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Ocean disposal in resident killer whale (*Orcinus orca*) Critical Habitat: Science in support of risk management

Immersion en mer dans les habitats essentiels des épaulards résidents (*Orcinus orca*) : la science à l'appui de la gestion des risques

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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

Resident killer whales in the coastal waters of British Columbia and Washington are heavily contaminated with persistent organic pollutants (POPs), including polychlorinated biphenyls (PCBs). The northern and southern populations of resident killer whales are listed, respectively, as threatened and endangered under the Canadian *Species at Risk Act* (SARA), which protects species at risk from being killed or harmed (section 32) and protects any part of their Critical Habitat from destruction (section 58). The Resident Killer Whale Recovery Strategy identified contaminants, reduced prey and disturbance (noise and physical) as threats to population recovery.

Sediment dredged from the lower Fraser River and other locations is periodically disposed of at marine sites in coastal British Columbia, both within and outside killer whale Critical Habitat. Ocean disposal is regulated by Environment Canada under the *Canadian Environmental Protection Act* (CEPA). Sediments contain complex mixtures of contaminants, and material intended to be disposed of at sea is screened for a select list contaminants. Because killer whales are long-lived and occupy a very high trophic level, they are at particular risk to accumulating high concentrations of PCBs and related compounds.

This report provides a response to the following questions identified by Fisheries and Oceans Canada SARA and Habitat Managers**:** *Are current Ocean Disposal Rejection/Screening Limits for environmental contaminants (including PCBs, mercury and PAHs) under CEPA 1999 adequate to prevent northern and southern resident killer whale Critical Habitat from destruction, as required by SARA Section 58?;* and, *Do PCBs in materials deposited in any area of killer whale habitat increase the risk of harm or mortality of northern and southern resident killer whales, as required by SARA Section 32?*

In order to answer these questions given the complexity of killer whale ecology and that of their primary prey (Chinook salmon), we developed a novel food web modeling tool. This effort includes the following components: i) the designation of seven geographic areas that relate to management-related priorities (e.g. Critical Habitat) and/or international boundaries in the NE Pacific Ocean; ii) an assignment of time spent in each of these areas by southern and northern resident killer whales and their prey (Chinook salmon and non-salmonid species) based on best available information; iii) the adaptation of sediment-biota PCB bioaccumulation models to killer whales and their prey; iv) a compartmentalized approach to modeling sediment-food web uptake of PCBs within each of the seven areas identified so as to be able to evaluate site-specific impacts of disposal operations; and v) a comparison of model outcomes to three established health effects thresholds for PCBs in marine mammals. The modeling approach is based on the distribution of PCBs among sediments, the water column, and biota, and estimates concentrations that will accumulate in animals throughout a lifetime of exposure. This model does not evaluate the existing PCB distribution and pathways in the BC abiotic environment, but rather predicts the incremental consequences to killer whales of altering bottom sediments to reflect PCB concentrations following a series of disposal scenarios.

Our model indicates that disposal of material with PCB concentrations lower than those in the ambient sediments in the disposal sites would not increase PCB delivery to killer whales. However, the disposal of sediments into Critical Habitat sites from some of the more contaminated sites for which data are available could increase the PCB concentrations in killer whales by as much as 8%. These predictions assume that the sediments and water were in equilibrium, and that the Strait of Georgia was a closed system. Some processes may decrease uptake of PCBs by the food web relative to the model result (e.g. burial by sedimentation and exchange with the open ocean), while others may increase the uptake (e.g. direct uptake of PCBs by food web during disposal operations).

Under current practices, the scrutiny and approval of disposal applications in the Pacific Region is constrained by shortcomings in analytical measurement standards and by the definition of the CEPA Action Level based on only effects at low trophic levels. High Resolution instrumental analysis would reduce Detection Limits by up to 350 times, and improve risk management assessment of disposal applications. The current CEPA Action Level is too high to protect killer whales because PCBs biomagnify. Disposal of sediment containing PCBs at a concentration matching the current CEPA Action Level could hypothetically lead to a 32-fold increase in PCB levels in male southern resident killer whales.

We derived a sediment PCB concentration range that would protect 95% of resident killer whales of 0.012 to 0.200 $\mu g \cdot kg^{-1}$, dry weight. Results reveal the profound vulnerability of killer whales to accumulation of persistent contaminants, since only 4/61 (6.6%) sediment sites for which we have PCB measurements in BC and Washington State fall below the least protective of these sediment values. This suggests that there continues to be widespread contamination of resident killer whale habitat by the legacy PCBs.

This newly developed food web model can be employed as a risk management tool in support of SARA protections for killer whales. We suggest that the ambient sediment PCB concentration becomes an important benchmark for a management-based evaluation of risks to killer whales and to killer whale Critical Habitat. Disposal of materials with PCB concentrations lower than ambient in Critical Habitat in areas of high sedimentation will not increase sediment PCB concentrations, might help to bury contaminated sediments, and should not lead to increased PCB concentrations in killer whales. The decision about whether or not to dispose of materials with PCB concentrations that exceed ambient levels in the marine environment, particularly in Critical Habitat, will have consequences for killer whales.

RÉSUMÉ

Les épaulards résidents des eaux côtières de la Colombie-Britannique et de l'État de Washington sont fortement contaminés par les polluants organiques persistants (POP), y compris les biphényles polychlorés (BPC). Les populations du Nord et du Sud d'épaulards résidents sont inscrites en tant que populations menacées et en voie de disparition respectivement en vertu de la *Loi sur les espèces en péril* (LEP) du Canada, laquelle interdit de tuer ou de harceler un individu d'une espèce inscrite (article 32) et protège l'ensemble de l'habitat essentiel de ces espèces contre toute destruction (article 58). La stratégie de rétablissement de l'épaulard résident indique que les contaminants, la réduction des proies et les perturbations (bruits et physiques) sont des menaces au rétablissement de la population.

Les sédiments dragués dans la partie inférieure du fleuve Fraser et d'autres emplacements sont immergés périodiquement dans des sites marins des eaux côtières de la Colombie-Britannique, tant à l'intérieur qu'à l'extérieur de l'habitat essentiel de l'épaulard. L'immersion en mer est réglementée par Environnement Canada en vertu de la *Loi canadienne sur la protection de l'environnement* (LCPE). Les sédiments contiennent des mélanges complexes de contaminants et le matériel destiné à l'immersion en mer est examiné afin d'établir une liste de contaminants précis. Étant donné que les épaulards sont des animaux longévifs et qu'ils occupent un niveau trophique très élevé, ils sont exposés à un risque particulier d'accumuler de fortes concentrations de BPC et de substances voisines.

Ce rapport fournit des réponses aux questions suivantes définies par les gestionnaires de l'habitat et de la LEP de Pêches et Océans Canada **:** *Les limites applicables à la sélection et au rejet en mer à des fins d'immersion actuelles concernant les contaminants environnementaux (incluant les BPC, le mercure et les hydrocarbures aromatiques polycycliques (HAP)) aux termes de la LCPE 1999 sont-elles adéquates pour empêcher la destruction de l'habitat essentiel des populations du Nord et du Sud d'épaulards résidents, conformément à l'article 58 de la LEP?; les BPC présents dans les matériaux déposés dans un secteur quelconque de l'habitat des épaulards augmentent-ils le risque d'effets néfastes ou de mortalité chez les populations du Nord et du Sud d'épaulards résidents, aux termes de l'article 32 de la LEP?*

Afin de répondre à ces questions, et étant donné la complexité de l'écologie de l'épaulard et de celle de sa principale proie (saumon quinnat), nous avons élaboré un nouvel outil de modélisation dans le réseau tropique. Cet effort comprend les composantes suivantes : i) la désignation de sept zones géographiques en lien avec les priorités en matière de gestion (p. ex. l'habitat essentiel) et/ou les limites internationales dans l'océan du Pacifique Nord-Est; ii) une attribution du temps passé dans chacune de ces zones par les populations du Sud et du Nord d'épaulards et leurs proies (saumon quinnat et espèces autres que les salmonidés) d'après les meilleures données disponibles; iii) l'adaptation de modèles de la bioaccumulation des BPC dans les sédiments et le biote chez les épaulards et leurs proies; iv) une approche compartimentée de modélisation des prélèvements de BPC dans les réseaux trophiques/sédiments pour chacune des sept zones définies afin d'être en mesure d'évaluer les impacts de l'activité d'immersion à des sites particuliers; v) une comparaison des résultats des modèles avec trois seuils d'effet sur la santé établis pour les BPC chez les mammifères marins. L'approche de modélisation est fondée sur la distribution des BPC dans les sédiments, la colonne d'eau et le biote et pour estimer les concentrations qui peuvent s'accumuler chez les animaux au cours de leur vie entière. Ce modèle n'évalue pas la distribution existante des BPC et les voies empruntées dans le milieu abiotique de la Colombie-Britannique, mais prédit plutôt les conséquences additionnelles pour les épaulards de la modification des sédiments des fonds marins pour refléter les concentrations de BPC après une série de scénarios d'immersion.

Notre modèle indique que l’immersion de matériaux contenant des concentrations de BPC inférieures à celles des sédiments ambiants dans les sites d’immersion ne devrait pas accroître les apports de BPC chez les épaulards. Cependant, l’immersion de sédiments dans les sites d’habitat essentiel à partir de certains des sites les plus contaminés au sujet desquels des données sont disponibles pourrait entraîner une augmentation des concentrations de BPC chez les épaulards pouvant atteindre jusqu’à 8 %. Ces prévisions supposent que les sédiments et l’eau étaient en équilibre et que le détroit de Georgie était un système fermé. Certains processus peuvent entraîner une diminution de l’absorption de BPC par le réseau trophique par rapport au résultat du modèle (p. ex. enfouissement par sédimentation et échange avec la pleine mer), tandis que d’autres peuvent l’augmenter (p. ex. une absorption directe de BPC par le réseau tropique pendant les activités d’immersion).

Selon les pratiques actuelles, l’examen minutieux et l’approbation des applications d’immersion dans la région du Pacifique sont limités par les lacunes des étalons de mesures analytiques et par la définition du seuil d’intervention aux termes de la LCPE, fondé seulement sur des effets à des niveaux tropiques inférieurs. Une analyse instrumentale à haute résolution permettrait de réduire les limites de détection jusqu’à 350 fois et d’améliorer l’évaluation de la gestion des risques des applications d’immersion. Le seuil d’intervention actuel en vertu de la LCPE est trop élevé pour protéger les épaulards, car les BPC subissent une bioamplification. L’immersion de sédiments contenant des BPC à une concentration correspondant au seuil d’intervention actuel aux termes de la LCPE pourrait hypothétiquement entraîner une augmentation 32 fois plus importante des concentrations de BPC dans la population du Sud d’épaulards résidents mâles.

Nous avons trouvé qu’une fourchette de concentration de BPC dans les sédiments allant de 0,012 à 0,200 $\mu g \cdot kg^{-1}$ en poids sec protégerait 95 % des épaulards résidents. Les résultats montrent la grande vulnérabilité des épaulards quant à l’accumulation de contaminants persistants, étant donné que seulement 4 sites de sédiments sur 61 (6,6 %) pour lesquels nous connaissons les concentrations de BPC en Colombie-Britannique et dans l’État de Washington affichent des concentrations inférieures aux concentrations les moins prudentes dans les sédiments. Ces données laissent entendre que la contamination à grande échelle de l’habitat de l’épaulard résident par les BPC traditionnels se poursuit.

Ce nouvel outil de modélisation peut servir d’outil de gestion des risques pour appuyer la protection de l’épaulard en vertu de la LEP. Nous proposons que les concentrations de BPC dans les sédiments ambiants deviennent un point de référence important dans l’évaluation des risques fondée sur la gestion pour les épaulards et leur habitat essentiel. L’immersion de matériaux contenant des concentrations de BPC inférieures aux concentrations ambiantes de l’habitat essentiel dans les milieux d'intense sédimentation n’augmentera pas les concentrations de BPC dans les sédiments, pourrait contribuer à enfouir les sédiments contaminés et ne devrait pas entraîner une augmentation des concentrations de BPC chez les épaulards. La décision d’immerger ou non les matériaux contenant des concentrations de BPC supérieures aux niveaux ambiants dans le milieu marin, particulièrement dans l’habitat essentiel, aura des conséquences sur les épaulards.

1. INTRODUCTION

The marine environment of southern British Columbia (BC), Canada, and northern Washington State (WA), USA, is valued for its abundant wildlife, physical beauty, recreational opportunities, and fishing. These waters are frequented by three ecotypes of killer whales: resident, transient, and offshore (Ford *et al.* 1998). Resident killer whales are further distinguished as northern residents (NRKW) that are often found in the waters off northeast Vancouver Island, BC, and southern residents (SRKW) that are often found in the waters off southeast Vancouver Island (Figure 1) (Ford *et al.* 1998).

In 2001, SRKWs were listed as Endangered under the Canadian *Species at Risk Act* (SARA; Government of Canada 2010b), and in 2005 under the United States *Endangered Species Act* (NOAA 2010). The NRKW population is listed as Threatened in Canada (Government of Canada 2010a). Critical Habitat has been identified for both populations (Figures 2 and 3) and an evaluation of the threats to both the individuals and their Critical Habitat is currently under way. As part of this process, Fisheries and Oceans Canada (DFO) is evaluating the impact of Disposal at Sea to resident killer whales and their Critical Habitats.

While Environment Canada (EC) oversees disposal at sea activities under the terms of the Canadian Environmental Protection Act (CEPA), consistent with the principles of the London Convention, DFO has historically provided EC with habitat-based advice on disposal permits. The purpose of this report is to provide DFO with science-based tools to better understand the contaminant-related risks associated with disposal operations, particularly as they relate to SARA, such that such DFO permit reviews are better positioned to protect killer whales and their Critical Habitat from contaminant impacts.

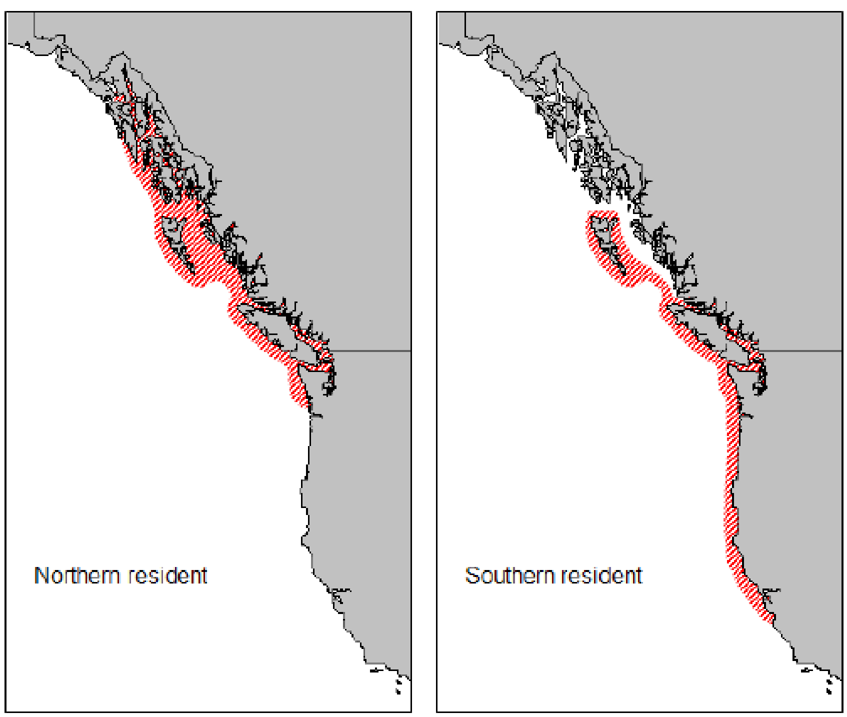


Figure 1: The currently-known geographical ranges of northern (left) and southern (right) resident killer whales (Ford 2006). The extent that the killer whales travel offshore is unknown.

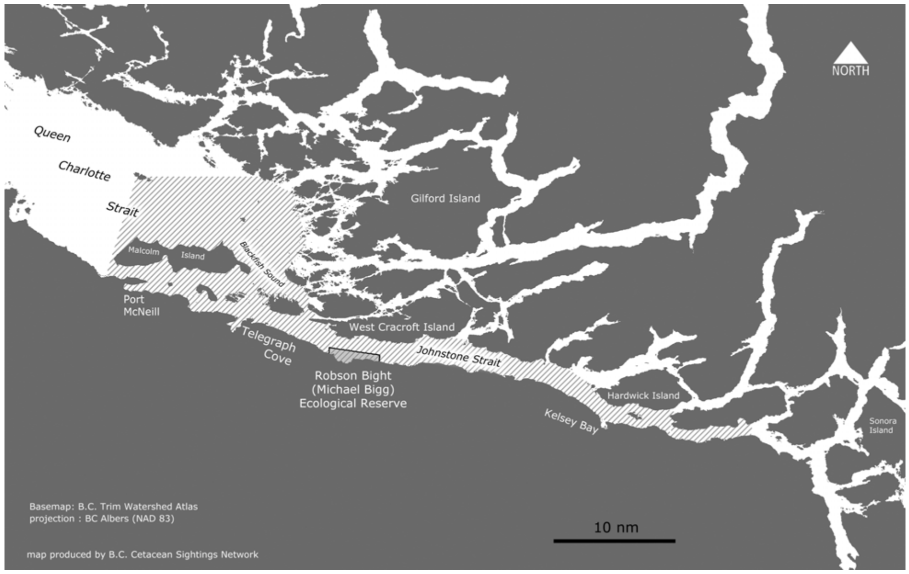


Figure 2: Northern resident killer whale Critical Habitat in British Columbia (Fisheries and Oceans Canada 2008).

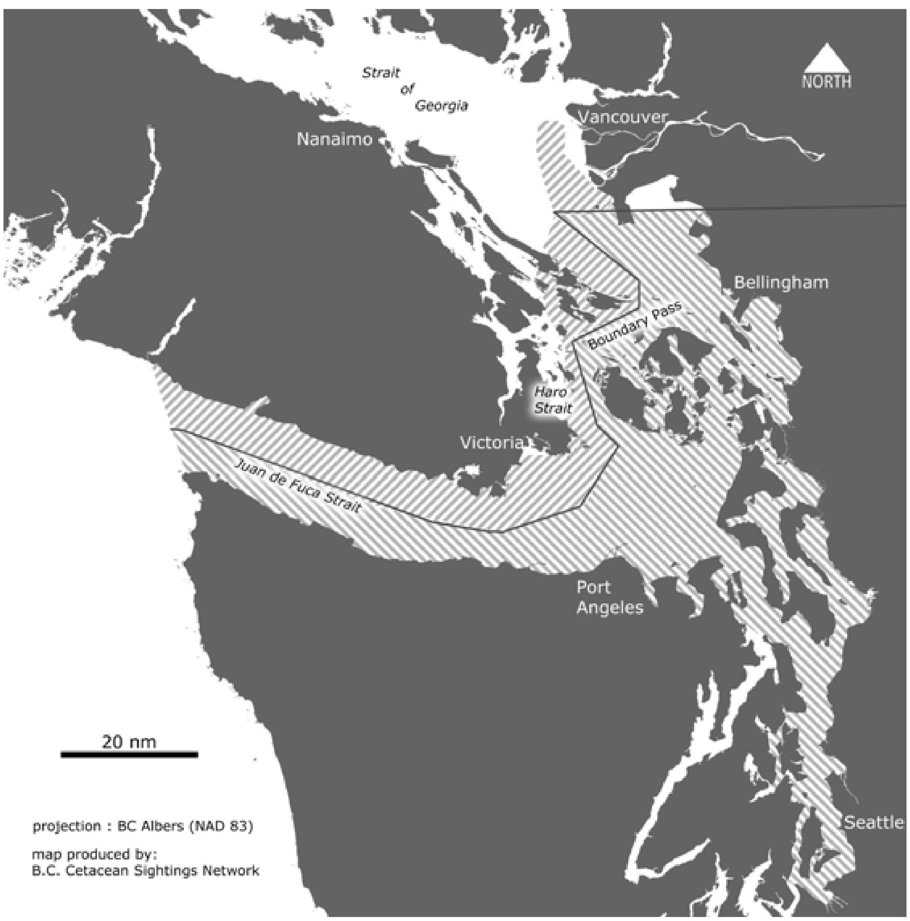


Figure 3: Southern resident killer whale Critical Habitat in Canada and the United States (Fisheries and Oceans Canada 2008). The hatched area in US waters depicts the approximate Critical Habitat under the US Endangered Species Act.

Critical Habitat is defined as the habitat necessary for survival or recovery of a listed wildlife species at risk, as identified in a Recovery Strategy or Action Plan. For resident killer whales, two such areas were identified largely based on their structure in funneling primary prey (Chinook salmon). Because Critical Habitat is so important for feeding, resident killer whales spend a considerable portion of their lives in these areas. This has implications for the disposal of potentially contaminated materials into that habitat. Under SARA, Critical Habitat is legally protected from destruction, and advice from Science is needed to justify management decisions designed to protect all of resident killer whale habitat under the *Fisheries Act*.

One of the three main threats that both populations of resident killer whales face is exposure to contaminants. PCB concentrations measured in adult northern and southern resident killer whales range from 9,300-146,000 $\mu g \cdot kg^{-1}$ lipid weight, with these killer whales among the most PCB-contaminated marine mammals in the world (Ross *et al.* 2000a). The PCB levels measured in the resident killer whales readily exceed thresholds that range from 10,000-77,000 $\mu g \cdot kg^{-1}$ PCB in blubber or liver lipids for the onset of adverse health effects determined for other marine mammals (Hall *et al.* 2006;Kannan *et al.* 2000;Reijnders 1986;Ross *et al.* 1996c).

PCBs do not generally cause outright mortality in exposed individuals, but rather are considered as ‘endocrine disruptors’ or ‘hormone mimics’. The health risks presented by PCBs are implied by the ‘weight of evidence’ from carefully controlled, mechanistic, single chemical, dosing studies (Ross 2000;Ross *et al.* 2000b;Ross and Birnbaum 2003). Such studies include laboratory animals (Morse 1995), accidentally exposed humans (Kamps *et al.* 1978), First Peoples and fishing cohorts exposed to high PCB concentrations through increased consumption of fish (Dewailly *et al.* 1993;Jacobson and Jacobson 1996), fish-eating seabirds (Tillitt *et al.* 1992), and high trophic level marine mammals (Ross *et al.* 1995).

The advantages of conducting lab-based studies include the demonstration that the chemical in question is causing an adverse effect, while the disadvantage is that such study designs lack environmental relevance (Figure 4). The advantage of field studies (e.g. using fish-eating birds and mammals) is that they incorporate real world relevance, but are confounded by the presence of many other contaminants of possible concern, and additional factors. By using a ‘weight of evidence’ for killer whales which considers the results of a variety of these study designs, one is better positioned to assess the risks associated with PCB exposures. Extrapolation and a ‘weight of evidence’ approach are rationalized by the conserved nature of many physiological systems across vertebrates (e.g. immune systems, reproductive systems, hormones, hormone receptors). Simply put, despite some obvious differences, killer whales share much with laboratory rodents and humans. In summary, a ‘weight of evidence’ approach offers a defensible means to:

- prioritize the many different POPs found in killer whales;
- guide an assessment of the potential for POP-related health effects;
- provide a list of possible biological outcomes related to POP exposure; and
- provide managers with benchmarks against which decisions and/or actions can be made.

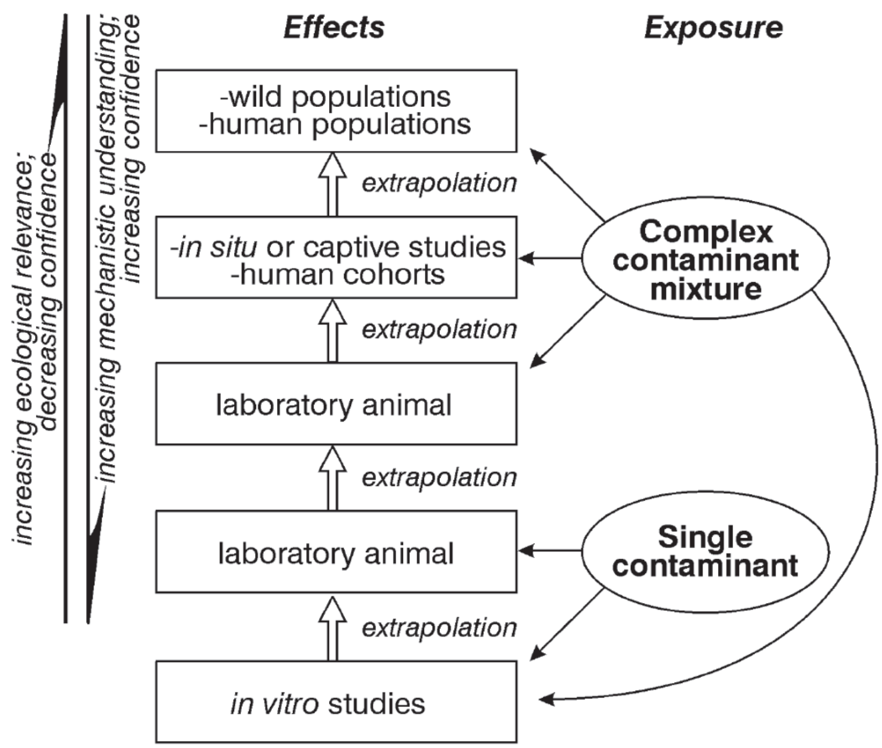


Figure 4: A 'weight of evidence' from multiple lines of research best enables a characterization of health risks associated with PCB exposure in killer whales. From (Ross and Birnbaum 2003).

Evidence that PCBs have adverse effects come from a number of studies and study designs:

1) associative evidence, where observations of mortalities, malformations, or reduced reproduction are observed in highly contaminated populations (De Guise *et al.* 1995;Delong *et al.* 1973;Mortensen *et al.* 1992);
2) correlative evidence, where correlations between physiological endpoints and PCB concentrations suggest a causal link between the two (Mos *et al.* 2006a;Tabuchi *et al.* 2006);
3) captive feeding studies, where two groups of marine mammals (harbour seals) have been fed fish diets containing different concentrations of contaminants over a long period, resulting in reduced reproduction, immune function and hormone levels (Brouwer *et al.* 1989;De Swart *et al.* 1996;Reijnders 1986;Ross *et al.* 1996c).

Characterizing the effects of PCBs on the health of killer whales is not easy, being constrained by legal, ethical, logistial and scientific hurdles. However, reduced reproductive success and higher-than-expected mortality among southern residents highlights concerns expressed in the Recovery Strategy, the need for additional research, and the rationale for a precautionary, 'weight of evidence' approach. Preliminary results of a genomics-based approach to evaluating the effects of PCBs on 16 physiological systems using biopsies of free-ranging killer whales strongly suggests that PCBs are affecting resident killer whales, although the implications for reproduction and mortality at the population level are unclear (Buckman *et al.* 2008; P.S. Ross, personal communication).

PCBs and DDT have dominated concerns about population-level impacts of contaminants on high trophic level wildlife. This reflects their high concentrations relative to other persistent contaminants in high trophic level wildlife, and their toxicity under a variety of study designs. A risk-based evaluation of 13 organochlorine contaminants found in BC harbour seals concluded that PCBs represented the overwhelming toxic concern, followed by DDT as a distant second (Mos *et al.* 2010).

The conservation-level implications of these findings are twofold. Firstly, where are these contaminants coming from and why are killer whales so contaminated? Secondly, what are the health implications for such findings? The answers to these questions are elusive due to complex habitat requirements of killer whales and the difficulties in conducting research on these large, protected mammals. However, since first reporting the high PCB levels in these killer whales, much has been done to generate insight into these questions.

The coastal waters of BC and WA are impacted by many anthropogenic activities such as shipping and transportation, pulp mill discharges, and mining and municipal wastewater effluents. This is primarily because the coastal waterways of BC and WA are surrounded by the large urban centres of metro Vancouver (population ~2.3 million; Statistics Canada 2010), Victoria (population ~352,000; Statistics Canada 2010), King County (including Seattle; population 1.9 million; US Census Bureau 2010), and Tacoma (population ~197,000; US Census Bureau 2010). The marine sediments in this region provide a record of historical contamination, as they represent a 'sink' for a variety of contaminants, including heavy metals such as lead and mercury (Johannessen *et al.* 2005a;Long *et al.* 2005;Macdonald *et al.* 1991), and a number of anthropogenic products or by-products such as dioxins and furans (Long *et al.* 2005;Macdonald *et al.* 1992), polychlorinated biphenyls (Johannessen *et al.* 2008a;Long *et al.* 2005), polycyclic aromatic hydrocarbons (Long *et al.* 2005;Yunker and Macdonald 2003), and industrial detergents (Shang *et al.* 1999).

Certain hydrophobic contaminants can partition into the organic carbon fraction of sediments. These include lead, mercury, some PAHs, PCBs, PCDDs, PCDFs, and organochlorine pesticides (MacDonald *et al.* 2003). PCBs and other contaminants with high octanol-water partition coefficients (Kow; Streets *et al.* 2006) and greater chlorination (Ross *et al.* 2000a) tend to bioaccumulate in lipids of biota and biomagnify (concentrations increase with trophic level in the food web), which is especially problematic for long-lived species such as killer whales (Krahn *et al.* 2007;Rayne *et al.* 2004;Ross *et al.* 2000a). Figure 5 depicts the fate of persistent organic contaminants in the environment and biota. Contaminants that do not biomagnify (i.e., PAHs, and cadmium) are not a concern for killer whales. Of emerging concern are PBDEs, which currently do not have Sediment Quality Guidelines, established models of their fate in the environment, and have many data gaps. Figure 6 depicts how total PBDE concentrations in harbour seals sampled on Gertrude Island in Puget Sound are increasing rapidly over time (Ross *et al.* 2008).

Metals do not generally biomagnify in aquatic food webs, although mercury (Hg) is a notable exception. In its organic (methylated) form, Hg can attain high concentrations in upper trophic levels and reach levels of biological concern in some species (Wolfe et al. 1998). For example, Hg in long-lived, high trophic level fish or marine mammals sometimes exceeds human consumption guidelines. The toxic risks associated with the relatively high Hg concentrations observed in some toothed marine mammals is unclear, since a gathering weight of evidence suggests that such species have evolved the ability to detoxify Hg (Koeman *et al.* 1973;Pelletier 1985). However, the detoxification potential in marine mammals may be superceded by the rapidly increasing levels of environmental Hg attributable to human activities over the last century or two (Jackson 1997;Johannessen *et al.* 2005a). There is at present no information on Hg concentrations in the killer whales of the NE Pacific Ocean, although increased evaluation of sediment and food web-associated Hg in killer whale habitat would be of value.

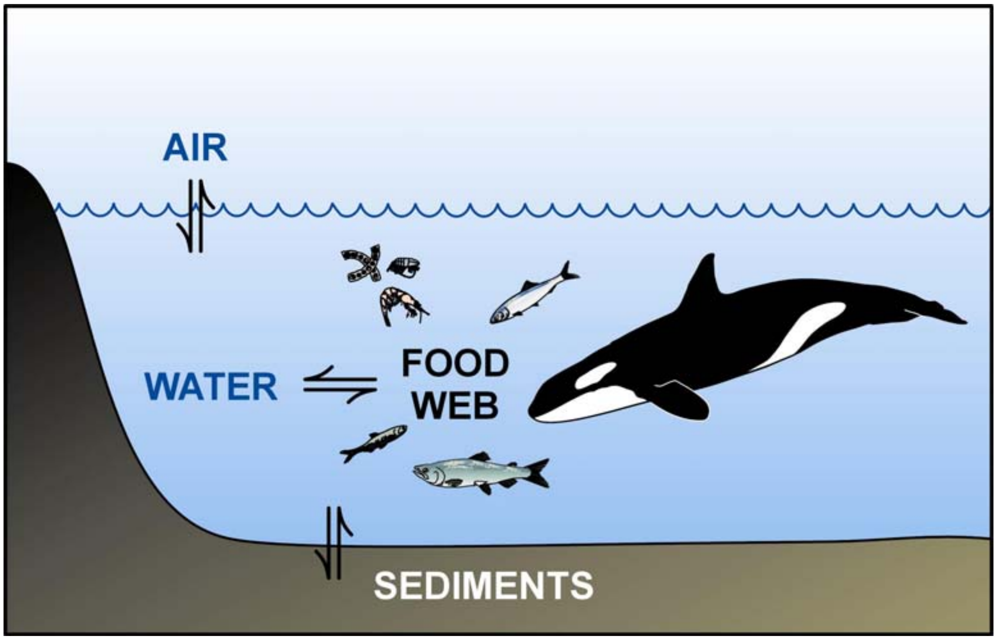


Figure 5: Persistent organic pollutants partition among compartments in the marine environment and are magnified in the food web (Ross, unpublished).

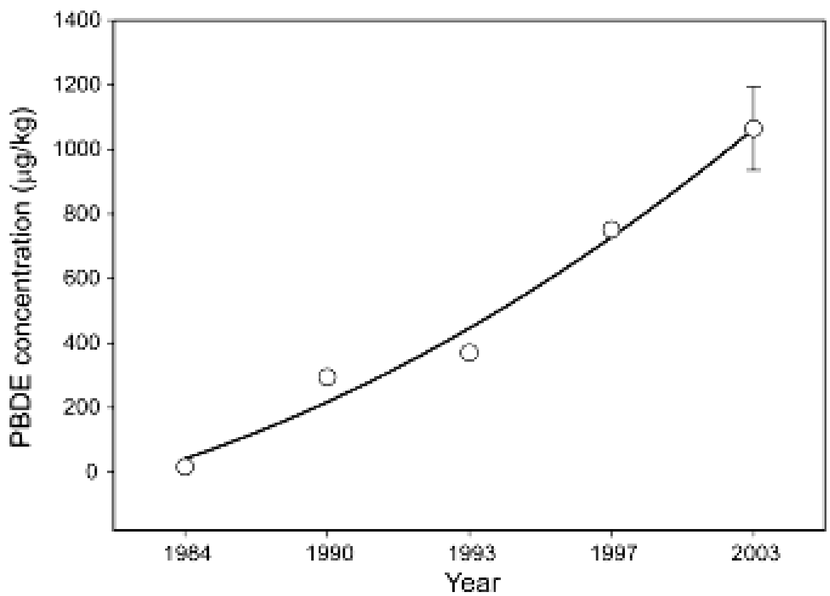


Figure 6: Total PBDE concentrations in harbour seals have increased on Gertrude Island in Puget Sound, underscoring concerns about this emerging contaminant (Ross et al. 2008).

Residue-based Sediment Quality Guidelines (SQGs) have been developed to account for bioaccumulation (Ingersoll and MacDonald 2003). A residue-based SQG is the maximum concentration of a compound or class of compounds in sediment predicted to produce tolerable levels in the tissues of marine organisms (i.e., below levels associated with adverse health effects) (Ingersoll and MacDonald 2003). Tissue residue guidelines (TRGs) are divided by biota-sediment accumulation factors (BSAFs) for specific compounds to provide the residue-based SQGs (Cook *et al.* 1992;NYSDEC 1999). Controversy has arisen over their use due to the large variation that exists among species, chemicals, and exposure conditions (Meador 2006). However, tissue concentrations used to identify toxic responses are generally less variable than responses derived from exposure concentrations, because of the reduction in toxicokinetic variability (Meador 2006). Residue-based SQGs also allow grouping of contaminants by mode of action, for both acute and chronic responses (McCarty and Mackay 1993). However, it can be difficult to determine realistic residue-based SQGs for compounds that can be metabolized (e.g., PAHs), especially when the metabolites also produce toxic responses (Meador 2006). The Canadian Council of Ministers of the Environment (CCME) set a mammalian tissue residue guideline for dioxin-like PCBs for the protection of wildlife consumers of aquatic biota at 0.79 ng $TEQ \cdot kg^{-1}$ diet wet weight (CCME 2001). This dioxin-like equivalent concentration has been used to approximate an equivalent total PCB guideline of 50 $\mu g \cdot kg^{-1}$ for the prey of wildlife consumers (Hickie et al. 2007).

1.1 *SPECIES AT RISK ACT* (SARA) AND *CANADIAN ENVIRONMENTAL PROTECTION ACT* (CEPA)

The *Species at Risk Act* (SARA) is a commitment (legal protection) by the Canadian federal government to preventing wildlife from going extinct which provides actions for species recovery. The Recovery Strategy for resident killer whales (Fisheries and Oceans Canada 2008) identified four threats, described Critical Habitat, listed activities likely to destroy Critical Habitat, defined recovery goals, and set objectives (including a contaminant objective). Disposal at sea was not flagged as a threat to Critical Habitat in the Recovery Strategy, but the analyses in this document relate to answering questions regarding disposal at sea's relevance to Critical Habitat destruction and the contaminant objective in the Recovery Strategy. The relevant sections of SARA for disposal at sea are sections 32 (risk of harm or mortality to individuals) and 58 (destruction of Critical Habitat) (see Appendix I).

Destruction of Critical Habitat is determined on a case by case basis. Destruction would result if part of the Critical Habitat were degraded, either permanently or temporarily, such that it would not serve its function when needed by the species. Destruction may result from a single or multiple activities at one point in time or from the cumulative effects of one or more activities over time. When Critical Habitat is identified in a Recovery Strategy or an Action Plan, examples of activities that are likely to result in its destruction will be provided. Thus this exercise examining the impacts of disposal at sea in Critical Habitat arose from the legal requirements for protection of the endangered SRKWs and threatened NRKWs.

Bioaccumulative contaminants present in the sediments of Critical Habitat of resident killer whales degrade habitat quality since a portion of those contaminants may not remain in the sediments but, rather, enter the food web and biomagnify at each trophic level, reducing prey quality, prey health, and ultimately killer whale health.

Environment Canada regulates Disposal at Sea in Canadian waters and ensures that the *London Convention of 1972* (London Convention 1996) is adhered to through a permit system under the *Canadian Environmental Protection Act* (CEPA), and in particular, the *Disposal at Sea*

Regulations (Porebski and Osborne 1998). Under CEPA 1999, Environment Canada is required to monitor representative disposal sites each year, which involves physical, chemical, and biological monitoring (Environment Canada 2006). From 1976 – 1991, chemical screening was the only criterion used to classify sediments to be disposed at sea (Canada Gazette 2001). However, effects-based chemical guidelines including toxicity, persistence, and bioaccumulation, are required to complement some chemical monitoring (Porebski and Osborne 1998). Currently, CEPA uses two Action Levels to evaluate material proposed to be disposed of at sea. Action Level Low is a chemical screening to determine whether contaminant levels are low enough to be of no concern (CEPA 2001;Environment Canada 2006). Lower Action Levels exist for mercury (750 $\mu g \cdot kg^{-1}$, dry weight), cadmium (600 $\mu g \cdot kg^{-1}$ dry weight), total PCBs (100 $\mu g \cdot kg^{-1}$ dry weight, Aroclor-based), and total PAHs (2,500 $\mu g \cdot kg^{-1}$ dry weight) (Environment Canada 2006). Any sediment with concentrations above the Lower Action Level is assessed with: (1) an acute lethality test, (2) two sub-lethal tests or (3) one sub-lethal test and one bioaccumulation test. If the acute lethality test or the other two tests fail to meet the criteria set out for those tests, then the sediments shall be considered to be above the Upper Level of the National Action List, and disposal at sea is prohibited (CEPA 2001;Environment Canada 2006).

