the consequent effects on individuals and populations, this study will also identify and assess potential interactions between threats to more accurately represent the natural system. The outputs of the PoE conceptual model are used to design and refine the population model in the second phase.

In the second phase, impacts identified in the PoE conceptual model that have sufficient data are parameterised and a quantitative population viability analysis (PVA) is conducted to assess the cumulative effects, building upon the methods and results of previous work (DFO 2007; Lacy et al. 2017; Murray et al. 2019). Existing literature and data are used to parameterise the impact of each threat on killer whale vital rates. These quantitative values and relationships are used to define the inputs to a population model describing the combined impact on population persistence through time. The model structure builds upon an existing PVA model developed for RKW by Murray et al. (2019).



1. Scoping

a. The goal of the assessment (e.g., population recovery);

b. The range of the population(s) to be assessed;

c. The threats (stressors) to the population to be considered;

d. The existing knowledge of the population(s);

e. Desired endpoint measures (e.g., population size, birth rate, etc.);

2. Develop a Pathways of Effects (PoE) Conceptual Model

Create a Pathways of Effects model linking the threats to the endpoint measures. Provide supporting information/data sources for each linkage pathway in the model, including expert elicitation where published information is unavailable. Document and incorporate levels of uncertainty. If possible, determine potential interaction types among threats (e.g. synergistic, additive, nonlinear, etc.).

3. Population Model and Scenario Testing

Quantify threat-endpoint relationships identified in conceptual model through literature review, expert elicitation, and quantitative models. Obtain demographic data on population. Input data into population model and test the outcomes of different threat combinations on endpoint measures. Compare modelled and observed population dynamics scenarios.

Figure 2. Steps in the cumulative effects assessment framework (adapted from Clarke Murray et al. 2019).

1.4.1 Assumptions

- Impacts from threats to population vital rates (mortality and fecundity), based on the best available information at the time of the assessment, are assumed to be accurately described in the Pathways of Effects model.
- The analysis assumes impacts only from the focal threats examined (reduced prey availability, disturbance, and contaminants), and does not consider the effects of broader impacts such as changing climate conditions and increasing human populations. These broader impacts may influence and interact with the focal threats and have their own impacts, but were out of scope at this time.

- The way that impacts are parameterised in the PVA model is assumed to represent the impacts of the entire threat (e.g., impacts of PCBs represents the Contaminants threat).
- Projections of the population dynamics of the WCT population into the future assume that current threat levels remain the same. It is also assumed that no threat mitigation measures or management actions are taken.
- The population model chosen to be used for future projections is assumed to be an effective surrogate for the real population dynamics for the WCT population.

1.4.2 Objectives

The current working paper has four major objectives:

1. Develop a Pathways of Effects (PoE) conceptual model to visually represent threat-impact pathways for the WCT population limited to the primary threats identified by the Species at Risk Program (DFO 2007).
2. Quantify threat linkage pathways identified in the PoE model by determining the best available and most recent data or information from data analysis, literature review and expert elicitation. This information will be used to develop and parameterise a quantitative PVA model.
3. Assess the cumulative effects acting on the WCT population by running single and cumulative PVA model scenarios to evaluate and compare the effects of each scenario to the observed population dynamics.
4. Identify uncertainties and sensitivities in data and methods and highlight knowledge gaps for future research.

1.4.3 Scope

The aim of the current assessment is to evaluate the cumulative effects of anthropogenic threats on the WCT population of BKW. The study is limited to considering the primary threats identified in the Recovery Strategy (DFO 2007) and Action Plan for the Transient Killer Whales (in prep). The effects of low probability but high impact events, such as catastrophic oil spills, are not included in the quantitative assessment. Future changes in anthropogenic activities and climate change are also not included or assessed. Potential mitigation measures and management actions will not be evaluated but this assessment can be used as a tool to appraise future changes and mitigation measures once the cumulative effects model has been reviewed.

2 PATHWAYS OF EFFECTS CONCEPTUAL MODEL

The overall PoE model describes what we understand about the mechanisms through which priority threats affect individuals in the WCT population based on available evidence (Figure 3). Linkage pathways connect threats to effects on population parameters (fecundity and mortality); they can connect directly, or manifest through interactions between threats. Sections 2.1 to 2.5 outline the supporting evidence for the linkages identified in the PoE model illustrated in Figure 1, where each circled number (e.g., ①) identifies a specific linkage pathway from a threat to an effect on a population parameter. These circled numbers are included in relevant parts of this section to identify where the supporting evidence is located.

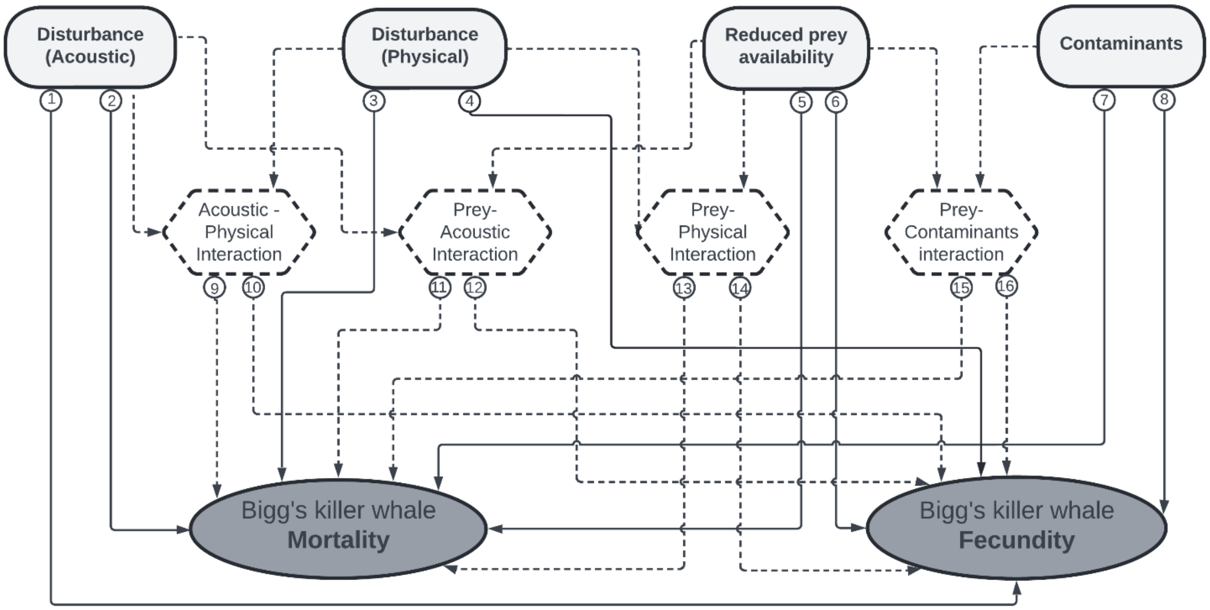


Figure 3. Overall Pathways of Effects (PoE) conceptual model for the West Coast Transient population of Bigg's killer whales showing linkage pathways that are supported by evidence. Eight linkage pathways from the four threats connect directly to effects on population parameters (mortality and fecundity) and eight additional linkages connect through four threat interactions.

2.1 DISTURBANCE (ACOUSTIC)

2.1.1 Background

While there are natural sources of noise in the ocean, such as wind, tides, currents, precipitation, surface waves, and animals, the background sound levels in the ocean have increased over the last several decades in the near-shore and the open ocean due to the presence of anthropogenic noise sources (Andrew et al. 2002; Burnham 2017; Hildebrand 2009). Sound travels much further in water than in air, with propagation in the ocean affected by water temperature, salinity, acidity, and mixing regimes, plus the topography and composition of the substrate (Burnham 2017; Burnham and Duffus 2023; Burnham et al. 2023; Francois and Garrison 1982; Hamilton 1980; Jensen et al. 2011). Increased sea temperature is predicted to increase the influence of already increasing anthropogenic ambient noise (Burnham 2017; Ilyina et al. 2010).

Noise disturbance can be chronic, such as ongoing low frequency shipping noise, or acute, such as the high intensity, intermittent and unpredictable noise from military sonar, construction blasting, seismic exploration, and other short-term, loud noises (Burnham 2017; Nowacek et al. 2007; Southall et al. 2021; Southall et al. 2019; Weilgart 2007) that can cause hearing loss, physical trauma, and disorientation in marine mammals (Abgrall et al. 2009; Simmonds et al. 2004; Southall et al. 2021; Southall et al. 2019). The impact on whales depends on the frequency, intensity, and signal structure of the sound (Murray et al. 2019; National Research Council 2003; Nowacek et al. 2007; Richardson et al. 1995; Simmonds et al. 2004; Southall et al. 2021; Southall et al. 2019; Weilgart 2007).

2.1.1.1 Species at Risk classification: chronic and acute noise

Canada's Species at Risk draft recovery strategy for the WCT population of BKW describes acute and chronic noise in their threat classification table, adapted below in Table 2.

Table 2. Anthropogenic threat classification table for chronic and acute noise, adapted from DFO (2007).

Category	**Chronic noise**	**Acute noise**
Stressor category	Habitat degradation	Disturbance
General stressor	Vessel noise	Intensive impulsive sound
Specific stress	Masking of communication signals, inability to forage successfully	Seismic surveys, military sonar, underwater explosions, pile driving
Effect	Physiological and physical harm	Behavioural effects, Physiological impairment and possible physical harm (from military sonar & underwater explosions only)
Occurrence	Current	Current
Frequency	Continuous, with seasonal variability	Recurrent
Causal certainty	Plausible but requires further study	Expected
Severity	Unknown	Low at current frequency
Level of concern	Moderate	High because of potential to expand

2.1.2 Chronic vessel noise from shipping

The most significant contributor to ocean ambient noise is low-frequency (from 5-1,000 Hz) energy from shipping, mainly from propellers (cavitation and singing) and propulsion (Veirs et al. 2016). Generally, shipping noise increases with shipping traffic and, given commercial shipping traffic has experienced a dramatic increase in volume, the associated noise has also increased. In addition to increased volume, the composition of vessel types have changed over time, particularly the proportion of bulk carriers and container ships. Bulk carriers decreased in proportion from 40.0% in the Salish Sea (1977-1997) to 24.3% (1998-2019), whereas over the same period, container ships increased from 21.4% to 41.7%. This increase is estimated to have resulted in a 38% increase in distance travelled in RKW critical habitat by container ships in in the Salish Sea (Taylor and Mayer 2023). Given that container ships are the noisiest type of commercial vessel (Veirs et al. 2016), the combination of increased shipping volume and a change in the dominant type of vessel has increased the level of chronic noise experienced by marine mammals in this area. How this noise disturbance might affect killer whales is discussed in the next sections.

2.1.3 Hearing and vocalisations in WCT killer whales

Killer whales use vocalisations with sounds largely in the range of 500 Hz to 15 kHz for communication and 15 to 100 kHz for echolocation (Heise et al. 2017; Vagle et al. 2021). Clicks are used for echolocation, orientation, and detection of prey; whistles are used for communication within a group; and pulses are used for both communication with the group and for coordination (Deecke et al. 2005). Social calls range in frequency from 0.1 kHz (click burst) to 75 kHz for ultrasonic whistles, whereas echolocation clicks range from 22 to 80 kHz (Southall et al. 2019). Echolocation use in WCT killer whales was not found to vary between foraging, milling, resting, socializing, slow travelling, and travelling behaviours, although they were observed to use it nearly ten times more in nearshore environments than offshore (Barrett-Lennard et al. 1996). As the hearing range of WCT killer whales and their prey overlap, they avoid alerting potential prey to their presence by travelling without audible echolocation or social calls, using cryptic calls made up of isolated clicks and infrequent, short trains of clicks to resemble random noise (Barrett-Lennard et al. 1996), and are nearly silent when actively hunting, except for after a kill or when interacting with the water surface (Riesch and Deecke 2011). WCT killer whales have been detected successfully hunting in darkness in Alaska using passive listening to home in on prey noises (Deecke et al. 2013).

2.1.4 Acoustic masking ①②⑨⑩⑪⑫

Masking is the interference of acoustic communication, echolocation, and passive listening caused by other noise sources, reducing the ability to detect relevant sounds (Clark et al. 2009). This could be from continuous background noise such as from shipping, or more acute human-derived noise sources (OSPAR Commission 2009). Masking can occur when the volume of a sound of interest does not exceed the threshold of hearing due to the presence of background noise (signal to noise ratio; Clark et al. 2009; Williams et al. 2020), as well as if the interfering noise is made in the same frequency as the sound of interest (Clark et al. 2009). Masking can affect marine mammals by obscuring or completely blocking communication and prey sounds, increasing the energetic demand of an individual communicating, or by causing physiological stress (Burnham and Duffus 2023; Smith 2017). Acoustic masking from chronic noise can also interfere with echolocation signals used for navigation and discrimination, and can affect coordination and social communication signals used by marine mammal-eating killer whales (DFO 2021b). In the PoE, masking affects population parameters through two interactions in the PoE model, a prey-acoustic interaction and an acoustic-physical interaction.

2.1.4.1 Acoustic masking and prey availability ⑪⑫

This section describes how the acoustic disturbance threat interacts with the reduced prey availability threat through acoustic masking (the *Prey-Acoustic* interaction in the PoE model). The primary prey of WCT is harbour seal, harbour porpoise and Steller's sea lion, but also include Dall's porpoise and California sea lion (Ford et al. 2013). Harbour seals and California sea lions can hear sounds in frequencies as low as 1 kHz (Deecke 2003), harbour porpoises hear well up to 140 kHz and Pacific white-sided dolphins up to 128 kHz (Deecke 2003). Killer whales hear best between 0.018 and 42 kHz, but can hear up to 115 kHz (Branstetter et al. 2021; Branstetter et al. 2017; Deecke 2003; Miller 2006; Thornton et al. 2022), and use passive listening to detect prey sounds. Although killer whale hearing range overlaps with the pinniped and dolphin species they hunt, it does not overlap with the upper extent of Dall's and harbour porpoises, who can communicate at much higher frequencies (Figure 4). Porpoises are thought to have evolved this 'acoustic crypsis' strategy to avoid detection by predators (Sørensen et al. 2018).

If low frequency shipping noise overlaps with the frequencies used by marine mammal-eating killer whales and their prey, acoustic masking of prey sounds by vessel noise can occur (Figure 4; Table 3), hindering prey detection and successful capture, especially at longer distances (Smith 2017), effectively reducing prey availability (Sato et al. 2021). Ships and boats, including pleasure crafts, bulk carriers, container ships, cargo and vehicle carriers, tankers, military, fishing, passenger, research, tug, and miscellaneous vessels, can produce broadband noise from 0.0115 kHz – 40 kHz (Veirs et al. 2016), which overlaps the entire killer whale communication range. In addition, some echosounders have peak energy emissions within the frequency range used by BKW prey species (Burnham et al. 2022, Table 3). Marine mammal-eating killer whales cannot compensate for vessel noise by increasing their call amplitude as fish-eating killer whales do (see Holt et al. 2011; Holt et al. 2009), without increasing the risk of being detected by their marine mammal prey (Barrett-Lennard et al. 1996; Holt et al. 2009; Smith 2017).

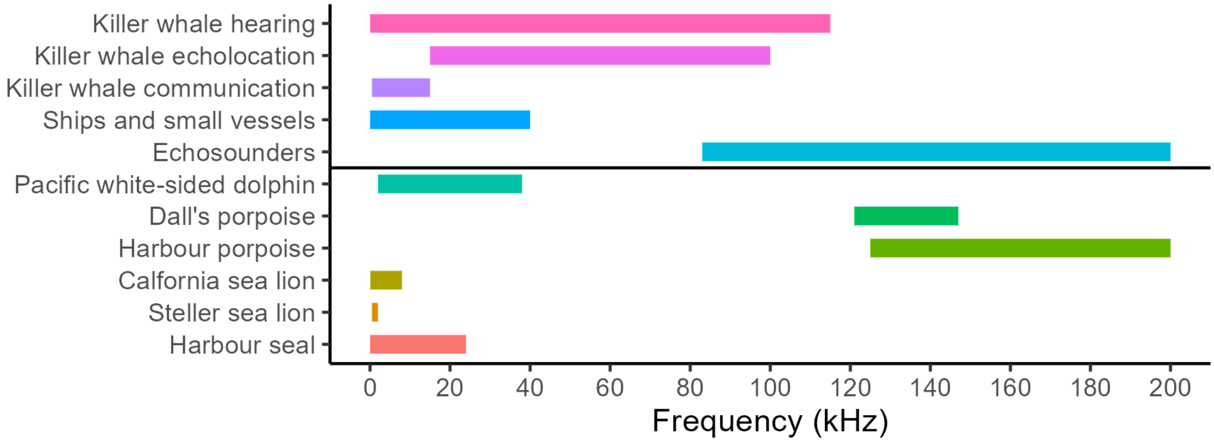


Figure 4. Hearing range and echolocation of killer whales compared to the sound frequencies produced by ships and small vessels, echosounders and BKW prey species. Note that this does not indicate sound intensity.

Harbour seals can use a mating system called ‘Lekking’, where males aggregate in certain areas and attract mating females with auditory and physical displays (Boness et al. 2006). During mating season, males spend more time in the water away from haul outs, and in certain areas produce loud noises as part of their display (Hayes et al. 2004). In BC, these mating calls have a diel pattern, occurring more at night (Nikolich et al. 2018). This is thought to be an anti-predation strategy, but may also reduce the potential for masking of their signals by vessel noise (Matthews et al. 2020; Nikolich et al. 2018).

Table 3. Sound production and level produced by Bigg’s Killer Whale prey species.

Species	**Sound production**	**Sound level**
Harbour seals (Phoca vitulina)	Produce underwater sounds from 0.02 (roar) to 24 kHz (roar) (Southall et al. 2019)	Source production level for the underwater roars produced by male harbour seals at one meter was approximately 140 dB re 1µPa. The low-frequency calls lasted 5-10 seconds, with the most energy between 100 – 800 Hz (Casey et al. 2016)
Steller sea lion (Eumetopias jubatus)	Produce underwater sounds from 0.5 to 2 kHz (belch) (Southall et al. 2019)	Information not found.
California sea lions (Zalophus californianus)	Produce underwater sounds from < 0.08 (sweep) to 8 kHz (bark, bang) (Southall et al. 2019)	At a distance of approximately 2 m, the source production level of an adult male California sea lion bark in air at 1 kHz was 103 dB re 0.0002/cm^2 and 90 dB at 0.5 and 2 kHz, and the barks of bulls emitted in air are similar to those underwater (Schusterman 1974)

Species	Sound production	Sound level
Harbour porpoise (Phocoena phocoena)	Produce click/echoic sounds with range of medium frequency from 125 to 200 kHz (Southall et al. 2019)	Harbour porpoise high-frequency clicks, measured in captivity, had a duration of ~100 μs, a peak frequency of ~130 kHz, an inter-click interval of ~60 ms, and a maximum source level of 172 dB re 1 μPa pp at 1 m (Villadsgaard et al. 2007). The mean source level for Canadian Harbour porpoises was 178±4 dB re 1 μPa (peak-peak) (Kyhn et al. 2013)
Dall's porpoises (Phocoenoides dalli)	Produce click/echoic sounds with range of medium frequency from 121 to 147 kHz (Southall et al. 2019)	Dall's porpoises produced a mean source level of 187±7 dB re 1 μPa (peak-peak) (Kyhn et al. 2013).
Pacific white-sided dolphins (Lagenorhynchus obliquidens)	Produce social calls from 2 to 20 kHz (whistle whole range); and click/echoic sounds with a range of peak (dominant) frequencies from 22 to 38 kHz (Southall et al. 2019)	Source levels of clicks from the Pacific white-sided dolphin measured in captivity were 170 dB (Evans 1973, cited in Rasmussen et al. (2002)).

2.1.4.2 Acoustic masking and navigation ①②⑨⑩

Masking can limit the ability of cetaceans to use acoustic cues for navigation, orientation, and wayfinding, and can reduce the ability to accurately form cognitive maps of their surroundings (Burnham and Duffus 2023). For WCT killer whale coordination and social communication signals, navigational acoustics and echolocation use could be affected (DFO 2021b). In the PoE model, the pathways of effect from impaired navigation from acoustic masking are through a direct route (increased risk of stranding) and an interaction (increased risk of ship strike).

Acoustic masking and accidental stranding ①②

WCT killer whales minimise the use of echolocation calls to avoid warning prey when foraging, milling, resting, socializing and travelling, but when in a nearshore environment echolocation use increases nearly ten times compared to offshore, as they need to be sure of their location to avoid stranding or colliding with the shoreline (Barrett-Lennard et al. 1996; Riesch and Deecke 2011). Increased risk of accidental stranding is one potential effect of acoustic masking from human noise sources (OSPAR Commission 2009) that could result in injury or death, although adults usually avoid accidental stranding by abandoning pursuit when prey seek refuge on rocky shores (Towers et al. 2020).

Acoustic masking and ship strike ⑨⑩

The *Acoustic-Physical* interaction in the PoE model describes how the acoustic disturbance threat, through acoustic masking, could interact with the physical disturbance threat to increase risk of ship strike. Ship strikes can kill or cause serious physical trauma to marine mammals that could contribute to population decline (Schoeman et al. 2020). In theory, the ability of killer whales to hear and locate vessels could be impaired by acoustic masking, increasing their risk of vessel strike. However, evidence for this theory has not been well documented, as this requires an understanding of complex factors such as the generation and propagation of vessel noise from different vessel types, particularly in surface waters, and in different areas and the quantification of diverse behavioural responses (Erbe et al. 2019), and isolating effects from acoustic masking from other factors. It is expected that the specific acoustic signature of a vessel plays a role in how well marine mammals can detect it (Leal et al. 2015) and, while there is evidence that vessel slowdowns reduce fatal ship-strikes to marine mammals (e.g., Laist et al. 2014; Vanderlaan and Taggart 2007), to what degree the associated reduction in noise and acoustic masking from slowdowns contributes to this outcome is not understood.

2.1.5 Other responses to noise ①

Marine mammals have complex and varied responses to noise disturbance that can include changes to surfacing and breathing patterns, swimming speed and direction (e.g., Lusseau et al. 2009; Parks et al. 2007), vocalisation (e.g., Dahlheim and Castellote 2016; Holt et al. 2009; Watkins 1986), heartbeat (Miksis et al. 2001), stress (Rolland et al. 2012), avoidance of noisy areas (Konrad Clarke et al. 2024; Morton and Symonds 2002; Richardson et al. 1995), disrupted feeding, breeding, nursing, and migration (Croll et al. 2001; Foote et al. 2004; Huntington et al. 2015; Walker et al. 2019; Williams et al. 2014). Chronic exposure to vessels appears to exacerbate stress in RKW (Smith 2017), and change vital rates and stress hormone levels in other cetaceans (Burnham and Duffus 2023; Rolland et al. 2012). Resident killer whales have been known to avoid areas with constant high amplitude background noise (Konrad Clarke et al. 2024; Morton and Symonds 2002). Although changes in behaviours such as diving, socialising and foraging as a response to noise have some energetic cost to killer whales (Noren et al. 2009; Williams et al. 2002; Williams et al. 2015), what this cost is, remains unclear. A recent study measuring killer whale respiration rates and oxygen intake during different behaviours (McRae et al. 2024) may help researchers start to understand these energetic costs.

2.1.6 Quantifying behavioural disturbance

Despite ongoing research, there is a still a lack of knowledge allowing scientists to quantify the biological significance of noise disturbance to marine mammals at the population level (Erbe et al. 2019). To quantify impacts to marine mammal population parameters such as fecundity, complex behavioural responses to noise need to be translated to an increase in energy expenditure. An added complexity for attempts to link behavioural change to changes in vital rates is that animals can adapt in the short term; a study that modelled responses of dolphins to increased vessel noise resulted in a biologically insignificant response due to the population adapting their behaviour (New et al. 2013). Population Consequences of Disturbance (PCoD) models can be used to link short-term behaviour changes to long-term effects on population dynamics (New et al. 2020), and can incorporate life-history traits, disturbance source characteristics and environmental conditions (Keen et al. 2021). A PCoD model has been used for SRKW (Tollit et al. 2017) but can be limited by the availability of information and data needed for inputs.

Other types of models have been developed to explore this area, for example a model to examine movement responses of marine mammals to noise disturbance and estimate impacts from reduced foraging based on a dose-response function (Joy et al. 2022). Another type of method to quantify the effect of shipping noise on killer whale vital rates uses economic ‘shock’ methodology, comparing RKW populations before and after sudden changes in shipping linked to human global and political dynamics (Taylor 2021a, 2021b; Taylor and Mayer 2023). This method uses a large dataset of port landings (Lloyds list) going back to 1977 to calculate vessel kilometres travelled in SRKW and NRKW habitat over time, then assesses the effects of negative “noise shocks” (abrupt increases in shipping traffic and associated noise) to the carrying capacity of RKW habitat. The disturbance of behaviours such as foraging, socialising and reproduction due to noise shocks were connected to effects on fertility and mortality. The study found lower killer whale births in higher vessel kilometer habitats, and that SRKW habitats were more affected than NRKW habitat (Taylor 2021a, 2021b; Taylor and Mayer 2023). It was concluded that the reduction in fertility and increased mortality in SRKW from noise shocks was sufficient to fall below replacement levels leading to declines of the population (Taylor 2021a, 2021b; Taylor and Mayer 2023).

2.1.7 Summary of linkage pathways to BKW population parameters

Two pathways in the PoE model connect the acoustic disturbance threat directly to mortality and fecundity, and four pathways connect to mortality and fecundity through two interactions (Figure 5).

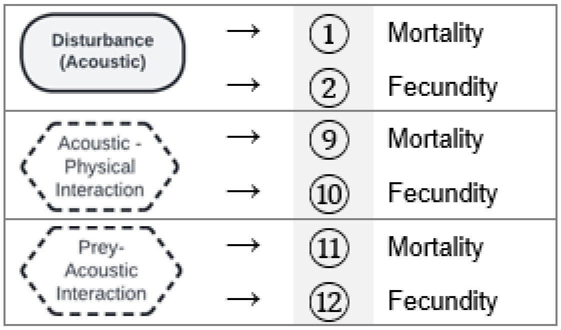


Figure 5. Summary of linkage pathways in the PoE that connect from the Disturbance (acoustic) threat to effects on population parameters directly or through interactions with other threats.

2.2 DISTURBANCE (PHYSICAL) – PRESENCE AND STRIKES

2.2.1 Background

The physical presence threat includes two types of physical disturbance from vessels that cause effects in the PoE model: effects from the physical presence of a vessel, and effects from vessel strikes.

2.2.2 Vessel presence ④⑬⑭

Vessels, especially those engaged in whale watching, can seek out, follow and observe killer whales, sometimes for extended periods of time (e.g., Moore et al. 2021). WCT can be affected by vessel presence through increased stress (psychological and/or nutritional), displacement, and/or reduced fitness from the disruption of normal behaviours, energy expenditure is increased with reduced rest and foraging time, and by needing to change travel patterns when vessels are present (Smith 2017). This has been documented for Northern Resident killer whales (NRKW), where the important social behaviour of beach rubbing was negatively affected by vessel presence due to the learned avoidance over time of those areas (Konrad Clarke et al. 2024). In a meta-analysis of whale-watching impact studies, Senigaglia et al. (2016) found that cetacean travel paths consistently became more convoluted in the presence of whale-watching boats and that many species also changed their activity budget, with animals more likely to travel than rest or forage. These changes decrease their opportunity to obtain energy inputs from prey consumption. Each of these behavioural changes can cause increased energy expenditures in the presence of whale-watching boats.

2.2.2.1 Species at Risk classification for physical disturbance

Canada’s Species at Risk draft recovery strategy for the WCT population of BKW describes this threat in their threat classification table, adapted below in Table 4.

Table 4. Anthropogenic threat classification table for vessel presence, adapted from DFO (2007).

Category	Physical disturbance
Stressor category	Disturbance
General stressor	Recreational activities Whale-watching operations
Specific stress	Interruption of foraging and social behaviours
Effect	Possible displacement
Occurrence	Current
Frequency	Continuous, with some seasonal variability
Causal certainty	Expected but requires further study
Severity	Unknown
Level of concern	High

2.2.2.2 Vessel presence and prey availability ⑬⑭

The interaction between vessel presence and prey availability is represented in the PoE model with the *Prey-Physical* interaction. Vessel presence is hypothesised to interfere with hunting in BKW based on observations of harbour seals utilizing nearby vessels as haulouts to avoid capture (Schmunk 2015; Smith 2017).

There is little information about how BKW respond to disturbance, as individuals or as a population. Studies on dolphin populations have found that open populations that are not food-limited, such as BKW, are more resilient to disturbance and less likely to change their behavioral time budgets and motivations at low frequencies of disturbance (New et al. 2020). An additional challenge is the difficulty isolating the effects of vessel presence from the effects of vessel noise, and other changes in behaviour due to external factors.

2.2.3 Vessel strikes ③④⑨⑩

If a moving vessel strikes a whale, it can result in blunt force trauma and lacerations that can injure or kill, and have the potential to affect all age classes and ecotypes, especially those that frequent waters near human populations and shipping lanes (Kelley et al. 2021; Raverty et al. 2020). Expected increases in commercial shipping and cruise ship traffic could also increase the risk of vessel strikes, and the risk is likely increased in areas where vessels need to navigate confined passages such as routes in and out of industrial developments (DFO 2021b).