PCB concentrations in most surface sediments of the Strait of Georgia and Puget Sound determined from a limited number of samples (Figure 7) are below CEPA Action Level Low for total PCBs (100 $\mu g \cdot kg^{-1}$ dry weight) and the CCME ISQG for total PCBs (21.5 $\mu g \cdot kg^{-1}$ dry weight). The Figures shows widespread distribution of legacy PCBs in the coastal marine sediments of BC and Washington, which needs to be taken into account when evaluating contamination of dredge spoil. In addition, local hotspots are evident in Figure 7, potentially reflecting continued inputs from historical urban and industrial PCB sources. Only some of the data displayed in Figure 7 was used in this report (Table 4).

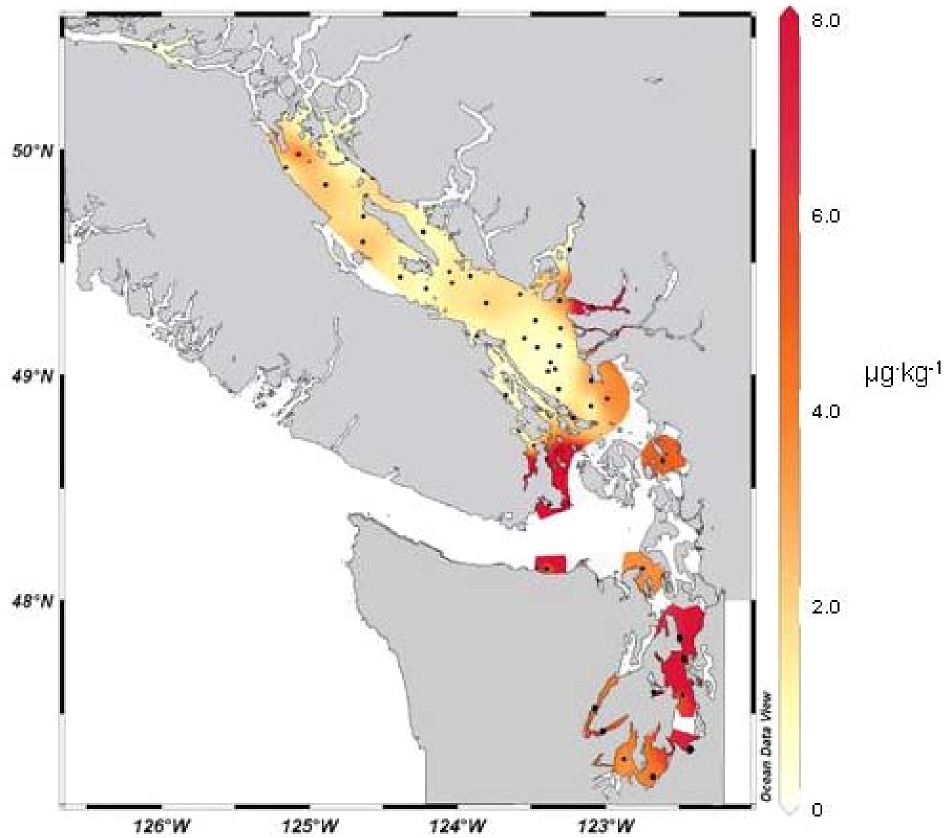


Figure 7: Contour plot of PCB concentrations (µg·kg^{-1}, dry weight) in the Strait of Georgia, with Canadian data from (Grant et al. 2010) and US data from Puget Sound Ambient Monitoring Program (Washington State Department of Ecology 2002).

1.2 REQUEST FOR ADVICE

This report provides a response to the following Request for Advice solicited by Fisheries and Oceans Canada**:**

> *Are current Ocean Disposal Rejection/Screening Limits for environmental contaminants (including PCBs, mercury and PAHs) under CEPA 1999 adequate to prevent northern and southern resident killer whale Critical Habitat from destruction, as required by SARA Section 58?*

A secondary question concerning threats to individual resident killer whales listed under SARA followed:

> *Do PCBs in materials deposited in any area of killer whale habitat increase the risk of harm or mortality of northern and southern resident killer whales, as required by SARA Section 32?*

Additional, more technical questions were addressed:

- What contaminants, or concentrations thereof, in disposal materials might represent destruction of killer whale Critical Habitat as per SARA Section 58, or present a risk of harm or mortality to individuals as per SARA Section 32?

- Are current analytical standards as applied in the Pacific Region under Disposal at Sea operations sufficient to enable science-based advice under the terms of SARA protection orders and/or permitting in Critical Habitat?
- What are the estimated threshold concentrations of contaminants (notably PCBs) in sediments or disposal materials that would be considered as adequate to prevent negative effects on killer whale health and destruction of Critical Habitat?
- Can we detect a contribution of ambient sediment-associated and/or disposal-associated PCBs in killer whale Critical Habitat using a food web bioaccumulation modeling approach?
- Can we attribute PCBs in northern and southern resident killer whale habitat areas used by killer whales and their prey?

Questions considered that have more bearing on management and policies include:
- Are new thresholds for sediment quality and/or disposal screening necessary to protect resident killer whale Critical Habitat (SARA Section 58) and/or health of individuals (SARA Section 32)?
- Is there adequate information to develop a set of basic guiding principles for disposal practices and/or disposal site selection that would reduce contaminant risks to killer whale Critical Habitat to avoid Section 58 destruction and/or killer whale health to avoid Section 32 harm or mortality?

We used a tiered approach to answer the questions posed. Tier 1 of this report investigates the impact of disposal at sea at the five disposal at sea sites in southern and northern resident killer whale Critical Habitat. Tier 2 investigates the threats from this activity outside Critical Habitat but within the summer feeding habitat. Tier 3 evaluates threats outside general range of resident killer whales, in order to provide advice on chemical regulation, and global/background sources (e.g., atmospheric transport from Asia) of contaminants imported into the salmon/coastal food web (Christensen *et al.* 2008;Cullon *et al.* 2009a;Noël *et al.* 2008). The impacts of Tier 3 are managed under the Stockholm Convention. The focus of this report is restricted to those contaminants deemed persistent, bioaccumulative, and toxic (PBT), for which only PCBs presently have disposal guidelines. This, combined with the high level of concern regarding PCBs in SARA-listed killer whales, provides the foundation for this report.

1.3 OBJECTIVES OF STUDY

British Columbia's killer whales have been identified as among the world's most PCB-contaminated marine mammals, surpassing the endangered St Lawrence beluga whales (*Delphinapterus leucas*) by a factor of 2-5 times (Ross *et al.* 2000a). Legal, logistical and ethical constraints prevent mechanistic toxicological studies from being carried out on killer whales, and constrain our ability to determine the precise health impacts of their very high PCB burdens. The list of obstacles for an assessment of population-level consequences of high PCB exposures is long.

Killer whales are:
- exposed to a complex mixture of contaminants;
- long-lived, meaning that they are exposed to a cumulative history of chemical use;
- have large habitat needs as do their primary prey (Chinook salmon);
- difficult to study, such that collecting blood (or many other tissue samples) for toxicological evaluation is not possible;

- protected under the terms of SARA in BC waters.

Both northern and southern resident killer whales were listed under SARA/COSEWIC because their populations are threatened. With the development of accurate population and demographic estimates through photo-identificationin the early 1970s (Bigg 1982;Bigg *et al.* 1990;Ford *et al.* 1994;Ford *et al.* 2000a) it has been found that the northern resident populations experienced a 2.44% increase in population numbers per year between 1974 and 2003 compared to just 0.71% per year between 1973 and 2003 for the southern residents (Fisheries and Oceans Canada 2008). This is explained by southern residents having lower female age at sexual maturity (as indicated by estimated female age at first successful calf), apparently reduced reproductive females among their peers, and higher mortality rates, compared to northern residents (Olesiuk et al. 1990). A shortage of chinook salmon has been highlighted as a major driver of birth and mortality rates among resident killer whales (Ford et al. 2010), although PCBs could exacerbate this phenomenon through a variety of mechanisms (Ross *et al.* 2000a;Ross 2006).

PCBs have been implicated in the disruption of endocrine and immune systems in pinnipeds (De Swart *et al.* 1994). Such observations explain at least partly the increased incidence of reproductive impairment (De Guise *et al.* 1995;Helle *et al.* 1976) and disease outbreaks (Ross *et al.* 1996c) in free-ranging populations of seals and whales. There are a number of established effects of PCBs in mammals, including reproductive impairment (Addison 1989;Subramanian *et al.* 1987), immunotoxicity (Brouwer *et al.* 1989;De Swart *et al.* 1996;Mos *et al.* 2006b;Ross *et al.* 1995;Ross *et al.* 1996b), skeletal abnormalities (Bergman *et al.* 1992;Ross *et al.* 1996b;Ross *et al.* 2000a), and endocrine disruption (Brouwer *et al.* 1989;De Swart *et al.* 1996;Ross *et al.* 1996b;Ross *et al.* 2000a;Tabuchi *et al.* 2006). PCBs have been linked to cancer in both humans (Bertazzi *et al.* 2001) and California sea lions (Ylitalo *et al.* 2005), and are listed as probable human carcinogens by the US EPA and International Agency for Research on Cancer (ATSDR 2000).

Mechanistic evidence for PCB-related effects requires single chemical dosing experiments, these providing a dose-dependent series of thresholds. While such studies are understandably based on laboratory animal studies, they provide a means of characterizing PCB-related health risks in killer whales (see Table 1).

Two major captive feeding studies of harbour seals were carried out in the Netherlands in the early 1980s (contaminant effects on reproduction) and early 1990s (contaminant effects on immune system). These studies had the advantage of the control over study animals, which helped to ensure comparability in terms of condition, nutrition and sample access (see Table 2). In addition, studies of free-ranging harbour seals and bottlenose dolphins have generated more insight into the effects of PCBs on marine mammal health (see Table 2). These studies also minimized confounding factors.

While PCBs represent but one chemical class found in complex environmental mixtures, they have widely been viewed as the pre-eminent concern at the top of aquatic food webs in the northern hemisphere over the past three decades (Ross and Birnbaum 2003). In British Columbia, a comprehensive risk-based evaluation of different POPs in harbour seals clearly identified the PCBs as the top concern (Mos *et al.* 2010; see Table 3). While a similar exercise has not yet been conducted in killer whales, this ranking is not expected to differ markedly.

Table 1: PCB effects thresholds in mammals. NOAEL=No Observed Adverse Effects Level; EC50=Estimated Concentration at which 50% of population experiences effects; LW=lipid weight.

Effect	**Species**	**Chemical**	**Tissue Residue Concentration (LW)**	**Reference**
NOAEL	Rhesus monkey	PCBs	0.9 mg·kg^{-1}	1
NOAEL	Mink	PCBs	9.0 mg·kg^{-1}	2
Impaired reproduction	Mink	PCB (Aroclor 1254)	12.3 mg·kg^{-1} *	3
Reduced survival and growth	Mink kits following in utero exposure	PCB (Aroclor 1254)	20.0 mg·kg^{-1} *	4,5
EC50 for litter size and kit survival	mink kit following in utero exposure	PCBs	40-60 mg·kg^{-1}	6

- *based on assumed lipid content of 10% in liver*
- *1-(Schantz et al. 1991), 2-(Kihlstrom et al. 1992), 3-(Platonow and Karstad 1973), 4-(Wren et al. 1987b), 5-(Wren et al. 1987a), 6-(Leonards et al. 1995).*

Table 2: POP-related health effects have been characterized in a series of captive and free-ranging studies of marine mammals. These studies have largely implicated the PCBs as the dominant cause of reported effects.

Species	**Health Endpoint Affected**	**PCB Estimated Effects Concentration (LW)**	**Reference**
Harbour seal	Reproduction Vitamin A and thyroid hormones	25 mg·kg^{-1}	1,2
Harbour seal	Immune function - Natural killer cell activity - T-cell function - Antibody responses Vitamin A and thyroid hormones	17 mg·kg^{-1}	3,4,5,6
Bottlenose dolphin	mortality	10 mg·kg^{-1}	7
Harbour seal	EC_5 Immune function Vitamin A and thyroid hormones Thyroid hormone receptors	1.3 mg·kg^{-1}	8

1-(Reijnders 1986), 2-(Brouwer et al. 1989), 3-(Ross et al. 1996a), 4-(Ross et al. 1996c) , 5-(De Swart et al. 1994), 6-(De Swart et al. 1996), 7-(Hall et al. 2006), 8-(Mos et al. 2010).

Table 3: Comparative risk quotients (CRQs) ranking polychlorinated biphenyl (ΣPCB) and organochlorine pesticide concentrations in harbour seals sampled in British Columbia and Washington State, expressed relative to rodent toxicity reference values (TRVs) for immunotoxicity (Top table) and endocrine disruption (Lower table). This exercise clearly identifies the PCBs as the preeminent contaminant of concern at the top of the regional food web, based on a combination of presence (concentration in seals) and toxicity (as evaluated using mechanistic laboratory animal studies). From Mos et al. (2010).

	Average blubber concentration ± SEM (μg/kg lipid weight)	TRV_{rodent} (μg/kg/day)	CRQ
ΣPCB	2499.2 ± 580.6	500	7.5
Dieldrin	9.4 ± 0.6	13[1]	1.09
ΣDDT	1031.1 ± 137.9	10500	0.15
ΣChlordane	96.2 ± 8.9	8000	0.02
ΣEndrin	3.7 ± 0.7	300[1, 2]	0.02
ΣHeptachlor	71.4 ± 7.7	19000	0.006
ΣHCH	54.7 ± 4.0	20000	0.004
ΣEndosulfan	4.2 ± 0.6	2100	0.003
HCB	5.7 ± 0.5	22000	0.0004
Aldrin	0.02 ± 0.0	250[2]	0.0001
Octachlorostyrene	1.3 ± 0.2	22000[3]	0.0001
Methoxychlor	1.7 ± 1.2	100000[2]	0.0000
Mirex	3.6 ± 0.5	NA	NC

CRQ = comparative risk quotient, representing a ratio between an average blubber contaminant concentration divided by an allometrically scaled rodent TRV; ; SEM = Standard error of the mean; NA = Not available; NC = Not calculated
[1] LOAEL divided by 10
[2] No data available for mouse, TRV is based on rat study
[3] TRV for HCB based on structural similarity

	Average blubber concentration ± SEM (μg/kg lipid weight)	TRV_{rodent} (μg/kg/day)	CRQ
ΣPCB	2499.2 ± 580.6	4000	0.80
ΣDDT	1031.1 ± 137.9	5400	0.24
ΣHeptachlor	71.4 ± 7.7	650[1, 2]	0.16
ΣChlordane	96.2 ± 8.9	1000[1, 2]	0.14
Dieldrin	9.4 ± 0.6	500	0.02
ΣHCH	54.7 ± 4.0	5000	0.01
ΣEndrin	3.7 ± 0.7	420[2]	0.01
HCB	5.7 ± 0.5	9500	0.0008
ΣEndosulfan	4.2 ± 0.6	10000	0.0005
Octachlorostyrene	1.3 ± 0.2	9500[3]	0.0002
Methoxychlor	1.7 ± 1.2	50000	0.0000
Aldrin	0.02 ± 0.0	630	0.0000
Mirex	3.6 ± 0.5	NA	NC

CRQ = comparative risk quotient, representing a ratio between an average blubber contaminant concentration divided by an allometrically scaled rodent TRV; SEM = Standard error of the mean; NA = Not available; NC = Not calculated
[1] LOAEL divided by 10
[2] No data available for rat, TRV is based on mouse study
[3] TRV for HCB based on structural similarity

PCBs have been extensively studied, such that the available literature provides substantial guidance on the source, transport, and fate mechanisms of this class of chemical in the environment. Their accumulation in marine food webs in British Columbia has provided insight into spatial and temporal aspects of accumulation in coastal food webs in killer whale habitat (Cullon *et al.* 2005;Cullon *et al.* 2008;Ross *et al.* 2004). An earlier modeling effort concluded that killer whales are particularly vulnerable to accumulating very high PCB concentrations in their tissues, reflecting their high trophic level, long lifespan, and limited metabolic ability to clear these contaminants from their bodies (Hickie *et al.* 2007).

Dioxins and furans represent a significant environmental concern in coastal British Columbia, but risks to killer whales have been limited by the 1989 implementation of source control and regulations (Hagen *et al.* 1997), and the relatively rapid apparent metabolism of dioxin-like contaminants by killer whales and their prey (Ross *et al.* 2000a). Mercury represents a concern to upper trophic levels, as organic forms of this metal biomagnify in food webs. While anthropogenic activities release large quantities of mercury into the biosphere, marine mammals have evolved to be able to detoxify mercury as a result of long-term exposure to natural geological background concentrations of this metal (Ikemoto *et al.* 2004). Polycyclic aromatic hydrocarbons (PAHs) have been implicated in liver tumours and other health effects in fish inhabiting industrialized harbours, but this complex class of contaminants is readily metabolized at lower levels of the food web, and killer whales are unlikely to be at significant risk of health impacts at current levels of dietary exposure.

Given the special vulnerability of killer whales to contamination by PCBs and related contaminants and their associated health effects, it is important that current *Canadian Environmental Protection Act* (CEPA) regulations for disposal at sea be critically evaluated in this regard, with an emphasis on contamination within the species' Critical Habitat. Studies, such as those by Hickie *et al.* (2007) and Natale (2007), have evaluated the protectiveness of sediment guidelines and regulations (e.g., CEPA Action Levels) for upper trophic level organisms and the results indicate that the guidelines and regulations are often not protective for biomagnifying contaminants. However, most sediment quality guidelines and regulations were not designed to protect against bioaccumulation, and do not consider upper trophic levels. To protect 95% of the population of male harbour seals in Burrard Inlet, Natale (2007) found that total PCB concentrations in sediments would need to be below 1.13 $\mu g \cdot kg^{-1}$ dry weight. This value is 20 times lower than the current CCME Interim Sediment Quality Guideline for total PCBs of 21.5 $\mu g \cdot kg^{-1}$ dry weight (CCME 1999). Sediment-associated contaminants may lead to bioaccumulation in organisms and biomagnification in the food web, and PCB-contaminated sediments figure prominently in decision-making for disposal at sea of dredged materials (Linkov *et al.* 2001;von Stackelberg *et al.* 2002b).

Canadian Sediment Quality Guidelines and Disposal at Sea Action Levels for contaminants were designed to be protective of benthic organisms, and do not take biomagnification into account. Thus, they were not designed to protect upper trophic level organisms, such as killer whales, from contaminants, and guidelines to do so currently do not exist. However, these are the only broadly available sediment quality criteria for the management and assessment of sediment contamination in Canada, and are routinely used in site-specific risk assessment and remediation efforts in order to protect aquatic biota. We therefore evaluated their value in protecting upper trophic level wildlife such as killer whales.

This study uses a food web bioaccumulation modeling approach developed by Gobas (1993) and Gobas and Arnot (2010) to determine if PCBs in dredged material disposed in resident killer whale Critical Habitat poses a threat to the whales via destruction of their Critical Habitat (Section 58 of the *Species at Risk Act*) or via harm to individuals (Section 32 of the Species *at Risk Act*). This model was chosen because it has been extensively tested in a variety of studies (e.g., Linkov *et al.* 2001;Linkov *et al.* 2002;von Stackelberg *et al.* 2002a;von Stackelberg *et al.* 2002b), has found good agreement with predicted and observed contaminant concentrations, and it is one of the best food web bioaccumulation models currently available. The Gobas model uses measured concentrations of PCBs in water and sediments and processes that control PCB bioaccumulation in the food web, such as the toxicokinetics of PCB uptake and elimination in different organisms, to estimate the resulting concentrations in biota as they biomagnify up the food web. The model assumes that steady state throughout the food web is achieved in balance with the prescribed sediment and water PCB concentrations, but does not specifically include other sources or sinks for PCB (e.g. air-sea exchange, ocean exchange, sediment burial).

Resident killer whales forage widely in BC coastal waters, and our challenge is to characterize the extent of use of Critical Habitat by killer whales and their prey, and estimate what component of their exposure to PCBs might come from the disposal of spoils in Critical Habitat.

In addition to the Critical Habitats of northern and southern resident killer whales (in Canada and the United States), we must also consider the wider foraging domain including Strait of Georgia, Queen Charlotte Strait, and the outer coast off Vancouver Island, BC.

Unfortunately, data and time constraints preclude the development of realistic scenarios that would incorporate, for example, variability in PCB concentrations in dredgeate, distribution of dredged sediment on the bottom after disposal, the potential release or sequestering of PCBs during transit of dredge spoil through the water, and a realistic ecosystem comprising benthic to surface domains over a stratified water column up to 420 m deep. Consequently, we have relied upon reasonable approximations of PCB levels in dredgeate, or have accepted established screening or evaluation criteria (i.e., sediment PCB concentrations from dredged areas in the Strait of Georgia and Burrard Inlet, and the analytical detection limit for PCBs currently used by Environment Canada) to project a range of outcomes from prescribed disposal activities. Basically, two major scenarios based on the baseline modeling were conducted: (a) *disposal at all sites within killer whale Critical Habitat* (i.e. dredge spoils have been limited to the full area of the disposal sites found within Critical Habitat), and (b) *disposal at all disposal sites within killer whale habitat* (i.e. dredge spoils containing prescribed PCB concentrations have been substituted for sediments at disposal found both inside Critical Habitat and outside Critical Habitat). Thus the results provide means to explore the implications of a range from relatively low to high dredge PCB concentrations. For example, sediment concentrations representing Burrard Inlet provide the worst-case scenario of what exposures could occur to killer whales and their prey if such dredgeate were placed into disposal sites inside Critical Habitat or at sites in killer whale habitat in general. Most sediments disposed in the Strait of Georgia comprise sandy material undergoing dynamic transport in the lower Fraser River, and as such, may help to ‘dilute’ or ‘cap’ more PCB-contaminated ambient sediments at disposal sites. The scenarios considered are described below.

Baseline Model Applications to Chinook Salmon and Killer Whales: The model was used to predict PCB concentrations in Chinook salmon and resident killer whales using prescribed sediment values based on observations or quality guideline values, and assuming 100% time spent by each of the two species in each of the seven areas evaluated. The resulting BSAF values (Chinook : sediment and killer whale : sediment) were employed in subsequent area-weighted approaches (actual time spent in each of the seven areas of interest) to predicting PCB concentrations in salmon and killer whales under a variety of scenarios. PCB concentrations were estimated for two major Chinook salmon stocks (South Thompson and Lower Fraser). This approach also enabled an attribution of PCBs in killer whales to each of the seven areas of interest. Killer whales had diet of 96% Chinook salmon, 2% halibut, and 2% sablefish.

Disposal at Sea Scenarios: Exploring the Implications of Disposal in Critical Habitat Disposal Sites: To evaluate the possible impacts of disposal operations on killer whale PCB accumulation, we undertook a series of sediment PCB substitutions for the surface area covered by those disposal sites found within Critical Habitat for both northern and southern resident killer whales. We assume here a one-time and permanent shift away from the ambient PCB concentrations found at those sites towards the substituted value. We used data from the literature and expert consultations to determine annual habitat distributions for Chinook salmon (Lower Fraser River and South Thompson River stocks) and northern and southern resident killer whales. Killer whales had diet of 96% Chinook salmon, 2% halibut, and 2% sablefish.

This modeling approach uses a life history based distribution of time spent by killer whales and Chinook salmon in their different habitat areas, while varying the input sediment PCB values weighted by the surface area of the disposal sites. In this series of exercises, eight different sediment PCB values were used, representing four empirically derived values from different dredge scenarios (sediment values from Puget Sound East Harbor Island, Victoria Harbour, Burrard Inlet and Strait of Georgia mean) and four methods/decision criteria values (CEPA Action Level Low, EC DL Minimum, CCME ISQG, and the LEACA DL).

Disposal at Sea Scenarios: Exploring the Implications of Disposal at All Sites Within Killer Whale Habitat: To evaluate the possible impacts of disposal operations on killer whale PCB accumulation, we undertook a series of sediment PCB substitutions for the surface area covered by those disposal sites found both inside Critical Habitat and outside Critical Habitat for both northern and southern resident killer whales. We assume here a one-time and permanent shift away from the ambient PCB concentrations found at those sites towards the substituted value. We used data from the literature and expert consultations to determine annual habitat distributions for Chinook salmon (Lower Fraser River and South Thompson River stocks) and northern and southern resident killer whales. Killer whales had diet of 96% Chinook salmon, 2% halibut, and 2% sablefish.

This modeling approach uses a life history based distribution of time spent by killer whales and Chinook salmon in their different habitat areas, while varying the input sediment PCB values weighted by the surface area of the disposal sites. In this series of exercises, eight different sediment PCB values were used, representing four empirically derived values from different dredge scenarios (sediment values from Puget Sound East Harbor Island, Victoria Harbour, Burrard Inlet and Strait of Georgia mean) and four methods/decision citeria values (CEPA Action Level Low, EC DL Minimum, CCME ISQG, and the LEACA DL).

Using hypothetical and realistic habitat distributions of Chinook salmon and killer whales to assess decision criteria and laboratory methods: We evaluated the implications of different sediment PCB values for hypothetical killer whales that spend 100% of their time in an area. In a similar fashion, documented habitat distribution and at field observations of realistic time spent by Chinook salmon and killer whale were modeled to evaluate the health impact of sediment PCB values. In both cases, Killer whales had a diet consisting of 96% Chinook salmon, 2% halibut, and 2% sablefish. Predicted outcomes were compared against available guidelines for prey (Chinook) or health effects thresholds (killer whales). In this series of exercises, eight different sediment PCB values were used, representing four empirically derived values from different dredge scenarios (sediment values from Puget Sound East Harbor Island, Victoria Harbour, Burrard Inlet and Strait of Georgia mean) and four methods/decision criteria values (CEPA Action Level Low, EC DL Minimum, CCME ISQG, and the LEACA DL).

Backward Scenarios: In an effort to derive new Sediment Quality Guidelines protective of killer whales, we conducted a backward application of the BSAF model. We used realistic habitat distributions for resident killer whales (and a diet of 96% Chinook salmon, 2% halibut, and 2% sablefish) and Chinook salmon, as well as hypothetical scenarios where the animals spend 100% of their time in the area.

One of the goals of the modeling process presented above was to evaluate the Interim Sediment Quality Guideline and Action Level Low for PCBs and their protectiveness of killer whales and their Critical Habitats. Thus three guidelines were used as model inputs: the CCME Interim Sediment Quality Guideline of 21.5 $\mu g \cdot kg^{-1}$ dry weight (CCME 1999); the CEPA Action Level Low of 100 $\mu g \cdot kg^{-1}$ dry weight (CEPA 2001); and the BCMWLAP Sediment Quality Criteria for sensitive species (SQC_{SCS}) of 120 $\mu g \cdot kg^{-1}$ dry weight (BCMWLAP 2004a). Estimated PCB concentrations in killer whales were compared with thresholds for toxicity in other marine mammals.

Several toxicity thresholds in other marine mammals were considered: the harbour seal toxicity threshold for PCBs of 17,000 $\mu g \cdot kg^{-1}$ lipid (Ross *et al.* 1996a); the toxicity threshold for bottlenose dolphins of 10,000 $\mu g \cdot kg^{-1}$ lipid weight (Hall *et al.* 2006); and the revised harbour seal toxicity reference value (TRV) of 1,300 $\mu g \cdot kg^{-1}$ lipid weight tissue residue in blubber (Mos *et al.* 2010). We also considered two toxicity thresholds in killer whale prey (i.e., Chinook salmon): the tissue residue guideline for fish-eating wildlife of 50 $\mu g \cdot kg^{-}1$ derived for PCBs from the CCME guideline for dioxin-like toxicity (Hickie *et al.* 2007), and the newly-derived value of 8 $\mu g \cdot kg^{-1}$, wet weight PCBs in killer whale prey in order for 95% of the killer whale population to fall below the 17,000 $\mu g \cdot kg^{-1}$ toxicity threshold (Hickie *et al.* 2007). PCB concentrations in killer whale prey below these two toxicity thresholds would reduce PCB concentrations in killer whales to levels deemed by the two groups to be protective from health effects. The resulting predicted sediment concentrations from these thresholds may be considered as ecologically relevant targets for management, which can guide remediation, pollution control, and suitability of disposal sites with respect to resident killer whales' Critical Habitat.

Additional goals of the food web bioaccumulation model are to (1) integrate current information and improve our understanding of PCB bioaccumulation in killer whale habitat, (2) identify areas where information is lacking to provide guidance for future research, and (3) communicate the results to the Habitat and Species at Risk Management Branch of Fisheries & Oceans Canada, the scientific community, and the general public.

1.4 DISPOSAL AT SEA IN BRITISH COLUMBIA

There are 15 Disposal at Sea sites in coastal BC (Figure 8) neglecting the Roberts Bank site, which is rarely used. Two ocean disposal sites exist within NRKW Critical Habitat, Hickey Point and Hanson Island, both of which are in Johnstone Strait (Environment Canada 2006). SRKW Critical Habitat also contains two ocean disposal sites, Sand Heads and Victoria (Environment Canada 2006). Roberts Bank site is also located in SRKW Critical Habitat but this site is only used infrequently by the Delta Port. Additional disposal sites are located outside the boundaries of killer whale Critical Habitat, but within their general habitat range (e.g., Point Grey disposal site; Environment Canada 2006).

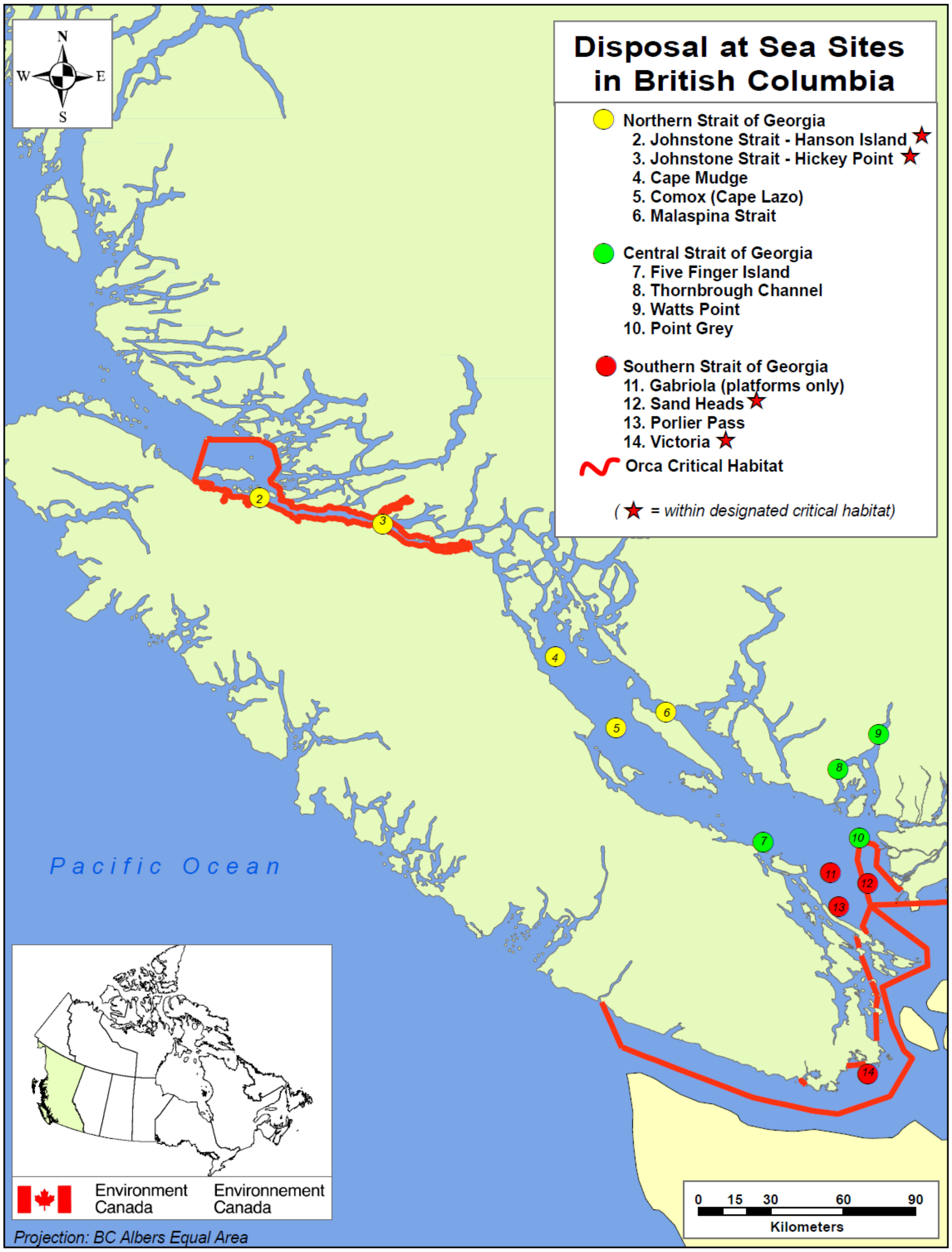


Figure 8: Disposal at Sea sites in British Columbia (map generated by Scott Lewis, Environment Canada, Disposal at Sea Program, 351 St. Joseph Blvd. 16th Floor, Gatineau, QC). Thick red lines indicate northern and southern resident killer whale Critical Habitat boundaries.

Disposal at sea sites are influenced by a number of factors that govern the fate and bioavailability of PCBs. Key abiotic factors influencing disposal sites are hydrology (tidal action and currents), sediment deposition rate, contaminant burial rate, and bioturbation. In addition to knowledge of sediment contaminant concentrations in background areas comparable to the disposal sites, it is essential to determine whether or not disposal sites increase the risk to biota from exposure to contaminants. Total PCB concentrations ($\mu g \cdot kg^{-1}$, dry weight; sum of all PCB congener concentrations detected) in surface sediments in SRKW Critical Habitat in BC (BCMWLAP 2001) and WA (Washington State Department of Ecology 2002) are shown in Figure 9. The dashed reference line in Figure 9 indicates the Action Level Low for total PCBs in dredged materials to be deposited at sea (CEPA 2001), the solid reference line indicates the Canadian Interim Sediment Quality Guideline (ISQG; CCME 1999), and the dotted reference line indicates the BC Sediment Quality Criteria for sensitive habitats (BC SQC_{SCS}; which applies to endangered and threatened species) (BCMWLAP 2004b). Action levels are basically screening levels to determine if the concentrations of contaminants in dredged materials are too high to be disposed at sea.

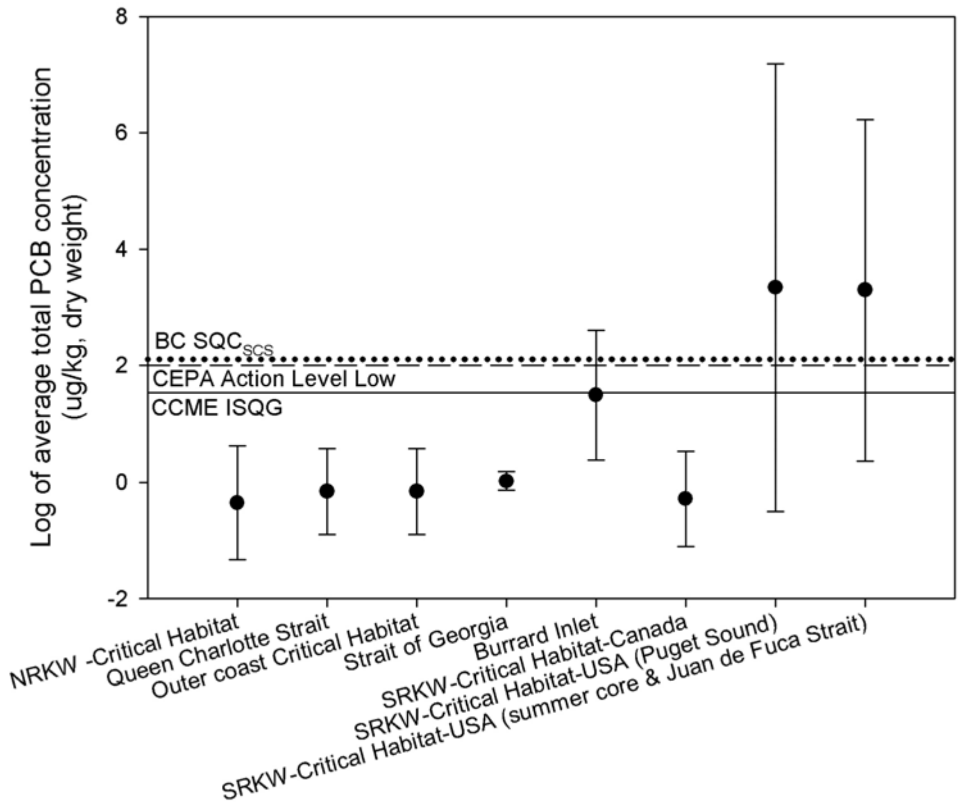


Figure 9: Log of mean total PCB concentration (µg·kg^{-1}, dry weight) and standard deviation in sediments located in areas of resident killer whale habitat. The dashed reference line indicates the log value of the CEPA Action Level Low for total PCBs (100 µg·kg^{-1}, dry weight) in dredged materials to be disposed at sea in Canada (CEPA 2001), the solid reference line indicates the log value of the Canadian Interim Sediment Quality Guideline for total PCBs (21.5 µg·kg^{-1}, dry weight) (CCME 1999), and the thick dotted line indicates the log value of the British Columbia Sediment Quality Criteria for sensitive habitats for total PCBs (120 µg·kg^{-1}, dry weight)(BCMWLAP 2004b) .Sample sites used to calculate the mean PCB concentration (and their references) are listed in Table 7.

Sites in resident killer whale habitat in BC have total PCB concentrations below the CEPA Action Level Low for disposal at sea materials (100 µg·kg^{-1}, dry weight; Figure 9), and the CCME Interim Sediment Quality Guideline (21.5 µg·kg^{-1}, dry weight). Sediments in SRKW Critical Habitat in the US (summer core, Juan de Fuca Strait, and Puget Sound) are much more contaminated with PCBs than Strait of Georgia sediments, and the average concentrations from those areas exceed the CEPA Action Level Low and CCME ISQG. However, many of the sites used to determine the Puget Sound average PCB concentration were from hot spots, thus the value represented likely overestimates average concentrations in Puget Sound.

An extensive study from 1997 to 1999 that included 300 sediment samples from various areas in Puget Sound found that PCB congener 153 was dominant, with concentrations of CBs 101, 118, 138 also very high (Long *et al.* 2005). The maximum PCB concentration from a sediment sample (from inner Everett Harbor) was 4658 $\mu g \cdot kg^{-1}$, which is 26 times the Effects Range-Median (ERM) value of 180 $\mu g \cdot kg^{-1}$ (Long *et al.* 2005); however, the mean concentration of total PCBs for all samples was much lower at 80 $\mu g \cdot kg^{-1}$ (Long *et al.* 2005). Fourteen Puget Sound sediment samples exceeded the ERM for total PCBs, and generally they were from urban/industrialized embayment's such as the Whidbey Basin, Everett Harbor, Elliott Bay, Commencement Bay, Sinclair Inlet, as well as central Puget Sound (Long *et al.* 2005). Mean concentrations from these areas exceeded the Effects Range-Low (ERL) value of 22.7 $\mu g \cdot kg^{-1}$ (Long *et al.* 2005). The study found that the lowest total PCB concentrations (many samples had no detectable PCBs) were from the Strait of Georgia, Admiralty Inlet, Hood Canal, and southern Puget Sound (Long *et al.* 2005).

A multi-year study determined that sediments are more contaminated with PCBs and other toxins in central Puget Sound than in the northern and southern areas of the sound (Long *et al.* 1999;Long *et al.* 2000;Long *et al.* 2002). The percent of each study area deemed to be degraded (i.e., exceeded Sediment Quality Guidelines) was 2.8% in the central sound, 1.3% in the northern sound, and 0.5% in the southern sound (Long *et al.* 1999;Long *et al.* 2000;Long *et al.* 2002). Total PCB concentrations in sediments exceeded the ERM (180 $\mu g \cdot kg^{-1}$) and SQS (12,000 $\mu g \cdot kg^{-1}$) in inner Everett Harbor (Long *et al.* 1999), Elliott Bay (Long *et al.* 2000), East and West Harbor Island (Long *et al.* 2000), the Duwamish River (Long *et al.* 2000), and the Thea Foss and Hylebos Waterways (Long *et al.* 2002). Total PCB concentrations in sediments exceeded the ERL (22.7 $\mu g \cdot kg^{-1}$) in Bellingham Bay, Everett Harbor, and Port Gardner Bay (Long *et al.* 1999).

Disposal at Sea sites range from net erosional to net depositional. In high energy erosional environments sediments are re-suspended in the water column, which can greatly increase the mobility of contaminants bound to the sediments (Apitz *et al.* 2005). Low energy environments are usually depositional with little re-suspension, and sediments tend to be fine-grained, with high sorptive ability, and slow advection and oxygen transport (Apitz *et al.* 2005). There are both erosional and depositional areas within resident killer whale Critical Habitat (see Figure 10 below).

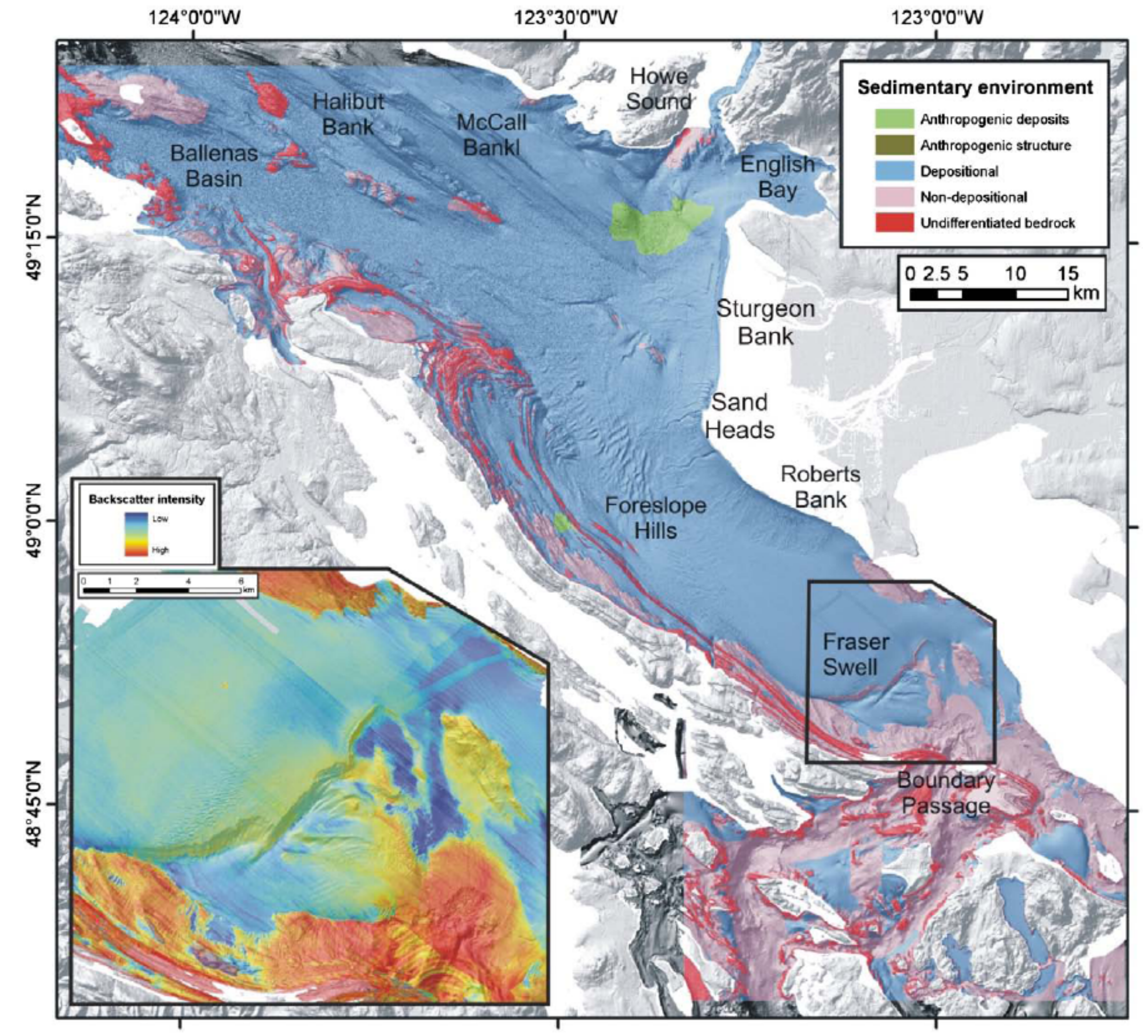


Figure 10: Map of the Strait of Georgia depicted with multibeam backscatter intensity (from Hill et al. 2008). Areas coloured blue represent low backscatter intensity, which correspond to fine-grained sediments and sediment accumulation. Areas coloured red represent stronger backscatter intensity, which correspond to coarser sediments or hard bottoms. Ocean disposal areas are referred to as anthropogenic deposits in the map legend, and are pale green in colour. The map shows that in the northern Strait of Georgia, sediments accumulate in most deep-water areas, whereas some ridges and basin margins are non-depositional. In the southern Strait of Georgia the deposition pattern is more complex as fine sediments that were deposited in the past are currently being winnowed and eroded, leaving areas of higher backscatter intensity.

As seen in Figure 10, sediment accumulation occurs in the main basin of the southern Strait of Georgia and in deep troughs between McCall and Halibut Banks (off the Sunshine Coast) (Hill *et al.* 2008). Sediment accumulation also occurs in Ballenas Basin, Howe Sound, English Bay, and Malaspina Strait (Hill *et al.* 2008). The Fraser Swell, the area from the mouth of the Fraser River across the Strait of Georgia to Boundary Passage, is a net erosional area that is being winnowed by current (Hill *et al.* 2008). South of the Fraser Swell there is limited deposition (Barrie *et al.* 2005). The sedimentation rate in the northern Strait of Georgia is less than 1 $cm{\cdot}yr^{-1}$, in the south/central Strait it is generally 1-2 $cm{\cdot}yr^{-1}$, except near the mouth of the Fraser River, where it is greater than 3 $cm{\cdot}yr^{-1}$, and may exceed 1 m per year (Hill *et al.* 2008;Johannessen *et al.* 2003).

The majority (~71%) of the subtidal area of the Strait of Georgia is greater than 50 m deep, and the bottom is primarily composed of soft sediments (Hill *et al.* 2008). Sediments in most of the central and southern Strait contain about 1% organic carbon, while north of Texada Island, where the influence of the Fraser River is much less, that percentage rises to 3-6% (Burd *et al.* 2008b;Johannessen *et al.* 2003). To the south of the Fraser River mouth, Fraser River sand mixes with fine sands eroded off the Gulf Islands to create a patchy distribution of sand and silt (Hill *et al.* 2008). The shallow sills and narrow passages that bound the Strait of Georgia at both ends provide little opportunity for fine sediment to accumulate, but these areas may provide important sources of suspended particles carrying contaminants (Johannessen *et al.* 2006), and water-soluble contaminants, which may spread into surrounding waters (Macdonald and Crecelius 1994). In the Strait of Georgia, bioturbation disturbs the top 5-15 cm of sediment (Johannessen *et al.* 2008a).

In Burrard Inlet, fast currents tend to keep particles suspended in the water column (Thomson 1981). However, dredging occurs at the First Narrows channel and Port Moody Arm to maintain navigation, which indicates that there is net deposition in these areas. Sediment TOC in Burrard Inlet is about 1.5-2% (McPherson *et al.* 2006). Even though a variety of pollutants are present in Burrard Inlet at higher levels than those measured at other load sites, the benthic community is not impoverished or dominated by opportunistic polychaetes but rather is dominated by small burrowing bivalves (McPherson *et al.* 2006).

Descriptions of the disposal sites follow below. General information on the disposal sites located within resident killer whale Critical Habitat is listed in Table 4.

Table 4: Characteristics of disposal at sea sites located inside southern (Sand Heads and Roberts Bank) and northern (Johnstone Strait Hanson Island and Hickey Point) resident killer whale Critical Habitat.

Site Characteristics	**Sand Heads**	**Roberts Bank**	**Victoria**	**Johnstone Strait – Hanson Island**	**Johnstone Strait – Hickey Point**
Latitude (N)	49°06.00’ [1]	49°00.70’ [2]	48°22.30’ [2]	50°33.50’ [1]	50°27.90’ [1]
Longitude (W)	123°19.50’ [1]	123°10.50’ [2]	123°21.90’ [2]	126°48.00’ [1]	126°04.90’ [1]
In use since	1974 [1]		1970 [1]	1980 [1]	1980 [1]
Depth (m)	70 [1]		100 [1]	470 [1]	270 [1]
Diameter (km)		1 [2]	1.85 [2]	1.85 [1]	1.85 [1]
Area (km^2)	0.89 [2]	2.69 [2]	2.69 [2]	2.69 [1]	2.69 [1]
Sedimentation velocity ($cm \cdot yr^{-1}$)	1.2 [3]	2.9 [3]			
Sediment accumulation rate ($g/cm^2/yr$)	1.3 [3]	2.7 [3]			
Surface mixed layer depth (cm)	7 [3]	12 [3]			
Mixing rate in the upper layer ($cm^2 \cdot yr^{-1}$)	20 [3]	12 [3]			
ΣPCBs in surface sediments ($\mu g \cdot kg^{-1}$)	0.48 -1.21 [3]	0.668 [3]			

Table References:

1. Environment Canada (2006)
2. Sean Standing (Environment Canada, Disposal at Sea Program, #201-401 Burrard St., Vancouver, BC V6C 3S5, pers. comm., 2010)
3. Johannessen *et al.* (2008a)

1.4.1 Sand Heads Disposal Site

The Sand Heads disposal site is in a very dynamic zone with significant tidal action (Environment Canada 2006). The site is bounded by the coordinates: 49°06.12' N 123°20.42' W, 49°06.31' N 123°18.83' W, 49°05.74' N 123°18.96' W, 49°05.22' N 123°19.64' W (Sean Standing, Environment Canada, Disposal at Sea Program, #201-401 Burrard St., Vancouver, BC V6C 3S5, pers. comm., 2010). The site is located in a submarine channel at the end of the Steveston Jetty and experiences occasional slope failures that lead to turbidity flows (Phil Hill, Natural Resources Canada, Geological Survey of Canada, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010). The turbidity flows carve out submarine channels, and can deliver sediment to the bottom of the slope, although most of the coarse, dredged disposal material stays in the channel (Phil Hill, Natural Resources Canada, Geological Survey of Canada, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010). The sedimentation rate in this area is variable, and the sedimentation rate is too rapid to permit bioturbation (Phil Hill, Natural Resources Canada, Geological Survey of Canada, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010). The sediments are composed mostly of sand near the end of the jetty, and the amount of sand decreases along the channel into the Strait of Georgia, where mud and silt predominate (Phil Hill, Natural Resources Canada, Geological Survey of Canada, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010).

Material received is almost exclusively sand and silt from annual maintenance dredging of navigation channels in the main arm of the Fraser River (Environment Canada 2006). Since 1974 296,544 m^3 of dredge material has been deposited (Environment Canada 2006). The most recent disposal activity since 1999 at Sands Heads is listed in Table 1 in Appendix II.

1.4.2 Roberts Bank Disposal Site

Tidal currents at the Roberts Bank site cause a predominant northward drift along the Fraser River delta slope. Mean flood tide velocities exceed 1.2 $m \cdot s^{-1}$ (Meulé 2005), so deposited material will move northward with the tidal current (Phil Hill, Natural Resources Canada, Geological Survey of Canada, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010). During peak currents, re-suspension of fine sand from the sea floor occurs to depths as great as 90 m (Kostaschuk *et al.* 1995). There is significant sediment accumulation at the river mouth, which exceeds 1 $m \cdot yr^{-1}$ (Hill 2010), which causes slope failures at the top of the slope (McKenna *et al.* 1992). The sediments are sandy, and the site is located in a sand wave field that has a high sedimentation rate, that experiences little if any bioturbation (Phil Hill, Natural Resources Canada, Geological Survey of Canada, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010).

The Roberts Bank disposal site only includes the area below the 40 m contour line and is not used routinely or considered available other than for Delta Port development (Sean Standing, Environment Canada, Disposal at Sea Program, #201-401 Burrard St., Vancouver, BC V6C 3S5, pers. comm., 2010). The most recent disposal at the site occurred in 2008, with 118,663 m^3 of material from the Lower Fraser River Delta Port deposited (Sean Standing, Environment Canada, Disposal at Sea Program, #201-401 Burrard St., Vancouver, BC V6C 3S5, pers. comm., 2010).

1.4.3 Victoria Disposal Site

The Victoria disposal site has little sediment cover, is composed mostly of silt, and experiences strong currents and a low sedimentation rate (Phil Hill, Natural Resources Canada, Geological Survey of Canada, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010). There is a high gravel and sand content, at both the disposal and reference locations, which suggests an erosional benthic environment (Environment Canada 2006).

The site has received approximately 296 544 m^3 of material since 1970 (Environment Canada 2006). The most recent disposal activity since 2000 occurred in 2002, when 230 m^3 of material was deposited from Blue Heron Basin on Vancouver Island, and in 2004 with 3,900 m^3 of material deposited from Victoria Harbour on Vancouver Island (Sean Standing, Environment Canada, Disposal at Sea Program, #201-401 Burrard St., Vancouver, BC V6C 3S5, pers. comm., 2010). Most of the deposited material is from maintenance dredging at marinas and commercial properties near Victoria. Sediment chemistry data show the site to be relatively uncontaminated and no further management has been recommended (Environment Canada 2006).

1.4.4 Johnstone Strait – Hanson Island Disposal Site

The Johnstone Strait-Hanson Island disposal site is located in a deep trough, and has received 225 853 m^3 of material since 1980 (Environment Canada 2006). The deposited material is from maintenance dredging at log handling facilities on northern Vancouver Island, and is composed primarily of wood waste with some naturally-distributed sediments (Environment Canada 2006). The sediments are hard and composed of rocks and coarse gravel, and the site experiences strong currents (Environment Canada 2006) thus can be considered erosional with a very low sedimentation rate (Phil Hill, Natural Resources Canada, Geological Survey of Canada, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010).

The site has been used extensively for disposal of dredged material and the most recent disposal activity since 2000 is listed in Table 2 Appendix II.

1.4.5 Johnstone Strait – Hickey Point Disposal Site

The Johnstone Strait-Hickey Point disposal site is located in a deep trough, with a bottom composed of rocks and sand with virtually no sedimentation and experiences strong currents (Phil Hill, Natural Resources Canada, Geological Survey of Canada, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010).

The site receives material from maintenance dredging of forest industry sites, the dredgeate being composed of wood waste, silt, clay, sand, and gravel (Environment Canada 2006). The site has received 183 694 m^3 of material since 1980 (Environment Canada 2006). The most recent disposal activity since 2000 is shown in Table 3 Appendix II.

1.4.6 Disposal Sites Outside Critical Habitat

There are several other disposal sites that fall outside of the boundaries of resident killer whale Critical Habitat but are still within the general habitat of resident killer whales and within the areas included in the food web bioaccumulation model. General characteristics of these sites are listed in Table 5. More information is available below for the Point Grey and Porlier Pass disposal sites. The Gabriola disposal site has not been in use since before the year 2000, thus details were not included in the table.

Table 5: Characteristics of disposal at sea sites located outside of resident killer whale Critical Habitat.

Site Characteristics	Point Grey	Porlier Pass	Cape Mudge	Comox	Malaspina	Five Finger Island	Thornborough Channel	Watts Point
Latitude (N)	49°15.40' [1]	49°00.20' [1]	49° 57.70' [2]	49° 41.70' [2]	49° 45.00' [2]	49° 15.20' [2]	49° 31.00' N [2]	49° 38.50' N [2]
Longitude (W)	123°21.10' [1]	123°29.80' [1]	125° 05.00' [2]	124° 44.50' [2]	124° 27.00' [2]	123° 54.70' [2]	123° 28.30' W [2]	123° 14.10' W [2]
In use since	1930's [1]	1978 [1]	1981 [2]	1977 [1]	1980 [2]	1978 [1]	1975 [2]	1976 [2]
Depth (m)	210 [1]	176 [1]	240 [2]	190 [1]	320 [2]	271 [1]	220 [2]	230 [2]
Diameter (km)	3.7 [2]	1.85 [2]	1.85 [2]	1.85 [2]	1.85 [2]	1.85 [2]	0.926 [2]	0.926 [2]
Area (km^2)	10.77 [2]	2.69 [2]	2.69 [2]	2.69 [2]	2.69 [2]	2.69 [2]	0.672 [2]	0.672 [2]
Sedimentation velocity ($cm \cdot yr^{-1}$)	0.35 [3]	0.3 [3]						
Sediment accumulation rate ($g/cm^2/yr$)	0.26 [3]	0.32 [3]						
Surface mixed layer depth (cm)	10 [3]	12 [3]						
Mixing rate in the upper layer ($cm^2 \cdot yr^{-1}$)	15 [3]	12 [3]						
ΣPCBs in surface sediments ($\mu g \cdot kg^{-1}$)	2.91 [3]	0.507 [3]						

Table References:

1. Environment Canada (2006)
2. Sean Standing (Environment Canada, Disposal at Sea Program, #201-401 Burrard St., Vancouver, BC V6C 3S5, pers. comm., 2010)
3. Johannessen *et al.* (2008a)

1.4.6.1 Point Grey Disposal Site

The Point Grey disposal site is the largest and oldest multi-user disposal site in BC (continuously used since the 1930's but officially in use since 1968; Environment Canada 2006), and the only disposal site in the Strait of Georgia where biological monitoring occurs (Burd *et al.* 2008a). The site is located on the slope of the Fraser River delta and experiences strong tidal currents, but is unaffected by waves (Phil Hill, Natural Resources Canada, Geological Survey of Canada, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010). The bottom sediments are approximately 99% mud (Phil Hill, Natural Resources Canada, Geological Survey of Canada, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010), and there is significant bioturbation (Johannessen *et al.* 2005a). The area around the site is depositional, with 20% of the sediment coming from the Fraser River discharge and 80% from dumping (Wilson and McKinnon 2003).

The amount of dredge material deposited at the site per year is 450,000 m^3 (Environment Canada 2006). The site receives dredge material comprised of wood waste and river silt from channels in the Port of Vancouver and ports used by the forest industry in the Fraser River (Environment Canada 2006). Historic evidence shows that material has been deposited outside and en route to the dump boundaries (Wilson and McKinnon 2003), and further evidence has shown that this problem continues (Yunker *et al.* 2000;Yunker 2000). The dumped material is significantly coarser than the surrounding sediments, yet the benthic invertebrate community is not notably different from that found outside the disposal site (Wilson and McKinnon 2003). The disposal site is dominated by a range of polychaete taxa, and burrowing and near-surface echinoderms and amphipods (Wilson and McKinnon 2003), which are sensitive to contaminants and organic material enrichment (Burd 2004). Sediment chemistry data show the site to be uncontaminated except for minor exceedances of cadmium, and no further management has been recommended (Environment Canada 2006). This disposal site is just on the edge of the boundary of SRKW Critical Habitat.

1.4.6.2 Porlier Pass Disposal Site

The Porlier Pass disposal site is located in a basin between bedrock ridges and glacial till ridges, and has relatively benign hydrodynamics (Phil Hill, Natural Resources Canada, Geological Survey of Canada, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010). The site has received 197 074 m^3 of material since 1978, most of it from maintenance dredging at sawmills and log handling facilities located on southern Vancouver Island (Environment Canada 2006). The area is depositional, and sediments have a high silt and clay content (Environment Canada 2006), and are composed of ~98% mud and there is high bioturbation (Phil Hill, Natural Resources Canada, Geological Survey of Canada, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010).

Metals and organics are not of concern except for a couple of marginal exceedances of CEPA guidelines by metal ions which showed toxic responses in benthic organisms (Environment Canada 2006). Further studies of the site and reference sites are recommended (Environment Canada 2006).

2. THEORY: MODEL DEVELOPMENT & PARAMETERIZATION

2.1 MODEL DEVELOPMENT

We develop here a conceptual framework that underlies the mechanics of our food web bioaccumulation model. The goal of the model is to address management issues regarding disposal at sea with respect to destruction or impacts to Critical Habitat of resident killer whales in British Columbia. In the model, PCB behaviour is simplified to the primary process controlling PCB fate in the food web of resident killer whales (e.g., congener specific partition coefficients). The management objectives outlined in Section 1.4 drove the model development, while the specific requests for advice posed to Science are listed in Section 1.2, and the model objectives are listed in Section 2.2. Simplifying assumptions have been made as described in Section 2.3. Functional relationships in the model used to describe the transfer of PCBs from sediments, water and air into various species in the areas included are discussed in Section 2.4. These functional relationships were parameterized as described in Section 2.4.

The food web bioaccumulation model was constructed using the most accurate information on PCB dynamics in the food web of resident killer whales, as well as data on biological, physical, and geochemical characteristics of the areas included in the model. Published data already in the literature were used to test and evaluate the accuracy of the model's predictions of PCB concentrations in biota (Section 2.4). The data set used was not as complete as it might have been given more time and resources, and it is possible that the results could change given more load site data, and non-hotspot data from the Puget Sound. Figure 11 is a flow chart that depicts the steps taken to format the food web bioaccumulation model and produce predictions of PCB concentrations in fish and killer whales.

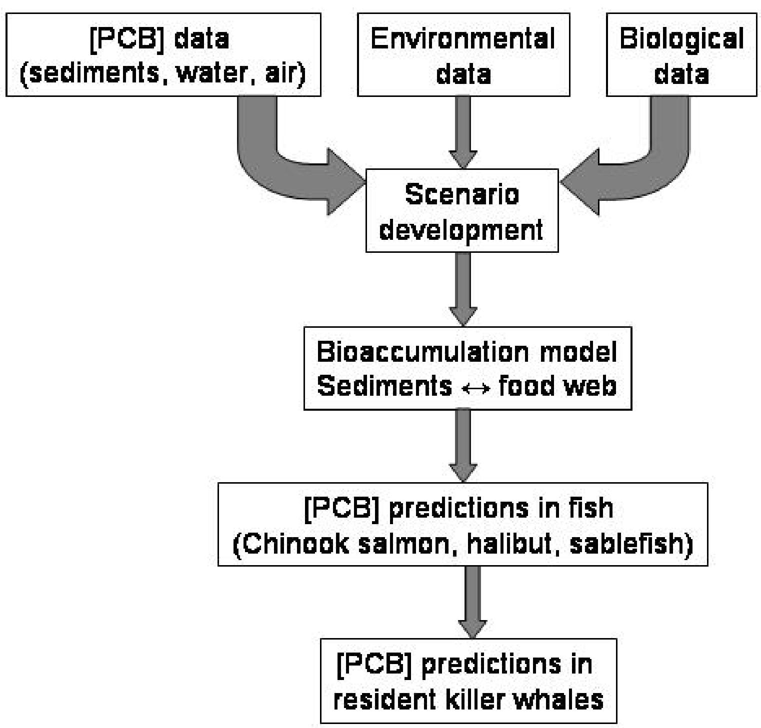


Figure 11: Flow chart depicting input into the food web bioaccumulation model required to make predictions of PCB concentrations in fish and killer whales.

2.2 GUIDING PRINCIPLES

Guiding principles are specific characteristics that shaped the development of the food web model, such as species that live in the areas included, PCB congeners found in the sediments, and information on space and time that affect the model. Major guiding principles are discussed below.

2.2.1 PCB Inputs to Resident Killer Whale Habitat

There are 209 theoretically possible PCB congeners, with 136 having been detected in killer whales in BC (Ross *et al.* 2000a). Properties of individual congeners vary, causing them to have different distributions in the environment, different levels of toxicity, and half-lives ranging from a few years to a hundred years. Even though PCBs are no longer used in Canada, they are persistent and are transported atmospherically from areas that continue to use them and cycling has produced stable concentrations in the environment (Addison and Smith 1998;Johannessen *et al.* 2008a;Tanabe *et al.* 1994).

PCBs enter killer whale habitat in a variety of ways: atmospheric deposition, urban runoff, sewage outfalls, ground water, watersheds such as the Fraser River, and smaller tributaries. Sediment PCB concentrations range from very low or non-detectable (outer coast) to extremely high as in Puget Sound's Everett Harbor (4658 $\mu g \cdot kg^{-1}$) (Long *et al.* 2005). It is important to capture the distribution of PCB congeners in the environment in the model. Empirical studies have found a wide range of congeners in resident killer whale habitat and biota; however, we have restricted those included in the model to the ones with the most data in the areas of interest (see Tables 13 and 14). Once the model has calculated concentrations of all congeners included, a total PCB (ΣPCB) concentration will be calculated which is the sum of the concentrations of the congeners included in the model.

The total toxic equivalent PCB concentration (TEQ) is also calculated by the food web bioaccumulation model. The TEQ is the sum of biota and PCB specific TEQs based on Toxic Equivalency Factors (TEFs) that are derived from various sources as:

$$TEQ = \Sigma(TEC_i) = \Sigma(TEF_i \cdot C_i) \qquad (2.1)$$

The TEQ represents the body burden or chemical dose that demonstrates a "dioxin" like mode of toxic action (most often observed in fish and mammals). The value is important because it indicates the toxicological significance of the PCB composition found in resident killer whale habitat. However, the model output may be an underestimation of the actual toxicity of the sediments because few PCB congeners included in the model have TEF values. Thus the calculated TEQs should not be used to assess the probability of exceeding TEQ-based threshold concentrations. Furthermore, other contaminants (i.e., PBDEs) have similar modes of action and they are not considered in the model's calculation of the TEQ. This also results in an underestimation of actual sediment toxicity. If data on other contaminants are available, then they could be included in future model analyses. Because of these weaknesses with TEQs, we disregarded TEQs in the food web bioaccumulation model to avoid under-predictions of PCB concentrations. Instead we used a toxicity threshold of 50 $\mu g \cdot kg^{-1}$ that was developed by Hickie *et al.* (2007). This value was derived from the tissue residue guideline for fish-eating wildlife of 0.79 $ng \cdot TEQ \cdot kg^{-1}$ (CCME 2001).

Hickie et al (2007) examined how the CCME dietary tissue residue guideline (TRG) for PCBs of 0.79 ng/kg TEQ derived to protect fish-eating wildlife compared to the use of the adverse health effects threshold for PCBs of 17 mg/kg lipid derived to protect marine mammals (Ross *et al.* 1996a). To do this, we first defined the relationship between PCB derived TEQs and total PCB concentrations in samples of chinook salmon from two British Columbia stocks. TEQs were calculated for 12 PCB congeners (77, 81, 105, 114, 118, 123, 126, 156, 157, 167, 169 and 189) using toxic equivalent factors (TEFs) derived for mammals (WHO 1998). Analysis of PCBs in chinook was done using high resolution gas chromatography (Cullon *et al.* 2009b).

TEQs and PCBs in chinook were linearly related as shown by the regression

$$\text{TEQ (ng/kg)} = 1.48*10^{-5} * \text{sPCB (ng/kg)} + 0.043 \quad (r^2 = 0.986,\ n = 12,\ p<0.001)$$

where PCBs ranged from 4,970 to 81,012 ng/kg wet wt (5 to 81 ug/kg wet wt). Based on this regression the TRG of 0.79 ng/kg TEQ equates to a sPCB concentration of 50 ug/kg wet wt in Chinook salmon. Hickie et al (Hickie *et al.* 2007) estimated that approximately 95% of an orca population feeding long-term on salmon at this PCB concentration would exceed would exceed the tissue threshold concentration of 17 mg/kg in blubber lipid. They then estimated that the diet concentration would have to be below 8 ug/kg wet wt for 95% of the orca population to fall below this tissue threshold.

2.2.2 The Structure of Killer Whale Food Webs

The structure of the resident killer whale food web is very complex and varies spatially and temporally. Not all species and interactions present in the food web were included in the model, which is a simplification of the real world and focuses on a few key species. Organisms at the same trophic level tend to have similar PCB concentrations, thus can be grouped as one trophic guild as long as the organisms included have similar feeding behaviours. One food web (coastal food web) was used in the Critical Habitat areas, Queen Charlotte Strait, and Strait of Georgia, but the outer coast area had a pelagic food web that differed slightly. Feeding behaviour is affected by prey abundance, prey size, and predator size, and the model is designed to account for these factors.

The following criteria were applied during the development of the food web structure for modeling PCB bioaccumulation in resident killer whale habitat:

1. Species of primary interest were included: northern and southern resident killer whales (*Orcinus orca*), Chinook salmon (*Oncorhynchus tshawytscha*), Pacific halibut (*Hippoglossus stenolepis*), sablefish (*Anoplopoma fimbria*), and Pacific herring (*Clupea pallasi*).
2. Species considered local to the areas considered in the model were included. These species forage primarily in the areas considered. For instance, resident killer whales have been documented to spend up to 12 months per year in the coastal waters of BC and WA, feeding on fish, principally salmonids (Ford *et al.* 1998).
3. Species from different trophic guilds relevant to the transfer and bioaccumulation of PCBs in the food web were included. Relevant trophic guilds include phytoplankton and algae, zooplankton (i.e., copepods), filter feeding invertebrates (i.e., mussels and oysters), benthic detritivores (i.e., amphipods, crabs, shrimp, and polychaetes), juvenile and adult forage and predatory fish, and resident killer whales.
4. Important trophic guilds were represented by one or two species to simplify the model and render calculations transparent.
5. Species with available empirical PCB concentration data were included to allow evaluation of the accuracy of the model predictions. PCB concentration data were available for Chinook salmon and northern resident killer whales.

We further minimized the number of species in the model to keep it simple and make model calculations more transparent. Simplifications of the food web (i.e., exact feeding preferences of fish) are consistent with evaluations of food webs that are sediment-driven (von Stackelberg *et al.* 2002b). Thus we only included the most abundant prey items for each fish species to represent their feeding behaviour. This approach produced a food web bioaccumulation model that included one category for phytoplankton, one category for zooplankton, eight invertebrate species (including detritivores and filter feeders), 12 fish species, and male, female, juvenile and newborn resident killer whales. Most of the data on ecology, feeding habits/diet composition and trophic position for fish and other aquatic biota were retrieved from www.fishbase.org (Froese and Pauly 2010) and www.sealifebase.org (Palomares and Pauly 2010), respectively. In addition, various peer-reviewed papers were consulted when information on life history parameters, prey items, and diet composition were unavailable in the web link sources. Weight and lipid content of Chinook salmon for killer whale Critical Habitats (i.e., Johnstone Strait, Strait of Georgia, and Puget Sound), for example, were obtained from Cullon *et al.* (2009a). The species that were included in the model and their feeding relationships are listed in Tables 23 and 24. Coastal and oceanic food webs are illustrated in Figures 16a and b, respectively.

2.2.2.1 Resident Killer Whales

Southern resident killer whales are composed of three pods: J, K and L. These pods range from Monterey Bay, California to Langara Island, BC, which is approximately 2000 km along the Pacific coast (Ford 2006). From early summer to late fall they are common off of southeastern Vancouver Island and Puget Sound (Ford 2006), and in July and August 90% of their time is spent in Critical Habitat in Canada and the US (Ford *et al.* 2010). In winter and spring SRKWs travel extensively in outer coastal waters (Ford *et al.* 2000b;Nichol and Shackleton 1996;Osborne 1999;Wiles 2004); however, J pod is often sighted in inshore waters all months of the year. K and L pods usually return to the Georgia Basin in May/June and leave in October/November, but from May to November all three pods make excursions to outer coastal areas for several days at a time (Ford 2006). From this information and from personal communication with John Ford (Fisheries & Oceans Canada, Pacific Biological Station, 3190 Hammond Bay Rd., Nanaimo, BC V9T 6N7) and Graeme Ellis (Fisheries & Oceans Canada, Pacific Biological Station, 3190 Hammond Bay Rd., Nanaimo, BC V9T 6N7), it was estimated that the annual distribution of SRKWs in the areas included in the food web bioaccumulation model are as follows (for modeling purposes it was assumed that annual pod distributions were the same for all pods):

- Time spent in outer coast is ~37% of the year.
- Time spent in Canadian Critical Habitat is ~18% of the year.
- Time spent in US Critical Habitat (summer core and Juan de Fuca Strait) is ~36% of the year.
- Time spent in US Critical Habitat (Puget Sound) is ~6% of the year.
- Time spent in the Strait of Georgia is ~3% of the year.

Northern resident killer whales range coastal waters from Glacier Bay, Alaska, to Gray's Harbor in Washington, which is approximately 1500 km along the Pacific coast (Ford 2006). During summer and fall they are often found in nearshore waters off northeastern Vancouver Island (Ford 2006). Like SRKWs, during winter and spring they travel extensively in outer coastal waters (Ford *et al.* 2000b;Nichol and Shackleton 1996;Osborne 1999;Wiles 2004). The Johnstone Strait Critical Habitat area is used by NRKWs all months of the year, but they are most often seen there from July-October, and are seen infrequently there from March-May (Ford 2006). On average 14.5% of the average 222 animals in the population were present in Critical Habitat from July-August (Ford *et al.* 2010). From the previous information and from personal communication with John Ford (Fisheries & Oceans Canada, Pacific Biological Station, 3190 Hammond Bay Rd., Nanaimo, BC V9T 6N7) and Graeme Ellis (Fisheries & Oceans Canada, Pacific Biological Station, 3190 Hammond Bay Rd., Nanaimo, BC V9T 6N7), it was estimated that the annual distribution of NRKWs in the areas included in the food web bioaccumulation model are as follows (for modeling purposes it was assumed that annual pod distributions were the same for all pods):

- Time spent in Critical Habitat is ~8% of the year.
- Time spent in Queen Charlotte Strait is ~17% of the year.
- Time spent in outer coast is ~75% of the year.

The distributions in the model areas described above for NRKWs and SRKWs were used in the "realistic" model scenarios, whereas "hypothetical" scenarios occurred when we considered the killer whales to spend 100% of their time in one of the model areas. This approach provides a range of scenarios that managers can evaluate.

To characterize the resident killer whale food web, published information on their diet was used to determine which fish species to include. Salmonid species comprise 96% of the diet of resident killer whales, of which 71.5% is Chinook salmon (Ford and Ellis 2006). The only non-salmonid species in their diet identified by Ford and Ellis (2006) were Pacific herring (*Clupea pallasi*), sablefish (*Anoplopoma fimbria*), yelloweye rockfish (*Sebastes ruberrimus*), quillback rockfish (*Sebastes maliger*), and Pacific halibut (*Hippoglossus stenolepis*). Ford and Ellis (2006) suspected that the herring and rockfish were not targeted as prey items but the halibut and sablefish were, as the rockfish were only partially eaten and discarded and the herring were likely consumed by salmon which were then consumed by the killer whales. Thus the main prey items of resident killer whales are Chinook salmon, and to a much smaller degree halibut, and sablefish. In "realistic" model scenarios we set the resident killer whale diet as: 96% Chinook salmon, 2% halibut, and 2% sablefish.

More recent data collection and analyses by Ford et al. (Ford *et al.* 2010) confirm the findings of (Ford and Ellis 2006). This study found that resident killer whales consumed 71% Chinook salmon, 24% chum salmon, and other salmonids comprised less than 3% each to the overall diet (Ford *et al.* 2010). However, significant variation in the percentages occurs seasonally, for example chum salmon are more important than Chinook in October and November (Ford *et al.* 2010). Upon further discussion with John Ford and Graeme Ellis (Fisheries & Oceans Canada, Pacific Biological Station, 3190 Hammond Bay Rd., Nanaimo, BC V9T 6N7), we refined the resident killer whale diet to include more species that they are likely consuming in winter months when little prey sampling studies are conducted. We considered the revised resident killer whale diet to be: 70% Chinook salmon, 15% other salmonids (10% chum, 5% coho), and 15% groundfish (3% halibut, 3% sablefish, 3% lingcod, 3% dover sole, 3% gonatid squid).

The majority of Chinook salmon consumed by SRKWs originates from the south Thompson River, but they also cosume a fair amount of south Fraser River Chinook (Ford *et al.* 2010). Resident killer whales consume approximately 75% ocean-type Chinook salmon, as stream-type Chinook migrate directly from natal rivers to the open ocean off the continental shelf and do not spend a significant amount of time in coastal waters (Ford *et al.* 2010). During winter when Chinook salmon abundance is low, ground fish such as sablefish may become prey items for resident killer whales and SRKW spend more time feeding on salmon in Puget Sound (Ford *et al.* 2010). During July and August they are likely eating close to 100% Chinook. During this time, SRKWs spend approximately 90% of their time in Critical Habitat, while NRKWs only spend 14.5% of their time in Critical Habitat during July and August (Ford *et al.* 2010). Both northern and southern resident killer whales leave Critical Habitat and head out of coastal areas, and have been found foraging at Swiftsure Bank, just outside the mouth of Juan de Fuca Strait, the extent of Critical Habitat (Ford *et al.* 2010). However, resident killer whales likely do not stray beyond the continental shelf to open ocean areas as salmon distribution is extremely patchy in those waters (John Ford, Fisheries & Oceans Canada, Pacific Biological Station, 3190 Hammond Bay Rd., Nanaimo, BC V9T 6N7, pers. comm., 2010).