2.2.3.1 Species at Risk classification: collision with vessel

Canada’s Species at Risk draft recovery strategy for WCT killer whales describes this threat as ‘collision with vessel’ in their threat classification table, adapted below in Table 5.

Table 5. Anthropogenic threat classification table for collision with vessels, adapted from DFO (2007).

Category	Collision with vessel
Stressor category	Accidental mortality
General stressor	High speed vessel traffic
Specific stress	Blunt force trauma and/or lacerations
Effect	Direct or indirect mortality (via infection)
Occurrence	Current
Frequency	Recurrent
Causal certainty	Demonstrated
Severity	Low
Level of concern	Low

The actual number of deaths and injuries to killer whales resulting from vessel strikes is not well known, but it has been reported that there were two mortalities and six injuries from vessel strikes between 2004 and 2016 (DFO 2021b). Vessel strikes have been documented to have contributed to the deaths of killer whales in four vessel strikes involving six killer whales between 1974 and 2016 (Raverty et al. 2020).

The bodies of dead killer whales can sink before detection (Ford et al. 1998; Murray et al. 2021), but those that are found can be necropsied to determine the cause of death. Records are available for 22 Bigg’s individuals (across all populations) necropsied from 2004-2018 (Lee et al. 2023; Raverty et al. 2020). Of these, 11 individuals had a cause of death that was not traumatic; the causes included environmental incidents (n=5), infection (n=1), failure to thrive (n=1), autolysis and a subcutaneous haematoma (n=1), and nutritional or metabolic issues (n=3). Three individuals died from trauma, and an additional 8 individuals had an unknown cause of death (Table 6).

Table 6. Details of 11 Bigg's killer whale deaths and the proximate cause of death, based on necropsy information from 2004-2018. Abbreviations for sex are female (F), male (M), and unknown (U). Source: Raverty et al. (2020) and Lee et al. (2023).

Necropsy or recovery date	Location	Age class	Sex	Ecotype	Proximate cause of death
2004-05-03	Oregon, USA	Adult	F	Bigg's	Unknown
2005-12-02	California, USA	Adult	F	Bigg's	Traumatic – unknown
2007-05-22	Washington, USA	Adult	F	Bigg's	Traumatic – Human interaction
2009-03-29	BC, Canada	Adult	M	Bigg's	Unknown
2009-04-05	California, USA	Calf	F	Bigg's	Unknown
2009-08-02	BC, Canada	Adult	F	Bigg's	Unknown
2010-06-14	Washington, USA	Adult	F	Transient 59 haplotype	Unknown
2010-07-23	Alaska, USA	Adult	U	Transient GAT haplotype	Unknown
2011-02-23	Oregon, USA	Sub adult	U	Bigg's	Unknown
2013-10-18	BC, Canada	Adult	F	Bigg's	Unknown
2018-11-14	BC, Canada	Adult	M	Bigg's	Trauma, possible vessel strike and aggression by conspecifics

2.2.4 Summary of linkage pathways to BKW population parameters

Two linkage pathways in the PoE model connect the Physical disturbance threat directly to mortality and fecundity, and four linkage pathways connect it to mortality and fecundity through two interactions (Figure 6).

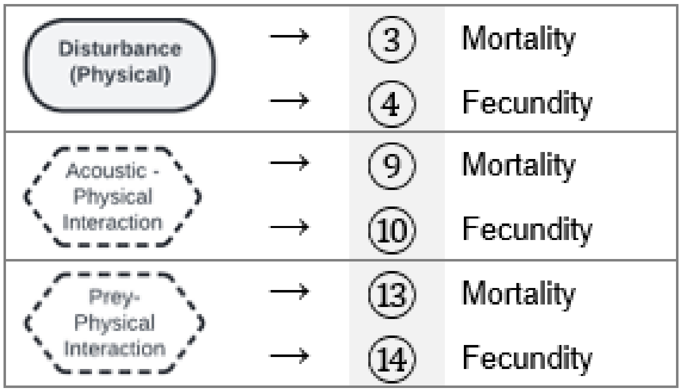


Figure 6. Summary of linkage pathways in the PoE that connect from Disturbance (physical) threat to effects on population parameters directly or through interactions with other threats.

2.3 REDUCTION IN PREY AVAILABILITY

2.3.1 Background

Bigg's killer whales consume a variety of marine mammals, a strategy that could reduce the risk that the decline of a single prey species affects the population. A decline across multiple prey species could result in a shift to a less desirable prey species in terms of energetic or nutritional requirements, and/or increased expenditure in capture or processing, leading to nutritional deficiencies. There is evidence of this in Alaskan populations in the 1990s (DFO 2009; Estes et al. 1998). BKW may starve before eating fish, even when that is the only food option (DFO 2007).

2.3.2 Prey species in the BKW diet

The WCT population of BKW has been documented to consume a variety of marine mammals in coastal waters. The species the WCTs most frequently prey upon is the harbour seal (DFO 2021b), and they adapt their distribution to align with harbour seal pupping and breeding times (Baird and Dill 1995). Of the predation events in the Canadian portion of the Salish Sea, WCTs consumed primarily harbour seal (66% of predation events) followed by harbour porpoise (17%), Steller sea lion (11%), Dall's porpoise (2%), and all other species (4%) (DFO-CRP and Bay Cetology unpublished data). There is no evidence that individual whales specialize on particular prey species (Ford et al. 2013).

2.3.3 Trends in prey species

2.3.3.1 Harbour seal and Steller's sea lion

The harbour seal population in British Columbia was depleted by extensive harvesting between 1879 and 1914, reducing the population to 20,000 animals, and then again from 1962 to 1968 further reducing numbers to 10,000 animals, before receiving protections in 1970 (Olesiuk 2010). The new protections allowed the population to recover to an approximate pre-hunt abundance of 100,000 in 2008 (Figure 7, left) (Olesiuk 2010). A subsequent assessment resulted in an estimated 85,400 harbour seals in 2019, with over 40% of animals found in the Strait of Georgia (DFO 2022). Given the uncertainty in the estimate, the stock in 2019 is either stable or has declined slightly relative to the 2008 assessment. The Salish Sea harbour seal population is estimated to be at equilibrium, totaling around 45,000 individuals. Population carrying capacity was believed to have been met in the mid-1990s (Ashley et al. 2020; DFO

2010; Jeffries et al. 2003). Harbour seals are central-place foragers who return to the same haulout location between trips, and use those haulout sites to rest, molt, raise pups, thermoregulate, and avoid predators (Brusa et al. 2024). Adult harbour seals have high site fidelity to haulout sites and forage in the surrounding 30 km, though foraging from haulout sites ranges from seals who stay within 5–10 km (Lowry et al. 2001) to those who travel up to 100 km (Brusa et al. 2024). During mating season (June to August), male display (lekking) behaviour happens both aquatically and terrestrially, while mating occurs in the water (Brusa et al. 2024).

The Steller sea lions in BC represent approximately one third of the Eastern Stock (extending from California to Alaska). Control programs and commercial harvests of Steller sea lion reduced the population to an estimated low of approximately 3,500 animals (Bigg 1984; Bigg 1985). The population has recovered since protections were introduced in the 1970s; non-pups increased at an average rate of 3.8% per annum and pup production at a rate of 4.8% per annum, resulting in more than a 4-fold increase in abundance since the species was protected in 1970 (Figure 7, right) (DFO 2021a; Olesiuk 2018). Recolonization and occupancy of new rookeries, as well as expansion of year-round and winter haulouts has also occurred. In 2017, the coastwide breeding season population was estimated to be 43,200 (38,700-48,200 95% CI) (DFO 2021a). The winter population can be higher as a result of an influx of animals from rookeries outside Pacific Canada (DFO 2021a).

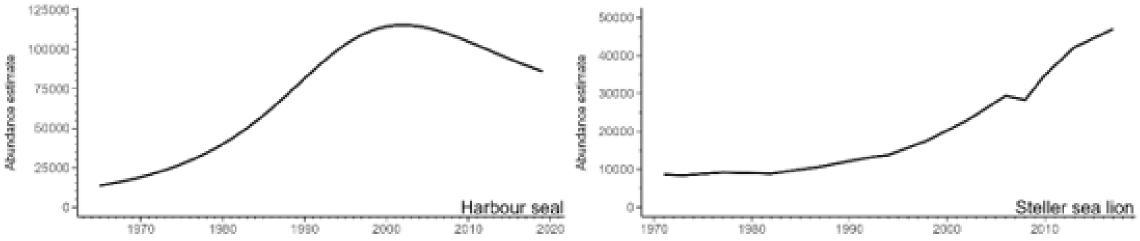


Figure 7. Population estimates for harbour seal (left) and Steller sea lion (right) through time.

2.3.3.2 Harbour porpoise

The harbour porpoise population has tripled since the 1990s (Figure 8) (Carretta et al. 2011), recovering from population lows that may be due to bycatch, vessel and noise disturbance, pollution, competition, or habitat loss (Jefferson et al. 2016). In 2015, the estimated abundance for the southern Salish Sea, Puget Sound, and Hood Canal regions was 11,233 porpoises (CV = 37%, 95% CI = 9,616 – 13,120) (Jefferson et al. 2016). The harbour porpoise population has been increasing along the Pacific coast (Evenson et al. 2016) and may be an increasingly important prey species in the BKW diet. Although this species has a small body size compared to other prey species, it has a higher energetic content per kilogram (Shields et al. 2018).

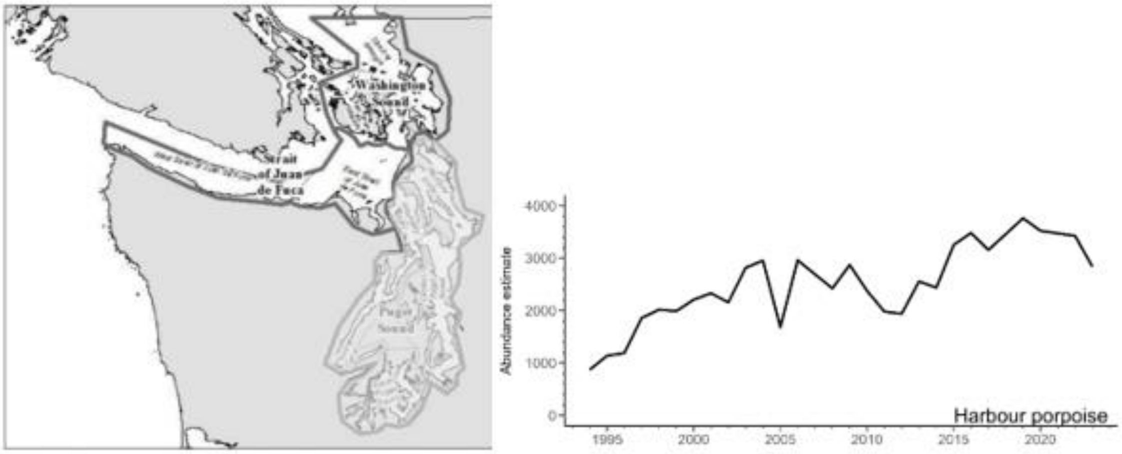


Figure 8. Left, harbour porpoise study area includes Washington Sound and Strait of Juan de Fuca regions (Washington Sound includes the Strait of Georgia and the San Juan Islands) and right, harbour porpoise abundance estimates over time (USDW, unpublished data).

2.3.3.3 Species at Risk classification: Decline in prey availability or quality

Canada's Species at Risk draft recovery strategy for WCT killer whales describes this threat in their threat classification table, adapted below in Table 7.

Table 7. Anthropogenic threat classification table for decline in prey availability or quality, adapted from DFO (2007).

Category	**Decline in prey availability and/or quality**
Stressor category	Consumptive use or culling
General stressor	Culling
Specific stress	Prey reduction
Effect	Lack of food
Occurrence	Current (local); Historical (range wide)
Frequency	Unknown (local); Continuous until early 1970s (range wide)
Causal certainty	Plausible
Severity	Low (local); High (range wide)
Level of concern	Low (based on current seal management and cetacean protections)

2.3.4 Effects of prey reduction on WCT killer whales ⑤⑥⑪⑫⑬⑭⑯

The WCT killer whale population and their marine mammal prey were greatly reduced historically in coastal BC, and both groups have been recovering due to the addition of protections in the 1970s. Because of this legacy, it can be difficult to assess what the impact of reduced prey could be on WCT population through direct study or through correlations since both are recovering from reduced population sizes. However, we can look to the resident killer whales as a comparison. Killer whales are unable to create large energy stores, so declines in prey can affect individuals quickly. For example, body condition in SRKW fluctuates even during a single year, increasing from May to September when prey is more abundant and decreasing in winter and early spring when prey is restricted (Stewart et al. 2021). Stewart et al. (2021) also found that killer whales with poor body condition were more likely to die, with the greatest risk among old whales (both male and female), followed by calves. In addition to the effect on mortality, for SRKW there is a strong correlation between fecundity and prey availability, with the probability of a female calving dropping by 50% after a year with low salmon abundance compared to high abundance (Ward et al. 2009). Also, there is some evidence of expanded summer ranges and decreased social cohesion among SRKW when prey density is low (Ward et al. 2009), which could reduce the probability of reproductive events. In other marine mammals, Steller sea lions were found to have decreased survivorship and fecundity when prey quality decreased (Trites and Donnelly 2003).

2.3.5 Interactions with other threats

In addition to directly affecting WCT population parameters, this threat affects WCT vital rates through three interactions in the PoE model. The interactions may not affect the abundance of prey directly, but may instead reduce the access that the WCT has to the prey, which has been called “reduced prey availability” to distinguish between the amount of prey present (abundance) and how accessible these prey are to the WCT population (availability). The nature of these interactions is described in section 2.6 - Other Threats:

• Prey - Acoustic interaction	• 2.1.4.1 Acoustic masking and prey availability ⑪⑫
• Prey – Physical interaction	• 2.2.2.2 Vessel presence and prey availability ⑬⑭
• Prey – Contaminants interaction	• 2.4.5.3 PCB toxicity and Prey availability

2.3.6 Summary of linkage pathways to WCT population parameters

Two pathways in the PoE model connect the prey availability threat directly to mortality and fecundity, and six pathways connect to mortality and fecundity through three interactions (Figure 9).

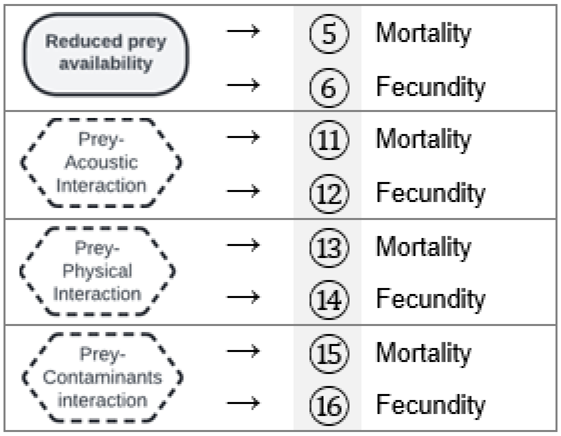


Figure 9. Summary of linkage pathways in the PoE that connect from Reduction in prey threat to effects on population parameters directly or through interactions with other threats.

2.4 CONTAMINANTS

2.4.1 Environmental exposure to contaminants

Contaminants enter the marine environment through routes that include storm water, wastewater, rivers, and atmospheric deposition (Cullon et al. 2009). Those that are not transported far from their sources have elevated levels closer to urban centres, whereas those with longer-range transport, such as Per- and polyfluoroalkyl substances? (PFOS), can be found at elevated levels in more remote areas (Lee et al. 2023). Though it follows that environmental exposure to different contaminants in killer whales is connected to the geographic range of the ecotype, drawing conclusions based on killer whale ecotype is challenging given that they are largely sympatric, ranges can vary each year and we lack full understanding of the areas used by populations.

2.4.2 Persistent Organic Pollutants and bioaccumulation

Most contaminants of concern for killer whales have the characteristics of Persistent Organic Pollutants (POPs), in that they are persistent in the environment, exposing organisms for extended times; they bioaccumulate and build up in body tissues; are toxic to organisms; and can have long-range transport in the environment (Hayes et al. 2022). Some legacy POPs are still a concern in the environment, despite being banned from manufacture since the 1970's. The levels of these types of contaminants increase with trophic level and bioaccumulate in body tissues over a lifespan. Given the extended lifespan and high trophic level of killer whales, their tissues can contain high contaminant burdens. For killer whale populations such as WCT that consume marine mammals, bioaccumulation becomes even more important as marine mammals have higher contaminant loads than the lower trophic level fish that resident killer whales consume (Lawson et al. 2020), resulting in far higher contaminant loads in WCT blubber (Lawson et al. 2020; Ross 2006; Ross et al. 2000). A further contributing factor to high contaminant levels in WCT is the particularly large amount of fat storage in individuals, allowing the accumulation of more contaminants than resident killer whales (Lee et al. 2023).

2.4.3 Contaminants of concern

Research into the effects of contaminants of concern to resident killer whales is ongoing (Environment and Climate Change Canada 2021). Contaminants identified to be of concern for

WCT are outlined in Table 8, these include emerging contaminants as well legacy contaminants that are no longer released.

Table 8. Contaminants of concern to killer whales and their potential effects. Effects: RD – Reproduction & development, IS – Immune system, ES – Endocrine system, CN – Carcinogenic.

Symbol	**Contaminant name**	**RD**	**IS**	**ES**	**CN**	**Ref**
APEs/Aps	Alkylphenol ethoxylates / Alkylphenols			✚		1
DDT	Dichloro-diphenyl-trichloroethane	✚	✚	✚		2
HBCCD	Hexabromocyclododecane					1
PAHs	Persistent polycyclic aromatic hydrocarbons				✚	2
PCDDs, PCDFs	Dioxins and Furans	✚	✚	✚	✚	2
PBBs, PBDEs	Polybrominated biphenyls, Polybrominated diphenyl ethers	✚	✚	✚	✚	2, 3, 4
PCBs	Polychlorinated biphenyls	✚	✚	✚		2
PCNs	Polychlorinated napthalenes	✚	✚	✚		6
PCPs	Polychlorinated paraffins			✚		2
PCTs	Polychlorinated terphenyls	✚		✚		2
PFAS PFOS	Per- and polyfluoroalkyl substances (e.g., PFOS and PFAS)	✚			✚	1
TBT, DBT	Tributyltin, Dibutyltin		✚			5

[1] Lee et al. (2023), [2] Grant and Ross (2002), [3] Ross (2006), [4] US EPA (2017), [5] Frouin et al. (2008), [6] Fernandes et al. (2022)

2.4.4 Species at risk classification

Canada’s Species at Risk draft recovery strategy for WCT killer whales describes the threat from contaminants in their threat classification table, adapted below in Table 9.

Table 9. Anthropogenic threat classification for 'pollution and changes in natural processes (food supply)', adapted from DFO (2007).

Category	**PBTs – legacy**	**PBTs emerging contaminants**
Stressor category	Pollution and changes in natural processes (food supply)	Pollution and changes in natural processes (food supply)
General stressor	Persistent Bioaccumulating Toxins PBTs	Persistent Bioaccumulating Toxins PBTs
Specific stress	Direct toxic effects and transfer (and bioaccumulation) of contaminants to killer whales through prey	Direct toxic effects and transfer (and bioaccumulation) of contaminants to killer whales through prey
Effect	Reproductive impairment, endocrine disruption, skeletal abnormalities, cancer, etc.	Reproductive impairment, endocrine disruption, skeletal abnormalities, cancer, etc.
Occurrence	Current	Current
Frequency	Continuous	Continuous
Causal certainty	Expected	Expected
Severity	High	High
Level of concern	High	High

2.4.5 Polychlorinated Biphenyls (PCBs)

Despite new information and data describing existing and emerging contaminants of concern for killer whales (Brown et al. 2022; Lee et al. 2023), there is only one model that connects the effects of a specific contaminant to mortality and/or fecundity of killer whales. The only model currently available is for PCBs (Hall et al. 2018), and consequently, this section will primarily focus on discussion of PCBs. PCBs are fat soluble and accumulate in body tissues, and exposure can result in chronic forms of immunotoxicity and neurotoxicity and reproductive impairment, making them of particular ecotoxicological concern for killer whales (Morra and Gobas 2017; Ross et al. 2000). PCB concentrations vary widely across individuals and populations of killer whales (Table 10).

Table 10. Comparison of mean PCB concentrations in samples taken from different killer whale populations.

Population	PCB concentration Mean (mgkg-1lw)	Citation
WCT (NE Pacific) – Males	132.4 ± 35.2 SE (n=16)	Guy (2018); Ross et al. (2013)
WCT (NE Pacific) – Females	104.8 ± 20.1 SE (n=17)	Guy (2018); Ross et al. (2013)
Alaska Transient	150 ± 31 SD (n=5)	Herman et al. (2005)
East Arctic (Canadian)	92 ± 10 SE (n=30)	Remili et al. (2023)
Eastern Canada	96 ± 31.2 (n=5)	Remili et al. (2023)
Alaska Resident	15 ± 6 SD (n=14)	Herman et al. (2005)
Alaska Offshore	66 ± 6.5 SD (n=2)	Herman et al. (2005)
Southern Resident (NE Pacific)	146 ± 33 SE	Guy (2018)
Northern Resident (NE Pacific)	37 ± 6 SE	Guy (2018)

2.4.5.1 Significance of diet and sex for PCB contaminant levels

Contaminant levels in male killer whales are higher than females due to the offloading of contaminants from mother to calf during gestation and lactation, reducing contaminant levels in the mother (e.g., Remili et al. 2023). The diet of a killer whale population is reflected in the PCB load of individuals in that population, and fatty acid signature analysis has found diet to be an even stronger predictor for PCB levels than sex in North Atlantic killer whale populations (Remili et al. 2023). Populations with fish dominant diets are on the lower end of PCB levels; contaminant load increases for populations with more mixed diets, and is highest in populations with pinniped and toothed whale dominant diets (Remili et al. 2023). The type of PCB congeners in killer whales also vary by diet, where primarily pinniped eating killer whales have a higher ratio of chlorinated PCB congeners than those mostly eating toothed whales, related to the ability of pinnipeds to metabolise less chlorinated PCB congeners (Remili et al. 2023).

2.4.5.2 The effect of PCB toxicity on killer whales ⑦⑧

PCBs are known to adversely affect the immune system and the endocrine system, and can cause a failure of reproduction in marine mammals through immunosuppression and disruption of endocrine function (Murphy et al. 2015). Effects to the immune system can increase the risk of infection, which has been observed in harbour porpoises, whose infection risk doubled with blubber PCB levels over 45 ug g^{-1} lipid (Hall et al. 2006). In pinnipeds, POP contamination of blubber can affect energy balance regulation through altering adipose function, important for fat deposition and mobilisation (Robinson et al. 2018). The threshold concentration of PCB exposure for the onset of toxic effects to marine mammals is estimated at 9 mg/kg lipid weight (lw); levels exceeding 41mg/kg lw present a high risk of reproductive failure (Dietz et al. 2019;

Helle and Olsson 1976; Jepson et al. 2016; Kannan et al. 2000). A threshold of 10mg/kg lw has been used for immunotoxic and hormonal imbalance effects for marine mammals (Dietz et al. (2019).

Recent research indicates marine sediments are now the main source of PCBs in the marine environment of the Salish Sea, rather than point sources and air pollution (Lee et al. 2023). Sediments are easier to sample and test for contaminants and methods have been developed for predicting PCB concentrations in resident killer whales using PCB concentrations in sediments of their important habitat areas (Alava 2019; Alava et al. 2012).

2.4.5.3 PCB toxicity and Prey availability interaction ⑮⑯

This section describes an interaction between the contaminant (PCB) threat and reduction in prey availability (the *Prey-Contaminants* interaction in the PoE model). The WCT population has shown consistent growth (Towers et al. 2019) despite evidence of very high concentrations of PCBs in their bodies, well above thresholds estimated for serious toxic effects (Dietz et al. 2019; Remili et al. 2023). In contrast, population declines have been observed in the fish-eating SRKW, which also have high PCB levels in their bodies. One reason for this difference may relate to differences in prey availability and the consequences of nutritional shortages, which can result in killer whales metabolising stored lipids in blubber to meet energy requirements (Lundin et al. 2016). Blubber provides buoyancy, supports locomotion, stores energy, and provides thermal insulation in cetaceans (Wang et al. 2015). The inner layer of blubber is where metabolic lipids are stored, ready to be used for energy when fasting (Samuel and Worthy 2004; Wang et al. 2015). Lipid soluble contaminants such as PCBs accumulate in blubber, with the PCB distribution in cetacean bodies correlated with tissue lipid content (Yordy et al. 2010). When lipids are metabolized from blubber, the previously stored contaminants are released and redistribute to other body tissues, where they can selectively mobilise into the bloodstream so that cetaceans experiencing nutritional shortages also increase their risk of health effects as a result of being exposed to contaminants (Yordy et al. 2010).

2.4.6 Summary of linkage pathways to population parameters

Two pathways in the PoE model connect the contaminants threat directly to mortality and fecundity, and two pathways connect to mortality and fecundity through one interaction (Figure 10). For more information on the four interactions in the PoE model, refer to Section 2.5.

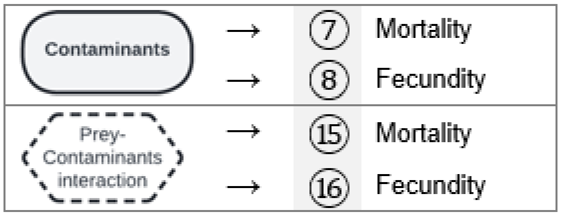
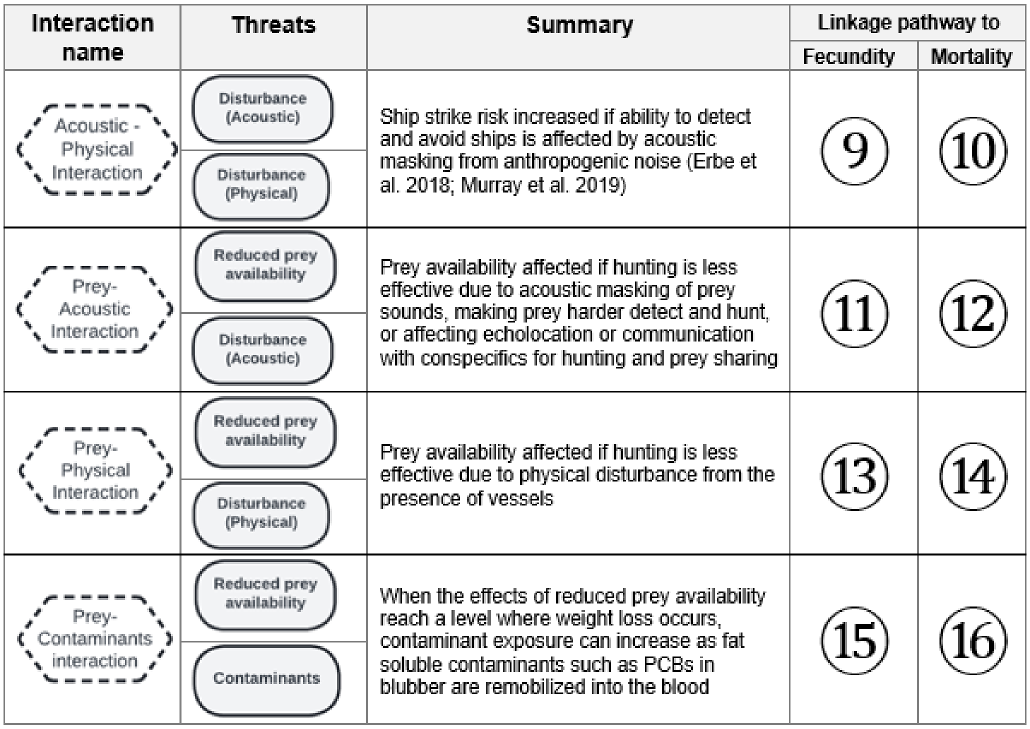


Figure 10. Summary of linkage pathways in the PoE that connect from Contaminants threat to effects on population parameters directly or through interactions with other threats.

2.5 SUMMARY OF THREAT INTERACTIONS IN THE POE MODEL

A threat or stressor interaction represents the way that two or more stressors interact to create an effect that may be different to that of each threat considered individually (Crain et al. 2008). For example, an additive interaction is where the effects of all stressors are the result of adding them together. In synergistic interactions, the effect of the stressors in the interaction is higher than expected if solely adding them together, and for antagonistic interactions the effect of the combined stressors is less than expected by adding them together (Clarke Murray et al. 2014). Outside of these three main interaction types are a range of other ways that stressors can interact in non-linear ways.

The PoE overview model for WCT killer whales identified four interactions between the identified focal threats (Figure 11); these have been introduced in the relevant sections but are also summarised below.