There is high variability in PCB concentrations in killer whales due to age, sex, reproductive status and birth order (Ross *et al.* 2000a;Ylitalo *et al.* 2001). Newborns have very low contaminant loads, but this quickly changes as they nurse from their mother and receive her contaminant load via lipid rich milk, and the contaminant load is especially high for first born calves (Ylitalo *et al.* 2001). One year old killer whales tend to be the most contaminated members of the population, and as killer whales grow and switch to a less contaminated fish diet, their PCB concentration is diluted (Ylitalo *et al.* 2001). At approximately 15 years of age, PCB concentrations in male killer whales tend to increase, whereas females transfer their contaminant burden to their offspring (Ylitalo *et al.* 2001). The mean lifetime of female killer whales is ~50 years and males ~29 years (Olesiuk *et al.* 1990).

2.2.2.2 Chinook Salmon

Chinook salmon (*Oncorhynchus tshawytscha*) are anadromous, most of their life is spent at sea and they return to natal streams to spawn (Healey 1991). They can accumulate PCBs from the water via gill uptake, and from dietary uptake (Qiao *et al.* 2000). While some PCB exposure may occur during their time in freshwater, estuarine and coastal environments, approximately 97-99% comes from global sources during their time outside of their natal streams, in marine waters (Cullon *et al.* 2009a). During the migration back to natal streams, Chinook salmon can lose more than 80% of their lipid reserves (Brett 1995), which concentrates their PCB burden as PCBs are lipid-soluble. SRKWs feed on Chinook salmon in waters that are relatively more contaminated, near-urban, and closer to natal streams than NRKWs, thus are likely eating fish that are more contaminated and have fewer lipids (Cullon *et al.* 2009a). Adult Chinook salmon primarily feed on forage fish, such as herring, sardine, anchovy, smelt, and groundfish, but also eat krill, squid, and crab (Brodeur 1990). Two food webs for Chinook were created, one that encompasses their diet while in continental shelf waters (coastal phase), and the other for when they are off the continental shelf (pelagic phase). In the Strait of Georgia, juvenile Chinook mainly eat herring, but they also consume crab megalops, amphipods, euphausiids, and insects (Healey 1980). The diet of juvenile Chinook further north in the Strait of Georgia is much less reliant on fish. While in their pelagic phase, Chinook salmon primarily eat gonatid squid (which are micronektonic), but will also eat mid-water fish and euphausiids (Pearcy *et al.* 1988).

There are two behavioural forms of Chinook salmon life history in BC, the “stream-type” and “ocean-type”, with the ocean-type being most common (Healey 1991). The stream-type Chinook rear in freshwater for a year or more and then migrate to the ocean where they travel extensively off the continental shelf for a year or longer before returning to their natal stream several months before they spawn (Healey 1991). The ocean-type Chinook usually migrate to the ocean as juveniles within three months of emergence and usually do not disperse more than 1,000 km from their natal river, and return to their natal river a few days or weeks before spawning (Healey 1983;Healey 1991). Approximately 75% of the Chinook salmon that resident killer whales eat are ocean-type, and 25% are stream-type (Ford *et al.* 2010).

Approximately 58% of Chinook salmon eaten by resident killer whales in all areas of the BC coast are composed of stocks from the Fraser River system (Ford *et al.* 2010). This predominance of Fraser River Chinook is especially pronounced in NRKW Critical Habitat (64%) and SRKW Critical Habitat (75%) (Ford *et al.* 2010). Of these Fraser River stocks, resident killer whales primarily eat South Thompson River and Lower Fraser River Chinook (Ford *et al.* 2010). South Thompson River Chinook migrate north from after leaving freshwater, and spend the least amount of time of any Chinook stock in southern BC (Gayle Brown, Fisheries & Oceans Canada, Pacific Biological Station, 3190 Hammond Bay Rd., Nanaimo, BC V9T 6N7, pers. comm., 2010). Fraser River Chinook stocks are the most prominent Chinook stock on the coast and once they enter saltwater they do not have northward migration, but are found at all life-stages in southern BC, from the Queen Charlotte Islands to Oregon, Puget Sound, and they also spend time offshore in the open ocean (Gayle Brown, Fisheries & Oceans Canada, Pacific Biological Station, 3190 Hammond Bay Rd., Nanaimo, BC V9T 6N7, pers. comm., 2010). To simplify the modeling process, we assumed that resident killer whales only eat South Thompson and Fraser River stocks of Chinook salmon.

Fishing mortality distribution tables (from 1985 to 2007) for Chinook salmon in different fishery regions were used as a proxy for the annual percent time Chinook spend in the model areas, and were provided by Gayle Brown (Fisheries & Oceans Canada, Pacific Biological Station, 3190 Hammond Bay Rd., Nanaimo, BC V9T 6N7). South Thompson River Chinook are represented by the lower Shushwap hatchery indicator stock, and Fraser River stocks by Chilliwack River hatchery stock (Gayle Brown, Fisheries & Oceans Canada, Pacific Biological Station, 3190 Hammond Bay Rd., Nanaimo, BC V9T 6N7, pers. comm., 2010). Both of these stocks are ocean-type Chinook (Gayle Brown, Fisheries & Oceans Canada, Pacific Biological Station, 3190 Hammond Bay Rd., Nanaimo, BC V9T 6N7, pers. comm., 2010). Fishery areas in the fishing mortality distribution tables were converted to areas in the model as follows:

- AABM (SEAK and WCVI) and ISBM (WA/OR coast) were considered equivalent to the outer coast area in the model.
- AABM (NBC) was considered equivalent to the Queen Charlotte Strait area in the model.
- ISBM (Geo St and Canada) were considered equivalent to the Strait of Georgia and SRKW Critical Habitat in Canada and NRKW Critical Habitat.
- ISBM (Puget Sound) was considered equivalent to SRKW Critical Habitat in the US (summer core and Juan de Fuca) and SRKW Critical Habitat in the US (Puget Sound).
- ISBM (Terminal) was not included because that fishery specifically targets fish as they re-enter their natal river, and they are likely only transiting that area and are not eating and accumulating contaminants.

Table 6, lists the average annual distribution (% time) South Thompson and Fraser River Chinook salmon spend in the areas included in the model, and were labelled “realistic” scenarios. Hypothetical scenarios occurred when we considered the salmon to occupy a model area for 100% of its life to obtain best and worst case results.

Table 6: Average annual distribution (% time) of South Thompson and Fraser River Chinook in the areas included in the model (Gayle Brown, Fisheries & Oceans Canada, Pacific Biological Station, 3190 Hammond Bay Rd., Nanaimo, BC V9T 6N7, pers. comm., 2010).

Area	**South Thompson Chinook**	**Fraser River Chinook**
Outer coast	80%	55%
Queen Charlotte Strait	8%	2%
NRKW Critical Habitat (CH)	3%	14%
Strait of Georgia	3%	8%
SRKW CH in Canada	3%	8%
SRKW CH in US (summer core and Juan de Fuca Strait)	2%	4%
SRKW CH in US (Puget Sound)	0.2%	9%

2.2.2.3 Chum Salmon

Chum salmon (*Oncorhynchus keta*) are benthopelagic and anadromous, as they inhabit coastal streams before moving to the ocean (Riede 2004). Migrating fry form schools in estuaries, and remain close to shore for a few months before dispersing into the ocean (Scott and Crossman 1973). The diet of juveniles and adults is composed mainly of copepods, tunicates, euphausiids, pteropods, squid, and small fishes (Scott and Crossman 1973). The diet is 17-40% pteropods, 17-60% euphausids, 52% fish, 10% salps, and10% miscellaneous (Birman 1960). The order of importance of food items is (1) amphipod / euphasid / pteropod / copepod, (2) fish, and (3) squid larvae (Kanno and Hamai 1971).Table 25 contains information on the values used in the model for chum salmon for the improved resident killer whale diet.

2.2.2.4 Coho Salmon

Coho salmon (*Oncorhynchus kisutch*) are demersal and anadromous (Riede 2004). They are found in oceans and lakes, and adults return to their natal rivers to spawn (Morrow 1980). Immature fish emerge in the spring and usually remain in fresh water for 1-2 years (sometimes up to 4 years) (Morrow 1980), and after that time they migrate at night to freshwater lakes or to the sea (Scott and Crossman 1973). Overall, Chinook salmon and coho salmon have a more coastal marine distribution along the continental shelf than do sockeye salmon, pink salmon, and chum salmon (Quinn 2005). When smolts reach the sea they remain close to the coast and feed on planktonic crustaceans, and as they grow they move farther out to sea and feed upon larger organisms (Morrow 1980) such as jellyfish, squid, and fishes (Coad and Reist 2004). Herring and sandlance comprise ~32% of their diet, amphipods ~34%, and crab megalops ~26% (Sandercock 1991). Adult coho and Chinook have very similar diets, except invertebrates comprise approximately one-fifth of the coho diet, and less than 3% for Chinook (Sandercock 1991). Table 25 contains information on the values used in the model for coho salmon for the improved resident killer whale diet.

2.2.2.5 Pacific Halibut

The maximum reported age of a Pacific halibut (*Hippoglossus stenolepis*) is 42 years (Armstrong 1996). It is one of the largest flatfish in the world, and the maximum reported size is 3 m and over 200 kg (Mecklenburg *et al.* 2002). This species lives near the bottom of the ocean, and adults spend the winter in deep waters (250-600 m) along the edge of the continental shelf, where spawning occurs in late January to mid-March (Armstrong 1996;Loher and Blood 2009a;Loher and Seitz 2008). British Columbian Halibut aggregate to spawn off Langara Island and Cape St. James (Skud 1977;St.Pierre 1984). In the summer they move to shallow coastal waters (<200 m deep) (Loher and Seitz 2008) to feed on fishes, crabs, clams, squid, and invertebrates (Hart 1973). Halibut can also move alongshore seasonally, and some of British

Columbia's summer biomass may join spawning groups in southern Alaskan waters, while halibut from Washington and Oregon may move north to Canadian waters (Loher and Blood 2009b;Loher and Blood 2009a).

2.2.2.6 Sablefish
Sablefish (*Anoplopoma fimbria*) are found on mud bottoms in waters deeper than 200 m (Allen and Smith 1988), with adults usually at the continental shelf-slope margin (Harvey 2009). They tend to be localized but some juveniles migrate more than 2,000 miles over 6-7 years (Armstrong 1996). They are a long lived species with a maximum reported age of 114 years (Beamish and MacFarlane 2000), and can reach up to 57 kg in weight (Eschmeyer *et al.* 1983), and one meter in length (Schirripa and Colbert 2005). Their diet is composed of crustaceans, worms, and small fishes (Clemens and Wilby 1961).

2.2.2.7 Lingcod
Lingcod (*Ophiodon elongates*) are demersal, ranging from the intertidal to depths of 475 m (Allen and Smith 1988), with adults typically found near rocks, and young found on sand or mud bottom of bays and inshore areas (Eschmeyer *et al.* 1983). They area oceanodromous (Riede 2004), and both migratory and non-migratory populations exist (Hart 1973). The average weight of lingcod is 30 kg (Stock and Meyer 2005), and the maximum reported age is 20 years (Miller and Geiber 1973). Young feed on copepods and other small crustaceans (Hart 1973); while adults mainly eat other fishes but they also take crustaceans, octopi, and squid (Clemens and Wilby 1961). Table 25 contains information on the values used in the model for lingcod for the improved resident killer whale diet.

2.2.2.8 Dover Sole
Dover Sole (*Microstomus pacificus*) are demersal, with a depth range from 10 - 1370 m (Russian Academy of Sciences 2000). They are found on mud bottoms (Eschmeyer *et al.* 1983), and move into deep water in winter (Eschmeyer *et al.* 1983). The average male weight is 245 g, and female weight is 508 g (Choromanski *et al.* 2005), and the maximum reported age is 45 years (Beverton *et al.* 1985). The diet of adults is 10.5-42.7% polychaetes, 41.4-84% ophiuroids, 3.5-14.5% mollusks, and 1.5-2.1% crustaceans (Gabriel and Pearcy 1981). Table 25 contains information on the values used in the model for dover sole for the improved resident killer whale diet.

2.2.2.9 Pacific Herring
Pacific herring (*Clupea pallasii pallasii*) populations in Puget Sound and the east side of the Strait of Georgia are non-migratory (Therriault *et al.* 2009). However, most herring populations in the Strait of Georgia are migratory and spend late spring, summer, and fall in feeding grounds (shelf waters <200 m deep) on the west coast of Vancouver Island (Tanasichuk 1997;Therriault *et al.* 2009). There is also a herring stock on the west coast of Vancouver Island, which comingles with the Strait of Georgia stock on the summer feeding grounds (Megrey *et al.* 2007). During the fall, herring form dense concentrations and then congregate in spawning areas in February and March (Therriault *et al.* 2009). Spawning occurs from February to May (mainly in March and April) and is concentrated on the east side of Vancouver Island between Saltspring and Denman islands (Therriault *et al.* 2009). Juvenile herring (at least one year of age) do not migrate until after their second summer in the Strait of Georgia (Therriault *et al.* 2009). Thus migratory herring populations spend approximately half the year in the Strait of Georgia. Adult herring feed primarily on zooplankton, larval invertebrates, and small fish (Iverson *et al.* 2002;Robinson 2000;Wailes 1936). West *et al.* (2008) found resident herring in Puget Sound had 3-9 times the PCB contamination level of herring in the Strait of Georgia, which is likely due to their year-round proximity to near-urban areas. Herring populations from northern British

Columbia are composed of three spawning stocks: Queen Charlotte Islands, Prince Rupert, and Central Coast (Megrey *et al.* 2007). These three populations spawn in locations different than the southern populations.

2.2.2.10 Gonatid Squid

Squid (*Gonatius sp.*) were included in the revised resident killer whale diet, and in the outer coast food web as the oceanic life stage of Chinook salmon feed predominantly (~70%) on gonatid squid, and to a lesser extent on fish and zooplankton (Brodeur 1990;Ito 1964).

2.2.2.11 Pollock

Pollock (*Theragra chalcogramma*) are small (max length 91 cm) (Eschmeyer *et al.* 1983) benthopelagic (depth range 0-1280 m), non-migratory fish (Fedorov *et al.* 2003) that can live up to 15 years old (Cohen *et al.* 1990). Pollock undergo diurnal vertical migrations (Cohen *et al.* 1990), and their diet is predominantly composed of krill (Anonymous 2001), but they also eat fish and crustaceans (Hart 1973).

2.2.2.12 Shiner Surfperch

Shiner surfperch (*Cymatogaster aggregata*) are small (max length 20 cm) (Morrow 1980) demersal, non-migratory fish (Eschmeyer *et al.* 1983) that can live up to 9 years of age (Shanks and Eckert 2005). Juveniles mainly eat copepods, and adults mainly eat various small crustaceans, mollusks, and algae (Morrow 1980).

2.2.2.13 Northern Anchovy

Most populations of Northern Anchovy (*Engraulis mordax*) remain off the west coast of Vancouver Island, and are unlikely to be significant forage fish in the Strait of Georgia (Therriault *et al.* 2009). Adult anchovy mainly feed on zooplankton such as euphausiids, copepods, and decapod larvae (Kucas 1986).

2.2.2.14 Benthic Invertebrates

Benthic organisms have a wide variety of feeding strategies (e.g., deposit feeding, suspension feeding, filter feeding, scavenging), thus process PCBs bound to organic matter in the sediments and water column (Burd *et al.* 2008a). While there is no direct link between zooplankton and PCBs in sediments, they make take up PCBs directly from the water column (Del Vento and Dachs 2002), or from re-suspended sediments.

2.2.3 Spatial Resolution of PCBs in the Food Web

The model was designed to focus on seven specific areas that make up the habitat of northern and southern resident killer whales in BC and WA (Figure 12). Areas were designated as:

1. *Outer coast:* The total size of this area is 107,878 km^2, which was determined using Fisheries & Oceans Canada's Mapster program (http://www-heb.pac.dfo-mpo.gc.ca/maps/maps-data_e.htm). The size of this area was based on resident killer whale distribution but also had to consider Chinook salmon distribution while offshore. This area could have been created to be much larger, but due to the lack of sediment samples in outer coast areas the size was restricted. The average sediment PCB concentration for this area is 0.695 ± 0.182 $\mu g \cdot kg^{-1}$, dry weight.
2. *Queen Charlotte Strait:* The total size of this area is 1,858 km^2, which was determined using Fisheries & Oceans Canada's Mapster program (http://www-heb.pac.dfo-mpo.gc.ca/maps/maps-data_e.htm). This area was meant to capture the coastal waters north of NRKW Critical Habitat to the northern tip of Vancouver Island. The average sediment PCB concentration for this area is 0.695 ± 0.182 $\mu g \cdot kg^{-1}$, dry weight.

3. *NRKW Critical Habitat:* The total size of this area is 904.61 km^2 (Waleed Elmarimi, Fisheries & Oceans Canada, Pacific Regional Headquarters, 401 Burrard St., Vancouver, BC V6C 3S4, pers. comm., 2010). The total size of the two disposal at sea sites in this area is 5.38 km^2 (Environment Canada 2006), thus the disposal at sea sites comprise 0.59% of this area. The average sediment PCB concentration for this area is 0.442 ± 0.105 $\mu g \cdot kg^{-1}$, dry weight.
4. *Strait of Georgia:* The total size of this area is 4,641.8 km^2, which was determined using Fisheries & Oceans Canada's Mapster program (http://www-heb.pac.dfo-mpo.gc.ca/maps/maps-data_e.htm). The total size of the seven disposal at sea sites in this area is 24.22 km^2 (Sean Standing, Environment Canada, Disposal at Sea Program, #201-401 Burrard St., Vancouver, BC V6C 3S5, pers. comm., 2010), thus the disposal at sea sites comprise 0.52% of this area. The average sediment PCB concentration for this area is 1.05 ± 0.1693 $\mu g \cdot kg^{-1}$, dry weight.
5. *SRKW Critical Habitat in Canada:* The total size of this area is 2,495.52 km^2 (Waleed Elmarimi, Fisheries & Oceans Canada, Pacific Regional Headquarters, 401 Burrard St., Vancouver, BC V6C 3S4, pers. comm., 2010). The total size of the three disposal at sea sites in this area is 6.27 km^2 (Sean Standing, Environment Canada, Disposal at Sea Program, #201-401 Burrard St., Vancouver, BC V6C 3S5, pers. comm., 2010), thus the disposal at sea sites comprise 0.25% of this area. The average sediment PCB concentration for this area is 0.518 ± 0.151 $\mu g \cdot kg^{-1}$, dry weight.
6. *SRKW Critical Habitat in the USA (summer core and Juan de Fuca Strait):* The total size of this area is 4,690 km^2 (Lynne Barre, National Marine Fisheries Service, 7600 Sand Point Way, NE, Seattle, WA 98115, pers. comm., 2010). The average sediment PCB concentration for this area is 74 ± 0.1 $\mu g \cdot kg^{-1}$, dry weight.
7. *SRKW Critical Habitat in the USA (Puget Sound):* The total size of this area is 2,230 km^2 (Lynne Barre, National Marine Fisheries Service, 7600 Sand Point Way, NE, Seattle, WA 98115, pers. comm., 2010). The average sediment PCB concentration for this area is 2196 ± 6972 $\mu g \cdot kg^{-1}$, dry weight.

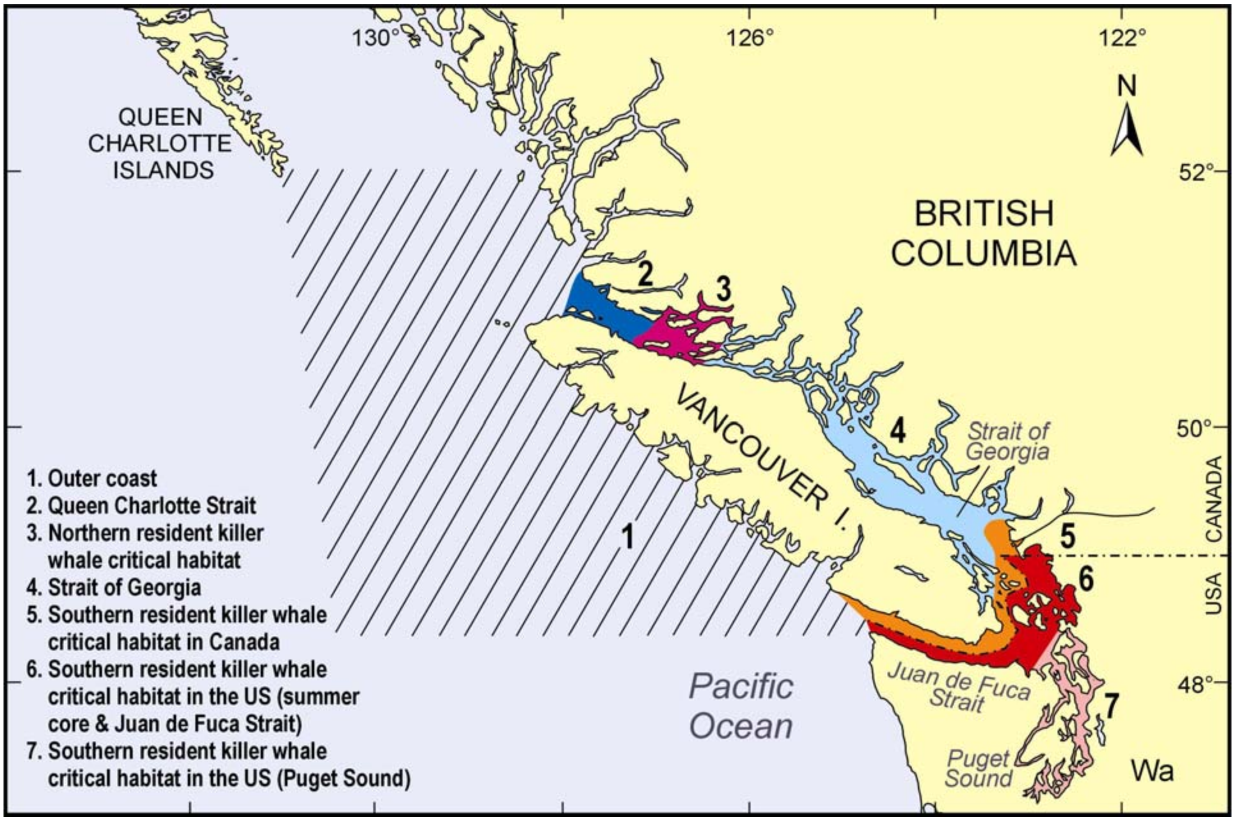


Figure 12: The seven areas included in the food web bioaccumulation model. Designated Critical Habitat for northern (Area 3) and southern (Area 5) resident killer whales in British Columbia and in the US (Areas 6 and 7) are also depicted in the figure (map created by Patricia Kimber, Tango Design).

These seven areas were designated because they represent areas of management or jurisdictional interest and relevance, they capture disposal at sea sites, they are areas that resident killer whales frequent, they have been previously studied and sediment concentrations and biota concentrations are available. The range of PCB concentrations in the different areas modelled result in significant variation in PCB concentrations in biota. Thus areas considered "PCB hot spots" tend to also have biota with high levels of PCBs (as long as the biota do not move significant distances). Organisms with limited mobility (e.g., certain invertebrates such as mussels, oyster, and polychaetes) are likely to reflect the PCB concentrations in their immediate environment. Hence, if they reside in a "hot spot" (i.e., Puget Sound), PCB concentrations are likely to be greater than concentrations in organisms that inhabit less PCB contaminated areas. However, several of the species included in the model have very wide foraging ranges that encompass several areas included in the model (e.g., Chinook salmon and killer whales). These species are exposed to widely variable sediment PCB concentrations, such that their tissue concentrations are proportional to the time spent in each area and the concentrations of PCBs present in those areas. The contributions from each area are then averaged to determine what the biota is exposed to.

The objective of the model is to charcaterize the main relationships between PCB concentrations in sediment, water and biota of the food-web. The model is based on the principle that PCB concentrations in all components of the food-web are related. The purpose of the food-web model is to characterize the most important relationships with the goal to assess the response of the PCB concentrations in killer whales to changes in PCB concentrations in the sediments. The model takes a long term view of the PCB distribution in the food-web, i.e. it determines the PCB concentration relationships at steady-state, a situation that is eventually achieved after an action such as PCB contaminated sediment disposal has occurred. The relationships between PCB concentrations in water, sediment and biota are based on a combination of empirical measurements where PCB concentration data are available and theoretical clalcuations where emprirical data are absent.

PCB sediment concentration monitoring programs have included a significant distribution of PCB sediment concentration hot spots throughout the Strait of Georgia and transboundary areas of Puget Sound (Grant *et al.* 2010). A fairly large number of independent sediment PCB concentration measurements have been collected from the region and can provide a reasonable representation of the spatial distribution of the PCB concentrations in the Critical Habitats. Table 7 indicates sites where sediment PCB data was obtained and then used in the food web model. The wildlife species included in the model are distributed over large areas of the Critical Habitats in the Strait of Georgia-Puget Sound region and most of them are year-round residents of the region, except for the oceanic stage of Chinook salmon and winter foraging areas of resident killer whales. The model accounts for PCB binding to organic carbon in the water column and sediments, which causes the PCBs to lose their bioavailability. To determine the ability of the overall model to mimic reality, model results were compared to available empirical data. The model performance was determined by comparing predicted PCB concentrations in biota to those measured empirically, and the model uncertainty was characeterized and considered in the interpretation of the model results. The model does not aim to predict PCB concentrations in the sediments from air-sea exchange and other sources. This is justified in our view as the model is applied to sediment disposal where it is clear what the source of the PCBs in the sediments is.

The data provided in Table 7 wer used as baseline to reconstruct PCB data for missing PCB congeners. The only exception is for NRKW Critical Habitat, Queen Charlotte Strait, and the Outer coast areas as data was complete and did not require congener re-constructions.

Table 7: Sediment sampling sites in the areas included in the resident killer whale food web bioaccumulation model. Area 1 is the outer coast, area 2 Queen Charlotte Strait (no samples available from this area), area 3 is northern resident killer whale Critical Habitat, area 4 is the Strait of Georgia, area 5 is southern resident killer whale Critical Habitat in Canada, area 6 is southern resident killer whale Critical Habitat in the US (summer core and Juan de Fuca), area 7 is southern resident killer whale Critical Habitat in the US (Puget Sound). Note that none of the sites have sediment total PCB concentrations that exceed the CEPA guideline of 100 μg·kg^{-1}, dry weight.

Area	Site #	Sample Site	Total PCB (μg·kg^{-1}, dw)	Latitude	Longitude	Ref
1	1	Queens Sound	2.04	51° 54' 12.60" N	128° 21' 27.84" W	1
1	2	SW Calvert Island	0.06	51° 25' 19.38" N	128° 16' 15.48" W	1
3	3	Queen Charlotte Strait / Malcolm Island	1.15	50° 42' 56.64" N	127° 05' 06.42" W	1
3	4	W Cracroft Island, Johnstone Strait	0.15	50° 28' 58.26" N	126° 14' 51.60 W	1
4	5	Bazan Bay	1.40	48° 37' 50.22" N	123° 24' 1.20" W	2
4	6	GVRD 7	0.51	49° 1' 59" N	123° 13' 12" W	2, 3, 4
4	7	GVRD 18	0.37	48° 51' 45.72" N	123° 05' 40.86" W	2, 4
4	8	GVRD 2	2.91	49° 19' 54.12" N	123° 18' 30.54" W	2, 3, 4
4	9	GVRD 20	2.06	49° 19' 9.72" N	123° 48' 5.52" W	2, 4
4	10	GVRD 14	1.76	49° 21' 33.18" N	123° 34' 22.20" W	2, 4
4	11	GVRD 5	1.12	49° 09' 52.62" N	123° 32' 43.80" W	2, 3, 4
4	12	Williamson Landing	0.89	49° 27' 20.58" N	123° 28' 10.98" W	2
4	13	GVRD 19	0.68	49° 01' 7.20" N	123° 23' 5.40" W	2, 4
4	14	Decanso Bay	0.66	49° 10' 29.04" N	123° 51' 52.38" W	2
4	15	Howe Sound	0.66	49° 33' 24.78" N	123° 14' 23.64" W	2
4	16	Willy Island	0.61	48° 54' 44.76" N	123° 40' 12.48" W	2
4	17	GVRD 15	0.33	49° 07' 26.40" N	123° 27' 28.92" W	2, 4
4	18	Cowichan Bay	0.30	48° 45' N	123° 34' W	2
4	19	GVRD 21	5.00	49° 58' 48.00" N	125° 04' 18.00" W	2, 4
4	20	Powell River	2.71	49° 52' N	124° 34' W	2
4	21	GVRD 10	2.53	49° 50' 40.32" N	124° 53' 12.42" W	2, 4
4	22	Blubber Bay	1.67	49° 47' 57.00" N	124° 36' 48.78" W	2
4	23	GVRD 11	1.50	49° 42' 11.34" N	124° 38' 0.90" W	2, 4
4	24	Oyster River	0.44	49° 55' 17.40" N	125° 09' 19.80" W	2
4	25	Scuttle Bay	0.42	49° 54' 27.42" N	124° 38' 5.16" W	2
4	26	Hurtado Point	0.18	49° 57' 42.36" N	124° 44' 33.00" W	2
4	27	Manson Landing	0.11	50° 0' 15.00" N	124° 59' 18.00 W	2
4	28	GVRD 1	2.36	49° 35' 30.90" N	124° 38' 16.50" W	2, 3, 4
4	29	GVRD 17	1.76	49° 24' 30.48" N	124° 02' 2.58" W	2, 4
4	30	GVRD 9	1.59	49° 27' 26.46" N	124° 03' 8.88" W	2, 4
4	31	GVRD 8	1.41	49° 26' 20.28" N	123° 54' 35.40" W	2, 4

4	32	GVRD 12	1.39	49° 26' 12.06" N	124° 22' 55.32" W	2, 4
4	33	GVRD 22	0.93	49° 38' 11.52" N	124° 13' 38.52" W	2, 4
4	34	GVRD 13	0.81	49° 23' 3.36" N	124° 12' 36.66" W	2, 4
4	35	Lasqueti River	0.28	49° 28' 48.00" N	124° 12' 14.40" W	2
4	36	Oyster River	2.76	49° 56' 26" N	125° 04' 14" W	1
4	37	W Texada Island	5.06	49° 34' 2.22" N	124° 34' 57.48" W	1
4	38	S Gabriola Island	2.54	49° 14' 59.34" N	123° 42' 5.82" W	1
4	39	E Active Pass	0.98	48° 54' 0.00" N	123° 13' 59.40" W	1
4	40	Patricia Bay	14.82	48° 39' 17.70" N	123° 30' 16.68" W	1
5	41	Samuel Island	2.40	48° 48' 27.30" N	123° 13' 06.78" W	2
5	42	GVRD 3	1.21	49° 12' 28.02" N	123° 17' 59.52" W	2, 3, 4
5	43	GVRD 16	0.97	49° 01' 36.30" N	123° 20' 14.22" W	2, 4
5	44	GVRD 6	0.67	48° 56' 11.46" N	123° 18' 47.52" W	2, 3, 4
5	45	Deep Cove	0.56	48° 41' 07.98" N	123° 28' 39.00" W	2
5	46	GVRD 4	0.48	49° 07' 46.68" N	123° 18' 42.84" W	2, 3, 4
6	47	Ave. of San Juan Island, E Juan de Fuca Strait, and Admiralty Inlet	9.3	N/A	N/A	5
6	48	Discovery Bay	1.81	48° 02' N	122° 51' W	6
6	49	Makah Bay	1.84	48° 18' 36.00" N	124° 40' 12.00" W	6
7	50	Puget Sound Main	5.79	47° 45' N	122° 27' W	7
7	51	Commencement Bay	24271.0	47° 17' N	122° 25' W	5
7	52	Possession Sound, Gedney Island	3.29	48° 2' 19.14" N	122° 18' 59.15" W	8
7	53	S Port Townsend	14.2	48° 2' 24.61" N	122° 44' 36.67" W	8
7	54	Possession Sound	8.64	47° 54' 24.52" N	122° 20' 12.62" W	8
7	55	Shoreline of Elliott Bay	57.0	47° 37' 26.18" N	122° 22' 26.76" W	8
7	56	East Harbor Island	1870.0	47° 35' 4.70" N	122° 20' 44.88" W	8
7	57	Port Gamble Bay	6.68	47° 50' 10.64" N	122° 34' 42.67" W	8
7	58	Dabob Bay	6.68	47° 44' 4.74" N	122° 50' 38.69" W	8
7	59	Port of Olympia	29.4	47° 3' 5.90" N	122° 53' 45.17" W	8
7	60	Case Inlet	6.59	47° 16' 10.45" N	122° 51' 3.67" W	8
7	61	Hylebos Waterway	73.0	47° 16' 43.14" N	122° 23' 54.31" W	8

Table References:

1. Fisheries & Oceans Canada, unpublished data.
2. Grant *et al.* (2010)
3. Johannessen *et al.* (2008a)
4. Wright *et al.* (2008)
5. Washington State Department of Ecology (2002)
6. Wilson and Partridge (2007)
7. Pelletier and Mohamedali (2009)
8. Washington State Western Coastal Environmental Monitoring and Assessment Program (EMAP), unpublished data

Since killer whales are warm-blooded, air-breathing organisms, in which the chemical inhalation and exhalation are important routes for uptake and elimination of PCBs, PCB air concentrations were also incorporated in the food web models. Air concentrations of total PCBs were obtained from the near urban Saturna Island station to represent air concentration (9.3×10^{-6} ng·L^{-1}) in Critical Habitats within the Strait of Georgia, and the remote Ucluelet station for air concentration (8.9×10^{-6} ng·L^{-1}) in offshore habitat at the west coast of Vancouver Island (Noël *et al.* 2009). These PCB concentrations in air are very low and may not represent a significant source to the killer whale burden, but the model builds on the assumption that an increase in sediment PCBs associated with disposal would lead to a consequent increase in delivery of PCBs to the killer whale food web.

2.2.4 Steady-State Vs. Time Dependence

Steady state models assume that contaminant concentrations have enough time to partition between the water column, the sediments, and biota in the food web and reach a dynamic "equilibrium" (contaminant concentrations no longer change over time). However, seasonal changes and the effect of age on PCB concentrations can still be captured with a steady state approach by using the appropriate parameters. A steady state rather than time dependent approach was adopted for the resident killer whale food web bioaccumulation model because the time response of sediment PCB concentrations to changes in loadings and external conditions is slow compared to that in biota. The environmental half-life for PCBs has been estimated to range from a few years to 100 years (Jonsson *et al.* 2003;Sinkkonen and Paasivirta 2000), while the half life of PCB 126 in rainbow trout (a salmonid) ranges from 82-180 days (Brown *et al.* 2002). This assumption is valid for small aquatic organisms (e.g., plankton) as equilibrium between uptake and elimination is quickly reached; however, this process can be much longer for larger organisms (e.g., seals and killer whales), as their body burden often lags behind changing environmental conditions (Hickie *et al.* 2007). Thus steady-state models often overestimate concentrations in larger organisms because those concentrations are not likely to be reached in the short time-span that the model considers (Natale 2007). To maintain simplicity in the model we applied a steady state approach, and included different age classes for certain organisms in the food web to account for age specific differences in PCB concentration. The temporal response of PCB concentrations in the sediments is the "rate controlling" step in the model. PCB concentrations in sediments varied among the various exposure scenarios but were kept constant for each scenario.

Predictions of PCB concentrations in the food web are obtained by inputting measured sediment and water concentrations from each area into the model. In response to requests from management, a major goal of the model is to determine if disposal at sea has any impacts in resident killer whale Critical Habitat. The model is designed to predict the steady state concentrations in biota due to exposure to PCBs in air, water, and sediments. While the model cannot predict how quickly this equilibrium will be achieved, effort will be made to determine this based on sedimentation rates and bioturbation rates in the different areas modelled. A time-dependent model may be more appropriate for answering that question; however, that type of model is very complex and the development would take much longer than the timeline management requires.

2.3 MODEL DESCRIPTION

2.3.1 General Model Description

The development of the PCB bioaccumulation model of the coastal and oceanic food webs for Chinook salmon and killer whale Critical Habitats was based on the application of a food web bioaccumulation model for PCBs developed for San Francisco Bay, CA, USA (Gobas and Arnot

2010) and the previous Gobas (1993) model. The aim of this model is to characterize the relationship between the concentrations of PCBs in sediments and key biological species (i.e., Chinook salmon) in resident killer whale Critical Habitats for their role as a vector for biota exposure and eco-toxicological risk significance. The food web bioaccumulation model for resident killer whales is comprised of two modules – the science module and management module. The science module contains information used to calculate Biota Sediment Accumulation Factors (BSAFs) for PCB congeners and ΣPCBs (the congeners included for each area are listed in Tables 1-7 of Appendix III), such as internal and external variables, functional relationships, and data for the evaluation of model performance. The main output of the model is the BSAF, which characterizes the relationship between PCB concentrations in biota (C_B; g PCB·kg^{-1}, wet weight organism), to those in sediments (C_S; g PCB·kg^{-1}, dry weight sediment):

$$BSAF = C_B / C_S \tag{2.2}$$

The model calculates BSAF values (kg dry sediment/kg wet weight organism) for each PCB congener in every species included in the model. BSAF values are output as statistical distributions rather than a single point estimate, to allow for seasonal variation.

In the management module, BSAF values are used to “forward” and “backward” calculate PCB concentrations (Figure 13).

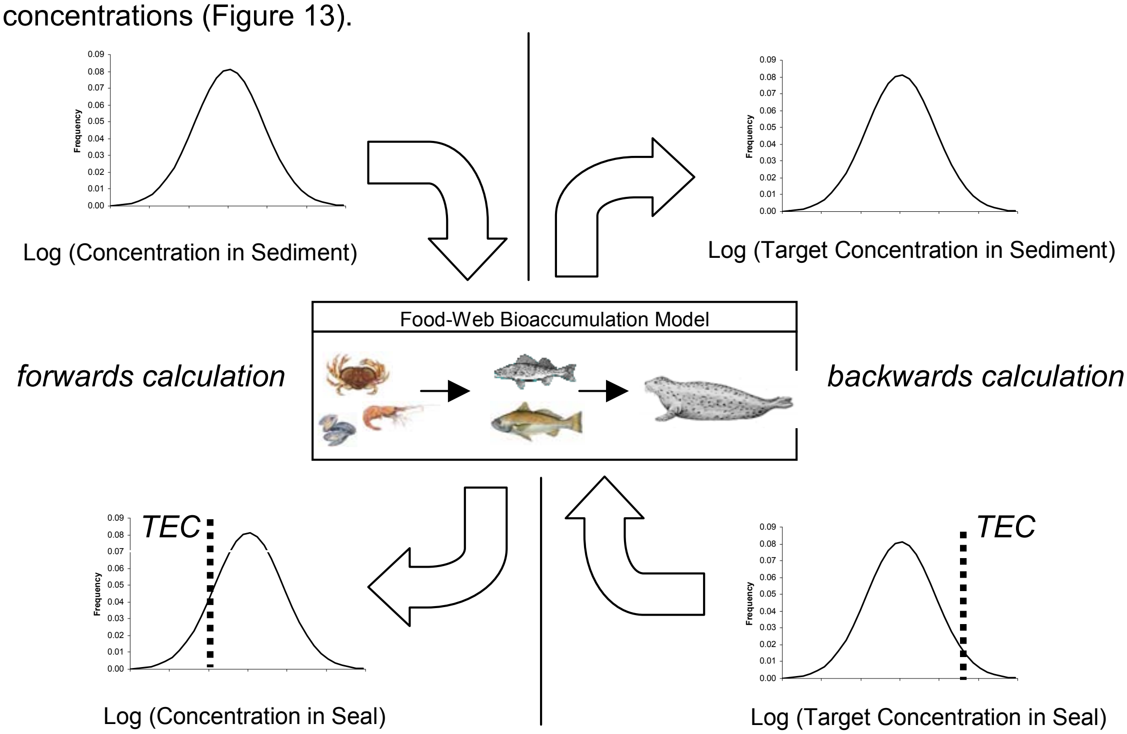


Figure 13: Illustration of the forward and backward applications of the BSAF in the food web bioaccumulation model for PCBs (taken from Gobas and Arnot 2010). TEC is the toxic effect concentration.

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

$$C_B = BSAF \cdot C_S \tag{2.3}$$

Backward calculations use PCB concentrations in biota (C_B) to predict PCB concentrations in sediments (C_S). This method is used to determine concentrations in sediments that are below thresholds for adverse health effects in biota. The equation for backward calculations is:

$$C_S = C_B / BSAF \tag{2.4}$$

The model uses various state variables (e.g., octanol water partition coefficient, lipid content, temperature, weight) to derive BSAFs. PCB concentration data are only used in the management module, where they are referred to as "external variables".

Several mathematical equations are used in the food web bioaccumulation model to describe PCB uptake and elimination in biota. Equations for air breathers (killer whales) are different than those for water breathers (plankton, benthic invertebrates, and fish), and the description of the model reflects this by being split into the two groups. Water breathing organisms absorb PCBs from the water via their respiratory surfaces as well as through their diet, while air breathing organisms absorb chemicals from their diet.

<u>2.3.2 Description of Food Web Bioaccumulation Model: Phytoplankton, Zooplankton, Aquatic Invertebrates, and Fish</u>

A conceptual representation of the main routes of PCB uptake and depuration in aquatic organisms that obtain oxygen from the water for ventilation is shown in Figure 14. The food web bioaccumulation model is based on the assumption that PCB exchange between an aquatic organism and the ambient environment can be sufficiently described by:

$$dM_B/dt=[W_B\cdot(k_1\cdot [m_O\cdot \Phi \cdot C_{WT,O} + m_P\cdot C_{WD,S}] + k_D\cdot \Sigma(P_i\cdot C_{D,i}))] - (k_2 + k_E + k_M)\cdot M_B \tag{2.5}$$

Where the mass (g) of the PCB congener in the organism is M_B, the net flux of PCB congener uptake and elimination by the organism at any point in time t (d) is dM_B/dt, the weight of the organism (kg) at time t is W_B, the elimination rate constant (L/kg·d) for uptake from the respiratory organ (i.e., gills or skin) is k_1, the fraction of respiratory ventilation of overlying water is m_O, the fraction of respiratory ventilation of sediment associated pore water is m_P, the fraction of the total chemical concentration in overlying water that is freely dissolved and can be absorbed via membrane diffusion is Φ (unitless), the total concentration ($g\cdot L^{-1}$) of the PCB congener in the water column above the sediments is $C_{WT,O}$, the freely dissolved PCB congener concentration ($g\cdot L^{-1}$) in the sediment associated pore/interstitial water is $C_{WD,S}$, the clearance rate constant (kg/kg·d) for chemical uptake via ingestion of food and water is k_D, the diet fraction consisting of prey item i is P_i, the PCB congener concentration ($g\cdot kg^{-1}$) in prey item i is $C_{D,i}$, the PCB elimination rate constant (d^{-1}) via the respiratory area (i.e., gills and skin) is k_2, the PCB elimination rate constant (d^{-1}) via excretion into egested feces is k_E, and the PCB metabolic transformation rate constant (d^{-1}) is k_M.

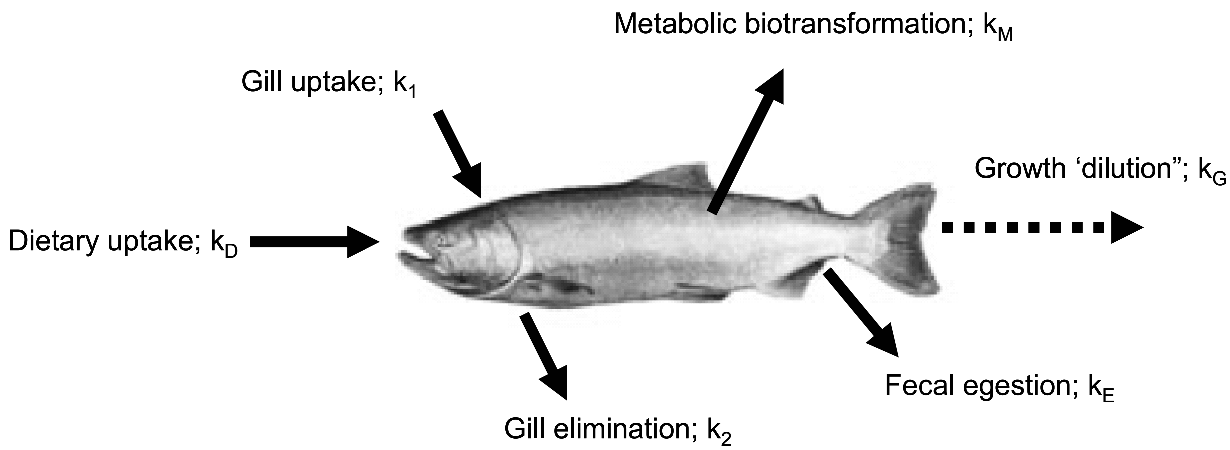


Figure 14: Conceptual diagram of major routes and associated rate constants of chemical (i.e., PCBs) uptake and elimination processes in fish (in this case Chinook salmon is used as an example).

Water column PCB concentrations for the model input ideally would come from the disposal site (von Stackelberg *et al.* 2002b). Since this information does not exist, it was assumed that the dredged sediments were in equilibrium with the overlying water. The predicted upperbound water PCB concentrations were estimated from the organic carbon-water partition coefficient, K_{oc}, and the distribution of organic carbon-normalized sediment concentrations for the dredged sediments (von Stackelberg *et al.* 2002b).

For phytoplankton and algae the value of k_D is zero, and k_E is considered insignificant. There are several other important assumptions in the model:

1. As long as differences in tissue composition and phase partitioning are accounted for, then PCB congeners are assumed to be homogeneously distributed in the organism (Ernst and Goerke 1976).
2. Organisms can be characterized as a compartment that experiences exchange with the surrounding environment (Branson *et al.* 1975). This assumption is most suitable for situations such as this where variations in PCB concentrations in sediments and water are rather slow over time.
3. PCB congeners can be eliminated via egg deposition or sperm ejection but lipid-normalized concentrations of PCB congeners within the organism remain the same. This process is captured with growth dilution associated with egg formation in the adult female, which is countered by uptake of PCBs from water and the diet. Thus the balance of these processes determines the ultimate PCB concentration in the female.

The steady state assumption ($dM_B/dt = 0$) simplifies equation 2.5 to:

$$C_B = (k_1 \cdot (m_O \cdot \Phi \cdot C_{WT,O} + m_P \cdot C_{WD,S}) + k_D \cdot \Sigma P_i \cdot C_{D,i}) / (k_2 + k_E + k_G + k_M) \qquad (2.6)$$

Where the organism's PCB congener concentration ($g \cdot kg^{-1}$, wet weight) (i.e., M_B/W_B) is C_B. It is reasonable to assume steady-state for organisms that have been exposed to the PCB congener for a long period of time and during their entire life. However, an implication of this assumption is that the organism's growth has to be described as a growth rate constant (k_G), which is $dW_B/(W_B \cdot dt)$. Inherent in the growth rate constant is that for the duration of time that the model applies, the organism's growth is a constant fraction of its body weight.

The bioaccumulation factor (BAF) is described by $C_B / C_{WT,O}$ while the wet weight BSAF is C_B/C_S, where the concentration ($g \cdot kg^{-1}$, dry sediment) in the bottom sediment is C_S:

$$BSAF = C_B / C_S \tag{2.7}$$

The primary output of the food web bioaccumulation model is the BSAF as it allows predictions of PCB concentrations in biota from the PCB concentration in the sediments. Submodels for k_1, k_2, k_E, k_M, k_G and Φ, used to determine the BSAF are described below.

Φ: PCBs are hydrophobic and preferentially bind to organic matter and organic carbon, rendering them unavailable for uptake by biota. Φ describes the ratio of the freely dissolved water concentration C_{WD} ($g \cdot L^{-1}$) to the total water concentration C_{WT} ($g \cdot L^{-1}$), and is estimated for non-ionizing PCBs as:

$$\Phi = C_{WD} / C_{WT} = 1 / (1 + x_{POC} \cdot D_{POC} \cdot \alpha_{POC} \cdot K_{OW} + x_{DOC} \cdot D_{DOC} \cdot \alpha_{DOC} \cdot K_{OW}) \tag{2.8}$$

Where concentrations of POC and DOC in the water ($kg \cdot L^{-1}$) are x_{POC} and x_{DOC} respectively. The disequilibrium factors for POC and DOC partitioning are D_{POC} and D_{DOC} respectively, and represent the degree POC-water and DOC-water distribution coefficients vary from POC-water and DOC-water equilibrium partition coefficients. Values greater than 1.0 for D_{POC} or D_{DOC} indicate distribution coefficients greater than equilibrium partition coefficients, values less than 1.0 indicate conditions where equilibrium has not been reached, and values equal to 1.0 indicate equilibrium partitioning. A variety of organic chemicals (including PCBs) show disequilibria between OC and water in several ecosystems (e.g., Gobas and Maclean 2003) but their values are difficult to predict. We used water and sediment concentration data from the areas of interest to characterize D_{POC} and D_{DOC} in the model. In equation 2.8 above, α_{POC} and α_{DOC} are proportionality constants that characterize the similar phase partitioning of POC and DOC in relation to octanol, and they can differ significantly between types of organic carbon. We assumed that α_{POC} was 0.35 with error bars equivalent to a factor of 2.5 (Seth *et al.* 1999), and α_{DOC} was 0.08 with error bars equivalent to a factor of 2.5 (Burkhard 2000).

k_1 and k_2: The rate the respiratory surface (e.g. gills and skin) absorb chemicals from the water is described by the aqueous uptake clearance rate constant k_1 ($L/kg \cdot d$). For fish, invertebrates, and zooplankton, it is a function of the ventilation rate G_V ($L \cdot d^{-1}$) and diffusion rate of PCBs across the respiratory surface area (Gobas 1993;Walker 1987):

$$k_1 = E_W \cdot G_V / W_B \tag{2.9}$$

Where the chemical uptake efficiency of the gills is E_W and the wet weight of the organism (kg) is W_B. The chemical uptake efficiency (E_W) is a function of the PCB congener's K_{OW} and was derived from a fish study (Gobas and Mackay 1987):

$$E_W = (A_{EW} + (B_{EW} / K_{OW}))^{-1} \tag{2.10}$$

Where the constants A_{EW} and B_{EW} are1.85 (± 0.13) and 155 (± 0.50), respectively. Calculations of G_V were based on an allometric relationship between wet weight and oxygen consumption (from a study of 200 fish species ranging in weight between $2.0 \cdot 10^{-5}$ and 60 kg under routine metabolic test conditions) (Thurston and Gehrke 1990), and on G_V data for zooplankton and aquatic invertebrates:

$$G_V = 1400 \cdot W_B^{0.65} / DO \tag{2.11}$$

Where the water's dissolved oxygen concentration (mg $O_2 \cdot L^{-1}$) is DO, which was obtained from the literature. A biphasic relationship for k_1 and k_2 based on a water-organic carbon two-phase resistance model was applied for algae, phytoplankton and aquatic macrophytes:

$$k_1 = (A_P + ((B_P / K_{OW}))^{-1} \tag{2.12}$$

Where the resistance to PCB uptake through the aqueous and organic phases of the algae or phytoplankton are described by the constants A_P and B_P (unit is time), respectively. Numerous data sets were evaluated to obtain A_P and B_P values for phytoplankton. We derived the constant B_P (default value = 5.5 (± 3.7)) by calibration to empirical k_2 values from various phytoplankton, algae and cyanobacteria species over a range of K_{OW} (Koelmans *et al.* 1993;1995;1999). We derived the constant A_P (default value = 6.0 (± 2.0) $\cdot$ 10^{-5}) from calibration to phytoplankton field BCF data (Oliver and Niimi 1988;Swackhammer and Skoglund 1993). The mean annual k_G value was 0.125 d^{-1} (Alpine and Cloern 1988;1992).

The elimination rate constant k_2 (d^{-1}) is similar to k_1 since they both involve water ventilation and membrane permeation:

$$k_2 = k_1 / K_{BW} \tag{2.13}$$

Where the biota-water partition coefficient is K_{BW} ($L \cdot kg^{-1}$, wet weight). PCB partitioning between biota and water is thought to occur in lipids, non-lipid organic matter (e.g., proteins and carbohydrates), and water. Each compartment has its own capacity to sorb PCB congeners, thus for every PCB congener in each organism the organism-water partition coefficient K_{BW} on a wet weight basis (ww) is:

$$K_{BW} = k_1 / k_2 = v_{LB} \cdot K_{OW} + v_{NB} \cdot \beta \cdot K_{OW} + v_{WB} \tag{2.14}$$

Where the lipid fraction (kg lipid/kg organism ww) is v_{LB}, the non-lipid organic matter (NLOM) fraction (kg NLOM/kg organism ww) is v_{NB}, and the water content (kg water/kg organism ww) of the organism is v_{WB}. The proportionality constant expressing the sorption capacity of NLOM to that of octanol is β, and the value used was 0.035 ±0.004 (Gobas *et al.* 1999). Thus the PCB sorption affinity of NLOM is ~3.5% that of octanol. Compared to lipid, the sorption affinity of NLOM is low but it can be important for controlling partitioning of organic chemicals in organisms with low lipid contents (e.g., phytoplankton).

To calculate the phytoplankton-water partition coefficient (K_{PW}), the value of NLOM in equation 2.14 was replaced by the proportionality constant of 0.35 for non-lipid organic carbon (kg NLOC/kg organism ww) (Skoglund and Swackhamer 1999):

$$K_{PW} = v_{LP} \cdot K_{OW} + v_{NP} \cdot 0.35 \cdot K_{OW} + v_{WP} \tag{2.15}$$

The BAF is a function of the k_1 and k_2 ratio, thus errors in determining G_V and E_W typically have little effect on the BAF since k_1 errors cancel out similar k_2 errors. Therefore the model is relatively insensitive to G_V and E_W parameterization error, and a single equation for a variety of species is able to represent ventilation rates and uptake efficiencies. Partitioning properties of the chemical (K_{BW}) play a more important role, which is reasonable because the main role of k_1 and k_2 is to describe the rate of equilibrium partitioning in the organism. Model sensitivity is most affected by k_1 and k_2 for substances taken up from water and food in similar quantities, and/or eliminated by gill ventilation at a rate similar to that for feces egestion, metabolic transformation, and growth dilution combined.

m_O, m_P: PCBs can be exchanged between sediment pore water and organism tissues when the organism spends time in close contact with bottom sediments (i.e., benthic fish and invertebrates). Due to sediment-water disequilibria, concentrations of freely dissolved PCBs in pore water can be greater than those in overlying water (Gobas and Maclean 2003), but the amount of pore water ventilated by benthic fish and invertebrates is often small because of its low oxygen concentration and food content. Even though little pore water is usually ventilated, it can have a significant effect on the BAF for PCBs with large sediment-water column disequilibria. Organisms with no direct pore water contact have an m_P of 0. For all organisms m_O is equal to 1 - m_P.

$C_{WD,P}$: Freely dissolved pore water PCB concentrations were estimated from bottom sediment PCB concentrations (Kraaij *et al.* 2002):

$$C_{WD,P} = C_{S,OC} / (10^{(\mathrm{Log}\ K_{ssw,co})}) \tag{2.16}$$

Where the freely dissolved pore water PCB concentration ($g \cdot L^{-1}$) is $C_{WD,P}$, the organic carbon normalized sediment PCB concentration (g/kg OC) is $C_{S,OC}$, and the organic carbon normalized suspended sediment-water distribution coefficient is log $K_{SSW,CO}$ (log $K_{SSW,CO}$ = log 0.52 $\cdot$ log K_{OW} + 3.02) (Mackintosh *et al.* 2006).

k_D and k_E: The dietary uptake clearance rate constant k_D (kg-food/kg-organism $\cdot$ d) describes the absorption rate of PCBs from the diet via the GIT, and is a function of dietary chemical transfer efficiency (E_D), feeding rate (G_D; $kg \cdot d^{-1}$), and organism weight (W_B; kg) (Gobas 1993):

$$k_D = E_D \cdot G_D / W_B \tag{2.17}$$

Empirical E_D values for aquatic invertebrates range from 0 to 100% (Bruner *et al.* 1994;Kukkonen and Landrum 1995;Landrum and Poore 1988;Lydy and Landrum 1993;Mayer *et al.* 2001;Morrison *et al.* 1996;Parkerton 1993;Wang and Fisher 1999) and from 0 to 90% for fish (Fisk *et al.* 1998;Gobas *et al.* 1988;Gobas *et al.* 1993b;Gobas *et al.* 1993a;Parkerton 1993). Due to the large variation in empirical data accurate models for dietary uptake rates are difficult to develop, but trends in E_D data can provide guidance. There is often a reduction in dietary uptake efficiency with increasing K_{OW} for high K_{OW} chemicals for invertebrates (Bruner *et al.* 1994;Parkerton 1993) and fish (Gobas *et al.* 1988;Parkerton 1993). Aquatic invertebrates and fish fed continuously have average dietary chemical transfer efficiency (E_D) of ~50% for chemicals with a log K_{OW} ranging from 4 – 6. This is in agreement with a two-phase resistance model for gut-organism exchange, also found by Gobas *et al.* (1988). PCB congener dietary absorption efficiencies were based on the lipid-water two phase resistance model:

$$E_D = (A_{ED} \cdot K_{OW} + B_{ED})^{-1} \tag{2.18}$$

Where for zooplankton, invertebrates and fish the constant A_{ED} equals 8.5 (± 1.4) $\cdot 10^{-8}$ and B_{ED} equals 2.0 (±0.6). A general bioenergetic relationship was applied for estimating feeding rates in fish and aquatic invertebrates (Weininger 1978):

$$G_D = 0.022 \cdot W_B^{0.85} \cdot e^{(0.06 \cdot Tw)} \tag{2.19}$$

Where the mean water temperature (°C) is T_W. Dietary uptake by filter feeding species has a unique mechanism described by:

$$G_D = G_V \cdot C_{SS} \cdot \sigma \tag{2.20}$$

Where feeding rate is a function of gill ventilation rate G_V ($L \cdot d^{-1}$), concentration of suspended solids C_{SS} ($kg \cdot L^{-1}$), and scavenging efficiency of particles from water σ (%).

PCB elimination by fecal matter egestion was expressed by k_E (d^{-1}), the fecal elimination rate constant (Gobas *et al.* 1993a):

$$k_E = G_F \cdot E_D \cdot K_{GB} / W_B \tag{2.21}$$

Where the fecal egestion rate is G_F (kg-feces/kg-organism · d) and the PCB partition coefficient between the GIT and organism is K_{GB}. G_F is a function of feeding rate and diet digestibility, which is a function of diet composition:

$$G_F = ((1-\varepsilon_L) \cdot v_{LD}) + (1-\varepsilon_N) \cdot v_{ND} + (1-\varepsilon_W) \cdot v_{WD}) \cdot G_D \tag{2.22}$$

Where dietary absorption efficiencies of lipid, NLOM and water are ε_L, ε_N and ε_W, respectively. The overall lipid, NLOM and water contents of the diet are v_{LD}, v_{ND}, and v_{WD}, respectively. Absorption efficiencies of lipid and NLOM in fish are approximately 90% and 50%, respectively (Gobas *et al.* 1999;Nichols *et al.* 2001).

Invertebrate absorption and assimilation efficiencies vary from 15 - 96% (Berg *et al.* 1996;Gordon 1966;Parkerton 1993;Roditi and Fisher 1999), and generally reflect the organism's dietary matrix (e.g., organic matter quantity and quality) and digestive physiology (e.g., feeding rates and gut retention time). Generally, species with low absorption efficiencies, like worms, consume poor quality sediment or detritus while maintaining high feeding rates to ingest sufficient nutrients. Lipid and non- lipid organic matter absorption efficiencies were set at 75% for aquatic invertebrates.

Zooplankton organic matter assimilation efficiencies range from 55 - 85% (Conover 1966), and are ~85% for carbon and phosphorus (Lehman 1993). We assumed zooplankton lipid and non-lipid organic matter absorption efficiencies were 72%. Water storage capacity has a negligible impact on the mechanism of biomagnification for PCBs, and its assumed absorption efficiency was 55% for zooplankton, invertebrate and fish.

K_{GB}: Is the PCB partition coefficient between the GIT contents and organism, and expresses the effect on phase partitioning properties resulting from digestion after ingestion:

$$K_{GB} = (v_{LG} \cdot K_{OW} + v_{NG} \cdot \beta \cdot K_{OW} + v_{WG}) / (v_{LB} \cdot K_{OW} + v_{NB} \cdot \beta \cdot K_{OW} + v_{WB}) \tag{2.23}$$

Where the lipid (kg lipid/kg digesta ww), NLOM (kg NLOM/kg digesta ww) and water (kg water/kg digesta ww) contents in the gut are v_{LG}, v_{NG}, and v_{WG}, respectively. Summing these fractions (i.e., total digesta) approaches 1 and depends on the absorption efficiency the dietary components:

$$v_{LG} = (1-\varepsilon_L) \cdot v_{LD} / ((1-\varepsilon_L) \cdot v_{LD} + (1-\varepsilon_N) \cdot v_{ND} + (1-\varepsilon_W) \cdot v_{WD}) \tag{2.24}$$

$$v_{NG} = (1-\varepsilon_N) \cdot v_{ND} / ((1-\varepsilon_L) \cdot v_{LD} + (1-\varepsilon_N) \cdot v_{ND} + (1-\varepsilon_W) \cdot v_{WD}) \tag{2.25}$$

$$v_{WG} = (1-\varepsilon_W) \cdot v_{WD} / ((1-\varepsilon_L) \cdot v_{LD} + (1-\varepsilon_N) \cdot v_{ND} + (1-\varepsilon_W) \cdot v_{WD}) \quad (2.26)$$

The bioaccumulation model in equation 2.6 depends on the ratio of k_D and k_E, which is $G_D/(G_F \cdot K_{GB})$, causing the feeding rate G_D (and hence G_F, eq. 2.22) and dietary uptake efficiency E_D model parameterization errors cancel out. If G_D and E_D are not characterized well, the model can still be expected to provide reasonable BAF and BSAF estimates, which is a nice feature because the variability and error in G_D and E_D are usually considerable.

k_G: Growth rates are highly variable among and within species because they are a function of factors such as size, temperature, prey availability, and quality. Reliable growth rate data were not available for most of the species in the food web bioaccumulation model, and instead we used the following generalized growth equations (Thomann 1989), to approximate the growth rate constant k_G (d^{-1}). For zooplankton and invertebrates the equation was:

$$k_G = I_{GR} \cdot W_B^{-0.2} \quad (2.27)$$

which is representative for temperatures around 10°C, and for fish species the equation was:

$$k_G = F_{GR} \cdot W_B^{-0.2} \quad (2.28)$$

With an average water temperature of ~15°C, the growth rate coefficient for invertebrates (I_{GR}) is 0.00035 and for fish (F_{GR}) is 0.0007.

k_M: The metabolic transformation rate constant k_M (d^{-1}) is the rate a parent compound is eliminated via metabolic transformation, and depends on the PCB congener and the species in question. Aquatic invertebrates and fish are very poor at metabolizing most PCB congeners, and we assumed k_M was negligible in these species.

A summary of abiotic model state variables is shown in Table 8, while Tables 9 and 10 summarize model state variables for phytoplankton and all other aquatic biota (i.e., zooplankton, invertebrates, and fish), respectively.

Table 8: A summary of abiotic model state variables requiring parameterization in the food web bioaccumulation model.

Definition	**Parameter**	**Units**
Mean air temperature	T_A	°C
Mean water temperature	T_W	°C
Dissolved oxygen concentration	DO	$mg\ O_2 \cdot L^{-1}$
Practical salinity units	PSU	unitless
Dissolved organic carbon content – water	OC_{WATER}	$kg \cdot L^{-1}$
Particulate organic carbon content – water	POC	$kg \cdot L^{-1}$
Concentration of suspended solids – water	C_{SS}	$kg \cdot L^{-1}$
Organic carbon content – sediment	$OC_{SEDIMENT}$	%
Chemical concentration – water	C_{WT}	$ng \cdot L^{-1}$
Octanol-water partition coefficient	K_{OW}	unitless
Octanol-air partition coefficient	K_{OA}	unitless
Non-lipid organic matter – octanol proportionality constant	β	unitless

Table 9: A summary of biotic state variables that require parameterization in the food web bioaccumulation model for phytoplankton.

Definition	Parameter	Units
Whole body lipid fraction	L	$kg \cdot kg^{-1}$
Whole body non-lipid organic carbon fraction	NLOC	$kg \cdot kg^{-1}$
Whole body water fraction	WC	$kg \cdot kg^{-1}$
Phytoplankton growth rate constant	K_G	d^{-1}
Constant A_P (equation 2.12)	A_P	d^{-1}
Constant B_P (equation 2.12)	B_P	d^{-1}

Table 10: A summary of model state variables that require parameterization in the food web bioaccumulation model for zooplankton, invertebrates, and fish.

Definition	Parameter	Units
Wet weight	W	kg
Whole body lipid fraction	L	$kg \cdot kg^{-1}$
Whole body non-lipid organic matter fraction	NLOM	$kg \cdot kg^{-1}$
Whole body water fraction	WC	$kg \cdot kg^{-1}$
Percentage of respired pore water	P_W	%
Invertebrate growth rate coefficient	I_{GR}	unitless
Fish growth rate coefficient	F_{GR}	unitless
Metabolic transformation rate constant	k_M	d^{-1}
Fraction of prey item in diet	P_i	unitless
Lipid absorption efficiency	ε_L	%
NLOM absorption efficiency	ε_N	%
Water absorption efficiency	ε_W	%
Constant A_{EW} (equation 2.10)	A_{EW}	unitless
Constant B_{EW} (equation 2.10)	B_{EW}	unitless
Constant A_{ED} (equation 2.18)	A_{ED}	unitless
Constant B_{ED} (equation 2.18)	B_{ED}	unitless

2.2.3 Description of Food Web Bioaccumulation Model: Killer Whales

Figure 15 is a conceptual overview of the primary PCB uptake and elimination routes in killer whales. PCB uptake occurs via inhalation and dietary uptake (expected to be the main source for killer whales). Elimination of PCBs in killer whales occurs via exhaled air, fecal matter, urine, and metabolism. Female killer whales can also transfer PCBs into calves and via lactation (Hickie *et al.* 2007;Ross *et al.* 2000a). Killer whale females give birth and nurse their calves for a period of approximately 12-24 months (Ford 2002). PCB concentrations can also be affected by growth periods. Many uptake and elimination processes happen at certain times of year and are non-continuous.

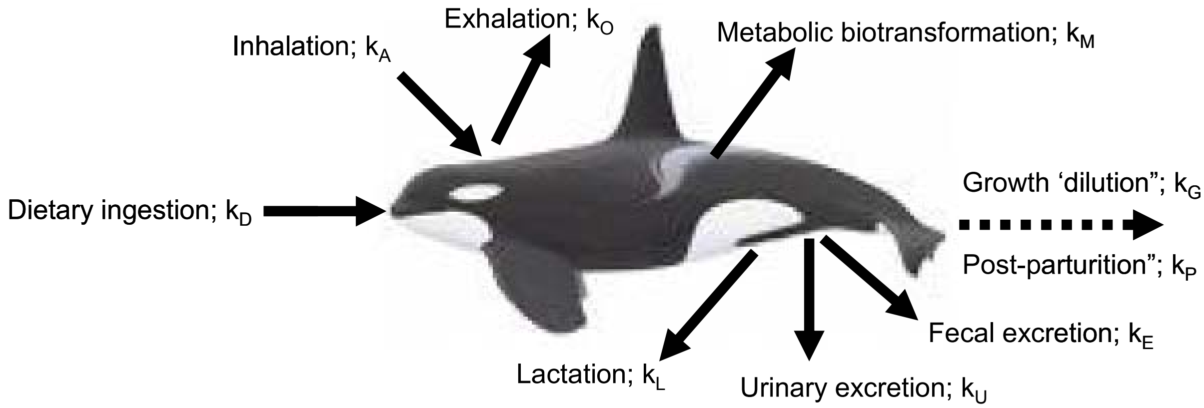


Figure 15: Conceptual diagram of the major routes and associated rate constants of chemical (i.e., PCBs) uptake and elimination processes of PCBs in the killer whale.

Certain PCB congeners can be metabolized in killer whales, although limited metabolic capacity/elimination in these long-lived animals contributes to sustained and prolonged PCB body burdens (Hickie *et al.* 2007;Ross *et al.* 2000a). To keep the model simple and capture uptake and elimination processes by killer whales, key PCB characteristics were considered. PCBs are lipophilic and accumulate to high concentrations in organism lipids. Killer whales have significant quantities of fat in their blubber (i.e., the lipid content of healthy killer whales is about 64% (Ross *et al.* 2000a)), and most PCBs are found in lipid tissues. PCBs tend to establish chemical equilibrium, which means that PCBs distribute equally between various parts of the organism's lipids (lipid-normalized concentration is approximately equal). This chemical equilibrium is of especially relevant to transfer of PCBs from female whales into their calves and lipid normalized concentrations in female whales will not change upon parturition. This means that the transfer of PCB mass from the mother to the calf upon giving birth is associated with a proportional drop in lipid mass of the mother, resulting in approximately the same lipid-normalized concentration. This transfer also occurs during lactation if one assumes that PCBs are equally distributed among fats in the nursing female, then during lactation there is no change in PCB concentration since proportional declines in PCB mass and lipid mass occur. However, offspring production and lactation do have a long-term concentration effect in killer whales because of growth dilution. These processes require that killer whales grow body mass in addition to any net (year-to-year) changes in mass. Growth dilution occurs gradually over the killer whale's life cycle and can be characterized as continuous. Uptake and elimination are represented by the following mass balance equation:

$$dC_{KW,l}/dt = k_A C_{AG} + k_D \cdot \Sigma(P_i \cdot C_{D,i}) - (k_O + k_E + k_U + k_G + k_P + k_L + k_M) \cdot C_{KW,1} \quad (2.29)$$

Where the lipid-normalized PCB congener concentration in the killer whale is $C_{KW,l}$, and the net change in lipid-normalized concentration over time t (d) is $dC_{KW,l}/dt$. The gaseous aerial concentration ($g \cdot L^{-1}$) is C_{AG}. The inhalation rate constant (L/kg lipid$\cdot d^{-1}$) is k_A. The clearance rate constant (kg/kg lipid$\cdot d^{-1}$) for PCB uptake via ingestion of food and water is k_D. The fraction of the diet consisting of prey item i is P_i and the concentration of the PCB congener ($g \cdot kg^{-1}$) in prey item i is $C_{D,i}$. The rate constant (d^{-1}) for PCB exhalation via the lungs is k_O. The rate constant (d^{-1}) for PCB congener elimination via excretion into feces is k_E. The rate constant for urinary PCB excretion is k_U. The rate constant for growth dilution is k_G, and it accounts for net growth increases year-to-year. The rate constant for PCB transfer into the calves is k_P, and it represents the lipid mass increase (equal to the calf's post-parturition lipid mass) during the gestation period. The rate constant for PCB transfer to the calf via lactation is k_L, and it represents the lipid mass increase of the female whale over the year that is transferred to the calf during lactation. k_G, k_P, and k_L (d^{-1}) are fixed annual proportional increases in body lipid weight (i.e., $dW_{KW,1}/(W_{KW,1} \cdot dt)$) where the weight of the lipids in the killer whale is $W_{KW,1}$. The rate constant for metabolic PCB congener transformation is k_M. At steady-state, we can simplify equation 2.29:

$$C_{KW,l} = (k_A C_{AG} + k_D \cdot \Sigma(P_i \cdot C_{D,i})) / (k_O + k_E + k_U + k_G + k_P + k_L + k_M) \qquad (2.30)$$

The lipid-normalized concentration can be used to calculate a whole organism wet weight based concentration in the killer whale C_{KW}:

$$C_{KW} = L_{KW} \cdot C_{S,1} \qquad (2.31)$$

During the year, considerable changes occur in the whole organism's lipid content thus the wet weight concentration is also expected to experience changes of the same magnitude. In the model this is captured by varying L_{KW}. Since killer whales have a high lipid content, non-lipid organic matter does not play a significant role in PCB storage.

The biota-sediment accumulation factor (BSAF; kg dry sediment/kg wet weight) is the ratio of PCB concentrations in the killer whale (C_{KW}) to that in sediments (C_S):

$$BSAF = C_{KW} / C_S \qquad (2.32)$$

BSAFs are a simple method to predict PCB concentrations in killer whales from PCB concentrations in sediments.

Submodels for calculating k_D, k_A, k_O, k_E, k_U, k_G, k_P, and k_L in the killer whale model are described as follows.

k_D and k_E: k_D (kg-food/kg- lipid · d) is the PCB dietary uptake clearance rate constant, which was estimated as a function of dietary chemical transfer efficiency E_D, feeding rate G_D ($kg \cdot d^{-1}$), and organism's lipid mass $W_{S,1}$ (kg):

$$k_D = E_D \cdot G_D / W_{S,1} \qquad (2.33)$$

To determine PCB congener dietary absorption efficiencies in male and female killer whales, we used the equation below (based on the lipid-water two-phase resistance model):

$$E_D = (A_{ED} \cdot K_{OW} + B_{ED})^{-1} \qquad (2.34)$$

For killer whales the constants A_{ED} and B_{ED} are 1.0 [± 0.17] $\cdot 10^{-9}$ and 1.025 [±0.00125], respectively.

k_E (d^{-1}) is the rate constant for PCB fecal excretion in killer whales, and was calculated as:

$$k_E = G_F \cdot E_D \cdot K_{GS,1} / W_{S,1} \qquad (2.35)$$

Where the fecal egestion rate is G_F (kg-feces/kg-organism · d) and the PCB partition coefficient between the GIT and killer whale lipids is $K_{GS,1}$. G_F is a function of feeding rate and diet digestibility, which itself is a function of diet composition:

$$G_F = ((1-\varepsilon_L) \cdot v_{LD} + (1-\varepsilon_N) \cdot v_{ND} + (1-\varepsilon_W) \cdot v_{WD}) \cdot G_D \qquad (2.36)$$

Where the dietary absorption efficiencies of lipid, NLOM, and water are ε_L, ε_N, and ε_W, respectively. The overall diet lipid, NLOM, and water contents are v_{LD}, v_{ND}, and v_{WD}, respectively. It was assumed that the absorption efficiencies of lipid and NLOM were approximately 98% and 75%, respectively for killer whales (Rosen *et al.* 2000;Rosen and Trites 2000).

The PCB partition coefficient $K_{GS,1}$ between the GIT contents and the body lipids of the killer whale is calculated as:

$$K_{GB} = (v_{LG} \cdot K_{OW} + v_{NG} \cdot \beta \cdot K_{OW} + v_{WG}) / K_{OW} \qquad (2.37)$$

Where the killer whale gut lipid (kg lipid/kg digesta ww), NLOM (kg NLOM/kg digesta ww), and water (kg water/kg digesta ww) contents are v_{LG}, v_{NG}, and v_{WG}, respectively. Summing these fractions (i.e., total digesta) approaches 1 and depends on each diet component's absorption efficiency:

$$v_{LG} = (1-\varepsilon_L) \cdot v_{LD} / ((1-\varepsilon_L) \cdot v_{LD} + (1-\varepsilon_N) \cdot v_{ND} + (1-\varepsilon_W) \cdot v_{WD}) \qquad (2.38)$$

$$v_{NG} = (1-\varepsilon_N) \cdot v_{ND} / ((1-\varepsilon_L) \cdot v_{LD} + (1-\varepsilon_N) \cdot v_{ND} + (1-\varepsilon_W) \cdot v_{WD}) \qquad (2.39)$$

$$v_{WG} = (1-\varepsilon_W) \cdot v_{WD} / ((1-\varepsilon_L) \cdot v_{LD} + (1-\varepsilon_N) \cdot v_{ND} + (1-\varepsilon_W) \cdot v_{WD}) \qquad (2.40)$$

k_A and k_O: The rate of PCB absorption from inhalation is described by k_A (L/kg lipid · d), the inhalation clearance rate constant:

$$k_A = E_A \cdot G_A / W_{S,1} \qquad (2.41)$$

Since inhalation and exhalation both utilize lung ventilation and pulmonary membrane permeation, the PCB elimination rate constant via exhalation k_O (d^{-1}) is related to k_A as:

$$k_O = k_A / K_{S,1A} \qquad (2.42)$$

Where the PCB congener partition coefficient between the killer whale's lipid biomass and air is $K_{S,1A}$ ($L \cdot kg^{-1}$, lipid), estimated from the octanol-air partition coefficient (K_{OA}) and the lipid density δ_L ($kg \cdot L^{-1}$) as:

$$K_{S,1A} = k_A / k_O = K_{OA} \cdot \delta_L^{-1} \qquad (2.43)$$

We calculated the urinary excretion rate constant k_U (d^{-1}) as:

$$k_U = G_U / (W_{S,1} \cdot K_{OW} \cdot \delta_L^{-1}) \qquad (2.44)$$

Where the urinary excretion rate ($L \cdot d^{-1}$) is G_U and the octanol-water partition coefficient is K_{OW}.

k_G, k_P, k_L: PCB elimination rate constants in killer whales for growth dilution, off-spring, and milk, represent PCB reduction in the lipid biomass of the whale that arises from the increase in lipid biomass due to growth, off spring production, and lactation. These rate constants are characterized by the proportional increase in lipid biomass over time:

$$dW_{KW,1} / (W_{KW,1} \cdot dt) \tag{2.45}$$

$dW_{KW,1}$ represents lipid mass increases attained during a year when calculating k_G, and it describes the calf's lipid mass at birth when assessing k_P. This lipid biomass is produced during the gestation period. $dW_{KW,1}$ describes the lipid mass transferred to the calf in milk during lactation (i.e., the product of lactation rate G_L ($L \cdot d^{-1}$) and duration of lactation t_L), when estimating k_L. For simplicity, we calculated the lipid biomass increase in female killer whales by summing lipid masses produced for growth, off-spring production, and lactation and described it as a fraction of the animal's lipid biomass generated over time.

k_M: Killer whales can metabolize certain PCB congeners, which can have a significant effect on the magnitude of PCB concentrations attained in the body. Because PCBs can have congener specific metabolic transformation patterns (Boon *et al.* 1987;1994;1997b), one can estimate a congener's metabolic transformation relative to a reference congener. PCB 153 is the dominant PCB congener in Harbour seals (Boon *et al.* 1987;1994;1997a), and PCB 153 and 138 dominate PCB congeners in resident killer whales of British Columbia (Ross *et al.* 2000a). However, for the aim of this model and because information for each PCB congener's metabolic transformation rate constant is scarce, we assumed that metabolic transformation rate constants (k_M) for each PCB congener were 0 d^{-1}. State variables for resident killer whales are summarized in Table 11.