Interaction name	Threats	Summary	Linkage pathway to	
			Fecundity	Mortality
Acoustic - Physical Interaction	Disturbance (Acoustic) Disturbance (Physical)	Ship strike risk increased if ability to detect and avoid ships is affected by acoustic masking from anthropogenic noise (Erbe et al. 2018; Murray et al. 2019)	⑨	⑩
Prey-Acoustic Interaction	Reduced prey availability Disturbance (Acoustic)	Prey availability affected if hunting is less effective due to acoustic masking of prey sounds, making prey harder detect and hunt, or affecting echolocation or communication with conspecifics for hunting and prey sharing	⑪	⑫
Prey-Physical Interaction	Reduced prey availability Disturbance (Physical)	Prey availability affected if hunting is less effective due to physical disturbance from the presence of vessels	⑬	⑭
Prey-Contaminants interaction	Reduced prey availability Contaminants	When the effects of reduced prey availability reach a level where weight loss occurs, contaminant exposure can increase as fat soluble contaminants such as PCBs in blubber are remobilized into the blood	⑮	⑯

Figure 11. Description of the four interactions identified in the overview Pathways of Effects (PoE) model for WCT Bigg's killer whales.

2.5.1 Natural mortality – threat interactions

2.5.1.1 Disease

Disease-causing agents such as viruses, bacteria, and parasites can cross species barriers more easily when species are closely related (DFO 2009). BKW may be at risk of infection from diseases that are endemic to their prey, due to their consumption of prey that are also marine mammals high in the food chain. Additionally, contamination entering the marine environment through sewage or runoff from terrestrial sources may expose killer whales and their prey to pathogens from livestock or domestic pets (DFO 2021b). Terrestrially-sourced pathogens include species such as *Brucella* spp., which is associated with reproductive tract lesions and encephalitis in cetaceans, as well as *Toxoplasma gondii* (DFO 2021b). *T. gondii* has been documented causing tissue cysts, encephalitis, lesions, and disseminated toxoplasmosis in cetaceans (Dubey et al. 2020), and is suspected to have contributed to the death of a young, male killer whale in captivity in 1988 (Costa-Silva et al. 2019).

Disease or biotoxin outbreaks are known to affect pinniped and cetacean populations around the world. Outbreaks of *Morbillivirus* have caused mass mortality events in dolphin and seals in other parts of the world (Aguilar and Borrell 1994; Kennedy et al. 2000) and in river otters in

Pacific Canada (Mos et al. 2003). If such an event were to occur in WCT prey species there is the potential to affect the WCT population (DFO 2021b) either through reduced prey abundance or directly by contracting the disease from their prey. According to Walther et al. (2002), the frequency of disease outbreaks is increasing. Further, as contaminated seals are more vulnerable to an outbreak of an infectious disease, the quality and/or quantity of WCT prey may be affected with increased contamination levels (DFO 2021b).

2.6 OTHER THREATS

2.6.1 Historic threats

Live capture fishery – Of the 68 killer whales removed as part of the live capture fishery in BC and Washington State waters from 1962-1977 (Bigg and Wolman 1975), five were BKW, which were removed between 1970-1975 (Asper and Cornell 1977).

Shooting – There was a period of time when killer whales in BC were shot indiscriminately, but evidence of this has been rarely observed since 1974 (Ford et al. 2000). Additionally, the fact that WCT killer whales are not fish eaters mean they are less likely to be involved in conflicts with commercial fishing operations.

2.6.2 Oil spills

Killer whales are vulnerable to low frequency, high consequence events such as large oil spills. As they live in small groups, the loss of only a few individuals can have drastic and long-lasting impacts (Myers et al. 2021). These types of impacts were seen in Alaskan BKW exposed to the Exxon Valdez oil spill in 1989, where 22 individuals of the genetically unique AT1 population were observed swimming through oiled waters (Fraker 2013). This resulted in the death of nine individuals in the following 18 months, likely due to inhalation of petroleum vapours (Matkin et al. 1999). The population continued to decline to the current population of 7 individuals observed in 2004-2019 (Matkin et al. 2008; Young et al. 2023). The severe decline had drastic impacts; there has been no recruitment and no births in this population since 1984 and the population it is not expected to recover (Muto et al. 2021).

2.6.3 Climate change

Though out of scope for this assessment, climate change is expected to affect marine mammals in multiple ways. Direct climate change impacts to killer whales, harbour porpoises, and harbour seals are not predicted (Gulland et al. 2022) but may result in a northward range shift (van Weelden et al. 2021). Mammal-eating killer whales may spend more time in the Arctic as the climate changes, as demonstrated by the increase in rake marks on bowhead whales (4% of landed whales bore marks between 1976-1992 compared to 8% 2002-2012). Pre-2000 reports of killer whales tended to range from June to August, while more recent observations suggest that killer whales are staying into September, and as late as mid-November, corresponding to sea ice advance (Stafford 2019).

Prey distribution and abundance for the WCT prey species, including California sea lions and harbour seals, are expected to change and result in reduced body condition for WCT prey (Gulland et al. 2022). Additionally, increased occurrence of harmful algal blooms are expected to increase infection disease outbreak frequency and extent with climate change (Gulland et al. 2022).

2.7 PATHWAYS OF EFFECTS DISCUSSION

The overview PoE represents the current knowledge of how identified threats could affect WCT fecundity and mortality. The most prominent node in the PoE model is ‘Reduced prey availability’, which connects to fecundity and mortality through two direct linkage pathways, and six pathways through three interactions (Prey-Acoustic, Prey-Physical, and Prey-Contaminants). Another important node is Acoustic disturbance, which connects to effects on fecundity and mortality through two direct linkage pathways, and six linkage pathways through two interactions (Acoustic-Physical and Prey-Acoustic). The next step is to determine which linkage pathways have the necessary data to be incorporated into the PVA model, as linkages need to be quantifiable for inclusion. Section 2.7.1 and Figure 12 illustrate the linkages which could be quantified (black lines) and those that could not (grey lines) with the data currently available. Section 3, Population Viability Analysis, describes how linkage pathways in this PoE model were quantified and incorporated into the PVA model.

Additional threats to the WCT population have been identified in the recovery strategy (DFO 2021b) that were not included in the current model, and include intense impulsive sound, chemical contaminants (both legacy and emerging) other than PCBs, biological pollutants, trace metals, toxic spills, and disease.

2.7.1 Pathways of effects model used in the PVA model

The PoE conceptual model provides a general understanding of the potential pathways of effects that selected threats might have on the WCT population parameters based on available literature and data (Figure 3). Only those linkages that can be parametrized with enough confidence can be incorporated into the PVA model. The linkage pathways (direct and interaction) that were not able to be parameterized, were removed from this assessment and are identified by grey lines in the pathways of effects model in Figure 12a. The structure of the modified PoE model representing the inputs to the PVA model comprises six linkages that could be parametrized (Figure 12b).

In the PVA model, the Disturbance (acoustic) threat is represented by the effects of acoustic masking from chronic shipping noise, including an interaction with prey availability. The disturbance (physical) threat is represented by vessel strikes. Prey availability is represented by prey abundance, based on a combined prey index of three species important in WCT diet. The contaminants threat only includes PCBs as it is the only contaminant with a quantified effect on fecundity and mortality, despite evidence that other contaminants are present in WCT killer whales.

Two interactions were parameterized for the PVA model, both of which include a threshold effect of low prey availability. When the prey index is below 1.0 (mean prey level), the Prey-Acoustic interaction is parameterized to decrease fecundity due to a reduction in prey detection, and the Prey-Contaminants interaction increases mortality and decreases fecundity when prey is below the threshold due to effects on the immune and endocrine systems, implemented in the population viability model as increased calf mortality. Although there is some evidence for the two interactions linked to Physical disturbance (Acoustic-Physical and Prey-Physical interactions) they were not able to be parameterized in the PVA model. The details of how threats were parameterized are described in the following PVA section, Section 3.

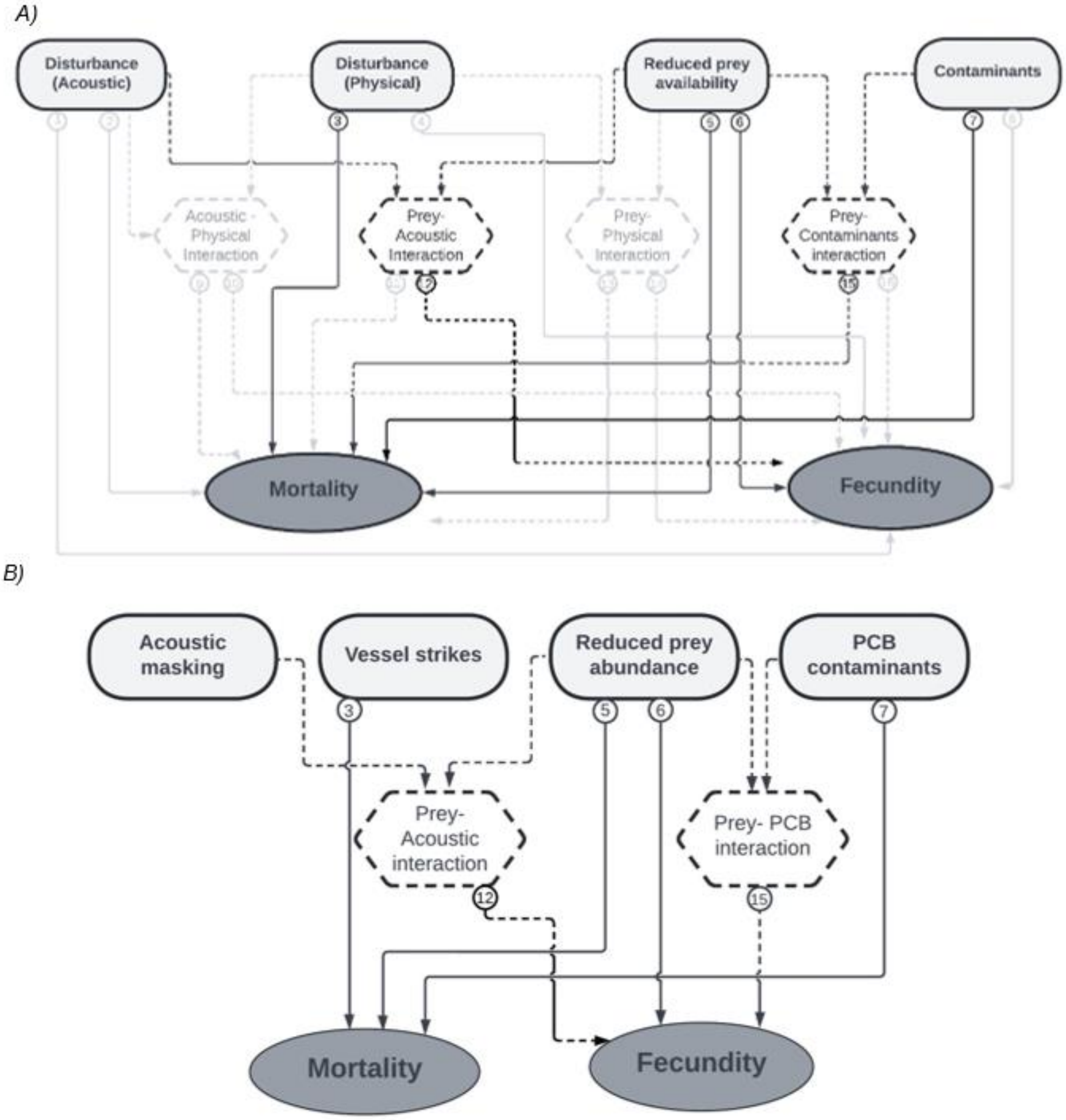


Figure 12. A) Visualisation of the linkages from stressors to effects within the overview Pathways of Effects (PoE) conceptual model, showing linkages that were not able to be quantified in light grey, and linkages which could be quantified in black. B) Visualisation of the Population Viability Analysis (PVA) conceptual model, showing the six quantifiable linkages from stressors to effects that were included in the PVA model (all other linkages removed).

3 POPULATION VIABILITY ANALYSIS

3.1 POPULATION MODEL - METHOD OVERVIEW

The population viability analysis is comprised of a series of steps outlined in Figure 13 and starts with establishing a baseline population trend from a relatively unimpacted population. Next, identified threats are examined and quantified, and their effects upon the baseline population trend are simulated. Different scenarios of threat combinations are examined to estimate how individual, interactive and cumulative threats impact the baseline population trend and are compared to the observed population trend. Sensitivity testing and projection can then estimate future population trends under different scenarios.

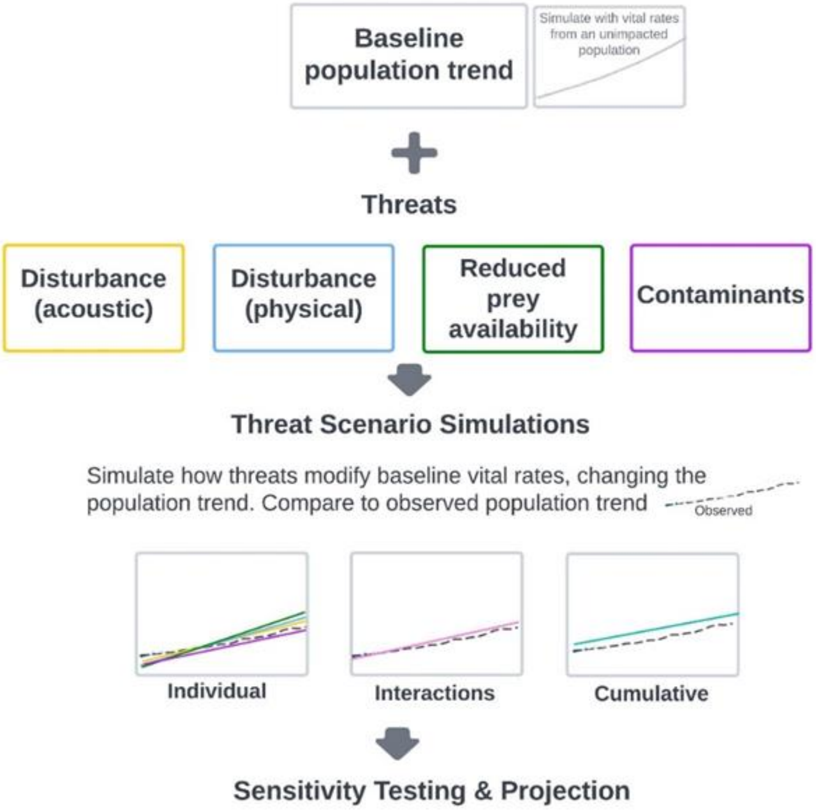


Figure 13. Overview of Population Viability Analysis (PVA) modelling approach.

3.1.1 Baseline and threat scenario simulations

Population Viability Analysis (PVA) was used to simulate the effects of the threats on the target population (Figure 13). The PVA was implemented using Vortex, an open access software which allows for individual-based simulation of populations based on demographic, environmental, and genetic stochastic inputs (Lacy and Pollak 2023). To run the two-sex stage-structured model, Vortex requires age-specific mortality and fecundity rates and a list of individuals with age at start of simulation, parentage, sex, and identity to be tracked over time (Figure 13). To allow the combination and addition of threat impacts, the fecundity and mortality rates for Southern Alaska Resident Killer Whales (SARKW) (Matkin et al. 2014) were used as a relatively-unimpacted baseline population separate from the threats already affecting the focal population (Murray et al. 2019). The SARKW are a population with similar life history to the WCT but removed from the threats experienced in the Salish Sea. While the SARKW population is not considered to be pristine as it is exposed to some anthropogenic impacts, and was notably impacted in 1989 by a major oil spill (Exxon-Valdez), these one-off impacts are not incorporated into the vital rates for SARKW as these anomalous deaths were excluded from the data analysis in Matkin et al. (2014). Using the rates from this outgroup population enables the systematic testing of individual threats.

Model scenarios were developed based on individual and cumulative threats. The threats (described in further detail in section 3.3) were included in the model as modifiers of the SARKW baseline vital rates (Figure 13). Simulations were run on each of the individual threats (noise, strikes, contamination), interactions between threats (noise-prey; contaminants-prey), and a combined, cumulative effects model including all four threats (Figure 13). To start the simulation, the list of individuals present in the Canadian Salish Sea subset (see section 3.2) in 2005 was used. The age, sex, and status of these individuals were input into Vortex and the SARKW baseline rates for mortality and fecundity were applied. At the beginning of the time series, immigration of WCT into the Salish Sea had been documented (Houghton et al. 2015; Shields et al. 2018; DFO-CRP and Bay Cetology unpublished data), so to account for this, ten whales were modelled to join the study area in the first two years, with a gradual decrease in immigration numbers for ten years. The baseline model is run with only this data, and each threat scenario is run as a modifier to the baseline model (see section 3.3 for more detail on threat quantification in the PVA).

Model simulations were run on each scenario 1,000 times and summary statistics were recorded for the population growth rate (*r*) and population size at each time step (N_t). Population growth rate (*r*) was quantified as the exponential rate of increase, according to Equation 1.

$$r = ln\left(\frac{N_{t+1}}{N_t}\right)$$

Equation 1

3.1.2 Model inspection and validation

A model inspection approach (Law and Kelton 1991) was used to examine the simulated population size, population growth rate, age structure, and sex ratio for each scenario and compared to the observed population data (Figure 13). Specifically, the predicted abundances resulting from threat-modified reference vital rates were compared to the abundance estimates from the mark-capture-recapture model (MCR; see section 3.2.1 Population Parameters) over the same time period (2005-2021). The assumption underlying this approach is that if we can define a model that replicates the realised dynamics for the WCT, we have constructed an appropriate model for the system.

In addition to visual inspection, we validated the comparison across models by calculating the root mean square error (RMSE) to quantify the differences between the Vortex abundances for each scenario (predicted) and the CJS MCR estimates (observed), according to Equation 2.

$$RMSE = \sqrt{\frac{1}{n}\sum_{i=1}^{n}([y_i - \hat{y}_i])^2}$$

Equation 2

where n is the number of years, y_i are the observed values for year *i* and $\hat{y}_i$ are the predicted values for year *i*. Lower RMSE values were indicative of the model scenarios better replicating the observed population trend (Hodson 2022).

3.1.3 Population projection

In order to demonstrate the use of the cumulative effects model, the Canadian Salish Sea subset was projected into the future. The full cumulative effects model, including all four threats (prey-PCB-noise-strike), began in 2021 and was projected 100 years into the future under two scenarios of prey availability. The first scenario assumed continued stable availability of prey to that observed in recent years. The values were defined by the mean and variation (SD) of the prey index from the time period 2005-2021. In Vortex, the annual prey index value was randomly sampled from normal distribution with mean 1.44 ± 0.11SD. The second prey availability scenario simulated a logarithmic increase in prey. The annual prey index trend

(1985-2021) was fitted to a logarithmic model and extrapolated, according to the logarithmic function, prey index = 0.0065*ln(x) +1.4539, where x represented year of the simulation.

3.1.4 Sensitivity analysis

Sensitivity analysis was conducted on key parameters in the model to test the impact of uncertainty in these parameters on the results of the study. The sensitivity of the model to the input parameters was investigated using sensitivity testing, “ST” scenarios in Vortex 10. Parameters that required sensitivity analysis included: noise impact value, prey index, PCB accumulation rate, and vessel strike probability. For each of the threats included in the cumulative effects model, single factor sensitivity testing of each parameter across its full range was conducted. This method systematically tests each parameter independently while holding the other parameters at the base level. The response of modelled population size (N) to variations across the range of the four threat parameters was recorded.

3.2 FOCAL POPULATION AND STUDY AREA

The West Coast Transients (WCT) have been photo-identified since the early 1970s and encounters with individuals are documented in an encounter database maintained by DFO Cetacean Research Program and Bay Cetology (Towers et al. 2019; DFO-CRP and Bay Cetology unpublished data). The database includes information about the individual (sex, age, dam (mother), etc.), if known, and the date and location of the sighting. Most WCT killer whales in the coastal subset of the population are observed each year but multi-year gaps in the sighting histories of some individuals generate uncertainty for years of birth and death. A population assessment for the WCT population has not been conducted since Ford et al. (2007). Life history parameters are not readily available and the information about resident killer whales is often used as a proxy. In order to conduct a quantitative cumulative effects assessment, recent, specific data about the population were required, including annual fecundity, mortality, and immigration rates as well as abundance.

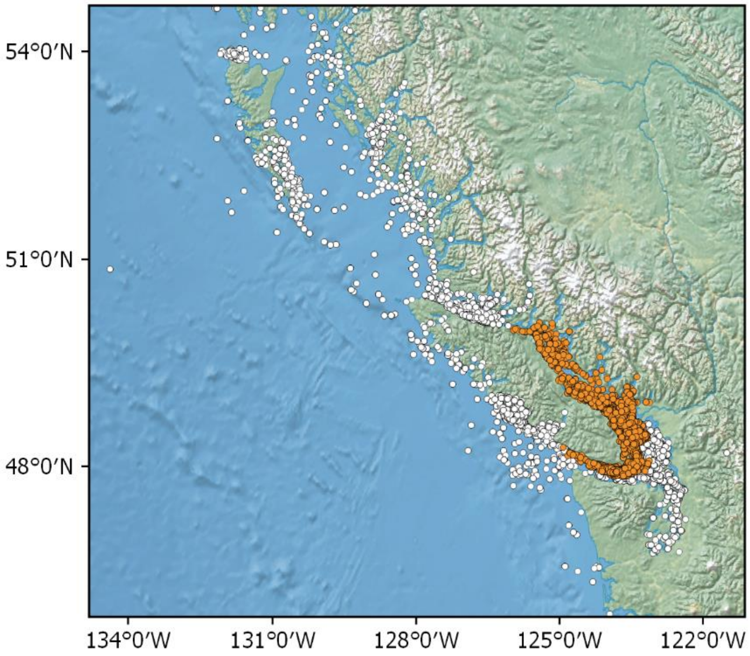


Figure 14. Map of encounters with WCT killer whales (DFO-CRP and Bay Cetology unpublished data). White circles: all encounters, 1970-2022. Orange circles: selection of encounters within the Canadian portion of the Salish Sea (CSS), 2005-2022.

The Canadian portion of the Salish Sea was chosen as the study area because of a relatively consistent survey effort and high likelihood of whales in the area to be affected by human activities. The entire encounter database for WCT (43,282 observations) was filtered for sightings of individuals within the Canadian portion of the Salish Sea (CSS) between 2005 and 2022 (Figure 14, Canadian Salish Sea subset, 16,409 observations). Therefore, this CSS dataset includes both individuals regularly observed (from the putative inner coast subpopulation or coastal subset) and some considered visitors to the area (from the outer coast subpopulation described under 2.1.1). The resulting dataset comprised 415 individuals (Figure 15). While this subset of individuals is not a biological population, it represents whales who have been documented using the study area at any time between 2005 and 2022, and therefore have been exposed to the high intensity and diversity of threats in the Salish Sea.

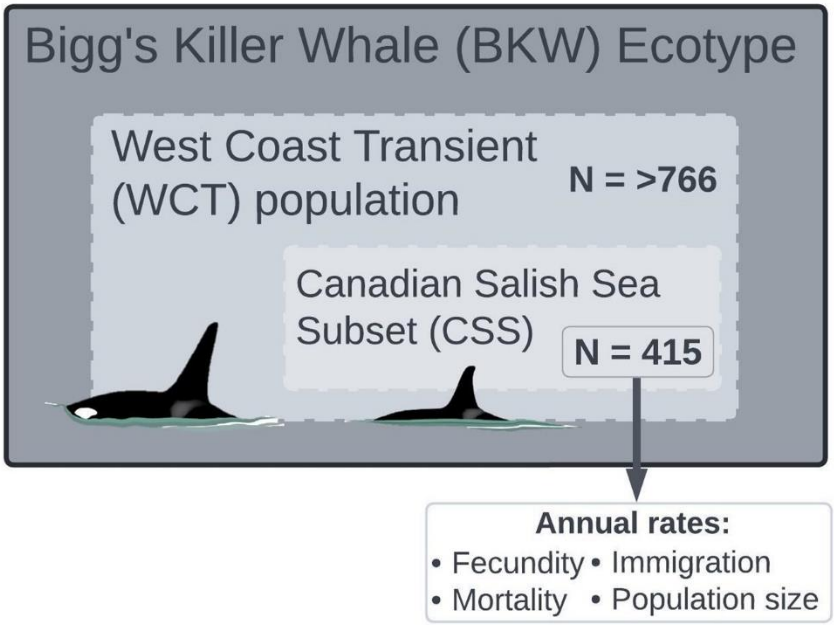


Figure 15. Data subset used in the estimation of population parameters (Canadian Salish Sea subset) and relation to West Coast Transient population (WCT) and Bigg's killer whale (BKW) ecotype.

3.2.1 Population parameters

Annual demographic rates (fecundity, mortality, and immigration) were needed to investigate relationships between population trends and prey. Annual population size was required to compare the threat scenario simulations to a measure of observed population size.

3.2.1.1 Fecundity

Annual fecundity rates (AFR) were defined as the number of births divided by the number of living reproductive females in each year in the Canadian Salish Sea (CSS) subset (DFO-CRP and Bay Cetology unpublished data). Life-history information (year of birth, reproductive status) was available for 377 of the 415 individuals in the CSS dataset. Two approaches were compared. An initial calculation was performed only on calves and females that had been observed in a given year. A second calculation also included calves “inferred” to have been born in a certain year based on data from the years immediately before and after, and females not observed but “inferred” to have been alive based on subsequent observations. The mean values from the two approaches for the fecundity over the 1985-2020 period were almost identical (0.1054 vs 0.1053, respectively, for the 10-50 age class). The second approach resulted in a larger sample size and a less variable number of females per year, and therefore was chosen to provide the annual rates.

AFR was estimated for two age groups, ages 10-50 and 10-30, with the 10-50 females representing the full range of reproductive females and 10-30 representing the most fertile age range. AFR for females aged 10 to 50 ranged from 0.036 to 0.222, with a mean 0.105. AFR for females aged 10 to 30 ranged from 0.048 to 0.312, with a mean of 0.148 (Figure 16).

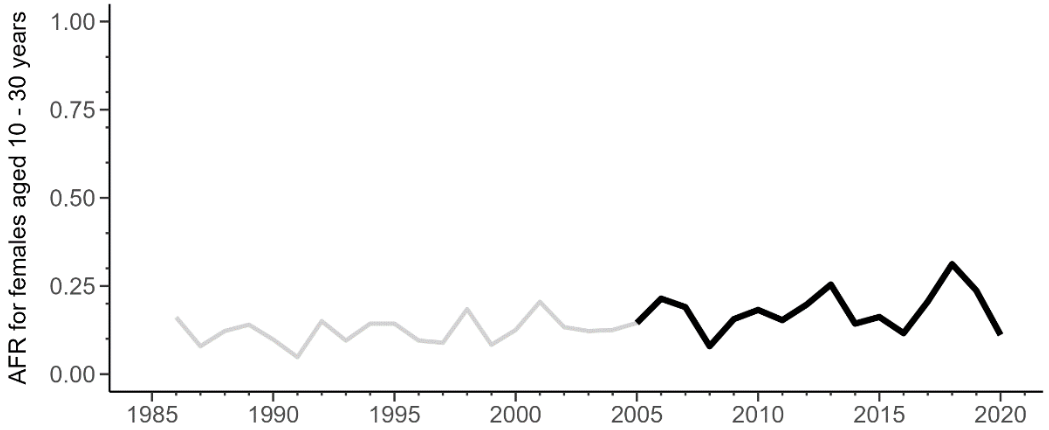


Figure 16. Annual fecundity rate (AFR) for females aged 10 - 30 years from individual encounters tracking dataset (DFO-CRP and Bay Cetology unpublished data), with time range for PVA highlighted in black.

3.2.1.1.1 Fecundity in the PVA

Fecundity rates were used in the quantification of the reduced prey availability threat (Section 3.3). In the PVA model, sex ratio at birth was assumed to be equal. Calves are dependent on their dams for 1 year after birth, so if a mother died in the model simulations in the first year after birth of the calf, her calf also died.

3.2.1.2 Mortality and abundance

Estimates of annual mortality rates and abundance required a different analytical approach. Cormack-Jolly-Seber (CJS) mark-capture-recapture (MCR) models were fitted to the annual capture histories of the 415 unique individuals of the CSS dataset from 2005 to 2022 using software MARK (White and Burnham 1999) through interface RMark (Laake 2013) in R (R Core Team, 2020). CJS models are a standard approach to estimate apparent survival rates (because permanent emigration cannot be distinguished from death in this framework) and recapture probabilities. Abundance is calculated by dividing the number of observed individuals each year by the recapture probabilities (Lebreton et al. 1992).

Several model formulations were compared using the small sample modification of Akaike's information criterion (AICc; Akaike 1973), with annual survival probability φ and annual recapture probability p being modeled as either constant (.) or varying annually over time (t). Goodness of fit was evaluated to check if the assumptions of CJS models were met, using Test 3.SR and Test 2.CT from package “R2ucare” (Gimenez et al. 2018). Test 3.SR explores whether animals marked for the first time have a different probability of being recaptured than previously encountered individuals (i.e., a transience effect) whereas Test 2.CT checks if recapture probability in the current year is affected by whether an animal was captured or not during the previous year (i.e., a trap dependence effect).

The CJS model with the strongest support had time-dependent survival and time-dependent recapture probabilities (Table 11). The second most-supported model had constant survival (0.958) and time-dependent recapture probabilities. Those two models yielded almost identical estimates of abundance Figure 17. Models with constant recapture had much lower support, likely reflecting the change in effort over time resulting from an increase in field research and whale-watching in the study area.

Table 11. Model selection for Cormack-Jolly-Seber (CJS) mark-capture-recapture analyses of WCT encounter data in the Canadian Salish Sea (2005-2021).