Table 11: A summary of model state variables that require parameterization in the food web bioaccumulation model for killer whales.

Definition	Parameter	Units
Wet weight	W	kg
Whole body lipid fraction	L	$kg \cdot kg^{-1}$
Whole body non-lipid organic matter fraction	NLOM	$kg \cdot kg^{-1}$
Whole body water fraction	WC	$kg \cdot kg^{-1}$
Mean homeotherm temperature	T_H	°C
Growth rate constant	K_G	d^{-1}
Fraction of prey item in diet	P_i	unitless
Lipid absorption efficiency	ε_L	%
NLOM absorption efficiency	ε_N	%
Water absorption efficiency	ε_W	%
Constant A_{ED} (equation 2.34 for killer whales)	A_{ED}	unitless
Constant B_{ED} (equation 2.34 for killer whales)	B_{ED}	unitless
Urine excretion rate	G_U	$L \cdot d^{-1}$
Metabolic transformation rate constant	k_M	d^{-1}

2.4 MODEL APPLICATIONS TO CHINOOK AND KILLER WHALES

2.4.1 General

Several site and species specific parameters are required in the food web bioaccumulation model, and were obtained from the scientific literature. If a species specific parameter was missing, an appropriate parameter from an equivalent species was used or was estimated. All model assumptions were documented.

The previous Gobas (1993) model and the model by Gobas and Arnot (2010) have had extensive use and testing to determine which parameters the model is most affected by. A general overview of relative sensitivity of the various parameters is shown in Table 12.

Table 12: Food web bioaccumulation model sensitivity to various parameters.

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

2.4.2 Physico-Chemical Properties of PCBS

Tables 13 and 14 summarize the PCB congener octanol-water (Log K_{OW}) and octanol-air (Log K_{OA}) partition coefficients used in the model areas. The tables contain the freshwater-based K_{OW} at the mean ambient water temperature of the areas of interest. These were used to calculate the saltwater-based K_{OW} values based on the approach of Xie *et al.* (1997), which were used to determine the PCB distribution between fish and water in the areas of interest. Freshwater-based K_{OW} values at 37.5°C were used to describe partitioning between lipids and aqueous media (e.g., urine) in killer whales. Also included in the table are K_{OA} values corrected to 37.5°C, which were used in the calculation of PCB transfer between killer whales and air, via their lungs.

Table 13: PCB congeners and properties' values used in the food web bioaccumulation model for northern resident killer whale Critical Habitat, Queen Charlotte Strait, and outer coast areas.

Chemical Name	Congener CAS #	Molecular Weight ($g \cdot mol^{-1}$)	LeBas Molar Volume ($cm^3 \cdot mol^{-1}$)	log K_{OW} (unitles)	log K_{OW} Temp corrected (37.5 °C) (unitless)	log K_{OA} Temp corrected (37.5 °C) (unitless)
PCB	8	223.1	226.4	5.42	4.96	6.83
PCB	18	257.5	247.4	5.62	5.12	6.82
PCB	28	257.5	247.4	5.99	5.47	7.29
PCB	31	257.5	247.4	6.11	5.60	7.39
PCB	33	257.5	247.4	5.98	5.47	7.40
PCB	44	292.0	268.4	6.16	5.63	7.96
PCB	49	292.0	268.4	6.30	5.76	7.61
PCB	52	292.0	268.4	6.26	5.72	7.64
PCB	56	292.0	268.4	6.39	5.80	8.16
PCB	60	292.0	268.4	6.49	5.91	8.55
PCB	66	292.0	268.4	6.36	5.81	8.58
PCB	70	292.0	268.4	6.46	5.90	8.25
PCB	74	292.0	268.4	6.46	5.91	8.41
PCB	87	326.5	289.4	6.72	6.15	8.51
PCB	95	326.5	289.4	6.43	5.86	8.28
PCB	99	326.5	289.4	6.73	6.16	8.58
PCB	101	326.5	289.4	6.68	6.16	8.25
PCB	105	326.5	289.4	7.20	6.62	8.90
PCB	110	326.5	289.4	6.68	6.11	8.48
PCB	118	326.5	289.4	6.97	6.39	8.74
PCB	128	361.0	310.4	7.18	6.59	9.16
PCB	132	361.0	310.4	6.90	6.36	8.94
PCB	138	361.0	310.4	7.59	7.04	9.05
PCB	141	361.0	310.4	7.13	6.59	9.22
PCB	149	361.0	310.4	6.99	6.44	8.94
PCB	151	361.0	310.4	6.96	6.42	8.99
PCB	153	360.88	310.4	7.28	6.65	8.78
PCB	156	361.0	310.4	7.37	6.85	9.74
PCB	158	361.0	310.4	7.23	6.71	9.43
PCB	170	395.5	331.4	7.56	7.00	9.89
PCB	174	395.5	331.4	7.43	6.83	9.62
PCB	177	395.5	331.4	7.41	6.81	9.73
PCB	180	395.5	331.4	7.57	6.95	9.51
PCB	183	395.5	331.4	7.52	6.92	9.88
PCB	187	395.5	331.4	7.49	6.89	9.71
PCB	194	429.77	352.4	8.18	7.56	10.46
PCB	195	430.0	352.4	7.87	7.25	10.45
PCB	201	430.0	352.4	7.92	7.31	10.26

Table 14: PCB congeners and properties' values used in the food web bioaccumulation model for Strait of Georgia, southern resident killer whale Critical Habitat in Canada, southern resident killer whale Critical Habitat in USA (Puget Sound), and southern resident killer whale Critical Habitat in USA (summer core and Juan de Fuca Strait) areas.

Chemical Name	Congener CAS #	Molecular Weight ($g \cdot mol^{-1}$)	LeBas Molar Volume ($cm^3 \cdot mol^{-1}$)	log K_{OW} (unitles)	log K_{OW} Temp corrected (37.5 °C) (unitless)	log K_{OA} Temp corrected (37.5 °C) (unitless)
PCB	8	223.1	226.4	5.42	4.96	6.83
PCB	18	257.5	247.4	5.62	5.12	6.82
PCB	28	257.5	247.4	5.99	5.47	7.29
PCB	44	292.0	268.4	6.16	5.63	7.96
PCB	49	292.0	268.4	6.30	5.76	7.61
PCB	52	292.0	268.4	6.26	5.72	7.64
PCB	66	292.0	268.4	6.36	5.81	8.58
PCB	74	292.0	268.4	6.46	5.91	8.41
PCB	95	326.5	289.4	6.43	5.86	8.28
PCB	99	326.5	289.4	6.73	6.16	8.58
PCB	101	326.5	289.4	6.68	6.16	8.25
PCB	105	326.5	289.4	7.20	6.62	8.90
PCB	110	326.5	289.4	6.68	6.11	8.48
PCB	118	326.5	289.4	6.97	6.39	8.74
PCB	128	361.0	310.4	7.18	6.59	9.16
PCB	138	361.0	310.4	7.59	7.04	9.05
PCB	149	361.0	310.4	6.99	6.44	8.94
PCB	151	361.0	310.4	6.96	6.42	8.99
PCB	153	360.9	310.4	7.28	6.65	8.78
PCB	156	361.0	310.4	7.37	6.85	9.74
PCB	170	395.5	331.4	7.56	7.00	9.89
PCB	177	395.5	331.4	7.41	6.81	9.73
PCB	180	395.5	331.4	7.57	6.95	9.51
PCB	183	395.5	331.4	7.52	6.92	9.88
PCB	187	395.5	331.4	7.49	6.89	9.71
PCB	194	429.8	352.4	8.18	7.56	10.5
PCB	203	430.0	352.4	7.95	7.33	10.4

2.4.3 Environmental Conditions of Areas Included in the Model

Tables 15 - 21 include the environmental condition input variables used in the seven model areas. The values are reported in the worksheet "Input-1" of the model. In water, PCBs can be freely dissolved or absorbed to particulate organic matter (POM) and dissolved organic carbon (DOC). These values were obtained from the literature or were estimated based on the relationship that most organic carbon (~80%) in water is in the form of DOC (Sophie Johannessen, Fisheries & Oceans Canada, Institute of Ocean Sciences, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010).

Table 15: Environmental input parameters for northern resident killer whale Critical Habitat area in the bioaccumulation food web model.

Parameters	**Input**		**Variability**	**Units**	**References**
Mean Water Temperature	8.67	±	0.5	°C	1
Mean Air Temperature	9	±	0	°C	2
Mean Homeothermic Biota Temperature	37.5	±	1	°C	3
Mean Water Temperature	281.82	±	1.35	K	
Mean Air Temperature	282.15	±	6	K	
Mean Homeothermic Biota Temperature	310.65	±		K	
pH of Water	7.7	±	0.071	Unitless	4
Practical Salinity Units (PSU)	30.38	±	1.34	Unitless	2
Dissolved Oxygen Concentration @ 90% Saturation (DO)	5	±	0	mg $O_2 \cdot L^{-1}$	5
Dissolved Organic Carbon Content - Water (OCwater)	7.26E-07	±	1.27E-07	$kg \cdot L^{-1}$	6
Particulate Organic Carbon Content - Water (POC)	1.56E-07	±	5.09E-08	$kg \cdot L^{-1}$	6
Concentration of Suspended Solids (Vss)	8.83E-07	±	1.80E-07	$kg \cdot L^{-1}$	6
Percentage of Organic Carbon - Sediment (OCsed)	4.27	±	0.021	%	7, 8
Density of Organic Carbon - Sediment (Docsed)	0.9	±		$kg \cdot L^{-1}$	9
Setschenow Proportionality Constant (SPC)	0.0018	±		$L \cdot cm^{-3}$	10
Molar Concentration of Seawater @ 35 ppt (MCS)	0.5	±		$mol \cdot L^{-1}$	10
Absolute Temperature (K)	273.16	±		K	
Ideal Gas Law Constant (Rgaslaw)	8.314	±		$Pa \cdot m^3/mol \cdot K$	

Table References:

1. http://www-sci.pac.dfo-mpo.gc.ca/osap/data/lighthouse/pinet.txt
2. Masson (2006)
3. Gobas and Arnot (2010)
4. Sophie Johannessen (Fisheries & Oceans Canada, Institute of Ocean Sciences, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010)
5. Foreman *et al.* (2006)
6. Johannessen *et al.* (2008b)
7. Burd *et al.* (2008b)
8. Johannessen *et al.* (2003)
9. Mackay (1991)
10. Xie *et al.* (1997)

Table 16: Environmental input parameters for Queen Charlotte Strait area in the bioaccumulation food web model.

Parameters	Input		Variability	Units	References
Mean Water Temperature	9.4	±	3.11	°C	1
Mean Air Temperature	10.8	±	1.1	°C	2
Mean Homeothermic Biota Temperature	37.5	±	1	°C	3
Mean Water Temperature	282.55	±	1.35	K	
Mean Air Temperature	283.95	±	6	K	
Mean Homeothermic Biota Temperature	310.65	±		K	
pH of Water	7.7	±	0.14	Unitless	4
Practical Salinity Units (PSU)	32.9	±	1.41	Unitless	1
Dissolved Oxygen Concentration @ 90% Saturation (DO)	6.5	±	0.87	mg $O_2 \cdot L^{-1}$	5
Dissolved Organic Carbon Content - Water (OCwater)	2.6E-07	±	0	$kg \cdot L^{-1}$	1 (estimated from POC reported)
Particulate Organic Carbon Content - Water (POC)	6.5E-08	±	2.12E-08	$kg \cdot L^{-1}$	1
Concentration of Suspended Solids (Vss)	2.17E-06	±	0	$kg \cdot L^{-1}$	Estimated as POC/3%
Percentage of Organic Carbon - Sediment (OCsed)	3.0	±	0	%	6
Density of Organic Carbon - Sediment (Docsed)	0.9	±		$kg \cdot L^{-1}$	7
Setschenow Proportionality Constant (SPC)	0.0018	±		$L \cdot cm^{-3}$	8
Molar Concentration of Seawater @ 35 ppt (MCS)	0.5	±		$mol \cdot L^{-1}$	8
Absolute Temperature (K)	273.16	±		K	
Ideal Gas Law Constant (Rgaslaw)	8.314	±		$Pa \cdot m^3/mol \cdot K$	

Table References:

1. Peña *et al.* (1999)
2. Environment Canada (www.climate.weatheroffice.ec.gc.ca)
3. Gobas and Arnot (2010)
4. Sophie Johannessen (Fisheries & Oceans Canada, Institute of Ocean Sciences, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010)
5. Tortell *et al.* (2005)
6. Conway *et al.* (2005)
7. Mackay (1991)
8. Xie *et al.* (1997)

Table 17: Environmental input parameters for the outer coast area in the bioaccumulation food web model.

Parameters	Input		Variability	Units	References
Mean Water Temperature	9.4	±	3.11	°C	1
Mean Air Temperature	10.8	±	1.1	°C	2
Mean Homeothermic Biota Temperature	37.5	±	1	°C	3
Mean Water Temperature	282.55	±	1.35	K	
Mean Air Temperature	283.95	±	6	K	
Mean Homeothermic Biota Temperature	310.65	±		K	
pH of Water	7.7	±	0.14	Unitless	4
Practical Salinity Units (PSU)	32.9	±	1.41	Unitless	1
Dissolved Oxygen Concentration @ 90% Saturation (DO)	6.5	±	0.87	$mg\ O_2 \cdot L^{-1}$	5
Dissolved Organic Carbon Content - Water (OCwater)	2.6E-07	±	0	$kg \cdot L^{-1}$	1 (estimated from POC reported)
Particulate Organic Carbon Content - Water (POC)	6.5E-08	±	2.12E-08	$kg \cdot L^{-1}$	1
Concentration of Suspended Solids (Vss)	2.17E-06	±	0	$kg \cdot L^{-1}$	Estimated as POC/3%
Percentage of Organic Carbon - Sediment (OCsed)	3.0	±	0	%	6
Density of Organic Carbon - Sediment (Docsed)	0.9	±		$kg \cdot L^{-1}$	7
Setschenow Proportionality Constant (SPC)	0.0018	±		$L \cdot cm^{-3}$	8
Molar Concentration of Seawater @ 35 ppt (MCS)	0.5	±		$mol \cdot L^{-1}$	8
Absolute Temperature (K)	273.16	±		K	
Ideal Gas Law Constant (Rgaslaw)	8.314	±		$Pa \cdot m^3/mol \cdot K$	

Table References:

1. Peña *et al.* (1999)
2. Environment Canada (www.climate.weatheroffice.ec.gc.ca)
3. Gobas and Arnot (2010)
4. Sophie Johannessen (Fisheries & Oceans Canada, Institute of Ocean Sciences, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010)
5. Tortell *et al.* (2005)
6. Conway *et al.* (2005)
7. Mackay (1991)
8. Xie *et al.* (1997)

Table 18: Environmental input parameters for the Strait of Georgia area in the bioaccumulation food web model.

Parameters	Input		Variability	Units	References
Mean Water Temperature	9.07	±	2.14	°C	1
Mean Air Temperature	9.25	±	8.13	°C	2
Mean Homeothermic Biota Temperature	37.5	±	1	°C	3
Mean Water Temperature	282.22	±	1.35	K	
Mean Air Temperature	282.4	±	6	K	
Mean Homeothermic Biota Temperature	310.65	±		K	
pH of Water	7.7	±	0.141	Unitless	4
Practical Salinity Units (PSU)	30.4	±	3.03	Unitless	1
Dissolved Oxygen Concentration @ 90% Saturation (DO)	4.11	±	2.03	mg $O_2 \cdot L^{-1}$	1
Dissolved Organic Carbon Content - Water (OCwater)	6.36E-07	±	1.19E-07	$kg \cdot L^{-1}$	5
Particulate Organic Carbon Content - Water (POC)	9.2E-08	±	4.96E-08	$kg \cdot L^{-1}$	5
Concentration of Suspended Solids (Vss)	2.62E-06	±	1.21E-06	$kg \cdot L^{-1}$	6
Percentage of Organic Carbon - Sediment (OCsed)	1.50	±	0.0071	%	7, 8
Density of Organic Carbon - Sediment (Docsed)	0.9	±		$kg \cdot L^{-1}$	9
Setschenow Proportionality Constant (SPC)	0.0018	±		$L \cdot cm^{-3}$	10
Molar Concentration of Seawater @ 35 ppt (MCS)	0.5	±		$mol \cdot L^{-1}$	10
Absolute Temperature (K)	273.16	±		K	
Ideal Gas Law Constant (Rgaslaw)	8.314	±		$Pa \cdot m^3/mol \cdot K$	

Table References:

1. Masson (2006)
2. Environment Canada (2009)
3. Gobas and Arnot (2010)
4. Sophie Johannessen (Fisheries & Oceans Canada, Institute of Ocean Sciences, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010)
5. Johannessen *et al.* (2008b)
6. Komick *et al.* (2009)
7. Burd *et al.* (2008b)
8. Johannessen *et al.* (2003)
9. Mackay (1991)
10. Xie *et al.* (1997)

Table 19: Environmental input parameters for the southern resident killer whale Critical Habitat in Canada area in the bioaccumulation food web model.

Parameters	Input		Variability	Units	References
Mean Water Temperature	9.07	±	2.14	°C	1
Mean Air Temperature	9.25	±	8.13	°C	2
Mean Homeothermic Biota Temperature	37.5	±	1	°C	3
Mean Water Temperature	282.22	±	1.35	K	
Mean Air Temperature	282.4	±	6	K	
Mean Homeothermic Biota Temperature	310.65	±		K	
pH of Water	7.7	±	0.141	Unitless	4
Practical Salinity Units (PSU)	30.4	±	3.03	Unitless	1
Dissolved Oxygen Concentration @ 90% Saturation (DO)	4.11	±	2.03	mg $O_2 \cdot L^{-1}$	1
Dissolved Organic Carbon Content - Water (OCwater)	6.36E-07	±	1.19E-07	$kg \cdot L^{-1}$	5
Particulate Organic Carbon Content - Water (POC)	9.2E-08	±	4.96E-08	$kg \cdot L^{-1}$	5
Concentration of Suspended Solids (Vss)	2.62E-06	±	1.21E-06	$kg \cdot L^{-1}$	6
Percentage of Organic Carbon - Sediment (OCsed)	1.50	±	0.0071	%	7, 8
Density of Organic Carbon - Sediment (Docsed)	0.9	±		$kg \cdot L^{-1}$	9
Setschenow Proportionality Constant (SPC)	0.0018	±		$L \cdot cm^{-3}$	10
Molar Concentration of Seawater @ 35 ppt (MCS)	0.5	±		$mol \cdot L^{-1}$	10
Absolute Temperature (K)	273.16	±		K	
Ideal Gas Law Constant (Rgaslaw)	8.314	±		$Pa \cdot m^3/mol \cdot K$	

Table References:

1. Masson (2006)
2. Environment Canada (2009)
3. Gobas and Arnot (2010)
4. Sophie Johannessen (Fisheries & Oceans Canada, Institute of Ocean Sciences, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010)
5. Johannessen *et al.* (2008b)
6. Komick *et al.* (2009)
7. Burd *et al.* (2008b)
8. Johannessen *et al.* (2003)
9. Mackay (1991)
10. Xie *et al.* (1997)

Table 20: Environmental input parameters for the southern resident killer whale Critical Habitat in USA (Puget Sound) area in the bioaccumulation food web model.

Parameters	Input		Variability	Units	References
Mean Water Temperature	10.3	±	0.0	°C	1
Mean Air Temperature	9.5	±	0.0	°C	1
Mean Homeothermic Biota Temperature	37.5	±	1	°C	2
Mean Water Temperature	283.45	±	1.35	K	
Mean Air Temperature	282.65	±	6	K	
Mean Homeothermic Biota Temperature	310.65	±		K	
pH of Water	7.7	±	0.0	Unitless	3
Practical Salinity Units (PSU)	30	±	0.0	Unitless	1
Dissolved Oxygen Concentration @ 90% Saturation (DO)	7.5	±	0.0	mg $O_2 \cdot L^{-1}$	1
Dissolved Organic Carbon Content - Water (OCwater)	1.00E-06	±	0.0	$kg \cdot L^{-1}$	1
Particulate Organic Carbon Content - Water (POC)	0.0	±	0.0	$kg \cdot L^{-1}$	1
Concentration of Suspended Solids (Vss)	2.4E-06	±	0.0	$kg \cdot L^{-1}$	1
Percentage of Organic Carbon - Sediment (OCsed)	1.74	±	0.0	%	1
Density of Organic Carbon - Sediment (Docsed)	0.9	±		$kg \cdot L^{-1}$	4
Setschenow Proportionality Constant (SPC)	0.0018	±		$L \cdot cm^{-3}$	5
Molar Concentration of Seawater @ 35 ppt (MCS)	0.5	±		$mol \cdot L^{-1}$	5
Absolute Temperature (K)	273.16	±		K	
Ideal Gas Law Constant (Rgaslaw)	8.314	±		$Pa \cdot m^3/mol \cdot K$	

Table References:

1. Pelletier and Mohamedali (2009)
2. Gobas and Arnot (2010)
3. Sophie Johannessen (Fisheries & Oceans Canada, Institute of Ocean Sciences, PO Box 6000, Sidney, BC V8L 4B2, pers. comm., 2010)
4. Mackay (1991)
5. Xie *et al.* (1997)

Table 21: Environmental input parameters for the southern resident killer whale Critical Habitat in USA (summer core and Juan de Fuca Strait) area in the bioaccumulation food web model.

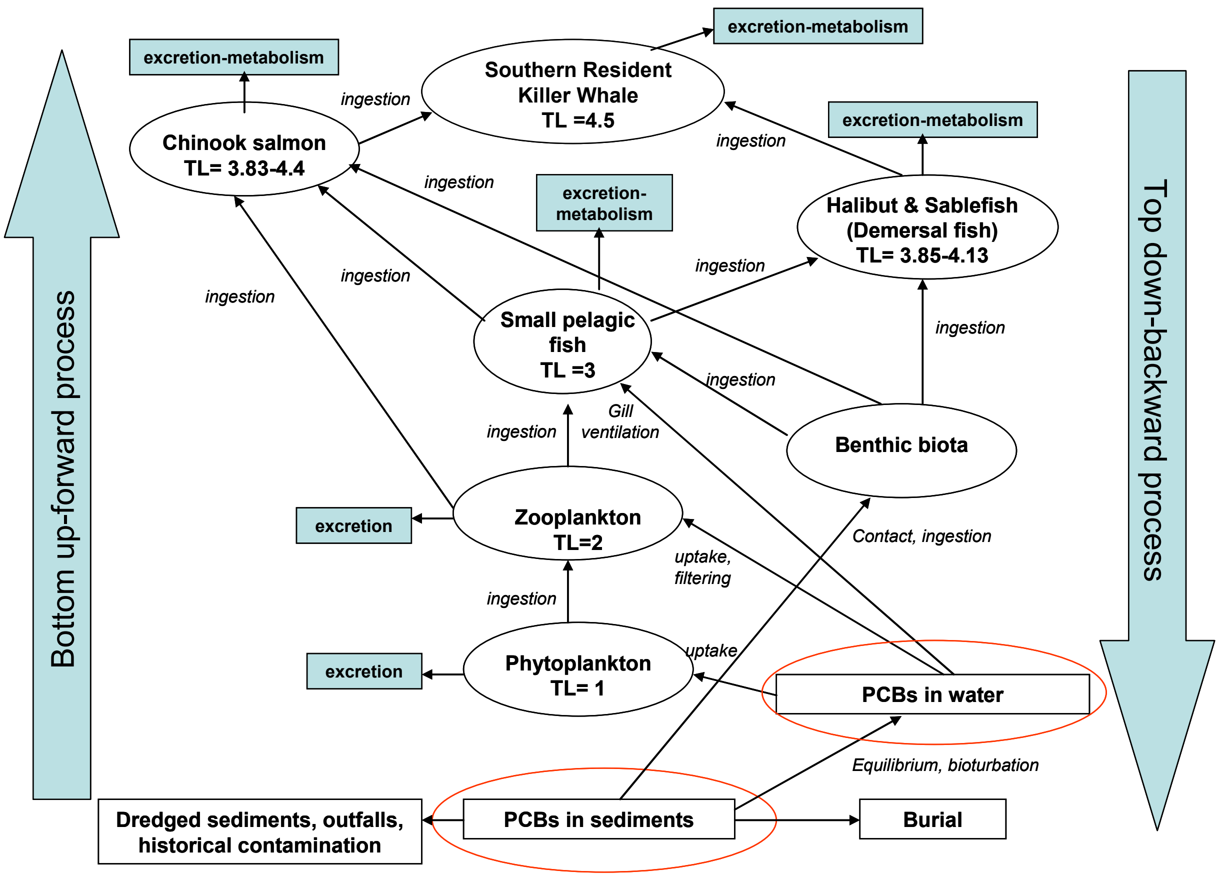
Parameters	Input		Variability	Units	References
Mean Water Temperature	12.5	±	0	°C	1
Mean Air Temperature	9.5	±	0	°C	2
Mean Homeothermic Biota Temperature	37.5	±	1	°C	3
Mean Water Temperature	285.65	±	1.35	K	
Mean Air Temperature	282.65	±	6	K	
Mean Homeothermic Biota Temperature	310.65	±		K	
pH of Water	7.63	±	0	Unitless	1
Practical Salinity Units (PSU)	31.3	±	0	Unitless	1
Dissolved Oxygen Concentration @ 90% Saturation (DO)	4.5	±	0	mg $O_2 \cdot L^{-1}$	1
Dissolved Organic Carbon Content - Water (OCwater)	1.00E-06	±	0	$kg \cdot L^{-1}$	2
Particulate Organic Carbon Content - Water (POC)	0	±	0	$kg \cdot L^{-1}$	2
Concentration of Suspended Solids (Vss)	6.00E-06	±	0	$kg \cdot L^{-1}$	1
Percentage of Organic Carbon - Sediment (OCsed)	1.14	±	0	%	2
Density of Organic Carbon - Sediment (Docsed)	0.9	±		$kg \cdot L^{-1}$	4
Setschenow Proportionality Constant (SPC)	0.0018	±		$L \cdot cm^{-3}$	5
Molar Concentration of Seawater @ 35 ppt (MCS)	0.5	±		$mol \cdot L^{-1}$	5
Absolute Temperature (K)	273.16	±		K	
Ideal Gas Law Constant (Rgaslaw)	8.314	±		$Pa \cdot m^3/mol \cdot K$	

Table References:

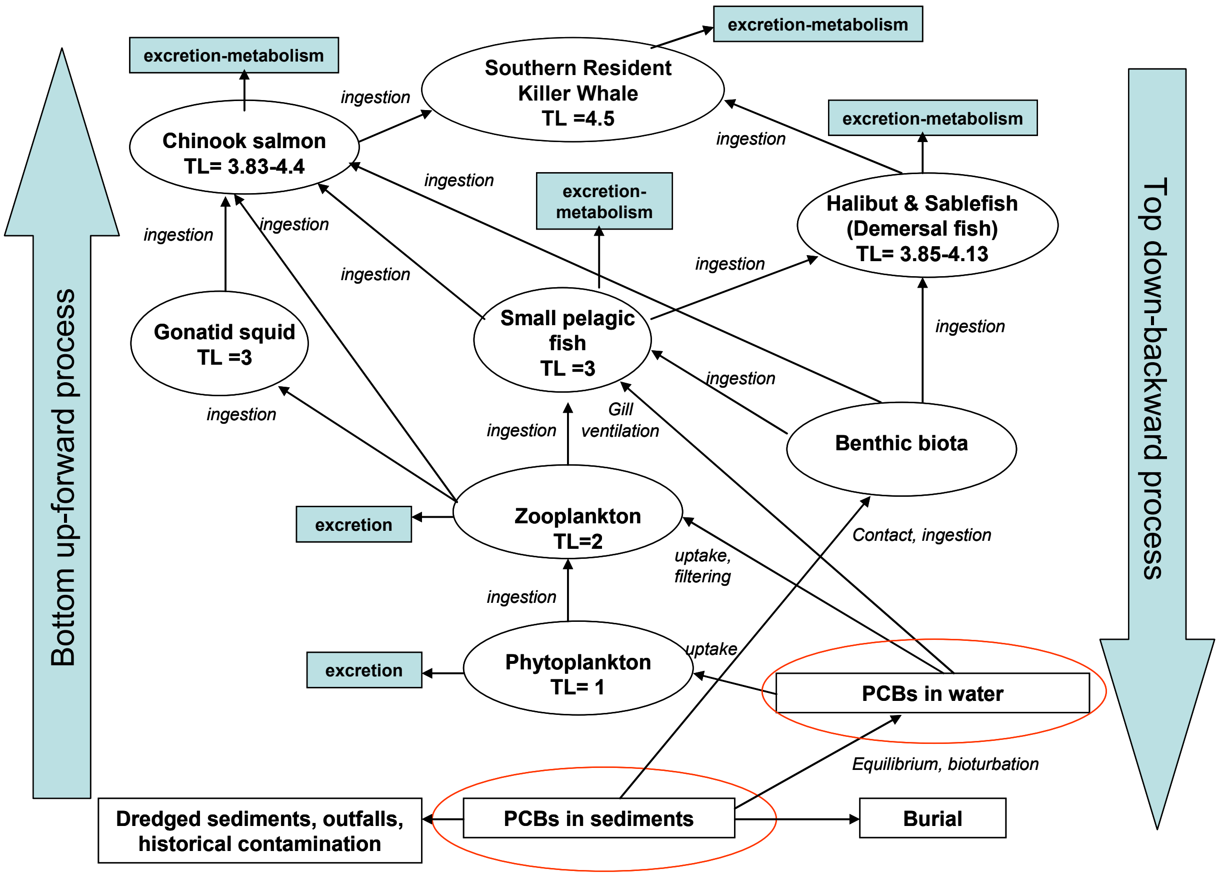
1. Wilson and Partridge (2007)
2. Pelletier and Mohamedali (2009)
3. Gobas and Arnot (2010)
4. Mackay (1991)
5. Xie *et al.* (1997)

2.4.4 Biological Variables in the Model

Figure 16a is a schematic diagram of organisms included in the coastal food web and the representative trophic interactions considered, while Figure 16b is that for the oceanic food web. The main difference between the two food webs is that Chinook salmon primarily feed on squid in the outer coast, rather than herring. Air concentrations of PCBs were also included but are not depicted in Figure 16a and b. Table 22 lists the biological and physiological parameters used in the food web bioaccumulation model. Tables 23 and 24 describe the feeding preferences of the species included in the model in the coastal and oceanic food webs, respectively.



(a)



(b)

Figure 16a & b: Conceptual diagram illustrating organisms included in the model and their trophic interactions and trophic level for coastal (a) and oceanic (b) food webs. The figure also highlights the pathways PCBs move from sediments and the water column to biota. In our model, measured data for PCBs in sediments and water were used an input variables for the killer whale food web model.

Table 22: General biological and physiological parameter definitions, values, and references used in the food web bioaccumulation model. E_D = dietary chemical transfer efficiency.

General aquatic species input parameter	Mean		SD	Units	Reference
Density of Lipids	0.9	±		$kg \cdot L^{-1}$	1
Non-Lipid Organic Matter Content (NLOM)	20%	±	0.01	%	
NLOM proportionality constant (MAF)	0.05	±	5.0E-03	Unitless	2 (modified from)
Fish Growth Rate Factor (PGR)	1.40E-03	±	7.0E-05	Unitless	3 (modified from)
Invertebrate Growth Rate Factor (IGR)	3.50E-04	±	3.5E-05	Unitless	3
Dietary absorption efficiency of lipid in benth-invertebrate (ε_L)	75%	±	0.02	%	4
Dietary absorption efficiency of NLOM in benth-invertebrate (ε_N)	50%	±	0.02	%	4 (modified from)
Dietary absorption efficiency of lipid in fish (ε_L)	92%	±	0.02	%	4
Dietary absorption efficiency of NLOM in fish (ε_N)	60%	±	0.02	%	4, 5
Dietary absorption efficiency of lipid in mammals (ε_L)	100%	±	0.02	%	5, 7, 8
Dietary absorption efficiency of NLOM in mammals (ε_N)	98%	±	0.02	%	1 (modified from)
E_D – Constant A - All feeding species except marine mammals	8.5E-08	±	1.4E-08	Unitless	1
E_D – Constant B - All feeding species except marine mammals	2.00	±	0.600	Unitless	1
E_D – Constant A - Mammals	1E-09	±	1.7E-10	Unitless	1
E_D – Constant B - Mammals	1.025	±	1.2E-03	Unitless	1
E_W - Constant A - Water absorption efficiency in fish & invertebrates	1.85	±	0.13	Unitless	1
Water digestion efficiency in marine mammals (E_W)	85%	±		%	1
Lung uptake efficiency in marine mammals(E_L)	0.7	±		Unitless	1
Mean homoeothermic temperature (marine mammals)	37.5	±	1.00	°C	1
Metabolic Transformation Rate Constant (k_M) - All species	0.00	±		$1 \cdot day^{-1}$	4
Particle Scavenging Efficiency (PSE)	100%			%	Default value

Table References:

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

Table 23: Feeding Preferences Matrix - dietary composition and trophic levels (TL) of 19 predator species / organisms in the Georgia Basin ecosystem. Prey species and their corresponding trophic levels are identified.

Coastal-Marine Food Web Species (Predators)		Prey[1] (Diet %)																				Sum
	TL	Det	Phy	Zoo	Pol-1	Pol-2	Mus	Oys	Amp	Mys	DCr	Shri	Sper	Herr	Wpol	Anch	WGr	Pmid	Sfish	Hal	Sal	
Zooplankton (Copepoda, *Neocalanus*)	2.0		100																			100
Polychaete-1 *(Neanthes succinea)*	2.1	90	5	5																		100
Polychaete-2 *(Harmothoe imbricata)*	2.1	30	35	35																		100
Blue mussel (*Mytilus edulis*)	2.3	15	60	25																		100
Oyster (*Crassostrea gigas*)	2.3	15	60	25																		100
Amphipods (*Themisto* sp.)	2.4	30	35	35																		100
Mysis sp.	2.5	10	45	45																		100
Dungeness crab (*Cancer magister*)	2.8	43	2	10	5	5	5	5	5	5		5	5	5								100
Crangon sp.*	2.9		30	30						40												100
Shiner surfperch (*Cymatogaster aggregata*)	3.2	5	10	10	10	10			20	15		20										100
Pacific Herring (*Clupea pallasi*)	3.0			98	1				1													100
Walleye pollock (*Theragra chalcogramma*)	3.0			95	2.5				2.5													100
Northern anchovy (*Engraulis mordax*)	3.1		20	20					15	25		20										100
White Spotted Greenling (*Hexagrammos stelleri*)	3.5			10					45	10	10	10	5	5		5						100
Plainfin midshipman (*Porichthys*	3.5	5			10	5			15	20	15	20	5			5						100

notatus)																						
Sablefish (*Anoplopoma fimbria*)	3.8			10	5				5		5	10	6	3	50	6						100
Halibut (*Hippoglossus stenolepis*)	4.0			1	1	1	1	1	1	1	1	14	10	5	47	10		5	1			100
Chinook salmon (*Oncorhynchus tshawytscha*)	4.0			5					1			4	20	25	25	20						100
Killer whale (*Orcinus orca*)	5.0																		2	2	96	100

[1]Legend prey species: Det = Detritud; Phy = Phytoplankton; Zoo = Zooplankton; Pol-1 = Polychaete-1; Pol-2 = Polychaete-2; Mus = Blue Mussels; Oys = Oyster; Amp = Amphipods; Mys = *Mysis*; DCr = Dungeness crab; Shri =Shrimp (*Crangon*); Sper = Shiner Surfperch; Herr = Pacific Herring; Wpol = Walleye Pollock; Anch = Northern Anchovy; WGr= Whitespotted Greenling; Pmid = Plainfin Midshipman; Sfish = Sablefish; Hal = Halibut; Sal = Chinook Salmon. In the models trophic position values for detritus (TL = 1) and phytoplankton (TL = 1) were assigned according to Vander Zanden and Rasmussen (1996).

Table 24: Feeding Preferences Matrix - dietary composition and trophic levels (TL) of 19 predator species / organisms in the outer coast area. Prey species and their corresponding trophic levels are identified.

	Prey[1] (Diet %)																					Sum
Offshore-Marine Food Web Species (Predators)	TL	Det	Phy	Zoo	Pol-1	Pol-2	Mus	Oys	Amp	Mys	DCr	Shri	Sper	Herr	Wpol	Anch	Sqd	Pmid	Sfish	Hal	Sal	
Zooplankton (Copepoda, *Neocalanus*)	2.0		100																			100
Polychaete-1 *(Neanthes succinea)*	2.1	90	5	5																		100
Polychaete-2 *(Harmothoe imbricata)*	2.1	30	35	35																		100
Blue mussel (*Mytilus edulis*)	2.3	15	60	25																		100
Oyster (*Crassostrea gigas*)	2.3	15	60	25																		100
Amphipods (*Themisto* sp.)	2.4	30	35	35																		100
Mysis sp.	2.5	10	45	45																		100
Dungeness crab (*Cancer magister*)	2.8	43	2	10	5	5	5	5	5	5		5	5	5								100
Crangon sp.*	2.9		30	30						40												100
Shiner surfperch (*C. aggregata*)	3.2	5	10	10	10	10			20	15		20										100
Pacific Herring (*Clupea pallasi*)	3.0			98	1				1													100
Walleye pollock (*T. chalcogramma*)	3.0			95	2.5				2.5													100
Northern anchovy (*Engraulis mordax*)	3.1		20	20					15	25		20										100
Gonatid squid (*Gonatius*)	3.5			50					3	5		5	9	9	9	9						100
Plainfin midshipman (*P. notatus*)	3.5	5			10	5			15	20	15	20	5			5						100
Sablefish (A. *fimbria*)	3.8			10	5				5		5	10	6	3	50	6						100
Halibut (*Hippoglossus stenolepis*)	4.0			1	1	1	1	1	1	1	1	14	5	5	43	5	4	5	1			100
Chinook salmon (*O. tshawytscha*)	4.0			5										10		14	71					100
Killer whale (*O. orca*)	5.0																		2	2	96	100

[1]Legend prey species: Det = Detritud; Phy = Phytoplankton; Zoo = Zooplankton; Pol-1 = Polychaete-1; Pol-2 = Polychaete-2; Mus = Blue Mussels; Oys = Oyster; Amp = Amphipods; Mys = *Mysis*; DCr = Dungeness crab; Shri =Shrimp (*Crangon*); Sper = Shiner Surfperch; Herr = Pacific Herring; Wpol = Walleye Pollock; Anch = Northern Anchovy; Sqd= Gonatid squid; Pmid = Plainfin Midshipman; Sfish = Sablefish; Hal = Halibut; Sal = Chinook Salmon. In the models trophic position values for detritus (TL = 1) and phytoplankton (TL = 1) were assigned according to Vander Zanden and Rasmussen (1996).