Model	**AICc**	**Delta-AICc**
φ(t) p(t)	4467.81	0
φ (.) p(t)	4474.40	6.59
φ (t) p(.)	4508.22	40.41
φ (.) p(.)	4519.09	51.28

Tests 3.SR and 2.CT were both statistically significant, indicating additional, unexplained heterogeneity in the data. Transience in the dataset could be due to the presence of rare individuals (e.g., from the outer coast sub-population) or could be explained by the arbitrary nature of the study area. For instance, an individual or group could have been observed more often in the neighbouring US part of the Salish Sea rather than in its Canadian portion, and therefore would be considered missing from our dataset in a given year even though it might have been located in close proximity to the boundaries of the study area. Genuine trap-dependency is unlikely in free-ranging cetaceans but could reflect differing patterns of site fidelity among killer whale groups, or it could be a result of the gregarious nature of the individuals since the social structure of WCT is not taken into account in the analysis.

To account for a possible transience effect, survival was modeled separately for two classes of individuals based on their time-since-marking: one for the year after first capture and another for all subsequent years. The survival estimates for the second time-since-marking class can be considered as unbiased in relation to potential transience (Pradel et al. 1997). For the trap dependency effect, a time-varying covariate was added to allow recapture probabilities to vary based on whether or not the animal had been captured in the previous year (Schleimer et al. 2019).

Comparing the standard CJS, the transience model and the trap-dependency model showed that the resulting abundance estimates were almost identical. Since those estimates are used as an index of survival to investigate relationships with prey availability but are not used in the PVA itself, the CJS model was retained for simplicity.

In the selected CJS model, population size (N) increased from 117 in 2005 to 267 in 2021 (Figure 17). Survival ranged from 0.88 to 1.0 over the time series, with a mean of 0.96 (Figure 18), and recapture probabilities were relatively constant over the time series (Figure 19).

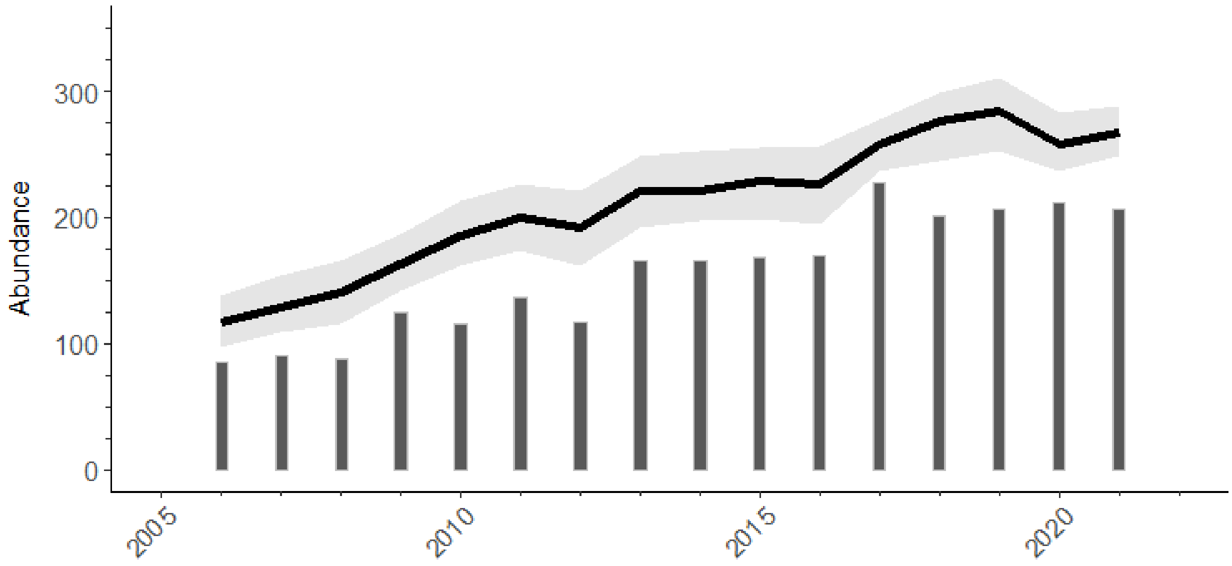


Figure 17. Annual abundance estimates for the Canadian Salish Sea subset from CJS model. Grey bars are the observed number of unique individuals in CSS study area; black line is the abundance estimated by the model with time-varying survival (with grey shaded area showing the 95% Confidence Interval from a non-parametric bootstrap.

3.2.1.2.1 Mortality and abundance in the PVA

Mortality estimates were calculated from the CJS survival rates and were used in the quantification of the reduced prey availability threat (Section 3.3). The estimates of annual abundance were used as the 'observed' population trend to compare threat scenario simulations.

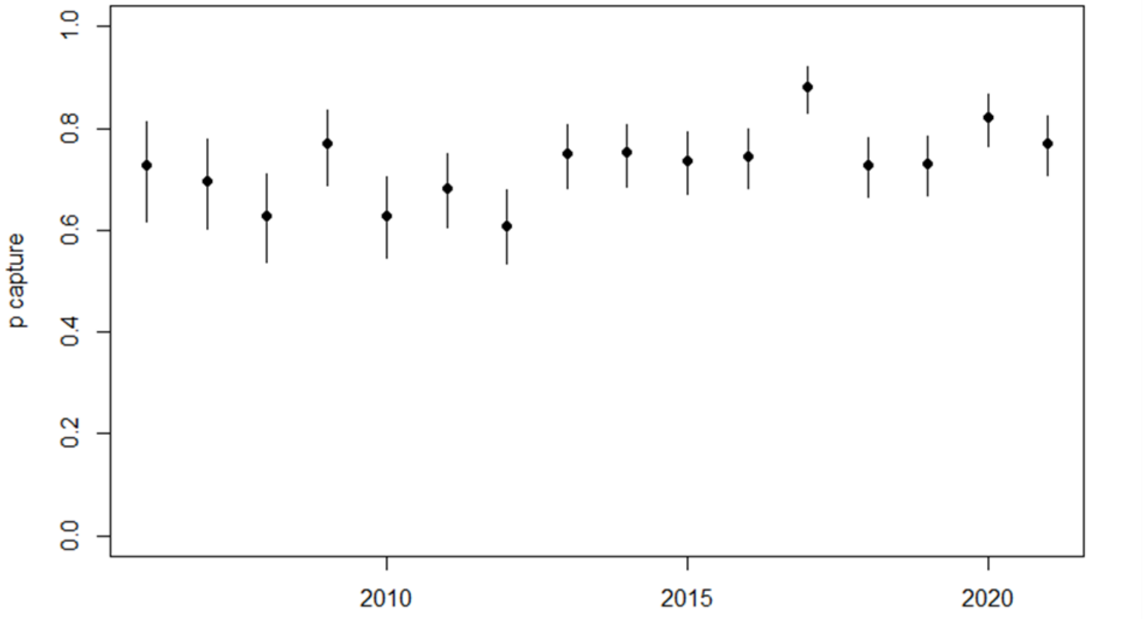


Figure 18. Annual survival rate for the Canadian Salish Sea subset estimated from the CJS model with time-varying survival and capture.

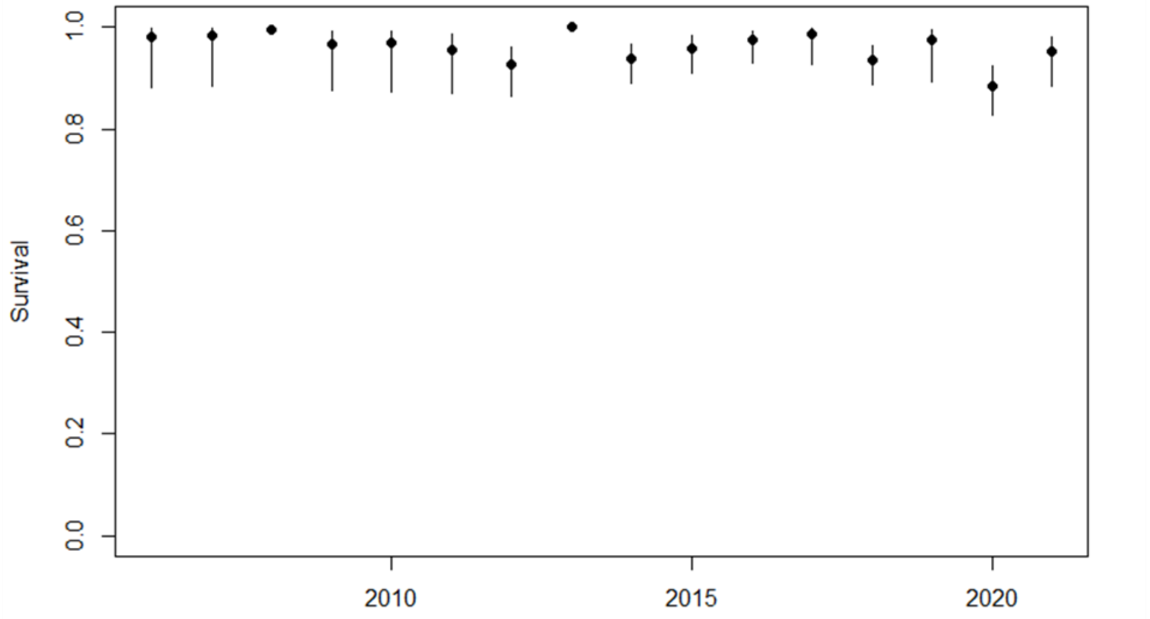


Figure 19. Annual probabilities of capture for the Canadian Salish Sea subset from CJS model with time-varying survival and capture.

3.2.1.3 Immigration

Examination of the discovery curves of individuals in the CSS dataset (Figure 20) shows that during the years 2005-2012, there was still a certain number of new non-calf individuals (i.e., individuals being observed in the CSS for the first time as juveniles or adults): 15 in 2005 and then an average of 4.9 per year. The observation of these new individuals suggests that increasing abundance in the study area was still driven by a combination of immigration and population growth (i.e., births). For the rest of the time series (2013-2021), the number of new non-calf individuals falls to 0-3 (average 1.4) per year and the discovery curve plateaus, indicating that over this period the increase in abundance was driven solely by births.

3.2.1.3.1 Immigration in the PVA

To account for immigration in the PVA, a fixed number of immigrants were applied using the Vortex ‘supplementation’ option in a manner that was informed by the raw data (Figure 20). Ten individuals (five females, five males) were added in the first two years and then decreasing by two immigrants every two years for the subsequent ten years, with no further immigrants for the rest of the time series.

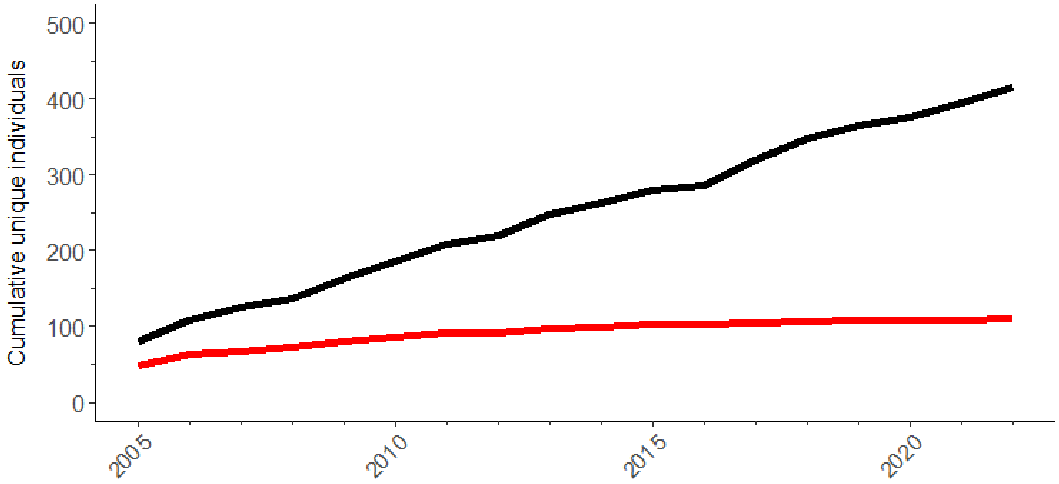


Figure 20. Discovery curve of new WCT individuals observed in the Canadian Salish Sea subset. Black line: all new individuals (calves and non-calves); red line: non-calves.

3.3 THREAT QUANTIFICATION

3.3.1 Disturbance

3.3.1.1 Noise

The effects of noise in the PVA model was quantified through a reduction in KW prey detection range (near seal haulouts) due to acoustic masking of prey sounds by vessel noise, using a vessel noise model, described in Burnham et al. (2023) that was applied to estimate masking of calls made by harbour seals.

3.3.2 Vessel noise

The shipping noise model used is based on the Range-dependent Acoustic Model (RAM) developed by Collins (1993) and refined by Aulanier et al. (2017). The model calculates the transmission losses as a function of range and direction from all vessel sources in the model domain for every time step in the model. The time step used was 30 minutes. The total received level (RL) at each grid point (approximately 400 by 400 m) in the model is estimated at up to 20 depths by summing the received levels from all available vessels in the domain. Inputs to the model include bathymetry (Haugerud 1999), sound speed profiles derived from temperature and salinity data obtained from the SalishSeaCast hydrodynamic model (Soontiens and Allen 2017; Soontiens et al. 2016); frequency dependent absorption predictions for pH 8 (Francois and Garrison 1982) and sound speeds for unconsolidated sediment (Hamilton 1980; Jensen et al. 2011). The vessels used as noise sources in the model were larger commercial vessels required to carry Class A Automatic Identification System (AIS) transceivers recorded by Canadian Coast Guard terrestrial receivers. The source levels for a range of vessel types (modeled in Burnham et al. 2023) were obtained from MacGillivray and Li (2018). The resulting vessel noise model output was RL at a single frequency of 125 Hz at all grid points in the model domain at each defined depth every 30 minutes for the period between February 2018 and March 2019. An example of the median RL for the period January to March 2019 at three different depths (7.5 m , 50 m and 100 m) are shown in Figure 21A.

Data from passive acoustic recorders with two years of continuous sound recordings (Feb 2018-March 2020) located at six sites in the Salish Sea (Figure 21A) were used to define the minimum observable noise levels as a function of frequency (Figure 21B). These levels represent the best possible conditions for killer whale calls and prey vocalizations to be detected.

3.3.2.1 Prey masking

Of the WCT prey species, only harbour seals had the required information about their calls to allow for masking calculations using the available model. The sound produced by harbour seals has a frequency range between 100 and 800 Hz with a source level (SL) of 140 dB re 1µPa @ 1m (Casey et al. 2016; Southall et al. 2019). From this information, assuming spherical spreading of the sound, and using the minimum ambient noise levels (NL) shown in Figure 21B it is possible to obtain a maximum detection range (R_{max}) using:

$$NL = SL - 20*\log(R_{max}) - \alpha*R_{max} \qquad \textit{Equation 3}$$

where NL, SL and α, represent the minimum ambient noise level, harbour seal source level, and frequency dependent sound absorption for the frequency range between 100 and 800 Hz, respectively. Using $NL = SL - 20*\log(R_{max}) - \alpha*R_{max}$ Equation 3, we find that the maximum detection range (R_{max}) for WCT whales to hear harbour seal calls in the absence of vessel noise is approximately 3.5 km. This assumes nothing about the sensitivity of BKW hearing in these frequencies, and the received noise levels (RL) are considered without weighting according to audiogram curves.

Using $NL = SL - 20*\log(R_{max}) - \alpha*R_{max}$ Equation 3 and replacing the minimum ambient noise level (NL) with the local RL obtained from the vessel noise model at a depth of 7.5 m, in the presence of commercial vessels, local detection ranges (R) can be calculated. The reduction in detection ranges can be presented as a percentage as $100*(R/R_{max})$ (method detailed in Burnham et al. 2023). Figure 22 shows this reduction using median modelled RL over four three-month periods in 2018 and 2019 for the Salish Sea.

Weighting the reduction in detection range by number of haulouts or by number of seals at each haulout, we can estimate the haulout and seals below the 50% detection range (Figure 23), i.e., haulouts and seals that would not be detected by whales further than 1.75 km away.

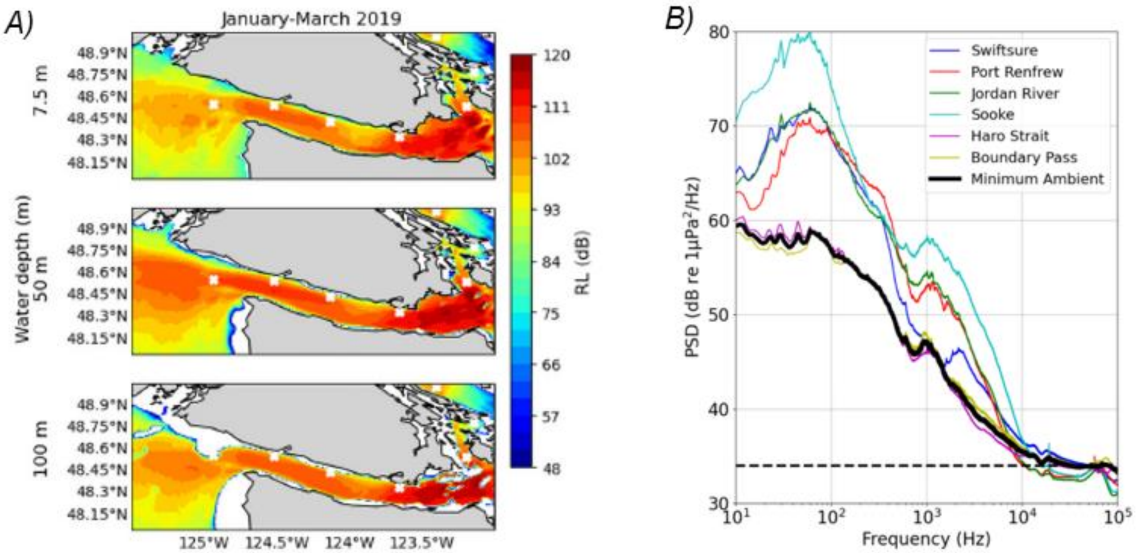


Figure 21. A) Median modeled Received Levels (RL) for January to March 2019 at three depths (7.5 m, 50 m, and 100 m) for the Salish Sea. White crosses mark Swiftsure Bank, Port Renfrew, Jordan River, Sooke, Haro Strait, and Boundary Pass. B) The minimum observed noise levels as function of frequency observed at six locations in the Salish Sea, adapted from Figure 3 in Burnham et al. (2023). The minimum frequency dependent noise levels used to estimate the maximum detection range is shown as the thick black line. The horizontal dashed line represents the minimum detectable noise floor. Harbour seal frequency range is 100-800Hz (140dB).

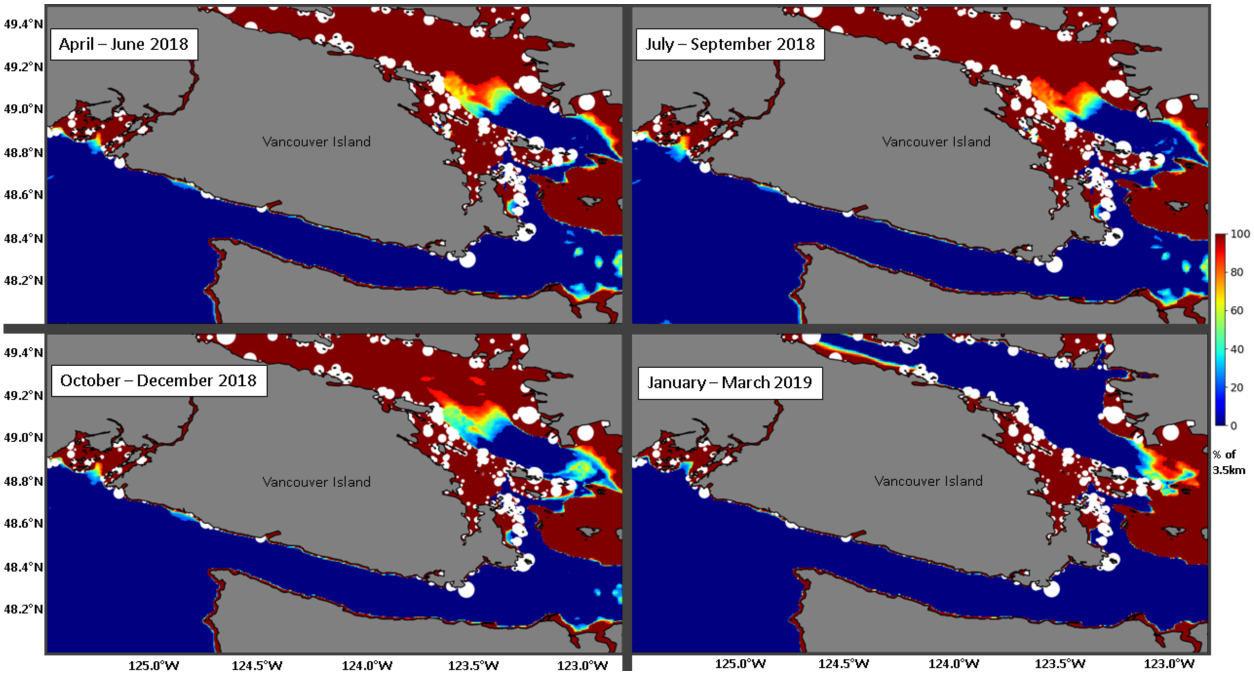


Figure 22. Harbour seal detection range as a percentage of maximum possible detection range due to commercial vessel noise estimated from three-month median values at a depth of 7.5 m. Dark red indicates no reduction in detection as a result of ship noise (full detection capacity, similar to that at minimum ambient conditions) and dark blue indicates full masking of prey noises by ship noise (i.e., harbour seal calls would be undetectable over background noise). Harbour seal haulouts, scaled by number of seals at each, are shown as white bubbles.

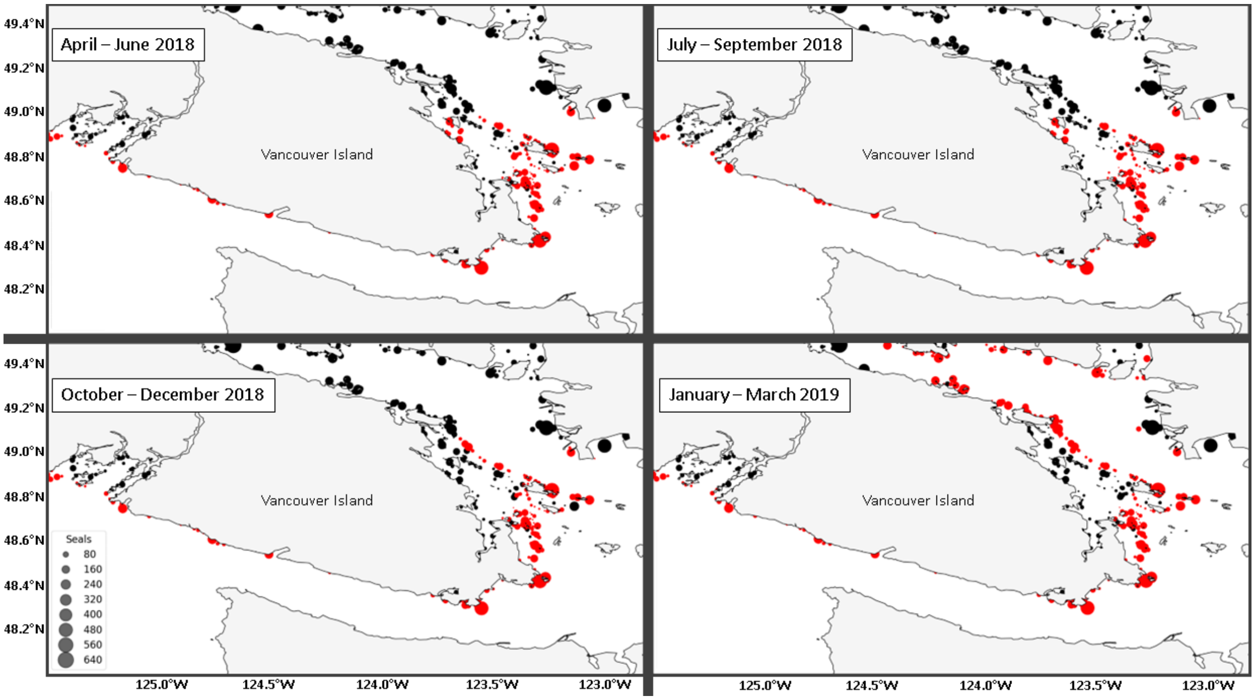


Figure 23. Harbour seal haulouts above (black dots) and below (red dots) the 50% detection range, relative to the 3.5 km maximum detection range due to median received level of ship noise masking their sounds. Haulouts are weighted by number of seals at each haulout, with larger circles indicating higher numbers of seals present (DFO 2022).

3.3.2.1.1 Noise in the PVA

The vessel noise model was used to estimate the noise levels that would result in a 50% reduction in prey detection from a 3.5 km maximum detection range, calculated using harbour seal vocalization parameters and the median received level of ship noise output from the model. Vessel noise affects WCT killer whales by increasing the difficulty to detect prey over background noise, forcing them to travel further to investigate possible food sources, and likely expend more efforts in finding and capturing prey to meet energetic requirements. The proportion of haulouts and harbour seals below this 50% detection threshold was 38.6% and 46.3%, respectively (Table 12). Under the assumption that the effect on demographic rates of reduced feeding activity is the same as a comparable reduction in prey (i.e., no behavioural compensation by killer whales), Lusseau et al. (2009) observed a 25% reduction in feeding activity when boats were present for NRKW.

Table 12. Percentage of harbour seals and harbour seal haulouts that were estimated to be below 50% communication range in three-month intervals, and grand mean.

Season	**Seals (%)**	**Haulouts (%)**
April – June	41.8	35.1
Jul - Sept	42.1	35.5
Oct – Dec	39.4	33.6

Season	Seals (%)	Haulouts (%)
Jan – Mar	61.8	50.1
Mean	46.3	38.6

The noise effect on prey availability was therefore estimated as a reduction in prey availability; reduction in feeding as a result of vessel noise (25%), multiplied by the reduction in prey detection range (50%), multiplied by the proportion of haulouts (or seals) affected (38.6% for haulouts and 46.3% for seals). This results in a 4.8% reduction in prey when calculated for haulouts and 5.8% for seals. This small reduction is applied directly to the prey index value in the noise scenario, so that a prey index value of 1.40 would be reduced to 1.33. Therefore, the effect of acoustic disturbance was modelled as a reduction in prey availability, and mediated by the carrying capacity effect on mortality. See Section 3.3.3. on Prey availability for details on how the reduced prey availability threat scenario was modelled.

In the PVA model, we tested a noise threat scenario for haulouts and for seals as well as a noise x prey threshold scenario (effect of noise applied under low prey abundance). The interaction of prey and noise was added through a threshold effect: when the prey index fell below 1.0 (mean prey level), the effect of noise is implemented in the model. The threshold effect is the same as that applied in the resident killer whale PVA (Murray et al. 2019), under the theory that disturbance may have a higher influence under low prey conditions.

3.3.2.2 Strikes

Over the 15 year period from 2004 to 2018 (see Table 6), three deaths in Bigg's killer whales of known were suspected to be due to trauma (likely vessel strike), approximately 0.2 per year (Lee et al. 2023; Raverty et al. 2020). There were another eight deaths in this ecotype where causes could not be determined (Lee et al. 2023; Raverty et al. 2020). If these undetermined fatalities are ascribed to vessel strike, an extreme or precautionary estimate of vessel strike probability is 0.73 per year.

3.3.2.2.1 Strikes in the PVA

Fatal vessel strikes were modelled as a mortality event in Vortex PVA software (Lacy and Pollak 2023) in two scenarios, with an annual probability of vessel strike = 0.2 or extreme = 0.73. The annual mortality event was assigned evenly between adult males and females. Changes in the frequency of vessel transits and the characteristics of ships (e.g., quieter ships may increase strike risk) could affect this probability in the future.

3.3.3 Contaminants

Killer whales are particularly vulnerable to legacy organic pollutants, including polychlorinated biphenyls (PCBs) and organochlorine pesticides (OCPs) given their long-life span, high trophic position, and limited ability to metabolize these persistent compounds. Despite the phase-out of PCBs under the international Stockholm Convention on Persistent Organic Pollutants (POPs), they continue to persist in the marine environment and will likely pose a health risk to Bigg's killer whales for a significant period creating a lag in impact (Hickie et al. 2007). A PCB accumulation/depuration model has been developed to link PCB levels to calf mortality in cetaceans (Hall et al. 2006; Hall et al. 2018). The logistic regression model developed in these studies (Hall et al. 2018) predicts calf survival based on the maternal PCB level. The levels of PCBs in killer whales measured from blubber samples in the field have been found to vary greatly between sexes and through time (Figure 24; Desforges et al. 2018; Guy 2018; Ross et al. 2013).

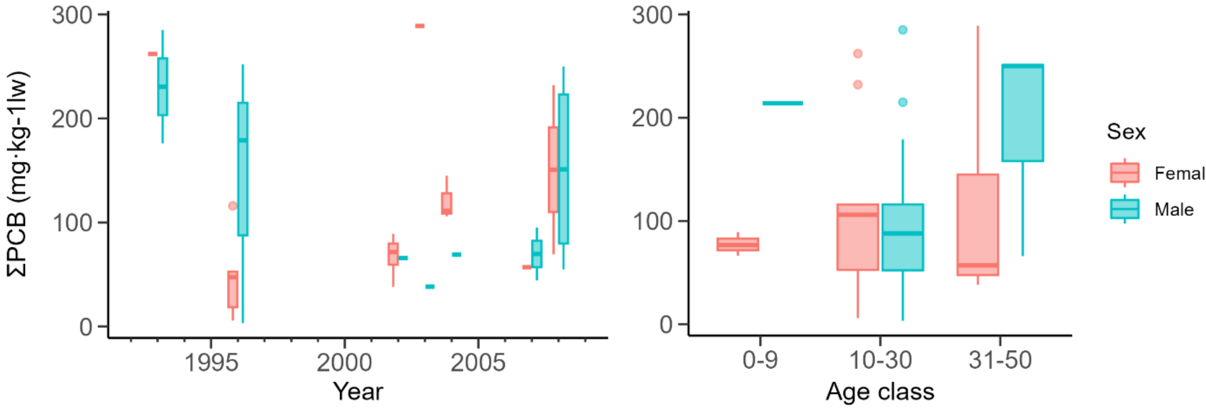


Figure 24. PCB contamination levels (mg kg^{-1} lw) in 33 West Coast Transient killer whale tissue samples by year (left) and age (right).

Data currently available for PCB levels measured from WCT killer whales with known sex was limited (n=27; Guy 2018; Ross et al. 2013), and the variability in PCB levels among individuals makes it difficult to parameterise the scenarios (Figure 24). To align with the Canadian Salish Sea subset of WCT killer whales, four PCB data points from individuals that were not part of this population subset were removed. PCB data for the remaining 29 individuals were used to calculate mean PCB levels for males and females by decade (1991-2000; and 2001-2010) to use in the simulation modelling (Table 13).

A correction factor of 1.75x was applied to the mean PCB concentration assigned to the modelled population to capture the contribution and associated risk of other POPs, including legacy OCPs. This 1.75x factor was derived from an endocrine disruption risk-based quotient for local harbour seals (*Phoca vitulina*) (Mos et al. 2010), a species that has been used previously as a surrogate to characterize Northeastern Pacific killer whale contaminant levels and risk. The correction factor was applied as a multiplicative factor to the PCB concentration value to conservatively estimate the effects of additional contaminants, referred to as 'PCB+' (Table 13).

The PCB+ model simulated the accumulation of PCBs in individuals over time, based on a set maternal transfer and accumulation rate. Females depurate (offload) an estimated 77% of PCB load to each calf during calving and nursing (Hall et al. 2006; Hall et al. 2018). The estimated accumulation rate can vary based on sources of PCBs in the environment and prey items (Guy 2018).

3.3.3.1.1 PCBs in the PVA

In Vortex, PCB+ contamination model scenarios were run by assigning the adjusted (1.75x) PCB+ levels (1991-2000; Table 13) to the starting model individuals. Previous simulation modeling tested accumulation rates ranging from 1, 2 or 6 mg kg^{-1} lw / year, with NRKW best explained by 1 mg kg^{-1} lw / year and SRKW by 6 mg kg^{-1} lw / year (Hall et al. 2018; Murray et al. 2019). Therefore, accumulation rate was set to either 1, 2, or 6 mg kg^{-1} lw / year in the current modeling. The accumulation and depuration model varied the PCB+ concentration in the model individuals by the accumulation rate through time, transferring a proportion of the PCB+ load from mother to calf. The effect was applied when calf PCB+ load exceeded the threshold and simulated a mortality event for that calf. The resulting simulated PCB+ concentrations for males and females were then compared to the observed tissue sample concentrations (Table 13) to determine the best accumulation rate for the Canadian Salish Sea subset.

Table 13. Mean PCB concentration values for males and females within the WCT population's Canadian Salish Sea subset (Ross et al., 2013; in Guy 2018). Individual PCB values were cross-referenced to ensure they only included values from members of the Canadian Salish Sea subset population (N=4), values without an assigned sex were also removed (N=4).

Scenario	**Time range**	**Female Mean (mg kg-1 lw)**	**Female SE (mg kg-1 lw)**	**Female sample size (N)**	**Male Mean (mg kg-1 lw)**	**Male SE (mg kg-1 lw)**	**Male sample size (N)**	**Source**
Original PCB	1991-2000	99.4	43.7	5	170.3	43.4	6	*Ross et al. 2013;* Guy 2018
Original PCB	2001-2010	116.9	26.0	10	104.0	28.8	8	Ross et al. 2013; Guy 2018
PCB+	1991-2000	173.9	76.4	5	298.1	76.0	6	Mos et al. 2010
PCB+	2001-2010	204.5	45.5	10	181.9	50.4	8	*Mos et al. 2010*

3.3.4 Prey availability

3.3.4.1 Trends in prey abundance

The amount of prey present (abundance) has been distinguished from how accessible prey are to the predator population (availability). The majority of the WCT diet is made up of four species: harbour seal, harbour porpoise, Steller sea lion, and Dall's porpoise (Ford et al. 2007; Ford et al. 1998). The relative proportions of each prey species were updated and estimated for the Canadian portion of the Salish Sea (

Table 14; DFO-CRP and Bay Cetology unpublished data). In order to analyse the relationship between WCT vital rates and prey abundance, concurrent time series data are required. Time series data are only available for harbour seals, Steller sea lions and harbour porpoises. These three prey species dominate the observed predation events (

Table 14). While Dall's porpoise makes up 2% of the diet, there is no trend information available (Houghton et al. 2015) and so it could not be included in the PVA.

Harbour seal abundance estimates were sourced from DFO (2022) and were available for 1965-2019, and projected forward to 2021. Most of the stock is in the Strait of Georgia region (42%). Coastwide abundance was used to calculate the trend in harbour seal population size as an estimate of prey available to WCT since the whales are highly mobile and can access prey across their range. The harbour seal index was calculated by standardising the coastwide abundance estimate data to the mean for the full available time range (1965-2019; Figure 25A).

Steller sea lion abundance estimates were sourced from DFO (2021a) using the estimates for the entire coast of BC from 1971-2017. Steller sea lions have not been surveyed each year, so years with missing data were interpolated using a polynomial model fit in accordance with DFO (2021a), and projected forward to 2021 in RStudio (2023.12.0+369). Abundance estimates were converted to an index by standardising data to the mean abundance from all available years (1971-2017; Figure 25B).

Harbour porpoise abundance data was collected by Washington Department of Fish and Wildlife from 1994 – 2023 (Matthew Hamer and Joseph Evenson, unpublished data). Abundance estimates were combined from three regions of interest: Juan de Fuca Strait, San Juan Islands, and the southern Strait of Georgia. The linear growth trend (Adjusted R-squared: 0.9076) from 1994 to 2004 was used to extrapolate values for 1985 to 1993 using RStudio (2023.12.0+369). Negative densities were changed to zero; note that these zeroes do not indicate extirpation but a density below detection by the surveys in the area of interest. To create the index, each year’s abundance value was standardised by the overall mean of the time series (1985-2023; Figure 25C).

The combined (or composite) prey index used the three individual prey indices weighted by the proportion that they have been found in WCT diet (

Table 14, Figure 25D). Time series data on abundance were only available for three prey species, so proportion was recalculated to sum to one for the relative proportion of the diet for each of harbour seal, harbour porpoise, and Steller sea lion in the combined index (

Table 14).

Table 14. Proportions of marine mammals in WCT Canadian Salish Sea predation events (DFO-CRP and Bay Cetology unpublished data), with recalculated proportions used in the PVA.

Marine mammal prey species	Percent of predation events observed	Recalculated proportion
Harbour seal	66%	0.702
Harbour porpoise	17%	0.181
Steller sea lion	11%	0.117
Dall’s porpoise	2%	-
Other	4%	-

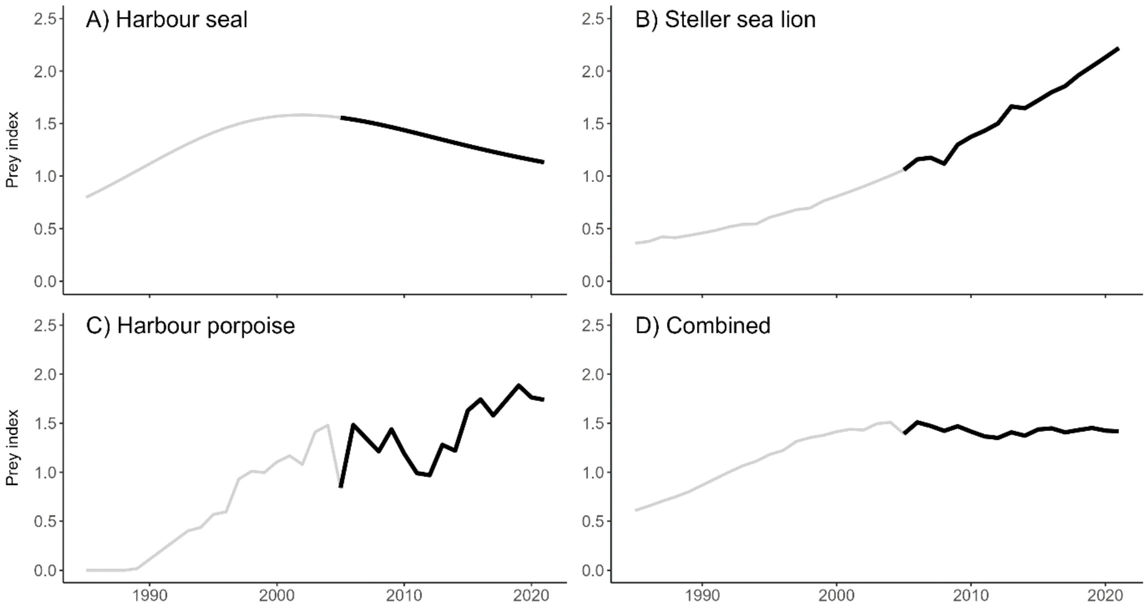


Figure 25. Prey indices for a) Harbour seal, b) Steller sea lion, c) Harbour porpoise, and d) Combined prey, with the range used in the PVA highlighted in black (2005-2021).

3.3.4.2 Prey abundance and vital rates

The relationships between prey abundance (described by the combined prey index, Figure 25D) and the CSS mortality and fecundity rates were examined. Over the years 2006-2021, the CSS population size (Figure 26A) increased while the combined prey index remained stable (Figure 26C), raising questions as to whether prey abundance could become limiting for killer whale population growth in the CSS subset.

Population size and vital rates before the impact of human activities are unknown. Carrying capacity is therefore difficult to estimate, but is likely influenced by a combination of prey abundance, habitat requirements, and limits imposed by the social structure of the populations (Murray et al. 2019). An analysis of the CSS population growth rates (Figure 26B) showed dampening oscillations which suggests that the WCT population might be approaching carrying capacity. The combined prey index (Figure 26C) shows decreasing prey population growth rates over time (Figure 26D). Together, the evidence suggests that WCT carrying capacity could be limited by prey production for the region.

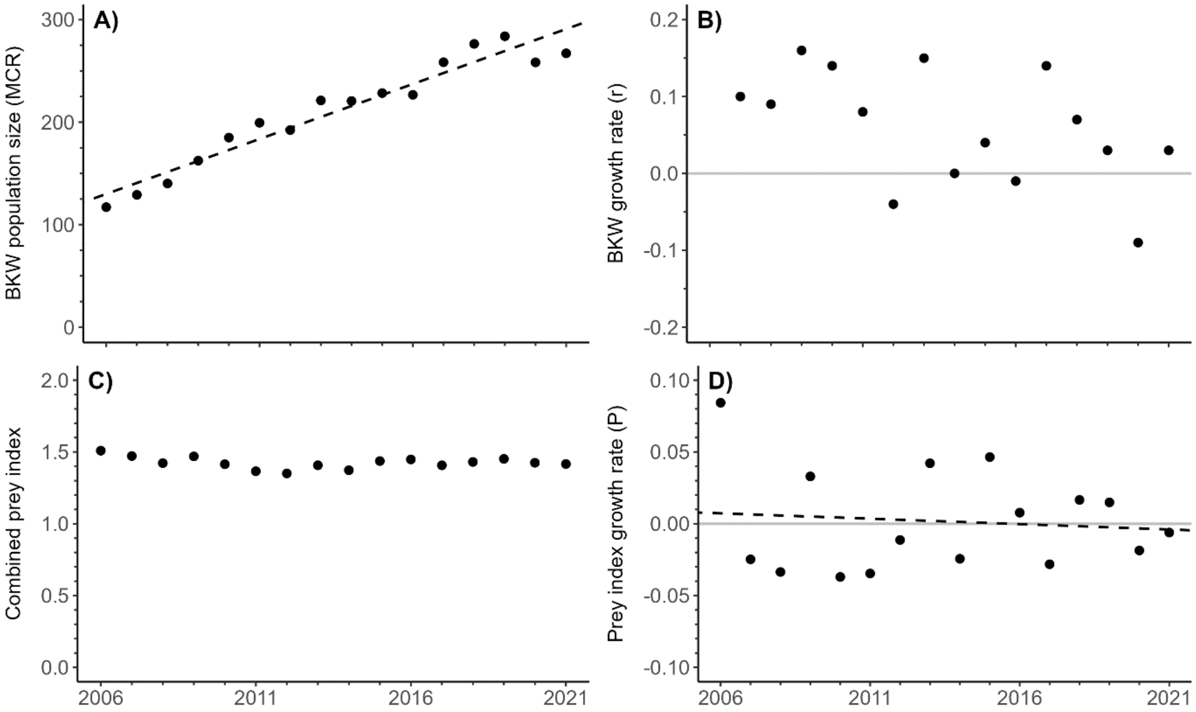


Figure 26. Analysis of A) CSS (BKW) abundance (N-CJS-MCR), B) CSS growth rate (r), C) combined prey index, and D) combined index growth rate (P). Dashed lines are fitted linear models, solid grey lines are for reference at zero.

We investigated how fecundity and mortality changed as abundance of WCT in the CSS increased, using both linear (GLM function in R) and non-linear regressions (gam function). There was no clear relationship between fecundity and abundance (Figure 27 left), and a weak linear increase in mortality as abundance increased (p = 0.10, Figure 27 right). Therefore, we implemented a density-dependent relationship in the PVA between mortality and abundance (expressed as a fraction of the carrying capacity).

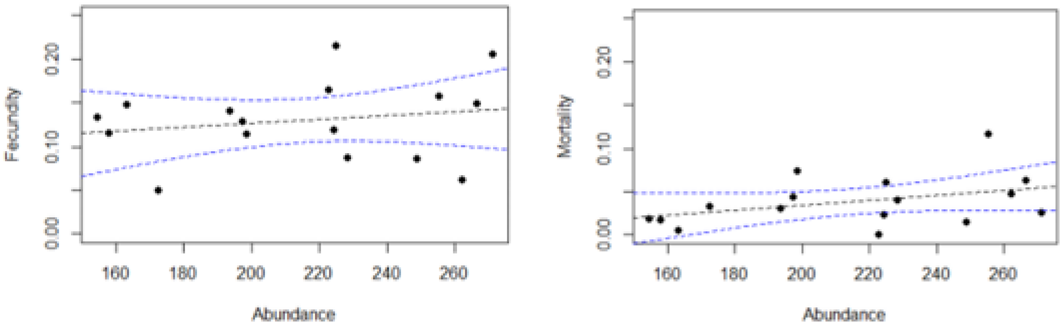


Figure 27. Relationship between CSS abundance and fecundity (left) and mortality (right). Black dashed line: linear regression. Blue dashed lines: 95% confidence interval around the regression line.

If we assume that carrying capacity in the CSS is proportional to the combined prey index, and since the prey index has been stable over the period 2005-2021, we can assume that K has been constant over this same range of years. Therefore, we attempted to estimate the carrying capacity K from the logistic density-dependence model (Equation 4),

$$r_t = r_{max}\left(1 - \frac{N_t}{K}\right)$$ *Equation 4*

with r_t being the realized growth rate at time *t*, r_{max} the intrinsic growth rate and N_t the abundance at time t. From this equation, we can estimate *K* from the observed growth rates for given levels of abundance (for this, we assumed that r_{max} was 5.5%, which is the rate observed under the baseline projections using the SARKW data). Observed growth rates (Figure 26B) were unrealistically high during the first half of the study period, likely due to the documented immigration of non-calf individuals, but were 2.6% on average from 2013-2021. Using the growth rates and abundance estimates for the last three years of the time series (2019-2021) yielded an estimate of K = 330. Using the averages of r_t and N_t over the entire 2013-2021 period yields K = 459. In other words, the slowing down of the growth rate in the most recent three years would suggest that the group of whales using the CSS are close to the carrying capacity of the area (about 270 whales vs 330 capacity), whereas trying to determine K from an earlier time period would suggest there is still more room for growth.

Vortex uses Equation 5 to model the effect of density dependence on survival:

$$S_N = \left(S_0 - \left((S_0 - S_K)\frac{N}{K}\right)\right)\frac{N}{N+A}$$ *Equation 5*

in which S_N is the survival rate when the population size is *N*, S_K is the survival when the population is at carrying capacity *K*, and S_0 is the survival at low densities when there is no Allee effect (A). Essentially, this equation applies a modifier to the baseline survival rate based on the ratio between current abundance and carrying capacity.

We used S_0 = 0.98, which is the survival rate of the SARKW (i.e., the highest observed survival rate, used in the baseline scenario). Since we do not know whether the population is currently at or close to carrying capacity, we do not have exact values for what survival would be at K. Therefore, to inform S_K, we considered two approaches. The first one was to infer the value of the mortality rate at the highest observed abundance from the linear relationship shown in Figure 27, i.e., 5.02% (and therefore S_K = 0.95). The second approach was to calculate the mortality that would result in a zero growth rate at carrying capacity by cancelling the natality among the CSS individuals. According to Equation 6, we know that:

$$r = b - d = PF - D$$ *Equation 6*

where *r* is the growth rate, *b* the birth rate, *d* is the mortality rate, *F* is the proportion of reproductive females in the population and *P* is the fecundity rate. Reorganizing this equation to find the mortality rate when the number of births equal the number of deaths, and using the observed proportion of females of 35% over the years 2013-2021 and the mean annual fecundity rate of 10% over the same period, we find an alternative value of mortality at *K* of 0.035 (i.e., S_K = 0.965).

3.3.4.2.1 Prey availability in the PVA

The resulting *K* and mortality values (S_0, S_K) were used in Vortex to define the density-dependent mortality rate as a function of the abundance relative to carrying capacity in any given year (using Equation 5). For future projections, some scenarios assume that carrying capacity will vary with time (e.g., an increase or decrease in prey abundance). Therefore, to relate the prey index (P_t) to a time-varying carrying capacity (K_t) for future projections, we used the largest value of the observed prey index during the 2005-2021 period, such that K_t = 218 x P_t if we assume K_{2021} = 330, and K_t= 304 x P_t if K_{2021} = 459. Combinations of plausible values for mortality and carrying capacity were tested in Vortex (Table 15) and the best fit values were used in the cumulative effects model.

Table 15. Mortality rates and carrying capacity (K) values tested in the prey availability threat scenarios in Vortex.

Mortality rates	Carrying capacity (K) (number of individuals)
0.035	330
0.050	459

3.3.4.3 Threat interactions

Two interactions between threats were explored in Vortex scenario simulations. The interaction of prey and noise was explored by adding a threshold effect of prey index below 1.0 (mean prey level) to implement the effect of noise. This threshold effect was applied in the RKW PVA under the theory that disturbance may have a higher influence under low prey conditions. Similarly, a prey and PCB contaminants interaction used a threshold effect of prey below 1.0 to implement the effect of PCBs. This interaction represents the possibility that contaminants are metabolised and have an effect on calf mortality only under low weight conditions when prey has been low. Threat interactions used a carrying capacity of 459 whales, with a 96.5% probability of survival at carrying capacity.

3.3.5 Cumulative threats

A cumulative effects scenario was created to include all four threats (Noise, Strikes, PCB Contamination, and Prey availability) with a probability of survival at carrying capacity of 96.5%. As in Murray et al. (2019; 2021), the individual threat scenarios, the parameters from the individual threat scenarios with the best fit and supporting evidence were applied to model the cumulative threats. Threat interactions were included where the fit of the scenario and knowledge to include it were highest.

4 PVA RESULTS

4.1 INDIVIDUAL THREAT SCENARIOS

The individual threat models were simulated in Vortex separately and then combined into a cumulative effects model that includes all threats (Figure 28). Each threat scenario is presented with the baseline scenario (SARKW rates) for comparison purposes. The Canadian Salish Sea population subset used in the PVA model was modeled over time using the CJS MCR model as a comparison to the baseline, and is included in the figures as “CJS MCR” (Figure 29A).The modeled growth rates, resulting population sizes, and probability of extinction for each of the individual scenarios are presented in Figure 29 and Table 17.

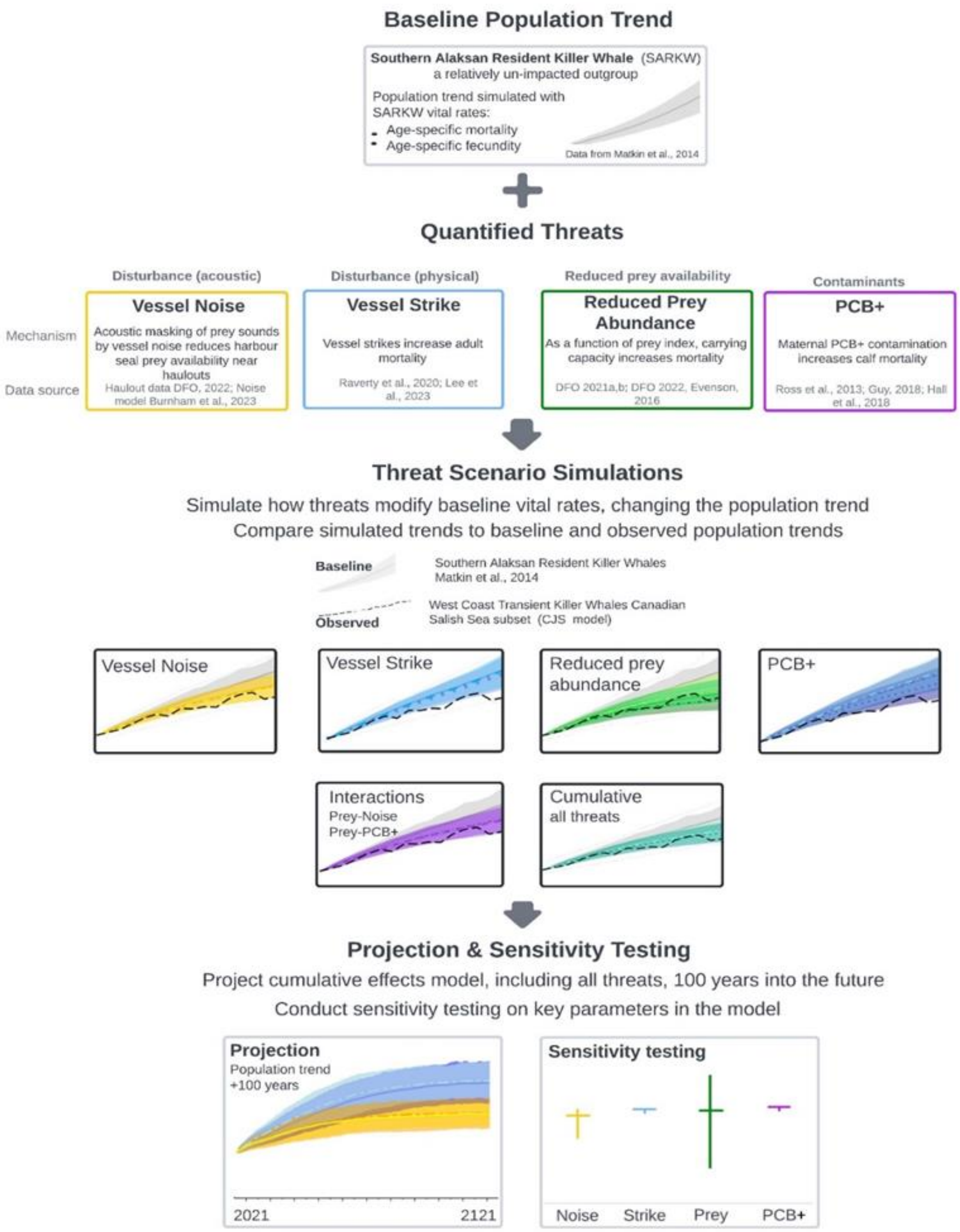


Figure 28. Detailed flow chart of the PVA modelling approach, source data, threats, and how the threats were represented in the PVA.

The effect of acoustic disturbance was explored through two scenarios for the reduction in detection of harbour seals or harbour seal haulouts (Figure 29B). The seal and haulout scenarios were similar to each other and overlapped with the observed population trend.

Vessel strikes were modelled under two scenarios of annual probability of fatal vessel strike, 0.2 from the estimated number of strikes, and 0.73, as an extreme estimate (Figure 29C). Vessel strike caused only a small depression in the population trend through time for both scenarios and overlapped with the baseline scenario trend.

The influence of density dependence (via prey availability) on annual mortality was implemented with carrying capacity (K) as a function of the prey index (Figure 29D). This threat had the largest effect on the simulated population, overlapping with the observed population trend. Different values of K and constant mortality were applied and the values that most closely aligned with the observed trend were 3.5% mortality and K = 459.

The PCB+ contamination threat showed some evidence of population depression in the later years but mostly overlapped with the baseline scenario trend. Multiple scenarios for contaminant accumulation rate were tested (1, 2, and 6 mg kg^{-1} lw per year). The best match between simulated and sampled PCB concentrations was the scenario with 6 mg kg^{-1} lw per year accumulation rate (Figure 29E, Table 16). The 6 mg kg^{-1} lw per year scenario PCB+ concentrations in the model aligned with the female field samples but overestimated the male field sample concentrations (Table 16).

Two interactions between threats were tested in the PVA (Figure 29F). The interaction effect of prey and PCBs had little effect on the simulated population, overlapping with the baseline simulation. This is a consequence of the threshold effect applied; the prey availability remained high during the modeled time frame so the PCB effect would not have been triggered. The interaction of noise and prey had a substantial effect, overlapping the observed population size. Again, this threshold effect would not have been triggered during a period of time with high prey availability, but the density dependence of prey remains active and depressed the population trend close to the observed.

All of the individual and interaction scenarios showed positive growth rates, ranging from the low 3.4% for prey effects on carrying capacity (Prey-K) to the highest, 6.82% for PCB+ 1mg (Table 17).

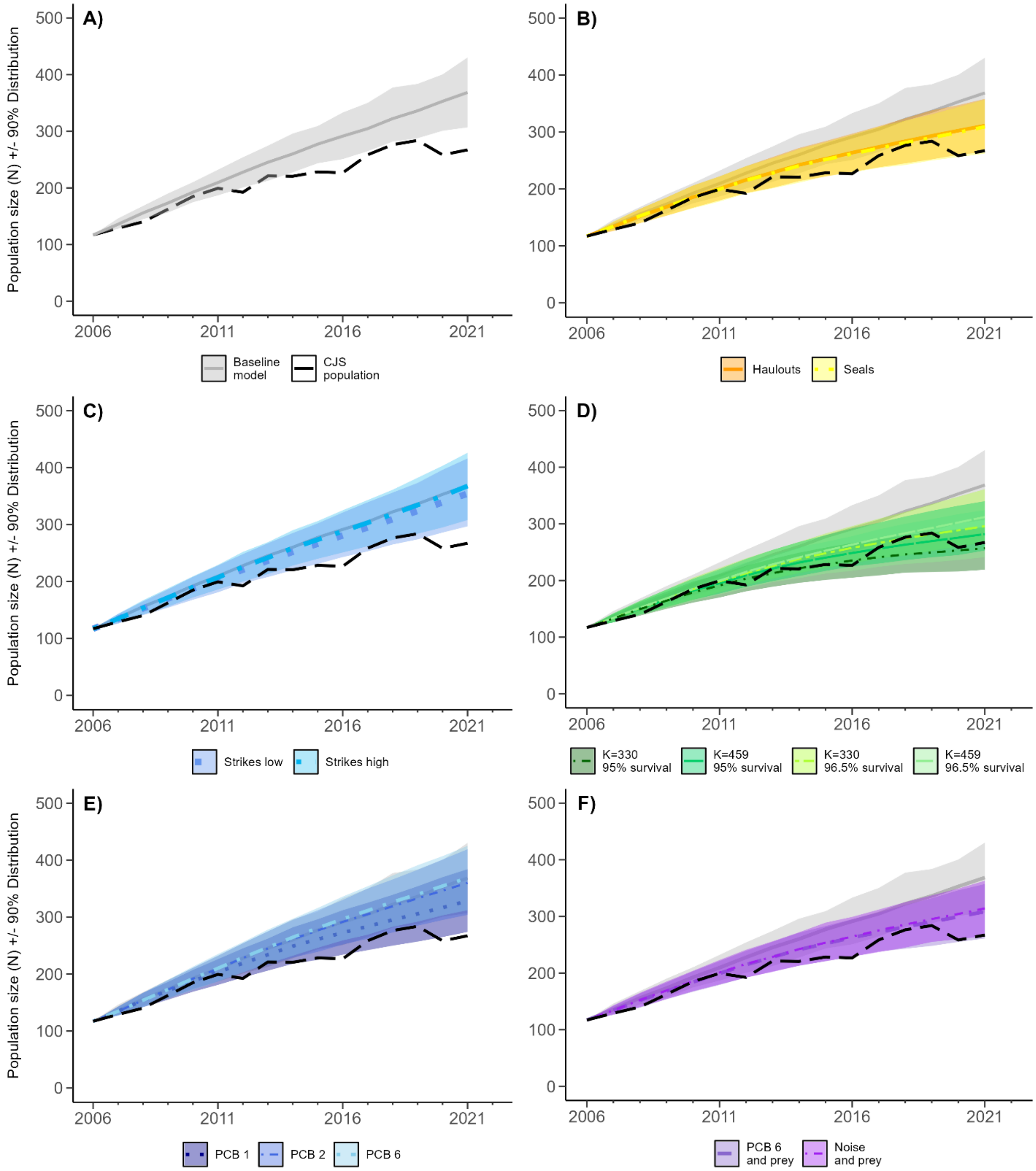


Figure 29. Results of the CSS subset PVA with the 90% distribution around the model mean. Panel A) shows the baseline model (grey) and the population trend from the CJS mark-capture-recapture (MCR) modelling (black dashed), which are included in each of the panels, B) acoustic disturbance modelled on impact on masking on number of seals and on seal haulouts, C) physical disturbance with ship strike risk, both high and low, D) impact of prey on carrying capacity via mortality, E) contaminants impacts at three levels of accumulation, and F) interactions between contaminants and prey, and noise and prey.