Table 25: Feeding Preferences Matrix - dietary composition and trophic levels (TL) of 22 predator species / organisms, incorporating updated data on new prey items for resident killer whales and redistribution of diet composition form some fish species. Prey species and their corresponding trophic levels are identified.

Coastal-Marine Food Web Species (Predators)	TL	Prey[1] (Diet %)																							Sum
		Det	**Phy**	**Zoo**	**Pol-1**	**Pol-2**	**Mus**	**Oys**	**Amp**	**Mys**	**DCr**	**Shri**	**Sper**	**Herr**	**Wpol**	**Anch**	**Dsol**	**Chum**	**Sqd**	**Coho**	**Lcod**	**Sfish**	**Hal**	**Sal**	
Zooplankton (Copepoda, *Neocalanus*)	2.0		100																						100
Polychaete-1 *(Neanthes succinea)*	2.1	90	5	5																					100
Polychaete-2 *(Harmothoe imbricata)*	2.1	30	35	35																					100
Blue mussel (*Mytilus edulis*)	2.3	15	60	25																					100
Oyster (*Crassostrea gigas*)	2.3	15	60	25																					100
Amphipods (*Themisto* sp.)	2.4	30	35	35																					100
Mysis sp.	2.5	10	45	45																					100
Dungeness crab (*Cancer magister*)	2.8	43	2	10	5	5	5	5	5	5		5	5	5											100
Crangon sp.*	2.9		30	30						40															100
Shiner surfperch (*Cymatogaster aggregata*)	3.2	5	10	10	10	10			20	15		20													100
Pacific Herring (*Clupea pallasi*)	3.0			98	1				1																100
Walleye pollock (*Theragra chalcogramma*)	3.0			95	2.5				2.5																100
Northern anchovy (*Engraulis mordax*)	3.1		20	20					15	25		20													100
Dover Sole (*Microstomus pacificus*	3.3				27	27	7.25	7.25	1	10	10	10													100
Chum salmon (*Oncorhynchus keta*)	3.4	12		24	0.5	0.5			9		2			17.5		17.5			17						100
Gonatid squis	3.5			50					3	5		5	9.3	9.3	9.3	9.3									100

(*Gonatius*)																							
Sablefish (*Anoplopoma fimbria*)	3.8	10	5				5		5	10	3	3	45	3	2.5		8						100
Coho salmon (*Oncorhynchus kisutch*)	4.2	26					34		4	4		16		8		8							100
Lingcod (*Ophiodon elongates*)	4.3						10	6.7	6.7	6.7			25		25		20						100
Halibut (*Hippoglossus stenolepis*)	4.0	1	1	1	1	1	1	10	14	14	5	5	38		1		5	1		1			100
Chinook salmon (*Oncorhynchus tshawytscha*)	4.0	5					1			4	10	25	25	10	10		10						100
Killer whale (*Orcinus orca*)	5.0														3	10	3	5	3	3	3	70	100

[1]Legend prey species: Det = Detritud; Phy = Phytoplankton; Zoo = Zooplankton; Pol-1 = Polychaete-1; Pol-2 = Polychaete-2; Mus = Blue Mussels; Oys = Oyster; Amp = Amphipods; Mys = *Mysis*; DCr = Dungeness crab; Shri =Shrimp (*Crangon*); Sper = Shiner Surfperch; Herr = Pacific Herring; Wpol = Walleye Pollock; Anch = Northern Anchovy; Dsol = Dove sole; Coho = Coho salmon; Sqd= Gonatid squid; Sfish = Sablefish; Chum = Chum Salmon; Lcod = Lingcod; Hal = Halibut; Sal = Chinook Salmon. In the models trophic position values for detritus (TL = 1) and phytoplankton (TL = 1) were assigned according to Vander Zanden and Rasmussen (1996).

2.5 MODEL IMPLEMENTATION IN EXCEL SPREADSHEET

We built the model in Microsoft Excel 2000 ®. The model includes submodels, data compilations, calculations (e.g., BSAFs), and results that were used in the evaluations of model performance analysis.

2.5.1 Forward Calculation: Total PCB Concentration Estimations in Fish and Wildlife

"Forward" calculations determine PCB concentrations in fish and wildlife (C_B) based on measured or predicted PCB concentrations in the sediment (C_S) (in this case sediment concentrations are the model input). Sediment PCB concentrations are in logarithmic format (log C_S) so that the lognormal distributions of sediment concentrations are able to be depicted as normal distributions of log C_S. The BSAF (model output) is also depicted in logarithmic format (log BSAF) based on the same reasoning. The calculation is:

$$\log C_B = \log C_S + \log BSAF \tag{2.46}$$

And C_B then follows as:

$$C_B = 10^{\log(C_B)} \tag{2.47}$$

Mathematically this is equivalent to:

$$C_B = BSAF \cdot C_S \tag{2.48}$$

Log C_B contains the propagation of variability and error from model input parameters (i.e., log C_S) and error in the model calculations (i.e., log BSAF). Uncertainty in organism concentrations is described by the geometric mean concentration's standard deviation (SD_{CB}), which is calculated from the log BSAF standard deviations (SD_{BSAF}) and the sediment concentration standard deviations (SD_{CS}):

$$SD_{CB} = \sqrt{(SD_{CS}^2 + SD_{BSAF}^2)} \tag{2.49}$$

C_B is calculated for each PCB congener and total-PCBs in the forward calculations, and its uncertainty is based on uncertainty in sediment total-PCB concentrations and BSAFs. Section 3.4 describes the variability and uncertainty in BSAF. Uncertainty is derived with model performance analysis involving comparison of observed and predicted total-PCB BSAFs.

Model predictions of C_B in the Management module are conducted for herring, sablefish, halibut, Chinook salmon, and killer whales. BSAFs are calculated for all species in the Science module of the model. All of the species in the Science module can be included in the Management module, but for simplicity, the number of species in the display of the model results was limited to those most relevant for management purposes.

Predicted species PCB concentration distributions can be used to determine the frequency they exceed target threshold PCB concentrations. An example is shown in Figure 17, which represents the biota log- normal PCB concentration distributions as cumulative distributions. The y-axis indicates the fraction of PCB concentrations expected to be larger or smaller than target values of interest.

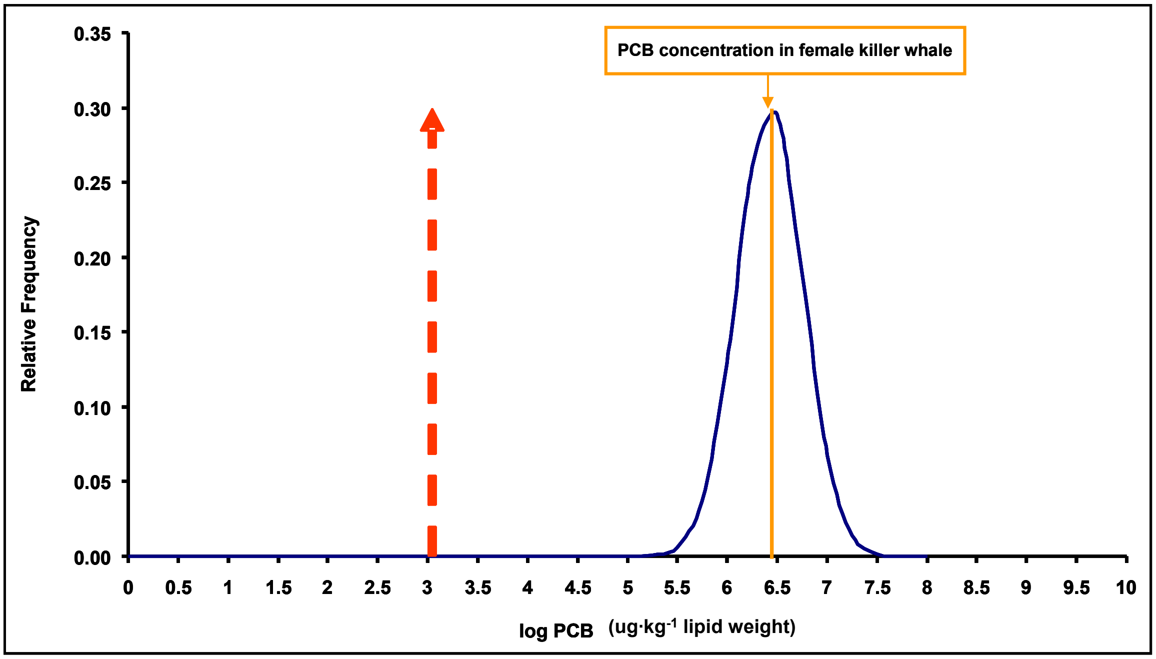


Figure 17: Example of a cumulative probability distribution of the ΣPCB concentration in female killer whales in relation to a threshold concentration (the red dashed line). The cumulative probability distribution illustrates the probability of exceeding the PCB target threshold concentration.

2.5.2 Forward Calculation: Characterizing the Potential for Ecological Effects to Fish and Wildlife

The food web bioaccumulation model can predict the frequency of occurrence of toxicological effects in species included in the model by comparing a species' ΣPCB concentration (of the congeners included; see Tables 1-7 in Appendix III) cumulative frequency distribution (i.e., normal probability density function) to accepted internal body residue concentrations associated with toxicological effects. The cumulative frequency distribution is the expected range of PCB concentrations in a species due to the sediment PCB concentration. The cumulative frequency distribution is able to depict the proportion of the population expected to have PCB concentrations exceeding the threshold concentration associated with the toxic effect, and which is expected to be negatively impacted by PCBs.

The model can make assessments of the toxicological significance of PCB concentrations in killer whales, and to do this we used several threshold effect concentrations for total PCBs: that for harbour seals of 17,000 $\mu g \cdot kg^{-1}$, wet weight (Hickie *et al.* 2007); that for bottlenose dolphins of 10,000 $\mu g \cdot kg^{-1}$, wet weight (Hall *et al.* 2006); that for harbour seals which was a revision of the Hickie *et al.* (2007) value of 1,300 $\mu g \cdot kg^{-1}$ lipid weight tissue residue in blubber (Mos *et al.* 2010); and the long-term PCB concentration in Chinook salmon of 8 $\mu g \cdot kg^{-1}$, wet weight for 95% of the killer whale population to fall below the toxicity threshold (Hickie *et al.* 2007).

An ecological risk index (ERI) is based on each effect concentration in each species:

$$ERI = C_B / C_{THRESHOLD} \qquad (2.50)$$

2.5.3 Backward Calculation: Estimating Total PCB Concentrations in Sediments from PCB Concentration in Fish and Wildlife

"Backward" calculations use PCB concentrations in fish or wildlife (C_B) to calculate the PCB concentration in sediments (C_S). This provides target sediment PCB concentrations that meet ecological criteria expressed as a PCB concentration C_B. The calculation is:

$$\log C_S = \log C_B - \log BSAF \tag{2.51}$$

Mathematically equivalent to:

$$C_S = C_B / BSAF \tag{2.52}$$

Where C_B is the external input variable and the model calculates the BSAF. Backwards calculations were conducted for ΣPCBs. Model error uncertainty is captured in backwards calculations in the uncertainty in the BSAF, which the model calculates as described above. When entering the biota PCB concentrations it is also possible to include accepted variability in the target biota concentration by combining the uncertainty in the BSAF and C_B to obtain a distribution of sediment PCB concentrations expected to produce the entered distribution of PCB concentrations in fish or wildlife. The backward calculation can be used to derive sediment target levels (e.g., new Sediment Quality Guidelines) using toxicity tissue thresholds reported for marine mammals (e.g., harbour seals) to protect predator species at the top of the food web such as killer whales.

3. METHODOLOGY

3.1 GENERAL

The PCB food web bioaccumulation model was tested and evaluated with a *model performance analysis* and an *uncertainty analysis*. The performance and uncertainty of the model have also been successfully tested in the original development and application (Gobas and Arnot 2010).

The *model performance analysis* (described in section 3.2) assesses the accuracy of model predictions, and compares model predicted BSAFs to independent, observed BSAFs of PCB congeners in biota.

The *uncertainty analysis* (described in section 3.3) evaluates error in model calculations, and is important for management purposes since it considers the magnitude of the model predictions. The *uncertainty analysis* uses calculated differences between observed and predicted BSAFs of ΣPCBs to assess the uncertainty of model calculations.

3.2 MODEL TESTING AND PERFORMANCE ANALYSIS

Model performance analysis compares each PCB congener's (i) model predicted sediment-receptor concentration relationship ($BSAF_{P,i}$), to the observed sediment-receptor concentration relationship ($BSAF_{O,i}$). Measured sediment and estimated water PCB congener concentrations were input parameters for calculation of PCB concentrations in biota, then $BSAF_{P,i}$ was calculated dividing the calculated biota concentration by the sediment concentration. Measured biota PCB concentrations were divided by measured sediment concentrations to obtain the $BSAF_{O,i}$. This measure of model performance was described quantitatively by the model bias (MB), which is species-specific:

$$MB_j = 10^{\left(\sum_{i=1}^{n} \frac{\left[\log(BSAF_{P,i} / BSAF_{O,i} \right]}{n} \right)} \quad (3.1)$$

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

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

$$MB_j{*} = 10^{\left(\sum_{i=1}^{m} \frac{\left[\log(BSAF_{P,\Sigma PCB} / BSAF_{O,\Sigma PCB} \right]}{m} \right)} \quad (3.2)$$

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

3.3 UNCERTAINTY ANALYSIS

Uncertainty in the model input parameters (i.e., log C_S) and in the model calculations (i.e., log BSAF) are propagated in the estimate of log C_B in terms of the standard deviation SD_{CB} of log C_B (i.e., the geometric mean concentration). Spatial distribution of sediment PCB congener concentrations were represented by the standard deviation (SD_{CS}) of the mean log C_S (i.e., of the geometric mean of the concentration in the sediments). The standard deviation (SD_{BSAF}) of log BSAF (i.e., geometric mean of the BSAF) was used to represent variability in log BSAF. The standard deviation (SD_{CB}) of log CB (i.e., geometric mean of the ΣPCB concentration in the biota) was used to express the effect of variability and error in sediment PCB congener concentrations and uncertainty in BSAF estimates of biota PCB congener concentrations. SD_{CB} is calculated from the standard deviations (SD_{BSAF}) of log BSAF estimates and the standard deviation (SD_{CS}) of log C_S as:

$$SD_{CB} = \sqrt{(SD_{CS}^2 + SD_{BSAF}^2)} \qquad (3.3)$$

Using empirical data to describe model calculation uncertainty improves model credibility since model calculations are directly compared to empirical data, but any problems with the empirical data are reflected in the model's uncertainty estimate. One important limitation of the empirical data is that there was limited spatial coverage. For example, biota concentrations were only obtained for some of the areas included in the model, thus the biota PCB concentrations may not accurately represent the concentrations in all areas. Also, empirical PCB concentrations likely do not accurately represent temporal variations in PCB concentrations. because of their limited temporal coverage. Thus it is beneficial to assess model uncertainty with a second method that attempts to incorporate geographical and temporal variations in PCB concentrations.

Uncertainty analysis through Monte Carlo simulation was considered but found to be problematic because of the interdependence of state variables and lack of data to define uncertainty distributions for several state variables. The interdependence of several state variables including feeding rates, growth rates, fecal egestion rates and feeding preferences caused inconsistencies in the energy and mass balance of the model. The associated error was deemed to be too large for the Monte Carlo simulations to provide meaningful estimates of model uncertainty. One of the advantages of using empirical observation to assess uncertainty is that it includes many sources of uncertainty while Monte Carlo simulation is limited to model parameterization uncertainty. Because uncertainty in observed ΣPCB concentrations reflect to some degree spatial variation in PCB concentrations in sediments, which is also specifically considered by SD_{CS} in Equation 3.3, the estimated uncertainty in C_B may be somewhat overestimated by this method.

3.4 MODEL APPLICATION

The model is able to make estimates of ΣPCB and PCB congener concentrations in organisms of the food web based on current sediment PCB concentrations, which are referred to as "forwards" calculations conducted in the management spreadsheet of the model. The model can also estimate sediment ΣPCB concentrations expected to meet a set of criteria of ecological and killer whale health relevance, which are referred to as "backwards" calculations also conducted in the management spreadsheet.

3.4.1 Forward Calculations

We calculated PCB congener concentrations of in nine fish species, and male and female killer whales from empirical sediment PCB congener concentrations.

First we compiled sediment concentration data from all areas included in the model (Queen Charlotte Strait lacked sediment PCB data, thus we assumed it had the same average sediment concentration as the outer coast area). Overall, 61 samples were available from several studies (references listed in Table 7). Statistical analysis of the concentration data was conducted on each congener to obtain the statistical distribution of sediment PCB concentrations, which were represented by log-normal distributions. The method was similar for ΣPCB, which is the sum of the PCB congener concentrations (described in section 2.3.1). A main assumption is that the sediment samples included from the literature adequately represent the spatial distribution of PCB concentrations to which biota are exposed. The Queen Charlotte Strait area is lacking any sediment PCB data and this is problematic, and highlights the need for further research. Sediment and water PCB congener concentrations included in the model to represent the areas are listed in Appendix III. PCB congener concentrations in biota were calculated from the sediment BSAFs and the PCB congener concentrations:

$$\log C_B = \log BSAF + \log C_S \qquad (3.4)$$

3.4.2 Backward Calculations

Sediment ΣPCB concentration expected to meet ΣPCB concentrations in fish and wildlife associated with various ecological risks was performed in the backwards calculation as:

$$\log C_S = \log (TEC) - \log BSAF - 1.96 \cdot (SD_{MB}) \qquad (3.5)$$

Where, TEC is the toxic effect concentration in biota and 1.96 is the confidence value to have 95% probability for the observations in a normal distribution to fall below the target sediment concentration, and SD_{MB} is the standard deviation or error of the model bias (MB) for biota obtained from the model testing/performance analysis. BSAFs were calculated in the forwards calculations based on the current composition of sediment PCB congeners in the areas included. The calculated sediment ΣPCB concentration (C_S) also assumes that the composition of PCB concentrations in the areas is the same as entered in forward calculations to represent current conditions. Thus ΣPCB concentrations can be used to calculate congener specific concentrations assuming that the PCB congener profile is similar to that in current sediments.

Target concentrations for ΣPCB in sediments were derived from ecological risk health effect thresholds (values summarized in Table 26) that were entered as log TEC in equation 3.5. To calculate log C_S, we subtracted log BSAF of ΣPCB and $1.96 \cdot (SD_{MB})$ from log TEC, and then log C_S was used to determine the target sediment ΣPCB concentration as the anti-log ($10^{\log C_S}$). The target sediment ΣPCB concentration represents the geometric mean sediment ΣPCB concentration required to meet ecological risk targets. Derivation of the target sediment concentrations introduced uncertainty in the BSAF and this represents the uncertainty in the model's calculation of the geometric mean sediment ΣPCB concentration that meets ecological risk targets. Of importance is that the model calculates a geometric mean target sediment ΣPCB concentration. Theoretically, many sediment concentration statistical distributions exhibit the same geometric mean, which indicates there are several different ΣPCB sediment concentration distributions consistent with ecological risk targets used in the model calculations. This is important from a management perspective since it suggests that a wide range of management options are potentially available for achieving ecological risk objectives.

Table 26: Health effect thresholds for total PCBs in marine mammals. All studies involved free-ranging or captive fed marine mammals, wherein PCBs represented the dominant concern and the contaminants which best correlated with observed effects.

Toxic Effect Concentrations (TEC)	**TEC (µg·kg^{-1} lipid)**	**Log TEC (µg·kg^{-1} lipid)**
Harbour seal PCB toxicity (Ross *et al.* 1996c)	17000	4.23
Bottlenose dolphin PCB toxicity (Hall *et al.* 2006)	10000	4.00
Revised harbour seal PCB toxicity (Mos *et al.* 2010)	1300	3.11

4. RESULTS AND DISCUSSION

4.1 EXPLORING DISPOSAL IN RESIDENT KILLER WHALE HABITAT

4.1.1 Model Applications to Chinook Salmon and Resident Killer Whales

Empiral sediment PCB values were used to predict PCB concentrations in hypothetical Chinook salmon and resident killer whales and to calculate the BSAF values for each of the areas of interest. These BSAFs are required for subsequent modeling calculations which rely on realistic geographical distributions of prey and predator, and contribute to our exploration of a variety of disposal scenarios or an assessment of decision criteria and laboratory methods. Summary tabulations are presented here for Chinook salmon (Table 27), and killer whales (males Table 28; females Table 29).

Using the BSAF from these exercises, we derived life history-based BSAF values based on predicted time spent by Chinook salmon and killer whales in each of the seven areas of interest. These more realistic outcomes are present for Chinook salmon (Lower Fraser stocks in Table 30; and South Thompson stocks in Table 31), and for killer whales (northern resident males and females in Table 32; southern resident males and females Table 33).

The lowest PCB concentrations in biota and the lowest BSAF value were predicted in the northern resident killer whale Critical Habitat (NRKW-Critical Habitat), while the highest PCB concentrations were predicted in the southern resident Critical Habitat in the USA (Puget Sound). This reflects a combination of ambient sediment PCB concentrations in the different areas, combined with relative time spent by Chinook salmon and resident killer whales in the different areas studied.

Predicted PCB concentrations for major diet items of resident killer whales, including Chinook salmon, halibut, and sablefish are provided in Appendix IV. Appendix IV also presents the relatively minor differences predicted for PCB congener concentrations in resident killer whales if the diet of killer whales were changed to include other species than those used in our primary models.

Table 27: Empirical PCB concentrations in sediments (C_S, mg·kg^{-1} dry weight) and predicted PCB concentrations in Chinook salmon (C_{fish}, mg·kg^{-1} wet weight), with calculated $BSAF_{fish}$, in assessed model areas (assuming 100% presence in model areas).

Model Areas	C_S	$BSAF_{fish}$	C_{fish}
NRKW -Critical Habitat	4.40E-04	26.5	0.010
Queen Charlotte Strait	6.95E-04	38.0	0.030
Outer coast	6.95E-04	68.0	0.050
Strait of Georgia	1.05E-03	63.0	0.100
SRKW-Critical Habitat in Canada	5.20E-04	63.5	0.03
SRKW-Critical Habitat in USA (Puget Sound)	7.40E-02	51.0	3.80
SRKW-Critical Habitat in USA (summer core & Juan de Fuca Strait)	6.10E-03	92	0.558

Table 28: Empirical PCB concentrations in sediments (C_S, mg·kg^{-1} dry weight) and predicted PCB concentrations in a male killer whale ($C_{KW\text{-}male}$, mg·kg^{-1} wet weight), with calculated $BSAF_{KW\text{-}male}$, in assessed model areas (assuming 100% presence in model areas; and a realistic diet: 96% Chinook salmon, 2% halibut; and 2% sablefish).

Model Areas	C_S	$BSAF_{KW\text{-}male}$	$C_{KW\text{-}male}$
NRKW -Critical Habitat	4.40E-04	7790	3.40
Queen Charlotte Strait	6.95E-04	11160	7.80
Outer coast	6.95E-04	19640	14.0
Strait of Georgia	1.05E-03	18800	20.0
SRKW-Critical Habitat in Canada	5.20E-04	18845	10.0
SRKW-Critical Habitat in USA (Puget Sound)	7.40E-02	15370	1140
SRKW-Critical Habitat in USA (summer core & Juan de Fuca Strait)	6.10E-03	27670	168

Table 29: Empirical PCB concentrations in sediments (C_S, mg·kg^{-1} dry weight) and predicted PCB concentrations in a female killer whale ($C_{KW\text{-}female}$, mg·kg^{-1} wet weight), with calculated $BSAF_{KW\text{-}female}$, in assessed model areas (assuming 100% presence in model areas; and a realistic diet: 96% Chinook salmon, 2% halibut; and 2% sablefish).

Model Areas	C_s	$BSAF_{KW\text{-}female}$	$C_{KW\text{-}female}$
NRKW -Critical Habitat	4.40E-04	1080	0.480
Queen Charlotte Strait	6.95E-04	1820	1.26
Outer coast	6.95E-04	3190	2.20
Strait of Georgia	1.05E-03	3045	3.20
SRKW-Critical Habitat in Canada	5.20E-04	3050	1.60
SRKW-Critical Habitat in USA (Puget Sound)	7.40E-02	2487	185
SRKW-Critical Habitat in USA (summer core & Juan de Fuca Strait)	6.10E-03	4475	27.0

Table 30: Realistic and total BSAF values for Lower Fraser River Chinook based on the observed distribution in the model areas.

Lower Fraser River Chinook areas	% Time spent per area	BSAF (100% presence)	BSAF per area
Queen Charlotte Strait	1.71	38.0	0.650
Outer coast	55.0	68.0	37.0
NRKW Critical Habitat	14.47	26.5	3.80
SRKW Critical Habitat in Canada	7.68	63.5	4.90
Strait of Georgia	7.68	63.0	4.90
SRKW Critical Habitat in USA (summer core & Juan de Fuca Strait)	4.07	92.0	3.75
SRKW Critical Habitat in USA (Puget Sound)	9.41	51.0	5.0
Total	100		60.0

BSAF per area = (% Time spent)*(BSAF)

Table 31: Realistic and total BSAF values for South Thompson Chinook based on the observed distribution in the model areas.

South Thompson Chinook areas	% Time spent per area	BSAF (100% presence)	BSAF per area
Queen Charlotte Strait	7.99	38.0	3.00
Outer coast	79.9	68.0	54.0
NRKW Critical Habitat	3.47	26.5	0.90
SRKW Critical Habitat in Canada	3.45	63.5	2.20
Strait of Georgia	3.45	63.0	2.20
SRKW Critical Habitat in USA (summer core & Juan de Fuca Strait)	1.63	92.0	1.50
SRKW Critical Habitat in USA (Puget Sound)	0.17	51.0	0.10
Total	100		64.0

BSAF per area = (% Time spent)*(BSAF)

Table 32: Realistic and total BSAF values for northern resident killer whales (males and females) based on field observed distributions in the model areas.

	Outer Coast	Queen Charlotte Strait	NRKW Critical Habitat	Total
% Time spent per area	**75.0**	**17.0**	**8.0**	**100**
		Northern Resident Killer Whale (male)		
BSAF (100% presence)	19640	11160	7790	
BSAF per area	14730	1900	620	1725 0
% of PCBs attributable to area	**85.4**	**11.0**	**3.6**	**100**
		Northern Resident Killer Whale (female)		
BSAF (100% presence)	3190	1820	1080	
BSAF per area	2390	310	86	2790
% of PCBs attributable to area	**85.7**	**11.1**	**3.1**	**100**

BSAF per area = (% Time spent)*(BSAF)

Table 33: Realistic and total BSAF values for southern resident killer whales based on field observed distributions in the model areas.

	Outer coast	SRKW Critical Habitat in Canada	Strait of Georgia	SRKW Critical Habitat in USA (Puget Sound)	SRKW Critical Habitat in USA (summer core & Juan de Fuca Strait)	Total
% Time spent per area	**37.0**	**18.0**	**3.0**	**6.0**	**36.0**	**100**
		Southern Resident Killer Whale (male)				
BSAF (100% presence)	19640	18845	18800	15370	27670	
BSAF per area	7266	3392	564	922	9960	22105
% of PCBs attributable to area	**32.9**	**15.3**	**2.6**	**4.2**	**45.1**	**100**
		Southern Resident Killer Whale (female)				
BSAF (100% presence)	3190	3050	3045	2490	4475	
BSAF per area	1180	550	90	150	1610	3580
% of PCBs attributable to area	**33.0**	**15.4**	**2.5**	**4.2**	**45.0**	**100**

BSAF per area = (% Time spent)*(BSAF)

4.1.2 Exploring the Implications of Disposal at Sites Only Within Critical Habitat

Predicting PCB concentrations in killer whales following the disposal of dredge materials into Critical Habitat is not easy. In this series of exercises, we modelled four hypothetical disposal scenarios equivalent to PCB measurements from existing sediment sampling sites including Puget Sound (east Harbor Island), Victoria Inner Harbour, Burrard Inlet average and the Strait of Georgia average; Tables 34 - 37). Since several habitat areas can be used by the two populations of resident killer whales, the PCB levels in killer whales are attributed to a contribution from different areas. These efforts assume a one-time and permanent shift from ambient current sediment PCB concentrations at disposal sites to one of these four new (substituted) values. The model employs a weighted average PCB concentration which is based on the relative surface area of the disposal sites nested within the wider area of the Critical Habitat.

We also evaluated two guideline values and two analytical detection limits in order to critically evaluate the utility of these values in supporting management decisions (Tables 34 - 37). Results reveal that the screening limit (CEPA AL Low) upon which permitting is based could render killer whales more vulnerable to PCB contamination (this is further evaluated in a hypothetical killer whale modelled in section 4.1.4). Results also suggest that detection limits for the PCB analyses used for Disposal at Sea permits (EC DL; USEPA Method 8080) also render killer whales vulnerable to heightened PCB contamination, as they would be of insufficient quality to measure potentially harmful PCB concentrations in dredge materials. The High Resolution Gas Chromatograph Mass Spectrometry (HRGC/MS) approach is used by specialized analytical laboratories; this is a more costly approach but produces congener-speecific results for PCBs and related compounds, and generates much lower detection limits. For this exercise, we used detection limits generated for marine sediments by the Laboratory of Expertise for Aquatic Chemical Analyses (LEACA) at the Institute of Ocean Sciences (Sidney BC), and determined that their detection limits for PCBs in sediments were up to 350 times lower than low resolution analyses, thereby enabling more (killer whale-) protective decision making. This detection limit was the only one sufficiently low that its use as the sediment background concentration resulted in a net reduction in predicted PCB concentrations in killer whales. Finally, we suggest that CCME guidelines to protect killer whales could be developed, as the current guidelines appear to only protect lower trophic levels.

Basic results indicate that while killer whales are vulnerable to the accumulation of very high concentrations of PCBs, the likely contribution from current disposal operations is low (Figure 18 and 19; Tables 34 and 35 for northern residents, Tables 36 and 37 for southern residents). We did not have access to measured PCB concentrations in dredge materials, but recent surveys are providing an excellent baseline for future modeling and/or assessment. These recent surveys reveal that sites in the Fraser River have PCB concentrations that are lower than the ambient disposal sites in the coastal waters (P. Mudroch, personal communication). Scrutiny of dredge materials can ensure that PCB delivery to killer whales is not exacerbated, since highly contaminated dredgeate can augment killer whale PCB burdens.

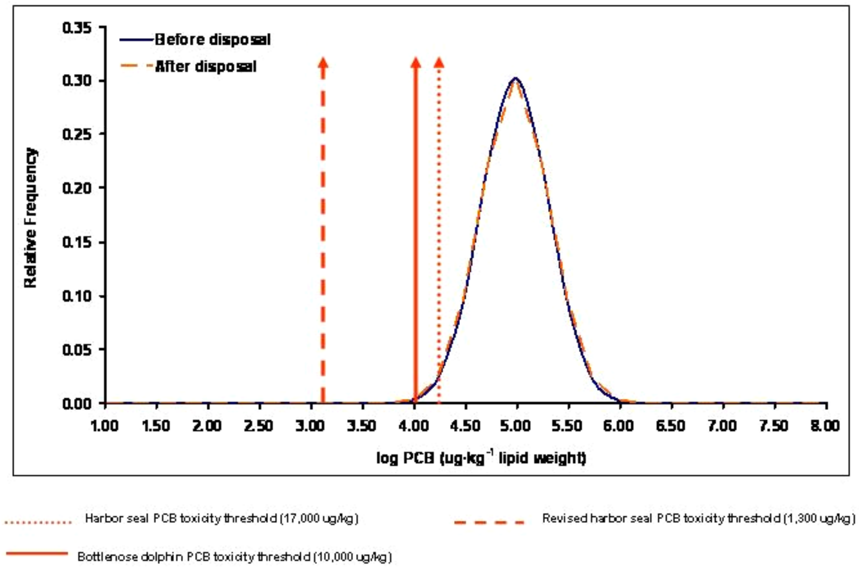


*Figure 18: Normal probability density distributions of predicted PCB concentrations for realistic habitat distributions for southern resident killer whales (SRKWs; males). Distribution lines reflect BEFORE (solid) and AFTER (dashed) a disposal scenario at sites within Critical Habitat consisting of sediments with a PCB concentration equal to the **CEPA Action Level Low**. The red dashed line represents the revised harbour seal PCB toxicity threshold (1,300 μg·kg^{-1} lipid; Mos et al. (2010)); the red solid line represents the bottlenose dolphin PCB toxicity threshold (10,000 μg·kg^{-1} lipid; Hall et al. (2006)); and, the red dotted line represents the previous harbour seal PCB toxicity threshold (17,000 μg·kg^{-1} lipid; Ross et al. (1996c)).*

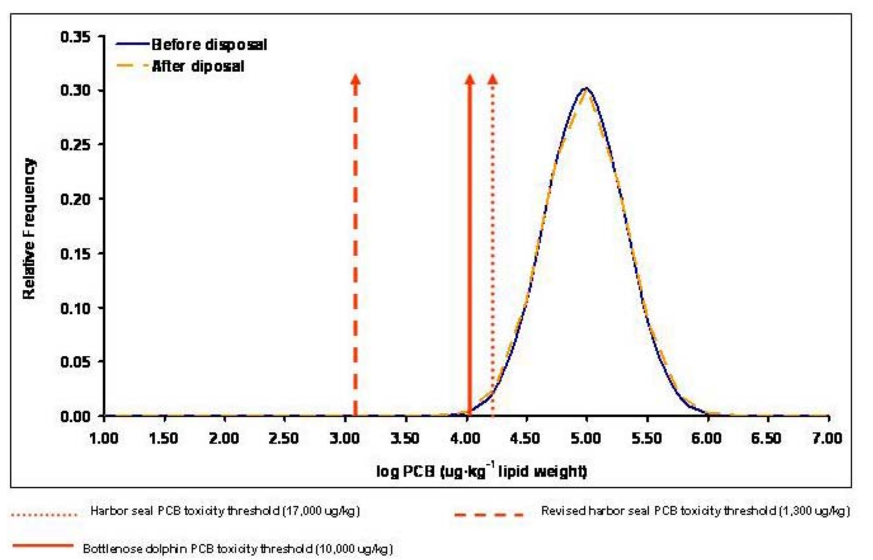
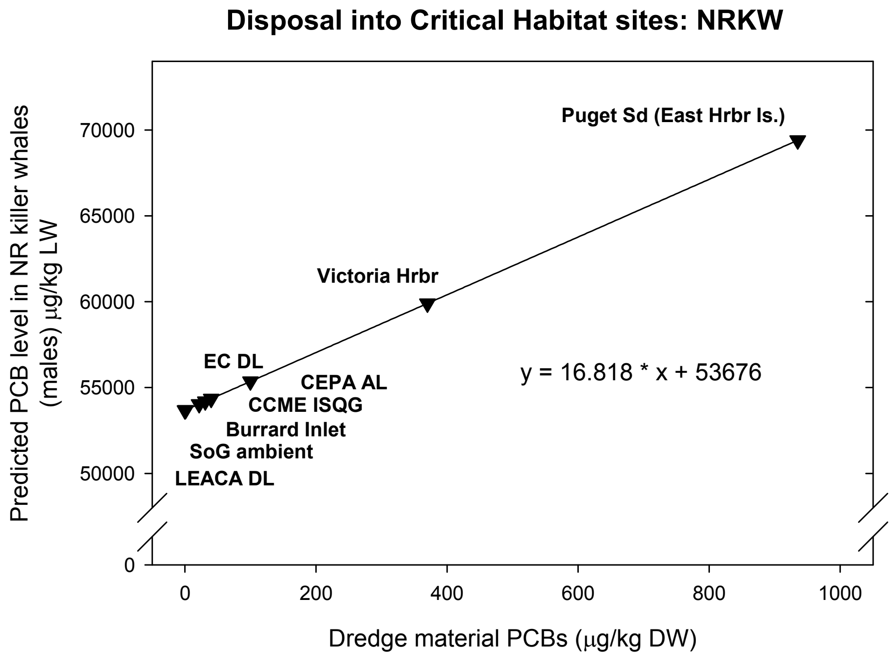


*Figure 19: Normal probability density distributions of predicted PCB concentrations for realistic habitat distributions for southern resident killer whales (SRKWs; males). Distribution lines reflect BEFORE (solid) and AFTER (dashed) a disposal scenario at sites within Critical Habitat consisting of sediments with a PCB concentration equal to the **Burrard Inlet average** empirical PCB concentration from (Grant et al., 2010). The red dashed line represents the revised harbour seal PCB toxicity threshold (1,300 μg·kg^{-1} lipid; Mos et al. (2010)); the red solid line represents the bottlenose dolphin PCB toxicity threshold (10,000 μg·kg^{-1} lipid; Hall et al. (2006)); and, the red dotted line represents the previous harbour seal PCB toxicity threshold (17,000 μg·kg^{-1} lipid; Ross et al. (1996c)).*

The implications of incremental increases in PCB concentrations in disposed dredgeate are apparent, as we present eight disposal scenarios in disposal sites found within Critical Habitat for both northern and southern resident killer whales (Figure 20). Highly contaminated dredge materials (using the example of Puget Sound East Harbor Island or Victoria) can increase the risk of PCB delivery to killer whales. These predicted increases assume that the sediments and water were in equilibrium, and that the Strait of Georgia was a closed system. However, some processes may decrease uptake of PCBs by the food web (e.g. burial by sedimentation and exchange with the open ocean), while others may increase the uptake (e.g. direct uptake of PCBs by food web during disposal operations). In addition, sediments do not provide an inexaustable supply of PCBs to the food web. The model, however, does reveal the incremental implications of the disposal of PCB-contaminated sediments into disposal sites, as determined by the sediment-water partition coefficients. In reality, a slow but steady burial of contaminated, dumped sediment by ambient sedimentation will in many cases reduce the availability of PCB to exchange with overlying water. In addition, if the concentration of PCB in coastal water is higher than that in the open ocean, which is predominantly if not always the case where there are local sources, then the Strait will export a proportion of the contaminant, making it unavailable to the Strait of Georgia food web. The replacement time of the water of the whole Strait is about one year, with much shorter replacement times for surface water (Pawlowicz *et al.* 2007) which limits the capacity of PCB evasion from local sediments to achieve steady state within the water.