Table 16. Contaminant (PCB+) accumulation rates (mg kg^{-1} lw) tested in the PVA and their resulting mean concentrations in the model (including the correction factor) at year 2021.

Scenario	**Male (mg kg^{-1} lw) (± SD)**	**Female (mg kg^{-1} lw) (± SD)**
PCB+ 1	227.29 (13.28)	76.65 (8.08)
PCB+ 2	255.38 (13.40)	94.92 (9.49)
PCB+ 6	367.71 (14.19)	162.96 (4.56)
Observed values, including correction factor (2001-2010)	204.5	181.9

Table 17. PVA results for each scenario, including the stochastic growth rate (r), population size (N) at model year 2021, and overall extinction risk.

Scenario	**Growth rate (r) + SD**	**Population size (N2021) + SD**	**Probability of extinction (P0)**
Baseline	0.0770 (SD = 0.0517)	373.29 (SD = 37.03)	0
Noise - seals	0.0643 (SD = 0.0565)	309.01 (SD = 36.86)	0
Noise - haulouts	0.0648 (SD = 0.057)	311.12 (SD = 35.81)	0
Strike risk – 0.20	0.0758 (SD = 0.0539)	367.68 (SD = 46.15)	0
Strike risk – 0.73	0.0733 (SD = 0.0538)	354.31 (SD = 46.54)	0
Prey – K mortality (5, 330)	0.0520 (SD = 0.0588)	256.92 (SD = 29.91)	0
Prey – K mortality (3.5, 330)	0.0614 (SD = 0.0579)	295.84 (SD = 34.47)	0
Prey – K mortality (5, 459)	0.0582 (SD = 0.0573)	281.91 (SD = 32.30)	0
Prey – K mortality (3.5, 459)	0.0649 (SD = 0.0559)	312.05 (SD = 36.35)	0
PCB+ – 1 mg/year	0.0760 (SD = 0.0554)	368.61 (SD = 42.91)	0

Scenario	Growth rate (r) + SD	Population size (N2021) + SD	Probability of extinction (P0)
PCB+ – 2 mg/year	0.0744 (SD = 0.0546)	360.06 (SD = 44.81)	0
PCB+ – 6 mg/year	0.0682 (SD = 0.0523)	328.58 (SD = 43.95)	0
Noise-Prey interaction	0.0654 (SD = 0.0565)	314.17 (SD = 37.07)	0
PCB+-Prey interaction	0.0641 (SD = 0.0569)	308.41 (SD = 37.39)	0
PreyK-noise-strike-pcb+ (K=459)	0.0642 (SD = 0.0569)	308.57 (SD = 37.21)	0
PreyK-noise-strike-pcb+ (K=330)	0.0613 (SD = 0.0579)	295.62 (SD = 36.06)	0

4.2 CUMULATIVE EFFECTS MODEL

The cumulative effects scenario included all four threats and the most evidence-based parameters for each: Prey availability effect on carrying capacity (K) mortality, acoustic disturbance effect on seals through prey availability, vessel strikes with 0.2 probability, and a threshold interaction between prey availability and PCB contaminants (Table 18). The cumulative model approaches the observed population trend closely and the WCT population size fits within the 90% distribution of the model estimates (Figure 30;Table 20).

Table 18. Details of the threat model parameters used in the cumulative effects model.

Threat model	WCT BKW CSS
Prey Availability	Carrying capacity is a function of prey index, which causes increased mortality as population approaches K. K value = 459 and mortality at K = 0.035
Vessel Strikes	0.2 probability of fatal vessel strike per year (Lee et al. 2023; Raverty et al. 2020)
Noise	Prey availability is expected to be reduced by 4.8% due to acoustic masking of prey sounds across haulouts (38.8% x 50% x 25%).
Contaminants (PCB+)	Calf survival based on maternal PCB+ concentration (Hall et al 2018). Females 173.9 mg/kg; Males 298.1 mg/kg; Accumulation rate 6 mg/kg/y; Depuration rate 0.77. When prey availability was low (less than mean index), the PCB+ impact was applied (threshold effect).

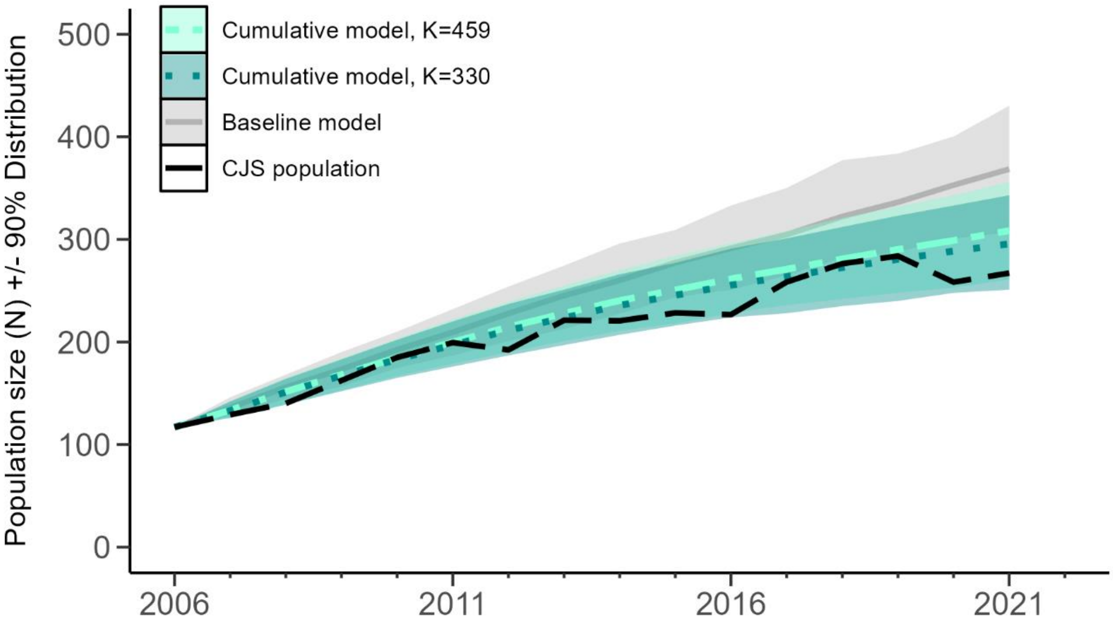


Figure 30. Cumulative effects scenario including all threats (Prey effect on carrying capacity via mortality, noise, strikes, and prey-PCB interaction, with probability of survival at carrying capacity of 96.5%), showing the model mean with the 90% distribution. Baseline model (grey) and CJS mark-capture-recapture model trend (black dashed line) included for comparison.

4.3 MODEL INSPECTION AND VALIDATION

Inspection of the results of the individual, interaction, and cumulative scenarios showed that the scenarios with the prey abundance effect on carrying capacity via mortality and the cumulative scenarios were closest to the observed (CJS) trend (Figure 29, Figure 30). Five scenarios exhibited the lowest RMSE values, which are indicative of the model scenarios better replicating the observed population trend (Hodson 2022). These five scenarios included three of the four density-dependence scenarios and both cumulative scenarios (Table 19).

Table 19. Root mean square error (RMSE) model validation results to quantify the differences between the threat scenarios (predicted) and the CJS MCR estimates (observed). Low RMSE values are indicative of the model scenarios better replicating the observed population trend.

Model	**RMSE**
Density dependence (K = 459), survival at K of 95%	11.40
Density dependence (K = 330), survival at K of 95%	13.73
Cumulative, with density dependence (K = 330), survival at K of 96.5%	15.15
Density dependence (K = 330), survival at K of 96.5%	15.97
Cumulative, with density dependence (K = 459), survival at K of 96.5%	20.02
Noise (based on seals)	20.30
Interaction: prey density dependence (K=459, survival at K of 96.5%) and PCB	20.40
Density dependence (K = 459), survival at K of 96.5%	21.57
Noise (based on haulouts)	21.62
Interaction: prey density dependence (K=459, survival at K of 96.5%) and noise	22.63
PCB (6 mg kg^{-1} lw / year)	29.36
Physical disturbance; 0.73 probability of fatal vessel strike	39.38
PCB (2 mg kg^{-1} lw / year)	45.64
Physical disturbance; 0.2 probability of fatal vessel strike	46.09
Baseline	47.94
PCB (1 mg kg^{-1} lw / year)	50.08

4.4 FUTURE PROJECTIONS

The projection of the cumulative effects model 100 years into the future used two values for carrying capacity (K), 330 and 459 individuals, with probability of survival at K of 96.5%. The differing prey trends had only small effects in the scenarios. The carrying capacity had the biggest effect on the simulated population trends. The higher carrying capacity scenarios showed a steady increase in the population over the first twenty years and then stable population size of approximately 400 individuals through the rest of the simulation (Figure 31; Table 17). The lower carrying capacity scenarios showed a small increase in the first few years and then a steady population size for the remainder of the simulation, approximately 300 individuals (Figure 31; Table 17). This means that for this subset of the WCT population in the Canadian Salish Sea, prey-mediated carrying capacity is the most important predictor of population size, distinguished in the first ten years of the simulation.

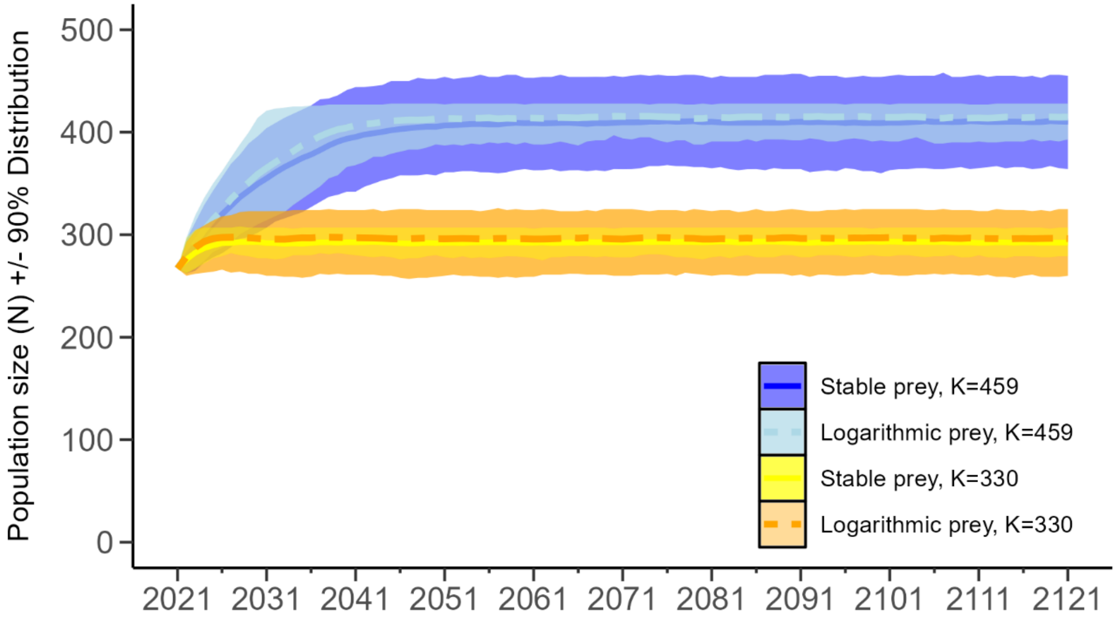


Figure 31. Future projections of WCT population size from 2021 through 2121 using two scenarios of prey index, stable prey levels or logarithmic increase, at two levels of carrying capacity. Projections used a probability of survival at carrying capacity of 96.5%.

Table 20. PVA results for each cumulative projection scenario, including the stochastic growth rate (r), population size (N) at model year 2121, and overall extinction risk.

Scenario	**Growth rate (r) ± SD**	**Population size (N_{2121}) ± SD**	**Probability of extinction (P_0)**
CE model projection – stable prey (K= 459)	0.0098 (SD = 0.041)	719.88 (SD = 103.66)	0
CE model projection – stable prey (K= 330)	0.0065 (SD = 0.041)	519.03 (SD = 79.01)	0
CE model projection – log increase prey (K= 459)	0.0100 (SD = 0.042)	731.95 (SD = 91.74)	0
CE model projection – log increase prey (K= 330)	0.0066 (SD = 0.041)	520.93 (SD = 69.33)	0

4.5 MODEL SENSITIVITY

The parameters for the four focal threats were systematically tested across their full plausible range, from minimum to maximum value (Table 21). The cumulative effects model was most sensitive to variation in the prey index (Figure 32). The prey index affects mortality rates indirectly through an effect on carrying capacity (*K*). Further, the effect of noise modifies the prey index to reduce prey availability and is used as a threshold for applying the effect of PCB+ contamination on calf mortality rates.

Table 21. Parameters used in sensitivity testing for each of the threats: the base value, minimum, maximum and increment.

Threat	Base	Minimum	Maximum	Increment
Strike	0.2	0.1	0.8	0.1
Noise	0.942	0.75	1.0	0.05
PCB+	2	1	10	2
Prey	1.44	0.5	3.0	0.16

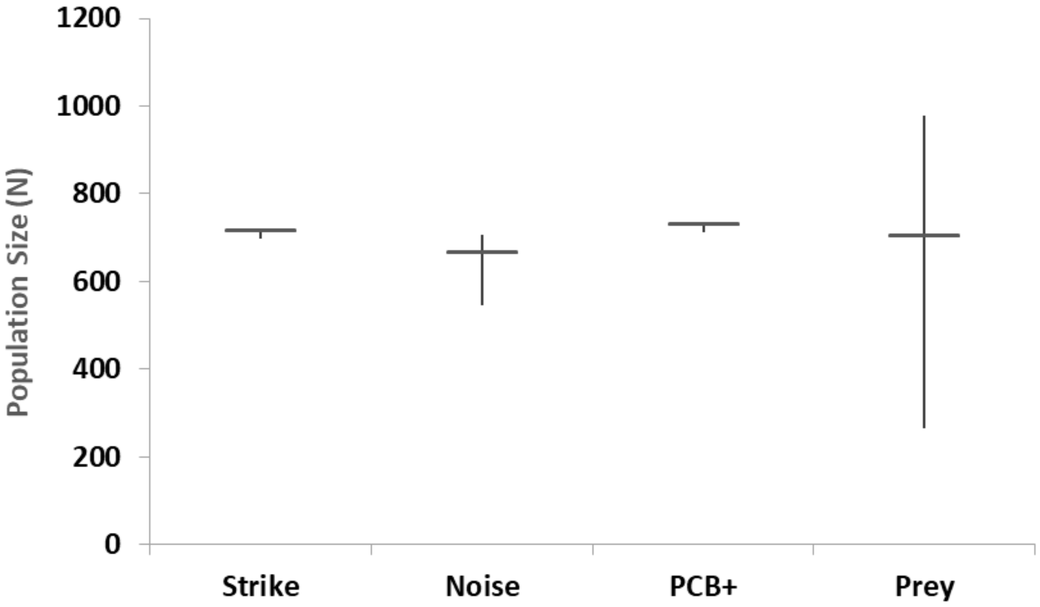


Figure 32. Sensitivity testing of the four threat parameters on WCT population size: vessel strike, acoustic disturbance, PCB+ accumulation rate, and prey index. Horizontal line marks the base value and the vertical lines mark the minimum and maximum population size.

5 DISCUSSION

5.1 ASSESSING CUMULATIVE EFFECTS

To examine the effects of anthropogenic threats on the threatened WCT population of Bigg's killer whale (BKW), this cumulative effects assessment (CEA) combined a Pathways of Effects (PoE) conceptual model with a quantitative Population Viability Analysis (PVA), as defined previously for the resident killer whale (RKW) ecotype (Clarke Murray et al. 2019). This systematic assessment of both individual and cumulative threats allows a detailed examination of the threats (and combination of threats) that may have an influence on the killer whale population into the future.

The evidence based PoE conceptual model is the first step of the CEA, and based on current knowledge, visualises the way threats affect WCT fecundity and mortality through linkage pathways providing the foundation for the assessment. The PoE included 16 pathway linkages from threats to effects on WCT population parameters, mortality and fecundity. Reduced prey availability is the most prominent threat in the PoE model and interacts with three other threats (Prey-Acoustic, Prey-Physical, and Prey-Contaminants interactions). Given all threats in the PoE model connect to at least one other threat, it is evident that threat interactions are crucial to understanding this system and these threats should not be considered in isolation. Although threat interactions are a commonly acknowledged component of cumulative effects assessment (Crain et al. 2008; Foley et al. 2017; Murray et al. 2020), it can be challenging to parameterise interaction effects as they are not always additive in nature, and can have non-linear or threshold effects.

The primary function of the PoE model is to provide a transparent process to scope and structure available knowledge in the form of effect linkages. The linkages identified as quantifiable then formed the basis of the PVA model. In this case, eight of the 16 linkages in the PoE model were quantifiable for inclusion in the PVA population model. A PoE conceptual model captures current knowledge and so is an iterative method that should be regularly updated and reassessed to include emerging knowledge and data. Once developed, the PoE model provides a structured method that can simplify and direct iterative updates, as well as clearly identifying data and knowledge gaps.

The PVA modelling for WCT systematically tested individual threats, their interactions, and a cumulative effects model to determine which best explains the observed population growth and, in turn, influences the long-term recovery of the population. This approach can be a type of sensitivity testing to determine which threats, individually and cumulatively, control the population dynamics. As in the previous RKW cumulative effects assessment (Clarke Murray et al. 2019), reference mortality and fecundity rates from the relatively unaffected Southern Alaska Resident Killer Whales (SARKW) population were used to parameterize the baseline scenario. The threats were then applied as modifications to the baseline rates and the threat scenarios compared to the observed population size. This approach assumes that SARKW are relatively unaffected by anthropogenic threats and that the mechanisms by which threats affect the WCT population are similar.

The individual threat scenarios included threat estimates specific to the WCT population and updated threat models:

- Acoustic disturbance – a newly developed vessel noise model was applied to estimate potential masking of the prey noises. The reduction in communication space compared to minimum ambient noise levels was estimated (Burnham et al. 2023). Acoustic parameters required were only available for harbour seal prey and the reduction of the detection range near harbour seal haulouts was calculated.
- Physical disturbance – fatal vessel strikes remain a low probability event and there were four incidents of documented vessel strikes on Bigg's killer whales between 1974 and 2016 (Raverty et al. 2020).
- Prey abundance – WCT killer whales in the Canadian Salish Sea consume species of pinniped and small cetaceans, with harbour seals dominating the predation event data. Time series data to compare with WCT population rates was limited to three species: harbour seal, Steller sea lion, and harbour porpoise, and a combined prey index was developed.
- Contaminants – no new data was available for the PCB contamination of WCT individuals, however, a new correction factor was applied to the PCB concentration values to conservatively estimate the effects of additional contaminants, such as PBDEs.

Prey availability had the greatest contribution to the cumulative effects model, with variations in the parameter causing the largest changes in sensitivity testing, as was the case for the resident killer whales (Clarke Murray et al. 2019). When combined with the remaining individual threats in a cumulative effects model, the modelled WCT population followed a similar trajectory and growth to the observed and had a low RMSE score, indicating good fit with the observed trend. The cumulative effects model for WCT includes the same threats as that for resident killer whales but applies them in a slightly different way. For the increasing WCT population, carrying capacity may limit the continued growth of the population with the expectation that prey species are at maximum levels and also nearing, or at, carrying capacity (DFO 2021a, 2022). The cumulative effects model was most sensitive to the values of the prey index, as carrying capacity was modelled as a function of the prey index, with resulting density dependence effects on fecundity and mortality. Projections of the outlook for the coastal subset of the WCT population are therefore highly dependent on prey availability and, as such, stressors that affect prey populations are indirectly linked to the WCT population.

5.2 ASSUMPTIONS AND UNCERTAINTIES

While the WCT killer whale population is a relatively well known and frequently observed population, there are comparably less data and literature than was available for Southern and Northen resident killer whales. The parameters of the WCT population itself had to be derived before modelling the individual and cumulative effects of threats. There were no recent population assessments to draw from, with the last assessment from 2007 (Ford et al. 2007), and therefore fecundity and mortality were estimated for the current work. Re-analysis of existing data was completed to estimate age-specific mortality and fecundity rates. New mark-capture-recapture (MCR) modeling was conducted to annual mortality rates and population size for the study area.

The mark-capture-recapture analysis was restricted to the subset of the WCT population that has been observed in the Canadian portion of the Salish Sea. Therefore, the assessments of population size and vital rates are limited to this subset of the population and should not be considered a full and complete population assessment for Species at Risk purposes. The modeling of the threats was dependent on data availability and as a consequence, included varying spatial and temporal scales. The effect of vessel strikes, prey availability, and contaminant threats used data at the coastwide scale, while the acoustic disturbance threat, represented by the vessel noise model, was limited to the Salish Sea and Juan de Fuca Strait (see Figure 1).

Prey availability is a key component of the cumulative effects model, suggesting that parameterisation of this component is critical. Time series data on trends in prey availability were limited to the two species of pinniped, harbour seal and Steller sea lion, and data on harbour porpoise was extrapolated from surveys in the US portion of the Salish Sea. There were no data on trends in Dall's porpoise populations. The prey preferences and degree of prey switching in WCT are unknown, but from the predation events observed, individuals of this population can successfully target a variety of prey species. A composite prey index captures this variability and allows a decrease in one prey species, like harbour seals in recent years, to be compensated by increases in other species, such as harbour porpoises.

The effect of acoustic disturbance on WCT is a significant knowledge gap. In the current cumulative effects model, acoustic disturbance was modelled as a reduction in communication space, i.e., masking of the sounds made by prey. However, a number of alternate functions and mechanisms of impact could be applied to this population. The acoustic range reduction does not consider the acoustic cues that may indicate the presence of prey, in addition to their calls. Conversely, vessel noise may benefit WCT hunting as the detection of the predator by the prey could be masked by vessel noise. The model used for acoustic masking was simplistic in its application in the population model, in that it assumed a single value reduction in the prey available to WCT. The knowledge and data on the effects of this threat are advancing rapidly (e.g., DFO 2017, 2025) so that more nuanced models may be applied in future applications. For example, a recent paper by Tennessen et al. (2024) suggested that shipping noise had differing implications for foraging male and female resident killer whales.

The effect of vessel presence on WCT killer whales is another knowledge gap. Anecdotal evidence suggests that WCT have used the presence of a vessel to assist hunts, targeting small cetaceans engaged in bow riding. In contrast, pinniped species have used nearby vessels to escape killer whale predation, using the vessels as emergency haulouts. Whale watching attention has been increasingly shifting to observing Bigg's whales to ease pressure on resident killer whales and as the WCT becomes more common in the Salish Sea. This shift may increase the amount of time WCT spend in the presence of vessels.

The representation of the contaminants threats has limitations and uncertainties. The model of accumulation and depuration of contaminants is only available for PCBs (Hall et al. 2018). The accumulation rate remains uncertain and difficult to estimate given there is a wide range of PCB toxin load in BKW whales in a relatively small number of blubber samples (N=33; Guy 2018; Ross et al. 2013). This variation may be the result of differential consumption of prey from habitats with varying pollutant levels. Harbour seals from Puget Sound show higher total toxic equivalents of PCBs, PCDDs and PCDFs than seals from the Strait of Georgia and Queen Charlotte Strait (Ross et al. 2004). Other contaminants of concern, such as PBDEs, are not included specifically in the contaminants threat model, although a new correction factor has been applied to account for impacts of other contaminants based on the endocrine disruption risk quotient for harbour seals (Mos et al. 2010).

Interactions between threats can complicate the understanding and management of threats (e.g., Brown et al. 2013). Additive interactions are the default assumption in cumulative effects research but other interactions such as synergistic and mitigative are common (Crain et al. 2008). Possible interactions and non-linearities were considered and identified in the Pathways of Effects model for each threat and included in the PVA model where evidence supported them. The cumulative effects model uses a threshold effect for the PCB contamination, where an increase in calf mortality is applied when the prey index was below the mean. This threshold effect helps explain why the more highly contaminated WCT population is increasing while the endangered SRKW continue to struggle. The synergistic effect of acoustic disturbance and prey availability was explored in the PVA scenarios but was not included in the cumulative effects model because of the uncertainty around the mechanism and direction of acoustic impact on WCT.

The future projections of WCT's continued population growth to carrying capacity assumes that the current levels of threats remain the same, without mitigation actions or management measures. Threats may increase, for example, the results from the vessel noise model are based on one year of vessel noise, and commercial vessel traffic in the study region has increased and is expected to increase further with new industrial developments (McWhinnie et al. 2021). In contrast, mitigation measures can be applied to decrease threats; increased ship traffic can increase the risk of vessel strike while the implementation of management measures such as ship slowdown areas can reduce the risk of fatalities. Further, the changing climate can affect threatened species in diverse ways that can contribute to cumulative effects (van Weelden et al. 2021). Threats can be affected by climate change; for example the propagation of sound through water is affected by changes in climate variables (see Ilyina et al. 2010). Direct and indirect effects on the food web mean that changing climate has the potential to affect the abundance of WCT pinniped and small cetacean prey (Kovacs et al. 2012). The addition and interaction of these changes therefore complicate the projection of population dynamics into the future.

5.3 CONCLUSIONS

This cumulative effects assessment further advances the field by applying the cumulative effects framework comprised of a Pathways of Effects conceptual model and a specific Population Viability Analysis simulation model to evaluate how the current state of threats may affect the future of the West Coast Transient population of Bigg's killer whales. The overview PoE model documents the current knowledge on how threats might affect WCT population parameters. Reduced prey availability and acoustic disturbance are the two most prominent threats in the PoE model as they both connect to effects on population parameters. The linkages that can be quantified with enough confidence and data to be incorporated into the PVA model are defined in the reduced, quantifiable PoE. The quantifiable PoE can also be used to identify knowledge and data gaps.

The cumulative effects PVA model captured the observed population dynamics effectively and includes all priority threats to the species, making it a useful tool. The model we have developed emphasised the importance of prey availability in another killer whale ecotype (Murray et al. 2019). The model can be used to explore the impacts of different mitigation and management options for individual threats on the population trajectory, for example to test whether a reduction in acoustic disturbance would be expected to alter the population trajectory and how long it may take for a change to be observable. The cumulative model defined for WCT includes advances over that applied in the RKW cumulative effects model as a result of ongoing research, such as underwater acoustic monitoring, pinniped and cetacean surveys, and contaminant impacts. Model use and refinement in future versions can help to adaptively inform and/or implement WCT recovery measures identified in the *Recovery Strategy for the Transient Killer Whale* (Orcinus orca) *in Canada* (DFO 2021b).

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7 REFERENCES CITED

Abgrall, P., Moulton, V.D., and Richardson, W.J. 2009. Updated Review of Scientific Information on Impacts of Seismic Survey Sound on Marine Mammals 2004-present. Can. Sci. Advis. Sec. Res. Doc. 2008/087.

Aguilar, A., and Borrell, A. 1994. Abnormally high polychlorinated biphenyl levels in striped dolphins (Stenella coeruleoalba) affected by the 1990–1992 Mediterranean epizootic. Science of The Total Environment **154**(2): 237-247. doi:https://doi.org/10.1016/0048-9697(94)90091-4.

Akaike, H. 1973. Information Theory and an Extension of the Maximum Likelihood Principle. In: Petrov, B.N. and Csaki, F., Eds., International Symposium on Information Theory, 267-281.

Alava, J.J. 2019. Legacy and emerging pollutants in marine mammals' habitat from British Columbia, Canada: Management perspectives for sensitive marine ecosystems. *In* Stewarding the Sound. CRC Press. pp. 87-114.

Alava, J.J., Ross, P.S., Lachmuth, C., Ford, J.K.B., Hickie, B.E., and Gobas, F.A.P.C. 2012. Habitat-Based PCB Environmental Quality Criteria for the Protection of Endangered Killer Whales (*Orcinus orca*). Environmental Science & Technology **46**(22): 12655-12663. doi:10.1021/es303062q.

Andrew, R.K., Howe, B.M., Mercer, J.A., and Dzieciuch, M.A. 2002. Ocean ambient sound: Comparing the 1960s with the 1990s for a receiver off the California coast. Acoustics Research Letters Online **3**(2): 65-70.

Ashley, E.A., Olson, J.K., Adler, T.E., Raverty, S., Anderson, E.M., Jeffries, S., and Gaydos, J.K. 2020. Causes of Mortality in a Harbor Seal (*Phoca vitulina*) Population at Equilibrium. Frontiers in Marine Science **7**: 319.

Asper, E.D., and Cornell, L.H. 1977. Live capture statistics for the killer whale (*Orcinus orca*) 1961-1976, in California, Washington and British Columbia. Aquatic Mammals **5**: 21-26.

Aulanier, F., Simard, Y., Roy, N., Gervaise, C., and Bandet, M. 2017. Effects of shipping on marine acoustic habitats in Canadian Arctic estimated via probabilistic modeling and mapping. Marine Pollution Bulletin **125**(1): 115-131.

Baird, R.W., and Dill, L.M. 1995. Occurrence and behaviour of transient killer whales: seasonal and pod-specific variability, foraging behaviour, and prey handling. Canadian Journal of Zoology **73**(7): 1300-1311. doi:10.1139/z95-154.

Baird, R.W., and Whitehead, H. 2000. Social organization of mammal-eating killer whales: group stability and dispersal patterns. Canadian Journal of Zoology **78**(12): 2096-2105. doi:10.1139/z00-155.

Barrett-Lennard, L.G., Ford, J.K.B., and Heise, K.A. 1996. The mixed blessing of echolocation: differences in sonar use by fish-eating and mammal-eating killer whales. Animal Behaviour **51**(3): 553-565. doi:https://doi.org/10.1006/anbe.1996.0059.

Bigg, M. A. 1982. An assessment of killer whale (*Orcinus orca*) stocks off Vancouver Island, British Columbia. Rep. Int. Whal. Commn. 32: 655-666.

Bigg, M. A. 1984. Sighting and kill data for the Steller lion (*Eumetopias jubatus*) and California sea lion (*Zalophus californianus*) in British Columbia, 1892-1982, with some records from Washington and Southeastern Alaska. DFO Canadian Data Report of Fisheries and Aquatic Sciences(460).

Bigg, M.A. 1985. Status of the Steller sea lion (*Eumetopias jubatus*) and California sea lion (*Zalophus californianus*) in British Columbia. Can. Spec. Publ. Fish. Aquat. Sci. 77: 20.

Bigg, M.A., and Wolman, A.A. 1975. Live-Capture Killer Whale (*Orcinus orca*) Fishery, British Columbia and Washington, 1962–73. Journal of the Fisheries Research Board of Canada **32**(7): 1213-1221. doi:10.1139/f75-140.

Boness, D.J., Bowen, W.D., Buhleier, B.M., and Marshall, G.J. 2006. Mating Tactics and Mating System of an Aquatic-Mating Pinniped: The Harbor Seal, *Phoca vitulina*. Behavioral Ecology and Sociobiology **61**(1): 119-130.

Branstetter, B.K., Felice, M., Robeck, T., Accomando, A.W., and Holt, M.M. 2021. Auditory masking in killer whales (*Orcinus orca*) with realistic signals and noise. The Journal of the Acoustical Society of America **150**(4_Supplement): A251-A251.

Branstetter, B.K., St. Leger, J., Acton, D., Stewart, J., Houser, D., Finneran, J.J., and Jenkins, K. 2017. Killer whale (*Orcinus orca*) behavioral audiograms. The Journal of the Acoustical Society of America **141**(4): 2387-2398.

Brown, C.J., Saunders, M.I., Possingham, H.P., and Richardson, A.J. 2013. Managing for Interactions between Local and Global Stressors of Ecosystems. PLOS ONE **8**(6): e65765.

Brown, T.M., Holbert, S., Gallilee, C., and Colbourne, K. 2022. Sediment Contaminant Concentrations from Disposal at Sea Sites at Point Grey and Sand Heads and inside and outside Southern Resident Killer Whale Critical Habitat, British Columbia in 2019. Canadian Data Report of Fisheries and Aquatic Sciences **1346**.

Brusa, J.L., Pearson, S.F., Raphael, M.G., and Gardner, B. 2024. Biotic and abiotic drivers of harbor seal (*Phoca vitulina*) fine-scale densities in the Salish Sea. Marine Mammal Science **107**(3): e13155.

Burnham, R. 2017. Whale geography: Acoustics, biogeography and whales. Progress in Physical Geography: Earth and Environment **41**(5): 676-685.

Burnham, R., and Duffus, D. 2023. Implication of Altered Acoustic Active Space for Cetacean Species That Result from Soundscape Changes and Noise Additions. Acoustics **5**(2): 444-461.

Burnham, R., Vagle, S., Van Buren, P., and Morrison, C. 2022. Spatial Impact of Recreational-Grade Echosounders and the Implications for Killer Whales. Journal of Marine Science and Engineering **10**(9): 1267.

Burnham, R.E., Vagle, S., Thupaki, P., and Thornton, S.J. 2023. Implications of wind and vessel noise on the sound fields experienced by southern resident killer whales *Orcinus orca* in the Salish Sea. Endangered Species Research **50**: 31-46.

Carretta, J.V., Forney, K.A., Lowry, M.S., Barlow, J., Baker, J.D., Johnston, D., Hanson, B., Brownell, R.L., Robbins, J., Mattila, D.K., Ralls, K., Muto, M.M., Lynch, D., Carswell, L., and Hill, M.C. 2011. U.S. Pacific marine mammal stock assessments 2010. Southwest Fisheries Science Center (U.S.).

Casey, C., Sills, J., and Reichmuth, C. 2016. Source level measurements for harbor seals and implications for estimating communication space. Proceedings of Meetings on Acoustics **27**(1).

Clark, C.W., Ellison, W.T., Southall, B.L., Hatch, L., Van Parijs, S.M., Frankel, A., and Ponirakis, D. 2009. Acoustic masking in marine ecosystems: intuitions, analysis, and implication. Marine Ecology Progress Series **395**: 201-222.

Clarke Murray, C., Mach, M.E., and Martone, R. 2014. Cumulative effects in marine ecosystems: scientific perspectives on its challenges and solutions. World Wildlife Fund Canada.

Clarke Murray, C., Hannah, L., Doniol-Valcroze, T., Wright, B., Stredulinsky, E., Locke, A., and Lacy, R. 2019. Cumulative Effects Assessment for Northern and Southern Resident Killer Whale Populations in the Northeast Pacific. DFO Canadian Science Advisory Research Document. p. x + 88.

Collins, M.D. 1993. A split-step Padé solution for the parabolic equation method. The Journal of the Acoustical Society of America **93**(4): 1736-1742.

Costa-Silva, S., Sacristán, C., Gonzales-Viera, O., Díaz-Delgado, J., Sánchez-Sarmiento, A.M., Marigo, J., Groch, K.R., Carvalho, V.L., Ewbank, A.C., and Colosio, A.C. 2019. Toxoplasma gondii in cetaceans of Brazil: a histopathological and immunohistochemical survey. Revista Brasileira de Parasitologia Veterinária **28**: 395-402.

Crain, C.M., Kroeker, K., and Halpern, B.S. 2008. Interactive and cumulative effects of multiple human stressors in marine systems. Ecology Letters **11**(12): 1304-1315.

Croll, D.A., Clark, C.W., Calambokidis, J., Ellison, W.T., and Tershy, B.R. 2001. Effect of anthropogenic low-frequency noise on the foraging ecology of Balaenoptera whales. Animal Conservation **4**(1): 13-27.

Cullon, D.L., Yunker, M.B., Alleyne, C., Dangerfield, N.J., O'Neill, S., Whiticar, M.J., and Ross, P.S. 2009. Persistent organic pollutants in chinook salmon (*Oncorhynchus tshawytscha*): Implications for resident killer whales of british columbia and adjacent waters. Environmental Toxicology and Chemistry **28**(1): 148-161.

Dahlheim, M., and Castellote, M. 2016. Changes in the acoustic behavior of gray whales *Eschrichtius robustus* in response to noise. Endangered Species Research **31**: 227-242.

Deecke, V.B. 2003. The vocal behaviour of transient killer whales (*Orcinus orca*): communicating with costly calls. Doctor of Philosophy, University of St Andrews, St Andrews, UK.

Deecke, V.B., Ford, J.K.B., and Slater, P.J.B. 2005. The vocal behaviour of mammal-eating killer whales: communicating with costly calls. Animal Behaviour **69**(2): 395-405.

Deecke, V.B., Shapiro, A.D., and Miller, P.J. 2013. Killers in the dark—Acoustic evidence for night-time predation by mammal-eating killer whales (Orcinus orca) in Alaska. *In* The Journal of the Acoustical Society of America. pp. 4008-4008.

Desforges, J.-P., Hall, A., McConnell, B., Rosing-Asvid, A., Barber, J.L., Brownlow, A., De Guise, S., Eulaers, I., Jepson, P.D., Letcher, R.J., Levin, M., Ross, P.S., Samarra, F., Víkingson, G., Sonne, C., and Dietz, R. 2018. Predicting global killer whale population collapse from PCB pollution. Science **361**(6409): 1373-1376.

DFO. 2007. Recovery Strategy for the Transient Killer Whale (*Orcinus orca*) in Canada. Species at Risk Act Recovery Strategy Series. Fisheries and Oceans Canada, Vancouver, vi + 46 pp.

DFO. 2009. Recovery Potential Assessment for West Coast Transient Killer Whales. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2009/039.

DFO. 2010. Population Assessment Pacific Harbour Seal (*Phoca vitulina richardsi*). DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2009/011.

DFO. 2017. Technical review of Roberts Bank Terminal 2 environmental impact statement and marine shipping supplemental report: effects on marine mammals. DFO Can. Sci. Advis. Sec. Sci. Resp. 2017/001.

DFO. 2019. Population Status Update for the Northern Resident Killer Whale (*Orcinus orca*) in 2018. DFO Can. Sci. Advis. Sec. Sci. Resp. 2019/025.

DFO. 2021a. Trends in abundance and distribution of Steller sea lions (*Eumetopias jubatus*) in Canada. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2021/035.

DFO. 2021b. Recovery Strategy for the Transient Killer Whale (*Orcinus orca*) in Canada [Draft]. *Species at Risk Act* Recovery Strategy Series. Fisheries and Oceans Canada, Vancouver. ix + 59 pp.

DFO. 2022. Stock assessment of Pacific harbour seals (*Phoca vitulina richardsi*) in Canada in 2019. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2022/034.

DFO. 2023. California Sea Lion Abundance Estimation in Canada, 2020–21. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2023/016.

DFO. 2025. Evaluation of a Proposed Approach for Offsetting Increases in Underwater Noise from Marine Shipping, Using Information on Southern Resident Killer Whales. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2025/001.

Dietz, R., Letcher, R.J., Desforges, J.-P., Eulaers, I., Sonne, C., Wilson, S., Andersen-Ranberg, E., Basu, N., Barst, B.D., Bustnes, J.O., Bytingsvik, J., Ciesielski, T.M., Drevnick, P.E., Gabrielsen, G.W., Haarr, A., Hylland, K., Jenssen, B.M., Levin, M., McKinney, M.A., Nørregaard, R.D., Pedersen, K.E., Provencher, J., Styrishave, B., Tartu, S., Aars, J., Ackerman, J.T., Rosing-Asvid, A., Barrett, R., Bignert, A., Born, E.W., Branigan, M., Braune, B., Bryan, C.E., Dam, M., Eagles-Smith, C.A., Evans, M., Evans, T.J., Fisk, A.T., Gamberg, M., Gustavson, K., Hartman, C.A., Helander, B., Herzog, M.P., Hoekstra, P.F., Houde, M., Hoydal, K., Jackson, A.K., Kucklick, J., Lie, E., Loseto, L., Mallory, M.L., Miljeteig, C., Mosbech, A., Muir, D.C.G., Nielsen, S.T., Peacock, E., Pedro, S., Peterson, S.H., Polder, A., Rigét, F.F., Roach, P., Saunes, H., Sinding, M.-H.S., Skaare, J.U., Søndergaard, J., Stenson, G., Stern, G., Treu, G., Schuur, S.S., and Víkingsson, G. 2019. Current state of knowledge on biological effects from contaminants on arctic wildlife and fish. Science of The Total Environment **696**: 133792.

Dubey, J.P., Murata, F.H.A., Cerqueira-Cézar, C.K., Kwok, O.C.H., and Grigg, M.E. 2020. Recent epidemiologic and clinical importance of Toxoplasma gondii infections in marine mammals: 2009–2020. Veterinary Parasitology **288**: 109296.

Environment and Climate Change Canada. 2021. 2020-2021 Southern Resident Killer Whale Contaminants Technical Working Group Accomplishments Report.

Erbe, C., Williams, R., Parsons, M., Parsons, S.K., Hendrawan, I.G., and Dewantama, I.M.I. 2018. Underwater noise from airplanes: An overlooked source of ocean noise. Marine pollution bulletin **137**: 656-661.

Erbe, C., Marley, S.A., Schoeman, R.P., Smith, J.N., Trigg, L.E., and Embling, C.B. 2019. The Effects of Ship Noise on Marine Mammals—A Review. Frontiers in Marine Science **6**.

Estes, J.A., Tinker, M.T., Williams, T.M., and Doak, D.F. 1998. Killer Whale Predation on Sea Otters Linking Oceanic and Nearshore Ecosystems. Science **282**(5388): 473-476.

Evenson, J.R., Anderson, D., Murphie, B.L., Cyra, T.A., and Calambokidis, J. 2016. Disappearance and return of harbor porpoise to Puget Sound: 20 year pattern revealed from winter aerial surveys. Technical Report. Washington Department of Fish and Wildlife, Wildlife Program and Cascadia Research Collective, Olympia, WA.

Fearnbach, H., Durban, J.W., Ellifrit, D.K., and Balcomb, K.C., III. 2011. Size and long-term growth trends of Endangered fish-eating killer whales. Endangered Species Research **13**(3): 173-180.

Fernandes, A.R., Kilanowicz, A., Stragierowicz, J., Klimczak, M., and Falandysz, J. 2022. The toxicological profile of polychlorinated naphthalenes (PCNs). Science of The Total Environment **837**: 155764.

Foley, M.M., Mease, L.A., Martone, R.G., Prahler, E.E., Morrison, T.H., Murray, C.C., and Wojcik, D. 2017. The challenges and opportunities in cumulative effects assessment. Environmental Impact Assessment Review **62**: 122-134.

Foote, A.D., Osborne, R.W., and Hoelzel, A.R. 2004. Whale-call response to masking boat noise. Nature **428**(6986): 910-910.

Ford, J.K., and Ellis, G.M. 1999. Transients: mammal-hunting killer whales of British Columbia, Washington State, and Southeast Alaska. UBC Press.

Ford, J.K.B., Ellis, G.E., and Balcomb, K.C. 2000. Killer whales: the natural history and geneology of *Orcinus orca* in British Columbia and Washington State 2nd edition. UBC Press, Vancouver, B.C.

Ford, J.K.B., Ellis, G.M., and Durban, J.W. 2007. An Assessment of the Potential for Recovery of West Coast Transient Killer Whales Using Coastal Waters of British Columbia. DFO Can. Sci. Advis. Sec. Res. Doc. 2007/088. iv + 34.

Ford, J.K.B., Ellis, G., and Stredulinsky, E.H. 2013. Information in support of the identification of critical habitat for Transient killer whales (*Orcinus orca*) off the west coast of Canada. DFO Can. Sci. Advis. Sec. Res. Doc. 2012/155. iv + 46 p.

Ford, J.K.B., Ellis, G.M., Barrett-Lennard, L.G., Morton, A.B., Palm, R.S., and Balcomb III, K.C. 1998. Dietary specialization in two sympatric populations of killer whales (*Orcinus orca*) in coastal British Columbia and adjacent waters. Canadian Journal of Zoology **76**(8): 1456-1471.

Fraker, M.A. 2013. Killer Whale (*Orcinus orca*) Deaths in Prince William Sound, Alaska, 1985–1990. Human and Ecological Risk Assessment: An International Journal **19**(1): 28-52.

Francois, R.E., and Garrison, G.R. 1982. Sound absorption based on ocean measurements. Part II: Boric acid contribution and equation for total absorption. The Journal of the Acoustical Society of America **72**(6): 1879-1890.

Frouin, H., Lebeuf, M., Saint-Louis, R., Hammill, M., Pelletier, É., and Fournier, M. 2008. Toxic effects of tributyltin and its metabolites on harbour seal (*Phoca vitulina*) immune cells in vitro. Aquatic Toxicology **90**(3): 243-251.

Gaydos, J.K., Balcomb, K.C., Osborne, R.W., and Dierauf, L. 2004. Evaluating potential infectious disease threats for southern resident killer whales, *Orcinus orca*: a model for endangered species. Biological Conservation **117**(3): 253-262.

Gimenez, O., Lebreton, J.-D., Choquet, R., and Pradel, R. 2018. R2ucare: An R package to perform goodness-of-fit tests for capture-recapture models. Methods in Ecology and Evolution **9**(7): 1749–1754.

Grant, S.C.H., and Ross, P.S. 2002. Southern Resident Killer Whales at Risk: Toxic Chemicals in the British Columbia and Washington Environment. Can. Tech. Rep. Fish. Aquat. Sci. 2412: xii + 111 p.

Gulland, F.M.D., Baker, J.D., Howe, M., LaBrecque, E., Leach, L., Moore, S.E., Reeves, R.R., and Thomas, P.O. 2022. A review of climate change effects on marine mammals in United States waters: Past predictions, observed impacts, current research and conservation imperatives. Climate Change Ecology **3**: 100054.

Guy, J.M.G. 2018. A risk analysis of legacy pollutants: PCBs, PBDEs and new emerging pollutants in Salish sea killer whales. PhD, Simon Fraser University, Environment: School of Resource and Environmental Management M.R.M thesis Report No 672 224pp.

Hall, A.J., Hugunin, K., Deaville, R., Law, R.J., Allchin, C.R., and Jepson, P.D. 2006. The Risk of Infection from Polychlorinated Biphenyl Exposure in the Harbor Porpoise (*Phocoena phocoena*): A Case–Control Approach. Environmental Health Perspectives **114**(5): 704-711.

Hall, A.J., McConnell, B.J., Schwacke, L.H., Ylitalo, G.M., Williams, R., and Rowles, T.K. 2018. Predicting the effects of polychlorinated biphenyls on cetacean populations through impacts on immunity and calf survival. Environmental Pollution **233**: 407-418.

Hamilton, E.L. 1980. Geoacoustic modeling of the sea floor. The Journal of the Acoustical Society of America **68**(5): 1313-1340.

Haugerud, R.A. 1999. Digital elevation model (DEM) of Cascadia, latitude 39N-53N, longitude 116W-133W. 99-369, U.S.G. Survey, Reston, VA.

Hayes, K.R.R., Ylitalo, G.M., Anderson, T.A., Urbán R, J., Jacobsen, J.K., Scordino, J.J., Lang, A.R., Baugh, K.A., Bolton, J.L., Brüniche-Olsen, A., Calambokidis, J., Martínez-Aguilar, S., Subbiah, S., Gribble, M.O., and Godard-Codding, C.A.J. 2022. Influence of Life-History Parameters on Persistent Organic Pollutant Concentrations in Blubber of Eastern North Pacific Gray Whales (*Eschrichtius robustus*). Environmental Science & Technology **56**(23): 17119-17130.

Hayes, S.A., Costa, D.P., Harvey, J.T., and le Boeuf, B.J. 2004. AQUATIC MATING STRATEGIES OF THE MALE PACIFIC HARBOR SEAL (*PHOCA VITULINA RICHARDII*): ARE MALES DEFENDING THE HOTSPOT? Marine Mammal Science **20**(3): 639-656.

Heise, K.A., Barrett-Lennard, L.G., Chapman, N.R., Dakin, D.T., Erbe, C., Hannay, D.E., Merchant, N.D., Pilkington, J.S., Thornton, S.J., Tollit, D.J., Vagle, S., Veirs, V.R., Vergara, V., Wood, J.D., Wright, B.M., Yurk, H. 2017. Proposed Metrics for the Management of Underwater Noise for Southern Resident Killer Whales. Coastal Ocean Report Series (2), Ocean Wise, Vancouver, 30p.

Helle, E., Olsson, M.T., and Jensen, S. 1976. PCB levels correlated with pathological changes in seal uteri. Ambio **5**(5/6): 261-263.

Herman, D.P., Burrows, D.G., Wade, P.R., Durban, J.W., Matkin, C.O., LeDuc, R.G., Barrett-Lennard, L.G., and Krahn, M.M. 2005. Feeding ecology of eastern North Pacific killer whales *Orcinus orca* from fatty acid, stable isotope, and organochlorine analyses of blubber biopsies. Marine Ecology Progress Series **302**: 275-291.

Hickie, B.E., Ross, P.S., Macdonald, R.W., and Ford, J.K.B. 2007. Killer Whales (*Orcinus orca*) Face Protracted Health Risks Associated with Lifetime Exposure to PCBs. Environmental Science & Technology **41**(18): 6613-6619.

Hildebrand, J.A. 2009. Anthropogenic and natural sources of ambient noise in the ocean. Marine Ecology Progress Series **395**: 5-20.

Hodson, T.O. 2022. Root-mean-square error (RMSE) or mean absolute error (MAE): when to use them or not. Geosci. Model Dev. **15**(14): 5481-5487.

Holt, M.M., Noren, D.P., and Emmons, C.K. 2011. Effects of noise levels and call types on the source levels of killer whale calls. The Journal of the Acoustical Society of America **130**(5): 3100-3106

Holt, M.M., Noren, D.P., Veirs, V., Emmons, C.K., and Veirs, S. 2009. Speaking up: Killer whales (*Orcinus orca*) increase their call amplitude in response to vessel noise. The Journal of the Acoustical Society of America **125**(1): EL27-EL32.

Houghton, J., Baird, R.W., Emmons, C.K., and Hanson, M.B. 2015. Changes in the Occurrence and Behavior of Mammal-Eating Killer Whales in Southern British Columbia and Washington State, 1987–2010. Northwest Science **89**(2): 154-169, 116.

Huntington, H.P., Daniel, R., Hartsig, A., Harun, K., Heiman, M., Meehan, R., Noongwook, G., Pearson, L., Prior-Parks, M., Robards, M., and Stetson, G. 2015. Vessels, risks, and rules: Planning for safe shipping in Bering Strait. Marine Policy **51**: 119-127.

Ilyina, T., Zeebe, R.E., and Brewer, P.G. 2010. Future ocean increasingly transparent to low-frequency sound owing to carbon dioxide emissions. Nature Geoscience **3**(1): 18-22.

Jefferson, T.A., Smultea, M.A., Courbis, S.S., and Campbell, G.S. 2016. Harbor porpoise (*Phocoena phocoena*) recovery in the inland waters of Washington: estimates of density and abundance from aerial surveys, 2013–2015. Canadian Journal of Zoology **94**(7): 505-515.

Jeffries, S., Huber, H., Calambokidis, J., and Laake, J. 2003. Trends and Status of Harbor Seals in Washington State: 1978-1999. The Journal of Wildlife Management **67**(1): 207-218.

Jensen, F.B., Kuperman, W.A., Porter, M.B., Schmidt, H., and Tolstoy, A. 2011. Computational ocean acoustics. New York, NY: Springer New York.

Jepson, P.D., Deaville, R., Barber, J.L., Aguilar, À., Borrell, A., Murphy, S., Barry, J., Brownlow, A., Barnett, J., Berrow, S., Cunningham, A.A., Davison, N.J., ten Doeschate, M., Esteban, R., Ferreira, M., Foote, A.D., Genov, T., Giménez, J., Loveridge, J., Llavona, Á., Martin, V., Maxwell, D.L., Papachlimitzou, A., Penrose, R., Perkins, M.W., Smith, B., de Stephanis, R., Tregenza, N., Verborgh, P., Fernandez, A., and Law, R.J. 2016. PCB pollution continues to impact populations of orcas and other dolphins in European waters. Scientific Reports **6**(1): 18573.

Jourdain, E., Barrett-Lennard, L.G., Ellis, G.M., Ford, J.K.B., Karoliussen, R., Towers, J.R., and Vongraven, D. 2021. Natural Entrapments of Killer Whales (*Orcinus orca*): A Review of Cases and Assessment of Intervention Techniques. Front. Conserv. Sci. **2**:707616

Joy, R., Schick, R.S., Dowd, M., Margolina, T., Joseph, J.E., and Thomas, L. 2022. A fine-scale marine mammal movement model for assessing long-term aggregate noise exposure. Ecological Modelling **464**: 109798.

Kannan, K., Blankenship, A.L., Jones, P.D., and Giesy, J.P. 2000. Toxicity Reference Values for the Toxic Effects of Polychlorinated Biphenyls to Aquatic Mammals. Human and Ecological Risk Assessment: An International Journal **6**(1): 181-201.

Keen, K.A., Beltran, R.S., Pirotta, E., and Costa, D.P. 2021. Emerging themes in Population Consequences of Disturbance models. Proc. R. Soc. B.2882021032520210325.

Kelley, D.E., Vlasic, J.P., and Brillant, S.W. 2021. Assessing the lethality of ship strikes on whales using simple biophysical models. Marine Mammal Science **37**(1): 251-267.

Kennedy, S., Kuiken, T., Jepson, P.D., Deaville, R., Forsyth, M., Barrett, T., van de Bildt, M.W., Osterhaus, A.D., Eybatov, T., Duck, C., Kydyrmanov, A., Mitrofanov, I., and Wilson, S. 2000. Mass die-Off of Caspian seals caused by canine distemper virus. Emerg Infect Dis **6**(6): 637-639. doi:10.3201/eid0606.000613.

Konrad Clarke, C.M., Stredulinsky, E., Toews, S., Rani, V., Bouvier-Brown, M., Smyth, D., Joy, R., and Thornton, S.J. 2024. Assessing the efficacy of ecological reserves: killer whale beach rubbing behaviour and vessel disturbance. Endangered Species Research **53**: 555-567.

Kotik, C., Durban, J.W., Fearnbach, H., and Barrett-Lennard, L.G. 2023. Morphometrics of mammal-eating killer whales from drone photogrammetry, with comparison to sympatric fish-eating killer whales in the eastern North Pacific. Marine Mammal Science **39**(1): 42-58.

Kovacs, K.M., Aguilar, A., Aurioles, D., Burkanov, V., Campagna, C., Gales, N., Gelatt, T., Goldsworthy, S.D., Goodman, S.J., Hofmeyr, G.J.G., Härkönen, T., Lowry, L., Lydersen, C., Schipper, J., Sipilä, T., Southwell, C., Stuart, S., Thompson, D., and Trillmich, F. 2012. Global threats to pinnipeds. Marine Mammal Science **28**(2): 414-436.

Kyhn, L.A., Tougaard, J., Beedholm, K., Jensen, F.H., Ashe, E., Williams, R., and Madsen, P.T. 2013. Clicking in a Killer Whale Habitat: Narrow-Band, High-Frequency Biosonar Clicks of Harbour Porpoise (*Phocoena phocoena*) and Dall's Porpoise (*Phocoenoides dalli*). PLOS ONE **8**(5): e63763.

Laake, J.L. 2013. RMark: An R Interface for analysis of capture-recapture data with MARK. AFSC Processed Rep. 2013-01, 25 p. Alaska Fish. Sci. Cent., NOAA, Natl. Mar. Fish. Serv., 7600 Sand Point Way NE, Seattle WA 98115.

Lacy, R.C., and Pollak, J.P. 2023. Vortex: A stochastic simulation of the extinction process. Chicago Zoological Society, Brookfield, Illinois, USA.

Lacy, R.C., Williams, R., Ashe, E., Balcomb III, K.C., Brent, L.J.N., Clark, C.W., Croft, D.P., Giles, D.A., MacDuffee, M., and Paquet, P.C. 2017. Evaluating anthropogenic threats to endangered killer whales to inform effective recovery plans. Scientific Reports **7**(1): 14119.

Laist, D.W., Knowlton, A.R., and Pendleton, D. 2014. Effectiveness of mandatory vessel speed limits for protecting North Atlantic right whales. Endangered Species Research **23**(2): 133-147.

Law, A.M., and Kelton, W.D. 1991. Simulation Modeling & Analysis. McGraw-Hill, New York, USA.

Lawson, T.M., Ylitalo, G.M., O'Neill, S.M., Dahlheim, M.E., Wade, P.R., Matkin, C.O., Burkanov, V., and Boyd, D.T. 2020. Concentrations and profiles of organochlorine contaminants in North Pacific resident and transient killer whale (*Orcinus orca*) populations. Science of The Total Environment **722**: 137776.

Leal, N., Leal, E., and Sanchez, G. 2015. Marine vessel recognition by acoustic signature. ARPN Journal of Engineering and Applied Sciences **10**(20): 9633-9639.

Lebreton, J.-D., Burnham, K.P., Clobert, J., and Anderson, D.R. 1992. Modeling Survival and Testing Biological Hypotheses Using Marked Animals: A Unified Approach with Case Studies. Ecological Monographs **62**(1): 67-118.

Lee, K., Alava, J.J., Cottrell, P., Cottrell, L., Grace, R., Zysk, I., and Raverty, S. 2023. Emerging Contaminants and New POPs (PFAS and HBCDD) in Endangered Southern Resident and Bigg's (Transient) Killer Whales (*Orcinus orca*): In Utero Maternal Transfer and Pollution Management Implications. Environmental Science & Technology **57**(1): 360-374.

Lowry, L.F., Frost, K.J., Ver Hoep, J.M., and Delong, R.A. 2001. MOVEMENTS OF SATELLITE-TAGGED SUBADULT AND ADULT HARBOR SEALS IN PRINCE WILLIAM SOUND, ALASKA. Marine Mammal Science **17**(4): 835-861.

Lundin, J.I., Ylitalo, G.M., Booth, R.K., Anulacion, B., Hempelmann, J.A., Parsons, K.M., Hanson, M.B., and Wasser, S.K. 2016. Modulation in Persistent Organic Pollutant Concentration and Profile by Prey Availability and Reproductive Status in Southern Resident Killer Whale Scat Samples. Environmental Science & Technology **50**(12): 6506-6516.

Lusseau, D., Bain, D.E., Williams, R., and Smith, J.C. 2009. Vessel traffic disrupts the foraging behavior of southern resident killer whales *Orcinus orca*. Endangered Species Research **6**(3): 211-221.

MacGillivray, A., and Li, Z. 2018. Vessel noise measurements from the ECHO slowdown trial: Final report. Document 01518. Version 3.0. Technical report by JASCO Applied Sciences for Vancouver Fraser Port Authority ECHO Program. Victoria, BC.

Matkin, C., Ellis, G., Saulitis, E., Barrett-Lennard, L., and Matkin, D. 1999. Killer whales of southern Alaska. North Gulf Oceanic Society, Homer, Alaska. 96 pp.

Matkin, C.O., Ward Testa, J., Ellis, G.M., and Saulitis, E.L. 2014. Life history and population dynamics of southern Alaska resident killer whales (*Orcinus orca*). Marine Mammal Science **30**(2): 460-479.

Matkin, C.O., Saulitis, E.L., Ellis, G.M., Olesiuk, P., and Rice, S.D. 2008. Ongoing population-level impacts on killer whales *Orcinus orca* following the Exxon Valdez oil spill in Prince William Sound, Alaska. Marine Ecology Progress Series **356**: 269-281.

Matkin, C.O., Durban, J.W., Saulitis, E.L., Andrews, R.D., Straley, J.M., Matkin, D.R., and Ellis, G.M. 2012. Contrasting abundance and residency patterns of two sympatric populations of transient killer whales (*Orcinus orca*) in the northern Gulf of Alaska. Fish. Bull. **110**(2):143–155.

Matthews, L.P., Fournet, M.E.H., Gabriele, C., Klinck, H., and Parks, S.E. 2020. Acoustically advertising male harbour seals in southeast Alaska do not make biologically relevant acoustic adjustments in the presence of vessel noise. Biology Letters **16**(4): 20190795.

McEntee, M.H.F., MacQueeney, M., Alvarado, D., and Mann, J. 2023. Infanticide and Sexual Conflict in Cetaceans. *In* Sex in Cetaceans: Morphology, Behavior, and the Evolution of Sexual Strategies. *Edited by* B. Würsig and D.N. Orbach. Springer International Publishing, Cham. pp. 199-224.

McRae, T.M., Volpov, B.L., Sidrow, E., Fortune, S.M.E., Auger-Méthé, M., Heckman, N., and Trites, A.W. 2024. Killer whale respiration rates. PLOS ONE **19**(5): e0302758.

McWhinnie, L.H., O'Hara, P.D., Hilliard, C., Le Baron, N., Smallshaw, L., Pelot, R., and Canessa, R. 2021. Assessing vessel traffic in the Salish Sea using satellite AIS: An important contribution for planning, management and conservation in southern resident killer whale critical habitat. Ocean & Coastal Management **200**: 105479.

Miksis, J.L., Grund, M.D., Nowacek, D.P., Solow, A.R., Connor, R.C., and Tyack, P.L. 2001. Cardiac response to acoustic playback experiments in the captive bottlenose dolphin (*Tursiops truncatus*). Journal of Comparative Psychology **115**(3): 227-232.

Miller, P.J.O. 2006. Diversity in sound pressure levels and estimated active space of resident killer whale vocalizations. Journal of Comparative Physiology A **192**(5): 449-459.

Moore, K.A., Burnham, R.E., Duffus, D.A., and Wells, P.G. 2021. An Assessment of Commercial Fleet Applications of Management Measures in Clayoquot Sound, British Columbia, Canada, Aimed to Mitigate Whale-watching Impacts. Tourism in Marine Environments **16**(4): 205-224.

Morin, P.A., McCarthy, M.L., Fung, C.W., Durban, J.W., Parsons, K.M., Perrin, W.F., Taylor, B.L., Jefferson, T.A., and Archer, F.I. 2024. Revised taxonomy of eastern North Pacific killer whales (*Orcinus orca*): Bigg's and resident ecotypes deserve species status. Royal Society Open Science **11**(3): 231368.

Morra, J., and Gobas, F. 2017. Recovering British Columbia’s Resident Killer Whales by Tackling Pollution. Report Prepared for the Department of Fisheries and Oceans Canada.

Morton, A.B., and Symonds, H.K. 2002. Displacement of *Orcinus orca* (L.) by high amplitude sound in British Columbia, Canada. ICES Journal of Marine Science **59**(1): 71-80.

Mos, L., Ross, P.S., McIntosh, D., and Raverty, S. 2003. Canine distemper virus in river otters in British Columbia as an emergent risk for coastal pinnipeds. Vet Rec **152**(8): 237-239.

Mos, L., Cameron, M., Jeffries, S.J., Koop, B.F., and Ross, P.S. 2010. Risk-based analysis of polychlorinated biphenyl toxicity in harbor seals. Integrated Environmental Assessment and Management **6**(4): 631-640.

Murphy, S., Barber, J.L., Learmonth, J.A., Read, F.L., Deaville, R., Perkins, M.W., Brownlow, A., Davison, N., Penrose, R., Pierce, G.J., Law, R.J., and Jepson, P.D. 2015. Reproductive Failure in UK Harbour Porpoises Phocoena phocoena: Legacy of Pollutant Exposure? PLOS ONE **10**(7): e0131085.

Murray, C.C., Hannah, L., and Locke, A. 2020. A review of cumulative effects research and assessment in Fisheries and Oceans Canada. Can. Tech. Rep. Fish. Aquat. Sci. 3357: vii + 51 p.

Murray, C.C., Hannah, L.C., Doniol-Valcroze, T., Wright, B., Stredulinsky, E., Locke, A., and R. Lacy. 2019. Cumulative Effects Assessment for Northern and Southern Resident Killer Whale Populations in the Northeast Pacific. DFO Can. Sci. Advis. Sec. Res. Doc. 2019/056. x. + 88 p..

Murray, C.C., Hannah, L.C., Doniol-Valcroze, T., Wright, B.M., Stredulinsky, E.H., Nelson, J.C., Locke, A., and Lacy, R.C. 2021. A cumulative effects model for population trajectories of resident killer whales in the Northeast Pacific. Biological Conservation **257**: 109124.

Muto, M. M., Helker, V. T., Delean, B.J., Young, N.C., Freed, J.C., Angliss, R.P., Friday, N.A, Boveng, P.L., Breiwick, J.M., Brost, B.M., Cameron, M.F., Clapham, P.J., Crance, J.L., Dahle, S.P., Dahlheim, M.E., Fadely, B.S., Ferguson, M.C., Fritz, L.W., Goetz, K.T., Hobbs, R.C., Ivashchenko, Y.V., Kennedy, A.S., London, J.M., Mizroch, S.A., Ream, R.R., Richmond, E.L., Shelden, K.E.W., Sweeney, K.L., Towell, R.G., Wade, P.R., Waite, J.M., and Zerbini, A.N. 2021. Alaska marine mammal stock assessments, 2020. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-421, 398 p.

Myers, H.J., Olsen, D.W., Matkin, C.O., Horstmann, L.A., and Konar, B. 2021. Passive acoustic monitoring of killer whales (*Orcinus orca*) reveals year-round distribution and residency patterns in the Gulf of Alaska. Scientific Reports **11**(1): 20284.

National Research Council. 2003. Ocean Noise and Marine Mammals. National Academies Press, Washington, DC. PMID: 25057640.

New, L., Lusseau, D., and Harcourt, R. 2020. Dolphins and Boats: When Is a Disturbance, Disturbing?. Frontiers in Marine Science **7**.

New, L.F., Harwood, J., Thomas, L., Donovan, C., Clark, J.S., Hastie, G., Thompson, P.M., Cheney, B., Scott-Hayward, L., and Lusseau, D. 2013. Modelling the biological significance of behavioural change in coastal bottlenose dolphins in response to disturbance. Functional Ecology **27**(2): 314-322.

Nielsen, M.L.K., Ellis, S., Towers, J.R., Doniol-Valcroze, T., Franks, D.W., Cant, M.A., Weiss, M.N., Johnstone, R.A., Balcomb III, K.C., Ellifrit, D.K., and Croft, D.P. 2021. A long postreproductive life span is a shared trait among genetically distinct killer whale populations. Ecology and Evolution **11**(13): 9123-9136.

Nielsen, M.L.K., Ellis, S., Weiss, M.N., Towers, J.R., Doniol-Valcroze, T., Franks, D.W., Cant, M.A., Ellis, G.M., Ford, J.K.B., Malleson, M., Sutton, G.J., Shaw, T.J.H., Balcomb, K.C., Ellifrit, D.K., and Croft, D.P. 2023. Temporal dynamics of mother–offspring relationships in Bigg's killer whales: opportunities for kin-directed help by post-reproductive females. Proceedings of the Royal Society B: Biological Sciences **290**(2000): 20230139.

Nikolich, K., Frouin-Mouy, H., and Acevedo-Gutiérrez, A. 2018. Clear diel patterns in breeding calls of harbor seals (*Phoca vitulina*) at Hornby Island, British Columbia, Canada. Canadian Journal of Zoology **96**(11): 1236-1243.

Noren, D.P., Johnson, A.H., Rehder, D., and Larson, A. 2009. Close approaches by vessels elicit surface active behaviors by southern resident killer whales. Endangered Species Research **8**(3): 179-192.

Nowacek, D.P., Thorne, L.H., Johnston, D.W., and Tyack, P.L. 2007. Responses of cetaceans to anthropogenic noise. Mammal Review **37**(2): 81-115.

Olesiuk, P.F. 2010. An assessment of population trends and abundance of harbour seals (*Phoca vitulina*) in British Columbia. DFO Can. Sci. Advis. Sec. Res. Doc. 2009/105. vi + 157 p.

Olesiuk, P.F. 2018. Recent trends in abundance of Steller sea lions (*Eumetopias jubatus*) in British Columbia. DFO Can. Sci. Advis. Sec. Res. Doc. 2018/006. v + 67 p.

Olesiuk, P.F., Bigg, M.A., and Ellis, G.M. 1990. Life history and population dynamics of resident killer whales (*Orcinus orca*) in the coastal waters of British Columbia and Washington State. Report of the International Whaling Commission, Special **12**: 209-243.

OSPAR Commission. 2009. Overview of the impacts of anthropogenic underwater sound in the marine environment. OSPAR Convention for the Protection of the Marine Environment of the North-East Atlantic. Biodiversity and Ecosystems Series, Publication Number 441/2009, 134 pp. Available.

Parks, S.E., Clark, C.W., and Tyack, P.L. 2007. Short- and long-term changes in right whale calling behavior: The potential effects of noise on acoustic communication. The Journal of the Acoustical Society of America **122**(6): 3725-3731.

Parsons, K.M., Durban, J.W., Burdin, A.M., Burkanov, V.N., Pitman, R.L., Barlow, J., Barrett-Lennard, L.G., LeDuc, R.G., Robertson, K.M., Matkin, C.O., and Wade, P.R. 2013. Geographic Patterns of Genetic Differentiation among Killer Whales in the Northern North Pacific. Journal of Heredity **104**(6): 737-754.

Pradel, R., Hines, J.E., Lebreton, J.-D., and Nichols, J.D. 1997. Capture-Recapture Survival Models Taking Account of Transients. Biometrics **53**(1): 60-72.

Rasmussen, M.H., Miller, L.A., and Au, W.W.L. 2002. Source levels of clicks from free-ranging white-beaked dolphins (*Lagenorhynchus albirostris* Gray 1846) recorded in Icelandic waters. The Journal of the Acoustical Society of America **111**(2): 1122-1125.

Raverty, S., St. Leger, J., Noren, D.P., Burek Huntington, K., Rotstein, D.S., Gulland, F.M.D., Ford, J.K.B., Hanson, M.B., Lambourn, D.M., Huggins, J., Delaney, M.A., Spaven, L., Rowles, T., Barre, L., Cottrell, P., Ellis, G., Goldstein, T., Terio, K., Duffield, D., Rice, J., and Gaydos, J.K. 2020. Pathology findings and correlation with body condition index in stranded killer whales (*Orcinus orca*) in the northeastern Pacific and Hawaii from 2004 to 2013. PLOS ONE **15**(12): e0242505.

Remili, A., Dietz, R., Sonne, C., Samarra, F.I.P., Letcher, R.J., Rikardsen, A.H., Ferguson, S.H., Watt, C.A., Matthews, C.J.D., Kiszka, J.J., Rosing-Asvid, A., and McKinney, M.A. 2023. Varying Diet Composition Causes Striking Differences in Legacy and Emerging Contaminant Concentrations in Killer Whales across the North Atlantic. Environmental Science & Technology.

Richardson, W.J., Greene Jr, C.R., Malme, C.I., and Thomson, D.H. 1995. Marine mammals and noise. Academic press.

Riesch, R., and Deecke, V.B. 2011. Whistle communication in mammal-eating killer whales (*Orcinus orca*): further evidence for acoustic divergence between ecotypes. Behavioral Ecology and Sociobiology **65**(7): 1377-1387.

Robinson, K.J., Hall, A.J., Debier, C., Eppe, G., Thomé, J.-P., and Bennett, K.A. 2018. Persistent Organic Pollutant Burden, Experimental POP Exposure, and Tissue Properties Affect Metabolic Profiles of Blubber from Gray Seal Pups. Environmental Science & Technology **52**(22): 13523-13534.

Rolland, R.M., Parks, S.E., Hunt, K.E., Castellote, M., Corkeron, P.J., Nowacek, D.P., Wasser, S.K., and Kraus, S.D. 2012. Evidence that ship noise increases stress in right whales. Proceedings of the Royal Society B: Biological Sciences **279**(1737): 2363-2368.

Ross, P.S. 2006. Fireproof killer whales (*Orcinus orca*): flame-retardant chemicals and the conservation imperative in the charismatic icon of British Columbia, Canada. Canadian Journal of Fisheries and Aquatic Sciences **63**(1): 224-234.

Ross, P.S., Ellis, G.E., Ikonomou, M.G., Barrett-Lennard, L.G., and Addison, R.F. 2000. High PCB concentrations in free-ranging Pacific killer whales, *Orcinus orca*: Effects of age, sex and dietary preference. Marine Pollution Bulletin **40**(6): 504-515.

Ross, P.S., Jeffries, S.J., Yunker, M.B., Addison, R.F., Ikonomou, M.G., and Calambokidis, J.C. 2004. Harbor seals (*Phoca vitulina*) in British Columbia, Canada, and Washington State, USA, reveal a combination of local and global polychlorinated biphenyl, dioxin, and furan signals. Environmental Toxicology and Chemistry **23**(1): 157-165.

Ross, P.S., Desforges, J.-P.W., Dangerfield, N.J., Crewe, N.F., Dubetz, C.P., Fischer, M.B., Fraser, T.L., and Ross, A.R. 2013. Blubber concentrations of PCBs, PBDEs, PCDDs and PCDFs from 1993 to 2009 in killer whales (*Orcinus orca*) from the North East Pacific Ocean. Unpublished.

Samuel, A.M., and Worthy, G.A.J. 2004. Variability in fatty acid composition of bottlenose dolphin (*Tursiops truncatus*) blubber as a function of body site, season, and reproductive state. Canadian Journal of Zoology **82**(12): 1933-1942.

Sato, M., Trites, A.W., and Gauthier, S. 2021. Southern resident killer whales encounter higher prey densities than northern resident killer whales during summer. Canadian Journal of Fisheries and Aquatic Sciences **78**(11): 1732-1743.

Schleimer, A., Ramp, C., Delarue, J., Carpentier, A., Bérubé, M., Palsbøll, P.J., Sears, R., and Hammond, P.S. 2019. Decline in abundance and apparent survival rates of fin whales (*Balaenoptera physalus*) in the northern Gulf of St. Lawrence. Ecology and Evolution **9**(7): 4231-4244.

Schmunk, R. 2015. 'Clever Seal Hops On Boat To Escape Killer Whales In B.C.'s Desolation Sound', *Huffpost*, 17 Aug.

Schoeman, R.P., Patterson-Abrolat, C., and Plön, S. 2020. A Global Review of Vessel Collisions With Marine Animals. Frontiers in Marine Science **7**:292.

Schusterman, R.J. 1974. Auditory sensitivity of a California sea lion to airborne sound. The Journal of the Acoustical Society of America **56**(4): 1248-1251.

Senigaglia, V., Christiansen, F., Bejder, L., Gendron, D., Lundquist, D., Noren, D.P., Schaffar, A., Smith, J.C., Williams, R., Martinez, E., Stockin, K., and Lusseau, D. 2016. Meta-analyses of whale-watching impact studies: comparisons of cetacean responses to disturbance. Marine Ecology Progress Series **542**: 251-263.

Shields, M.W., Hysong-Shimazu, S., Shields, J.C., and Woodruff, J. 2018. Increased presence of mammal-eating killer whales in the Salish Sea with implications for predator-prey dynamics. PeerJ **6**: e6062.

Simmonds, M., Dolman, S., and Weilgart, L. 2004. Oceans of noise. Whale and dolphin conservation Society.

Smith, C.E. 2017. Evaluating Social Network Dynamics of Bigg's Killer Whales (*Orcinus orca*) and Vessel Traffic within a Transboundary Region: Implications for Conservation Management. Ph.D., The University of Southern Mississippi.

Soontiens, N., and Allen, S.E. 2017. Modelling sensitivities to mixing and advection in a sill-basin estuarine system. Ocean Modelling **112**: 17-32.

Soontiens, N., Allen, S.E., Latornell, D., Le Souëf, K., Machuca, I., Paquin, J.-P., Lu, Y., Thompson, K., and Korabel, V. 2016. Storm Surges in the Strait of Georgia Simulated with a Regional Model. Atmosphere-Ocean **54**(1): 1-21.

Sørensen, P.M., Wisniewska, D.M., Jensen, F.H., Johnson, M., Teilmann, J., and Madsen, P.T. 2018. Click communication in wild harbour porpoises (*Phocoena phocoena*). Scientific Reports **8**(1): 9702.

Southall, B.L., Nowacek, D.P., Bowles, A.E., Senigaglia, V., Bejder, L., and Tyack, P.L. 2021. Marine Mammal Noise Exposure Criteria: Assessing the Severity of Marine Mammal Behavioral Responses to Human Noise. Aquatic Mammals **47**(5): 421-464.

Southall, B.L., Finneran, J.J., Reichmuth, C., Nachtigall, P.E., Ketten, D.R., Bowles, A.E., Ellison, W.T., Nowacek, D.P., and Tyack, P.L. 2019. Marine Mammal Noise Exposure Criteria: Updated Scientific Recommendations for Residual Hearing Effects. Aquatic Mammals **45**(2): 125-232.

Stafford, K.M. 2019. Increasing detections of killer whales (*Orcinus orca*), in the Pacific Arctic. Marine Mammal Science **35**(2): 696-706.

Stewart, J.D., Durban, J.W., Fearnbach, H., Barrett-Lennard, L.G., Casler, P.K., Ward, E.J., and Dapp, D.R. 2021. Survival of the fattest: linking body condition to prey availability and survivorship of killer whales. Ecosphere **12**(8): e03660.

Taylor, M.S. 2021a. Presidential Address: The orca conjecture. Canadian Journal of Economics/Revue canadienne d'économique **54**(4): 1459-1494.

Taylor, M.S. 2021b. Trade, Competitive Exclusion, and the Slow-Motion Extinction of the Southern Resident Killer Whales. National Bureau of Economic Research Working Paper Series No. 29014.

Taylor, M.S., and Mayer, F. 2023. International Trade, Noise Pollution, and Killer Whales. National Bureau of Economic Research Working Paper Series No. w31390.

Tennessen, J.B., Holt, M.M., Wright, B.M., Hanson, M.B., Emmons, C.K., Giles, D.A., Hogan, J.T., Thornton, S.J., and Deecke, V.B. 2024. Males miss and females forgo: Auditory masking from vessel noise impairs foraging efficiency and success in killer whales. Global Change Biology **30**(9): e17490.

Thornton, S.J., Toews, S., Burnham, R., Konrad, C.M., Stredulinsky, E., Gavrilchuk, K., Thupaki, P., and Vagle, S. 2022. Areas of elevated risk for vessel-related physical and acoustic impacts in Southern Resident Killer Whale (*Orcinus orca*) critical habitat. DFO Can. Sci. Advis. Sec. Res. Doc. 2022/058. vi + 47 p.

Tollit, D., Joy, R., and Wood, J. 2017. Estimating the effects of noise from commercial vessels and whale watch boats on Southern Resident Killer Whales. Report prepared for the ECHO Program of Vancouver Fraser Port Authority. S.C.N. America.

Towers, J.R., Keen, E.M., Balcomb-Bartok, K., Vonick, J., and Davis, D. 2020. Live Strandings of Bigg's Killer Whales (*Orcinus orca*) Along the West Coast of North America. Aquatic Mammals **46**(5): 466-477.

Towers, J.R., Hallé, M.J., Symonds, H.K., Sutton, G.J., Morton, A.B., Spong, P., Borrowman, J.P., and Ford, J.K.B. 2018. Infanticide in a mammal-eating killer whale population. Scientific Reports **8**(1): 4366.

Towers, J.R., Sutton, G.J., Shaw, T.J.H., Malleson, M., Matkin, D., Gisborne, B., Forde, J., Ellifrit, D., Ellis, G.M., Ford, J.K.B., and Doniol-Valcroze, T. 2019. Photo-identification Catalogue, Population Status, and Distribution of Bigg's Killer Whales known from Coastal Waters of British Columbia, Canada. Can. Tech. Rep. Fish. Aquat. Sci. 3311: vi + 299 p.

Trites, A.W., and Donnelly, C.P. 2003. The decline of Steller sea lions *Eumetopias jubatus* in Alaska: a review of the nutritional stress hypothesis. Mammal Review **33**(1): 3-28.

US EPA. 2017. Technical Fact Sheet – Polybrominated Diphenyl Ethers (PBDEs) November 2017. Office of Land and Emergency (5106P). EPA 505-F-17-015.

Vagle, S., Burnham, R., Thupaki, P., Konrad, C., Toews, S., Thornton, S.J. 2021. Vessel presence and acoustic environment within Southern Resident Killer Whale (*Orcinus orca*) critical habitat in the Salish Sea and Swiftsure Bank area. DFO Can. Sci. Advis. Sec. Res. Doc. 2021/058. x + 66 p .

van Weelden, C., Towers, J.R., and Bosker, T. 2021. Impacts of climate change on cetacean distribution, habitat and migration. Climate Change Ecology **1**: 100009.

van Weelden, C., Tixier, P., Doniol-Valcroze, T., Guinet, C., and Towers, J.R. 2025. Divergent killer whale populations exhibit similar acquisition but different healing rates of conspecific scars. Behavioural Ecology and Sociobiology **79**(39).

Vanderlaan, A.S.M., and Taggart, C.T. 2007. Vessel collisions with whales: the probability of lethal injury based on vessel speed. Marine Mammal Science **23**(1): 144-156.

Veirs, S., Veirs, V., and Wood, J.D. 2016. Ship noise extends to frequencies used for echolocation by endangered killer whales. PeerJ **4**: e1657.

Villadsgaard, A., Wahlberg, M., and Tougaard, J. 2007. Echolocation signals of wild harbour porpoises, *Phocoena phocoena*. Journal of Experimental Biology **210**(1): 56-64.

Walker, T.R., Adebambo, O., Feijoo, M.C.D.A., Elhaimer, E., Hossain, T., Edwards, S.J., Morrison, C.E., Romo, J., Sharma, N., and Taylor, S. 2019. Environmental effects of marine transportation. *In* World Seas: An Environmental Evaluation Volume III: Ecological Issues and Environmental Impacts. *Edited by* C. Sheppard. Academic Press. pp. 505-530.

Walther, G.R., Post, E., Convey, P., Menzel, A., Parmesan, C., Beebee, T.J., Fromentin, J.M., Hoegh-Guldberg, O., and Bairlein, F. 2002. Ecological responses to recent climate change. Nature **416**(6879): 389-395.

Wang, Z., Chen, Z., Xu, S., Ren, W., Zhou, K., and Yang, G. 2015. ‘Obesity’ is healthy for cetaceans? Evidence from pervasive positive selection in genes related to triacylglycerol metabolism. Scientific Reports **5**(1): 14187.

Ward, E.J., Holmes, E.E., and Balcomb, K.C. 2009. Quantifying the effects of prey abundance on killer whale reproduction. Journal of Applied Ecology **46**(3): 632-640.

Watkins, W.A. 1986. WHALE REACTIONS TO HUMAN ACTIVITIES IN CAPE COD WATERS. Marine Mammal Science **2**(4): 251-262.

Weilgart, L.S. 2007. A brief review of known effects of noise on marine mammals. International Journal of Comparative Psychology **20**(2): 159-168.

Westdal, K.H., Higdon, J.W., and Ferguson, S.H. 2017. Review of killer whale (*Orcinus orca*) ice entrapments and ice-related mortality events in the Northern Hemisphere. Polar Biology **40**(7): 1467-1473.

White, G.C., and Burnham, K.P. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study **46**(sup1): S120-S139.

Williams, R., Trites, A.W., and Bain, D.E. 2002. Behavioural responses of killer whales (*Orcinus orca*) to whale-watching boats: opportunistic observations and experimental approaches. Journal of Zoology **256**(2): 255-270.

Williams, R., Erbe, C., Ashe, E., Beerman, A., and Smith, J. 2014. Severity of killer whale behavioral responses to ship noise: A dose–response study. Marine Pollution Bulletin **79**(1): 254-260.

Williams, R., Cholewiak, D., Clark, C.W., Erbe, C., George, C., Lacy, R., Leaper, R., Moore, S., New, L., Parsons, C., Rosenbaum, H., Rowles, T., Simmonds, M., Stimmelmayr, R., Suydam, R.S., and Wright, A. 2020. Chronic ocean noise and cetacean population models. J. Cetacean Res. Manage. **21**(1): 85-94.

Williams, R., Wright, A.J., Ashe, E., Blight, L.K., Bruintjes, R., Canessa, R., Clark, C.W., Cullis-Suzuki, S., Dakin, D.T., Erbe, C., Hammond, P.S., Merchant, N.D., O'Hara, P.D., Purser, J., Radford, A.N., Simpson, S.D., Thomas, L., and Wale, M.A. 2015. Impacts of anthropogenic noise on marine life: Publication patterns, new discoveries, and future directions in research and management. Ocean & Coastal Management **115**: 17-24.

Yordy, J.E., Pabst, D.A., McLellan, W.A., Wells, R.S., Rowles, T.K., and Kucklick, J.R. 2010. Tissue-specific distribution and whole-body burden estimates of persistent organic pollutants in the bottlenose dolphin (*Tursiops truncatus*). Environmental Toxicology and Chemistry **29**(6): 1263-1273.

Young, N. C., Brower, A. A., Muto, M. M., Freed, J. C., Angliss, R. P., Friday, N. A., Boveng, P. L., Brost, B. M., Cameron, M. F., Crance, J. L., Dahle, S. P., Fadely, B. S., Ferguson, M. C., Goetz, K. T., London, J. M., Oleson, E. M., Ream, R. R., Richmond, E. L., Shelden, K. E. W., Sweeney, K. L., Towell, R. G., Wade, P. R., Waite, J. M., and Zerbini, A. N. 2023. Alaska marine mammal stock assessments, 2022. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-AFSC-474, 316 p.

8 APPENDIX

Table A1. Acronyms appearing the document and their meaning.

Acronym	**Meaning**
AFR	Annual fecundity rates
BC	British Columbia
BKW	Bigg’s Killer Whale
CEA	Cumulative Effects Assessment
CI	Confidence interval
CJS	Cormack-Jolly-Seber
CRP	Cetacean Research Program
DFO	Fisheries and Oceans Canada
CSS	Canadian Salish Sea (subset)
LW	Lipid weight
OCPs	Organochlorine pesticides
PAH	Polycyclic aromatic hydrocarbons
PCBs	Polychlorinated Biphenyls
PoE	Pathways of Effects
POPs	Persistent Organic Pollutants
PCoD	Population Consequences of Disturbance
PVA	Population Viability Analysis
RKW	Resident Killer Whale
SARA	Species at Risk Act
SARKW	Southern Alaska Resident Killer Whale
SD	Standard deviation
ST	Sensitivity testing
WCT	West Coast Transient (population)
WDFW	Washington Department of Fish and Wildlife