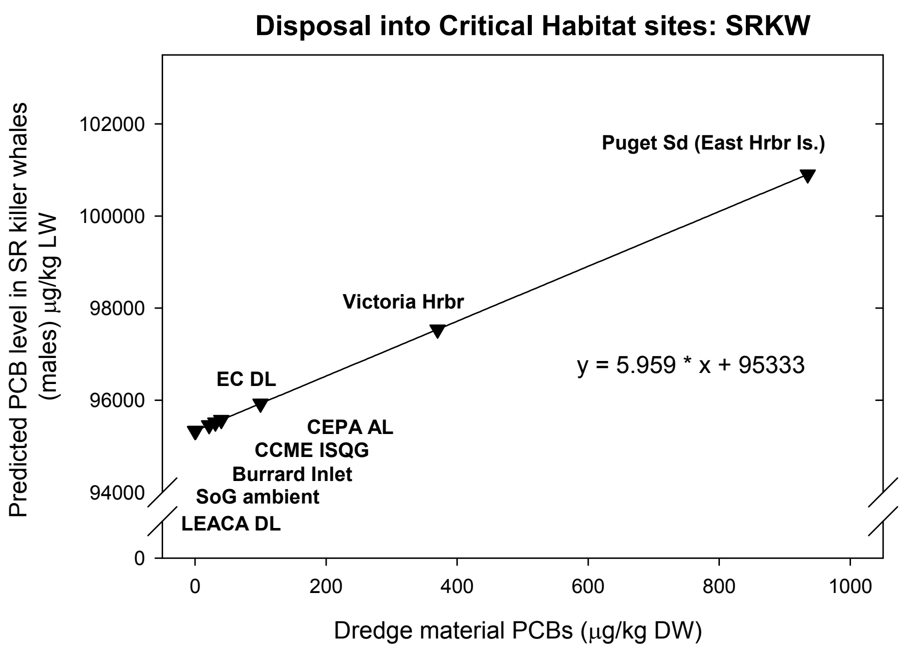


Figure 20: Predicted PCB concentrations in male northern resident killer whales (NRKWs) and southern resident killer whales (SRKWs) when sediments of different PCB concentrations are disposed of into ***all disposal sites (2 sites in NRKW CH and 3 sites in SRKW) in their respective Critical Habitats****. These predictions are based on theoretical scenarios where the sediment PCB values for the entire disposal site area nested within a broader Critical Habitat area are replaced with eight values, depicting four scenarios (PCB sediment values used from those measured in Puget Sound- East Harbor Island, Victoria Inner Harbour, Burrard Inlet average, and Strait of Georgia average) and four guidelines evaluated (current Detection Limit using EPA Method 8080, CEPA Action Level Low under Disposal at Sea regulations, CCME interim Sediment Quality Guidelines to protect aquatic biota, and the DFO LEACA Detection Limit for sediments).*

While ambient sedimentation buries contaminants, particularly rapidly near the Fraser River, mixing by benthic organisms (bioturbation) tends to delay this burial. Based on sedimentation rates (0.3 to 3 cm/yr) and depths of sediment surface mixed layers (4 cm to 25 cm) determined for 18 sites in the Strait of Georgia (Johannessen *et al.* 2005b;Macdonald *et al.* 1992), the half-life of PCB in the surface mixed layer, before it becomes permanently buried, is 10 ± 8 years (average ± s.d.). Therefore if a load of highly PCB-contaminated (relative to ambient) sediment was dumped in a given year and nothing further was done, the concentration of PCB would decline by approximately half in 2-20 years depending on the site, and after ~5 half lives would reach ambient PCB concentrations for the Strait. Figure 21 shows how a contaminant spike deposited at the surface of the sediment is affected by sedimentation without mixing (top panels) and sedimentation where benthic animals mix the sediments (bottom panels).

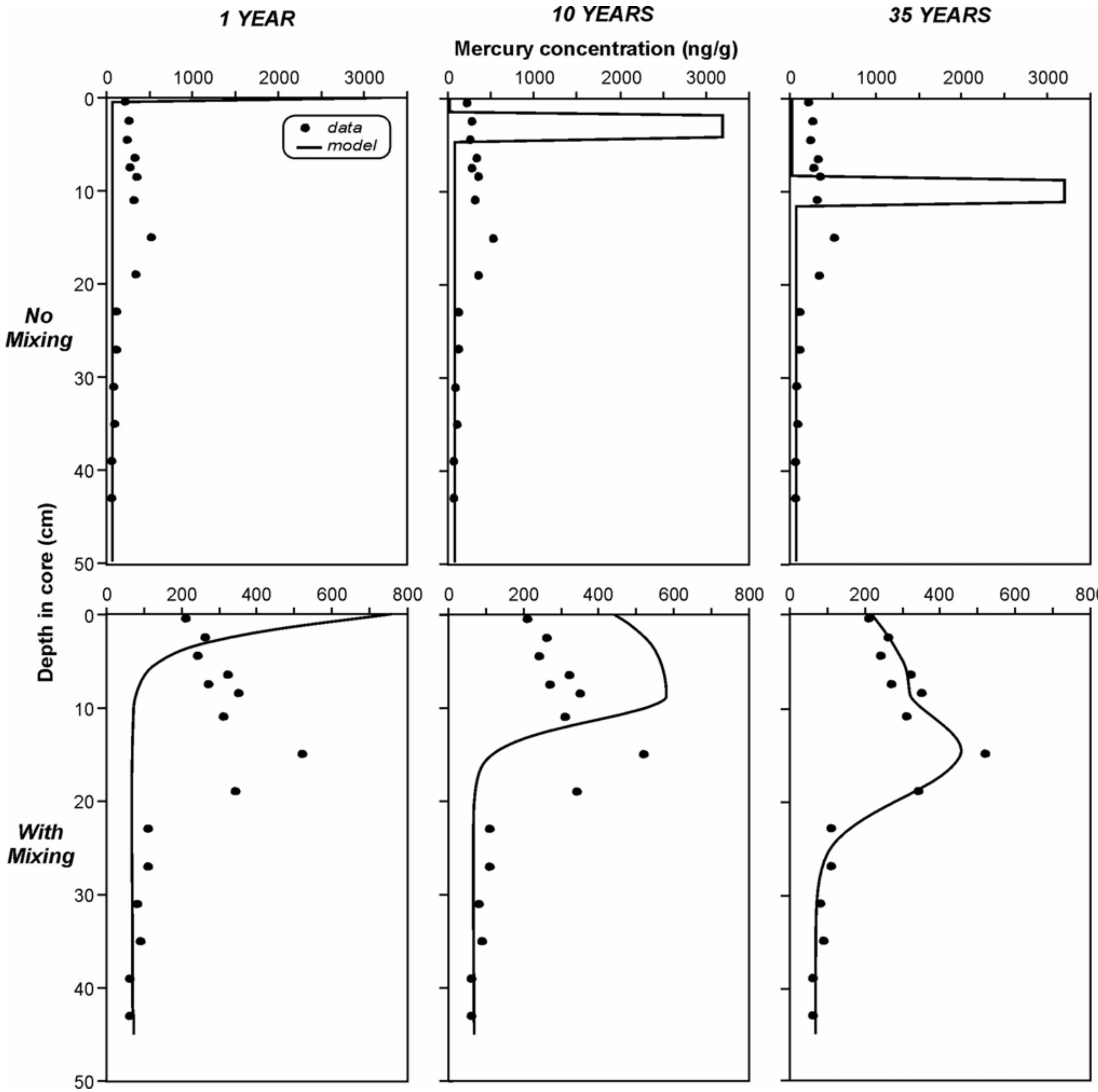


Figure 21: The effect of bioturbation (sediment mixing by benthic organisms) on the burial of a contaminant that enters in a single pulse, with time progressing along the panels from left to right. Without mixing (upper panels), the contaminant peak maintains its shape and magnitude, but is rapidly buried by subsequent, clean sediment. With a surface mixed layer (in this case about 10 cm), the peak is quickly smeared downwards, reducing its magnitude, but the mixing process maintains elevated surface concentrations for many years (half-lives of ~ 10 years). This example shows the burial of a pulse of mercury contamination (see (Johannessen et al. 2005a), but the circumstances of mixing and sedimentation apply equally to PCBs in material dumped onto bioturbated sediment.

This modeling approach employs an unrealistic one-time and permanent shift towards a single, static sediment PCB concentration, which then pervades the various ocean systems considered (outer coast, Queen Charlotte Strait, Strait of Georgia, etc.) or subregions within those systems (dump sites, Critical Habitat). However, the model does provide a realistic view of the consequent partitioning of PCB within such systems and its distribution within the foodweb up to and including killer whales, based on physical chemical properties of PCBs, reasonable statistical distributions of PCB in sediments, reasonable schemes of foraging and trophic interactions, and a plausible set of processes within the affected animals themselves, all based on a substantial body of literature. The findings, therefore, provide managers with a tool to carry out preliminary risk management evaluations based on a set of assumptions. The relationships between dredge material PCB content and predicted PCB levels in resident killer whales are depicted for Critical Habitat sites (Figure 20). For disposal at sites within Critical Habitat, therefore, this relationship can guide preliminary assessments of permit applications based on their PCB content as follows:

(1) $[PCB]_{NRKW} = 16.818 * x + 53676$ (Northern resident killer whales)
(2) $[PCB]_{SRKW} = 5.959 * x + 95333$ (Southern resident killer whales)

where 'x' represents the PCB concentration in dry weight ($\mu g \cdot kg^{-1}$) in dredge materials being considered for deposition at all disposal sites within Critical Habitat only.

A better predictive model for PCB concentrations in sediments following disposal would incorporate a sediment transport model which captures the spatial, temporal and constitutent physical and chemical features of disposal sites. This would be disposal site-specific, since such features vary among areas.

Another way to look at the above relationship would be to compare ambient sediment PCB concentrations with proposed dredge materials. PCB concentrations in dredged materials that are lower than the proposed receiving ambient sediment PCB concentrations would dilute the disposal site with cleaner sediments and not heighten the delivery of PCBs to killer whales. PCBs concentrations in dredged materials that are the same as the receiving ambient sediment PCB concentrations would elicit no effect in the model on PCB delivery to killer whales. Finally, PCB concentrations in dredge materials that exceed the ambient sediment PCB concentrations would have the capacity to increase the risk of heightened delivery of PCBs to killer whales, as described in the above equations and Figure 20. In the last case, alternative disposal strategies should be considered with the view of how best to reduce or eliminate re-cycling of PCB into biota through dredging activities.

The map of surface PCB concentrations in this report (Figure 6) illustrates the current distribution, but these concentrations are not static. The concentration of PCBs in the atmosphere is declining globally, as the use of these chemicals decreases, and that decline is reflected in sediment concentrations in the Strait of Georgia (Figure 22). Consequently, if it is determined that disposal should only be permitted where the PCB concentration in dumped material does not exceed the ambient concentration, it will be necessary to measure PCB concentrations at the disposal sites again every few years.

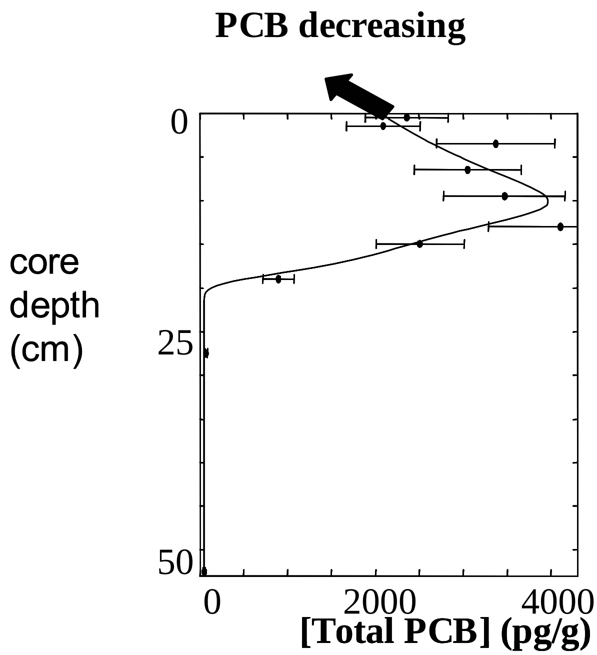


Figure 22: Total PCB concentration profile from a sediment core collected near Texada Is., BC, revealing historical trends for environmental PCBs in the coastal environment (modified from (Johannessen et al. 2008a)

*Table 34: Male northern resident killer whales: using an area-adjusted sediment PCB value for just the disposal sites situated **within Critical Habitat**, the implications of different disposal at sea scenarios for the PCB burden of killer whales are modelled. Four disposal scenarios were modelled, including empirical sediment values using the worst case data from US waters (Puget Sound East Harbor Island) and Canadian waters (Victoria Inner Harbour), as well as average values for Burrard Inlet and the Strait of Georgia. In addition, four test values were used in order to evaluate the efficacy of current decision-making tools: the Disposal at Sea regulatory screening limit for sediment PCBs (CEPA AL), the current analytical Detection Limit for permit proponents in Pacific Region (EC DL), the CCME Interim Sediment Quality Guideline to protect aquatic biota in the marine environment (CCME ISQG), and the Detection Limit for sediment PCBs at the DFO Laboratory for Expertise on Aquatic Analytical Chemistry (LEACA DL).*

Disposal scenario	PCB levels in dredge material or tested (µg·kg^{-1} DW)	NRKW PCB levels before disposal (µg·kg^{-1} LW)	NRKW PCB levels after disposal (µg·kg^{-1} LW)	**% increase in PCB levels in NRKWs**	Increase in population exceeding health thresholds		
					1,300 µg·kg^{-1}	10,000 µg·kg^{-1}	17,000 µg·kg^{-1}
Dredge scenarios							
Puget Sd-East Harbor Is.	935	53,684	69,401	**29.3**	NA	1.7	5.8
Victoria Harbour	370	53,684	59,899	**11.6**	NA	0.9	2.9
Burrard Inlet	31	53,684	54,196	**0.95**	NA	0	0
Strait of Georgia	1.0	53,684	53,693	**0.02**	NA	0	0
Methods/analysis scenarios							
CEPA AL	100	53,684	55,358	**3.12**	NA	0.2	0.6
Current EC DL	40	53,684	54,349	**1.24**	NA	0.2	0.6
CCME ISQG	21.5	53,684	54,038	**0.66**	NA	0	0
LEACA DL	0.119	53,684	53,678	**-0.01**	NA	0	0

Notes: NRKW, Northern Resident killer whales; PCBs, polychlorinated biphenyls; DW, dry weight; LW, lipid weight; NA, not applicable as all members of the population currently exceed health threshold; CEPA AL, Canadian Environmental Protection Act Action Level Low; EC DL, Environment Canada Detection Limit (minimum required for Disposal at Sea); LEACA DL, Laboratory for Expertise in Aquatic Chemical Analysis (DFO) Detection Limit; NA, not applicable as 100% of the population already exceeds the effects threshold.

Table 35: Female northern resident killer whales: using an area-adjusted sediment PCB value for just the disposal sites situated ***within Critical Habitat****, the implications of different disposal at sea scenarios for the PCB burden of killer whales are modelled. Four disposal scenarios were evaluated, including empirical sediment values modelled using the worst case data from US waters (Puget Sound East Harbor Island) and Canadian waters (Victoria Inner Harbour), as well as average values for Burrard Inlet and the Strait of Georgia. In addition, four test values were used in order to evaluate the efficacy of current decision-making tools: the Disposal at Sea regulatory screening limit for sediment PCBs (CEPA AL), the current analytical Detection Limit for permit proponents in Pacific Region (EC DL), the CCME Interim Sediment Quality Guideline to protect aquatic biota in the marine environment (CCME ISQG), and the Detection Limit for sediment PCBs at the DFO Laboratory for Expertise on Aquatic Analytical Chemistry (LEACA DL).*

Disposal scenario	PCB levels in dredge material or tested ($\mu g \cdot kg^{-1}$ DW)	NRKW PCB levels before disposal ($\mu g \cdot kg^{-1}$ LW)	NRKW PCB levels after disposal ($\mu g \cdot kg^{-1}$ LW)	**% increase in PCB levels in NRKWs**	Increase in population exceeding health thresholds		
					1,300 $\mu g \cdot kg^{-1}$	10,000 $\mu g \cdot kg^{-1}$	17,000 $\mu g \cdot kg^{-1}$
Dredge scenarios							
Puget Sd-East Hrbr Is.	935	8,287	10,357	**25.0**	0.4	10.8	5.6
Victoria Harbour	370	8,287	9,106	**9.88**	0.2	4.1	2.0
Burrard Inlet	31	8,287	8,355	**0.81**	0	0	0
Strait of Georgia	1.0	8,287	8,289	**0.01**	0	0	0
Methods/analysis scenarios							
CEPA AL	100	8,287	8,508	**2.66**	.05	1.0	0.5
EC DL minimum	40	8,287	8,375	**1.06**	0	0	0
CCME ISQG	21.5	8,287	8,334	**0.56**	0	0	0
LEACA DL	0.119	8,287	8,287	**-0.01**	0	0	0

Notes: NRKW, Northern Resident killer whales; PCBs, polychlorinated biphenyls; DW, dry weight; LW, lipid weight; NA, not applicable as all members of the population currently exceed health threshold; CEPA AL, Canadian Environmental Protection Act Action Level Low; EC DL, Environment Canada Detection Limit (minimum required for Disposal at Sea); LEACA DL, Laboratory for Expertise in Aquatic Chemical Analysis (DFO) Detection Limit; NA, not applicable as 100% of the population already exceeds the effects threshold.

Table 36: Male southern resident killer whales: using an area-adjusted sediment PCB value for just the disposal sites situated ***within Critical Habitat (three sites)****, the implications of different disposal at sea scenarios for the PCB burden of killer whales are modelled. Four disposal scenarios were evaluated, including empirical sediment values modelled using the worst case data from US waters (Puget Sound East Harbor Island) and Canadian waters (Victoria Inner Harbour), as well as average values for Burrard Inlet and the Strait of Georgia. In addition, four test values were used in order to evaluate the efficacy of current decision-making tools: the Disposal at Sea regulatory screening limit for sediment PCBs (CEPA AL), the current analytical Detection Limit for permit proponents in Pacific Region (EC DL), the CCME Interim Sediment Quality Guideline to protect aquatic biota in the marine environment (CCME ISQG), and the Detection Limit for sediment PCBs at the DFO Laboratory for Expertise on Aquatic Analytical Chemistry (LEACA DL).*

Disposal scenario	PCB levels in dredge material or tested (μg·kg^{-1} DW)	SRKW PCB levels before disposal (μg·kg^{-1} LW)	SRKW PCB levels after disposal (μg·kg^{-1} LW)	**% increase in PCB levels in SRKWs**	Increase in population exceeding health thresholds		
					1,300 μg·kg^{-1}	10,000 μg·kg^{-1}	17,000 μg·kg^{-1}
Dredge scenarios							
Puget Sd-East Hrbr Is.	935	95336	100904	**5.84**	NA	0.07	0.36
Victoria Harbour	370	95336	97537	**2.31**	NA	0.04	0.19
Burrard Inlet	31	95336	95517	**0.19**	NA	0	0
Strait of Georgia	1.0	95336	95339	**0.003**	NA	0	0
Methods/analysis scenarios							
CEPA AL	100	95336	95928	**0.62**	NA	0	0
EC DL minimum	40	95336	95571	**0.25**	NA	0	0
CCME ISQG	21.5	95336	95461	**0.13**	NA	0	0
LEACA DL	0.119	95336	95333	**-0.002**	NA	0	0

Notes: SRKW, Southern Resident killer whales; PCBs, polychlorinated biphenyls; DW, dry weight; LW, lipid weight; NA, not applicable as all members of the population currently exceed health threshold; CEPA AL, Canadian Environmental Protection Act Action Level Low; EC DL, Environment Canada Detection Limit (minimum required for Disposal at Sea); LEACA DL, Laboratory for Expertise in Aquatic Chemical Analysis (DFO) Detection Limit; NA, not applicable as 100% of the population already exceeds the effects threshold. Correction Note: The original PCB concentration predicted in male Southern Resident killer whales (PCB levels before disposal) was corrected by diving it to 6.5 (the antilog of the mean error bias), which was estimated after subtracting the empirical PCB mean concentration (146,300 μg·kg^{-1} LW) in males reported by Ross et al. (2000a) and the mean calculated (62,125 μg·kg^{-1} LW) from the data reported by Krahn et al. (2007) from the predicted PCB concentration in a logarithm format [$\mathbf{10^{\wedge\Sigma(\log PCBpredicted - \log PCBobserved)/n}}$].

Table 37: Female southern resident killer whales: using an area-adjusted sediment PCB value for just the disposal sites situated ***within Critical Habitat (three sites)****, the implications of different disposal at sea scenarios for the PCB burden of killer whales are modelled. Four disposal scenarios were evaluated, including empirical sediment values modelled using the worst case data from US waters (Puget Sound East Harbor Island) and Canadian waters (Victoria Inner Harbour), as well as average values for Burrard Inlet and the Strait of Georgia. In addition, four test values were used in order to evaluate the efficacy of current decision-making tools: the Disposal at Sea regulatory screening limit for sediment PCBs (CEPA AL), the current analytical Detection Limit for permit proponents in Pacific Region (EC DL), the CCME Interim Sediment Quality Guideline to protect aquatic biota in the marine environment (CCME ISQG), and the Detection Limit for sediment PCBs at the DFO Laboratory for Expertise on Aquatic Analytical Chemistry (LEACA DL).*

Disposal scenario	PCB levels in dredge material or tested (µg·kg^{-1} DW)	SRKW PCB levels before disposal (µg·kg^{-1} LW)	SRKW PCB levels after disposal (µg·kg^{-1} LW)	**% increase in PCB levels in SRKWs**	Increase in population exceeding health thresholds		
					1,300 µg·kg^{-1}	10,000 µg·kg^{-1}	17,000 µg·kg^{-1}
Dredge scenarios							
Puget Sd-East Hrbr Is.	935	49930	52849	**5.85**	NA	0.48	1.39
Victoria Harbour	370	49930	51084	**2.31**	NA	0.24	0.71
Burrard Inlet	31	49930	50025	**0.19**	NA	0	0
Strait of Georgia	1.0	49930	49931	**0.003**	NA	0	0
Methods/analysis scenarios							
CEPA AL	100	49930	50241	**0.62**	NA	0	0
EC DL minimum	40	49930	50053	**0.25**	NA	0	0
CCME ISQG	21.5	49930	49995	**0.13**	NA	0	0
LEACA DL	0.119	49930	49929	**-0.002**	NA	0	0

Notes: SRKW, Southern Resident killer whales; PCBs, polychlorinated biphenyls; DW, dry weight; LW, lipid weight; NA, not applicable as all members of the population currently exceed health threshold; CEPA AL, Canadian Environmental Protection Act Action Level Low; EC DL, Environment Canada Detection Limit (minimum required for Disposal at Sea); LEACA DL, Laboratory for Expertise in Aquatic Chemical Analysis (DFO) Detection Limit; NA, not applicable as 100% of the population already exceeds the effects threshold. Correction Note: The original PCB concentration predicted in male Southern Resident killer whales (PCB levels before disposal) was corrected by diving it to 1.9 (the antilog of the mean error bias), which was estimated after subtracting the empirical PCB mean concentration (146,300 µg·kg^{-1} LW) in males reported by Ross et al. (2000a) and the mean calculated (62,125 µg·kg^{-1} LW) from the data reported by Krahn et al. (2007) from the predicted PCB concentration in a logarithm format [$\mathbf{10^{\wedge}}^{\Sigma(\text{log PCBpredicted}-\text{log PCBobserved})/n}$].

4.1.3 Exploring the Implications of Disposal at All Disposal Sites (Inside and Outside of Critical Habitat)

Additional scenarios were evaluated to investigate the impact of disposal at sea into *all sites* within SRKW habitat (11 sites both inside and outside of Critical Habitat; 8 sites within the Strait of Georgia and 3 sites in SRKW Critical Habitat). These were conducted for male SRKWs (Table 38), and female SRKWs (Table 39).The same sediment PCB values tested in the previous Tables were also used in these scenarios. Since both disposal sites in northern resident killer whale habitat fall within Critical Habitat, a similar exercise was not carried out for this population, and the reader is referred back to section 4.1.2.

To evaluate the relative 'vulnerability' of non-Critical Habitat-based disposal sites, we conducted a comparative modeling exercise whereby disposal sites both within Critical Habitat and disposal sites both within and outside of Critical Habitat were subjected to the same eight scenarios evaluated above (Tables 38 – 39; Figure 23). Despite the majority of disposal sites being situated outside of Critical Habitat, their relative importance is not proportionate. For example, the increase in PCB levels in killer whales predicted from disposal in the Critical Habitat sites is 5.84% compared to 7.97% for both Critical Habitat and outside of Critical Habitat sites. This largely reflects the greater amount of time spent by southern resident killer whales in Critical Habitat than in areas outside of Critical Habitat such as the Strait of Georgia.

The relationships between dredge material PCB content and predicted PCB levels in resident killer whales is depicted for the southern resident killer whale disposal sites (Critical Habitat plus non-Critical Habitat sites; Figure 23).

For disposal at sites both within Critical Habitat and outside of Critical Habitat, the formula for southern resident killer whale becomes:

(3) $[PCB]_{SRKW} = 16.818 * x + 53676$ (Southern resident killer whales)

where 'x' represents the PCB concentration in dry weight ($\mu g \cdot kg^{-1}$) in dredge materials being deposited at all disposal sites.

Improved approaches to such an exercise would incorporate a sediment transport model which captures the spatial, temporal and constitutent physical and chemical features of disposal sites, and better predicts the resulting PCB concentration in sediments. A complementary means of assessing the implications of disposal of materials would entail a comparison of ambient sediment PCB concentrations with proposed dredge materials. PCB values in dredge materials that are lower than the proposed receiving ambient sediment PCB concentrations would presumably dilute the disposal site with cleaner sediments and not heighten the delivery of PCBs to killer whales. PCBs concentrations in dredge materials that are the same as the receiving ambient sediment PCB concentrations would presumably elicit no effect on PCB delivery to killer whales. Finally, PCB concentrations in dredge materials that exceed the ambient sediment PCB concentrations would likely increase the risk of heightened delivery of PCBs to killer whales, as described in the above equations and Figure 23. In the latter case, alternative disposal strategies are recommended.

*Table 38: Male southern resident killer whales: using an area-adjusted sediment PCB value for just the disposal sites situated **within Critical Habitat and in the Strait of Georgia (11 sites)**, the implications of different disposal at sea scenarios for the PCB burden of killer whales are modelled. Four disposal scenarios were evaluated, including empirical sediment values modelled using the worst case data from US waters (Puget Sound East Harbor Island) and Canadian waters (Victoria Inner Harbour), as well as average values for Burrard Inlet and the Strait of Georgia. In addition, four test values were used in order to evaluate the efficacy of current decision-making tools: the Disposal at Sea regulatory screening limit for sediment PCBs (CEPA AL), the current analytical Detection Limit for permit proponents in Pacific Region (EC DL), the CCME Interim Sediment Quality Guideline to protect aquatic biota in the marine environment (CCME ISQG), and the Detection Limit for sediment PCBs at the DFO Laboratory for Expertise on Aquatic Analytical Chemistry (LEACA DL).*

Disposal scenario	PCB levels in dredge material or tested (μg·kg^{-1} DW)	SRKW PCB levels before disposal (μg·kg^{-1} LW)	SRKW PCB levels after disposal (μg·kg^{-1} LW)	**% increase in PCB levels in SRKWs**	Increase in population exceeding health thresholds		
					1,300 μg·kg^{-1}	10,000 μg·kg^{-1}	17,000 μg·kg^{-1}
Dredge scenarios							
Puget Sd-East Hrbr Is.	935	95336	102932	**7.97**	NA	0.10	0.53
Victoria Harbour	370	95336	98338	**3.15**	NA	0.04	0.19
Burrard Inlet	31	95336	95582	**0.26**	NA	0	0
Strait of Georgia	1.0	95336	95338	**0.003**	NA	0	0
Methods/analysis scenarios							
CEPA AL	100	95336	96143	**0.85**	NA	0	0
EC DL minimum	40	95336	95655	**0.34**	NA	0	0
CCME ISQG	21.5	95336	95505	**0.18**	NA	0	0
LEACA DL	0.119	95336	95331	**-0.005**	NA	0	0

Notes: SRKW, Southern Resident killer whales; PCBs, polychlorinated biphenyls; DW, dry weight; LW, lipid weight; NA, not applicable as all members of the population currently exceed health threshold; CEPA AL, Canadian Environmental Protection Act Action Level Low; EC DL, Environment Canada Detection Limit (minimum required for Disposal at Sea); LEACA DL, Laboratory for Expertise in Aquatic Chemical Analysis (DFO) Detection Limit; NA, not applicable as 100% of the population already exceeds the effects threshold. Correction Note: The original PCB concentration predicted in male Southern Resident killer whales (PCB levels before disposal) was corrected by diving it to 6.5 (the antilog of the mean error bias), which was estimated after subtracting the empirical PCB mean concentration (146,300 μg·kg^{-1} LW) in males reported by Ross et al. (2000a) and the mean calculated (62,125 μg·kg^{-1} LW) from the data reported by Krahn et al. (2007) from the predicted PCB concentration in a logarithm format [$\mathbf{10}$^$^{\Sigma(\log PCBpredicted - \log PCBobserved)/n}$].

*Table 39**:** Female southern resident killer whales: using an area-adjusted sediment PCB value for just the disposal sites situated **within Critical Habitat and in the Strait of Georgia (11 sites)**, the implications of different disposal at sea scenarios for the PCB burden of killer whales are modelled. Four disposal scenarios were evaluated, including empirical sediment values modelled using the worst case data from US waters (Puget Sound East Harbor Island) and Canadian waters (Victoria Inner Harbour), as well as average values for Burrard Inlet and the Strait of Georgia. In addition, four test values were used in order to evaluate the efficacy of current decision-making tools: the Disposal at Sea regulatory screening limit for sediment PCBs (CEPA AL), the current analytical Detection Limit for permit proponents in Pacific Region (EC DL), the CCME Interim Sediment Quality Guideline to protect aquatic biota in the marine environment (CCME ISQG), and the Detection Limit for sediment PCBs at the DFO Laboratory for Expertise on Aquatic Analytical Chemistry (LEACA DL).*

Disposal scenario	PCB levels in dredge material or tested (µg·kg^{-1} DW)	SRKW PCB levels before disposal (µg·kg^{-1} LW)	SRKW PCB levels after disposal (µg·kg^{-1} LW)	**% increase in PCB levels in SRKWs**	Increase in population exceeding health thresholds		
					1,300 µg·kg^{-1}	10,000 µg·kg^{-1}	17,000 µg·kg^{-1}
Dredge scenarios							
Puget Sd-East Hrbr Is.	935	49930	53912	**7.97**	NA	0.70	2.05
Victoria Harbour	370	49930	51504	**3.15**	NA	0.24	0.71
Burrard Inlet	31	49930	50059	**0.26**	NA	0	0
Strait of Georgia	1.0	49930	49931	**0.003**	NA	0	0
Methods/analysis scenarios							
CEPA AL	100	49930	50353	**0.85**	NA	0	0
EC DL minimum	40	49930	50098	**0.34**	NA	0	0
CCME ISQG	21.5	49930	50019	**0.18**	NA	0	0
LEACA DL	0.119	49930	49928	**-0.005**	NA	0	0

Notes: SRKW, Southern Resident killer whales; PCBs, polychlorinated biphenyls; DW, dry weight; LW, lipid weight; NA, not applicable as all members of the population currently exceed health threshold; CEPA AL, Canadian Environmental Protection Act Action Level Low; EC DL, Environment Canada Detection Limit (minimum required for Disposal at Sea); LEACA DL, Laboratory for Expertise in Aquatic Chemical Analysis (DFO) Detection Limit; NA, not applicable as 100% of the population already exceeds the effects threshold. Correction Note: The original PCB concentration predicted in male Southern Resident killer whales (PCB levels before disposal) was corrected by diving it to 1.9 (the antilog of the mean error bias), which was estimated after subtracting the empirical PCB mean concentration (146,300 µg·kg^{-1} LW) in males reported by Ross et al. (2000a) and the mean calculated (62,125 µg·kg^{-1} LW) from the data reported by Krahn et al. (2007) from the predicted PCB concentration in a logarithm format [$10^{\wedge\Sigma(\log PCBpredicted - \log PCBobserved)/n}$].

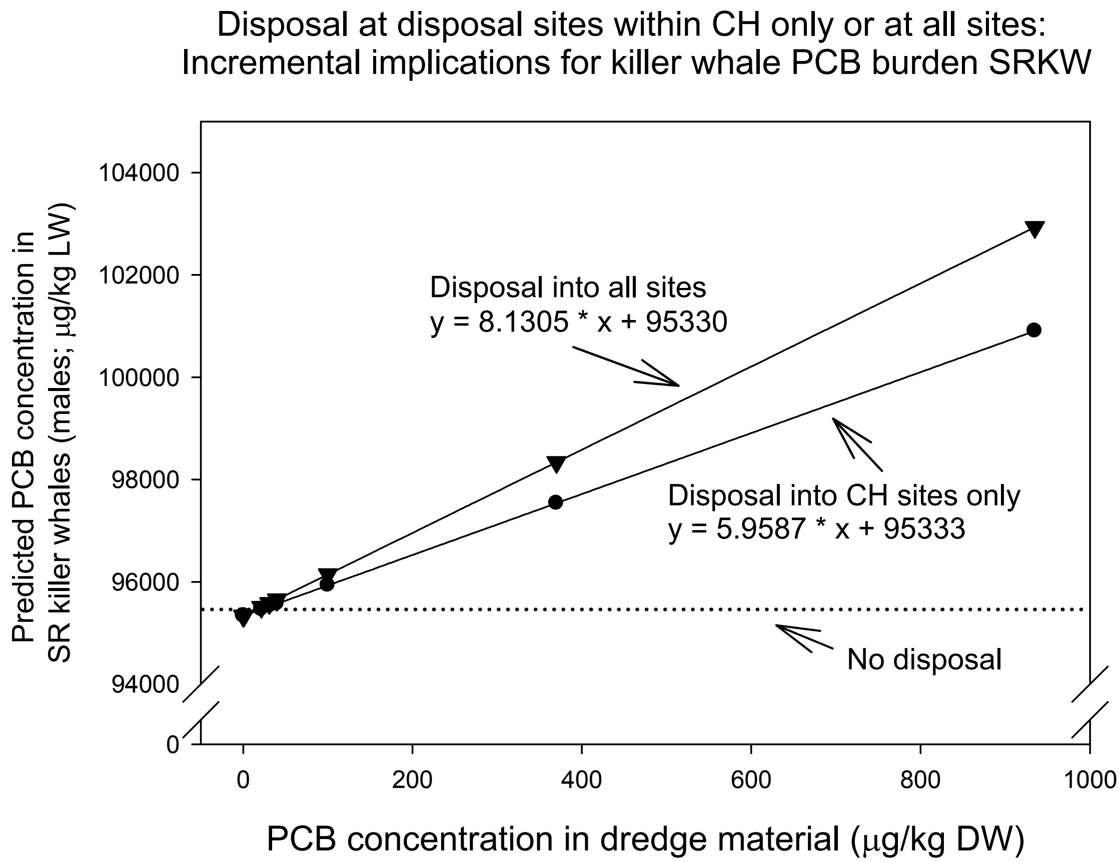


Figure 23: While both disposal sites in NRKW habitat are situated inside their Critical Habitat, the situation is different for SRKW habitat. 3 of 11 disposal sites are found within CH, leaving 8 outside of CH but within general SRKW habitat. This modeling evaluates the relative importance of the disposal sites within CH vs those outside of CH. The same eight scenarios used in Figure 22 are used here, but the sediment PCB values tested are modeled as replacing ambient within just the 3 disposal sites within CH (the lower line) or all disposal sites (the upper line; all 11 sites). The equations of these lines can be used to evaluate the implications of any disposal scenario when one is provided with a dredgeate PCB concentration. The dotted line represents the predicted PCB concentration in male SRKWs if there were no disposal activities, with the result being that ambient PCB sediment concentrations would result in the given value. The model assumes steady state and no influence of sedimentation over time.

4.1.4 Assessing Utility of Decision Criteria, Laboratory Methods and Impacts of Disposal

Killer whales are exceptionally vulnerable to PCB contamination as a result of their high trophic level, long lifespan, and inability to metabolize many of the congeners. Since PCBs partition readily into lipids and magnify in food webs, even relatively low concentrations in sediments or in lower trophic levels can lead to high concentrations in killer whales. The model, which attempts to describe the PCB concentration relationships between sediment, water and the food-web, is therefore suited to address how PCB concentrations in the food-web will respond to changes in concentrations in the sediments. The model does not address the sources of PCB contamination to the sediment. For example, it does not reveal how atmospheric sources of PCBs have contributed to current PCB concentrations in the sediments. In case of sediment disposal, it is clear that the PCBs are directly introduced into the system in association with sediment. The latter makes it unnecessary to develop a larger PCB environmental model. While direct measurements of tissue biopsies obtained from killer whales or tissue samples obtained from their prey provide insight into the nature of biomagnification, these do not enable an assessment of sources, disposal operations, or sediment quality in killer whale habitat.

The food web bioaccumulation model provides a way to characterize PCB uptake, accumulation, and magnification in killer whale food webs, in the context of a trophic system and its apex feeders responding to a system whose PCB concentrations are controlled by varying sediment PCB concentrations. This model also allows us to evaluate the current approaches to either measuring PCBs or assessing their concentrations under the terms of current disposal practices.

In this section, we use hypothetical killer whales to evaluate the implications of four dredge scenarios and four method/decision criteria. A hypothetical killer whale in this section is an animal that spends its entire life within a single area where PCB concentrations relate to a given static sediment PCB concentration. This limits area- and life history-based variation, and enables a more focused and sensitive evaluation of our eight test questions.

The PCB concentration in Chinook salmon, assuming 100% of their time is spent in the areas included in the model, exceeded the tissue residue guideline for fish-eating wildlife (50 $\mu g \cdot kg^{-1}$) and the Chinook concentration for 95% of the killer whale population to fall below the toxicity threshold of 8 $\mu g \cdot kg^{-1}$ for the ISQG and Action Level Low tested. Realistic scenarios based on the total BSAF values per annual habitat distribution showed that Lower Fraser River and South Thompson Chinook salmon also exceeded tissue residue guidelines for the ISQG and Action Level Low tested. For example, both Chinook salmon stocks exhibited PCB concentrations above tissue residue guidelines when the CEPA Action Level Low was tested.

This scenario was evaluated for male and female resident killer whales (Tables 40 and 41). Results generally reveal that current PCB concentrations in NE Pacific sediments are compatible with contaminated killer whales, in which the majority, if not all, of the population would be exposed to health risks (Tables 40 and 41; Figure 24). In addition, current decision criteria under CEPA or CCME do not protect killer whales (nor were they designed to), with these criteria benefiting from re-evaluation in the context of high trophic level wildlife. Improved analytical methods will enable a better assessment of implications for killer whales (LEACA Detection Limit vs EC Detection Limit minimum currently used). Of note is that only 1 of the 40 sediment sites for which PCB data are currently available for the Strait of Georgia (Grant et al, in prep) have PCB values above the minimum Detection Limit currently used under CEPA Disposal at Sea review in the Pacific Region. By using High Resolution GC/MS (as with LEACA), 40 out of 41 sites would have detectable PCB concentrations, thereby enabling a more precise assessment of PCB fate in the killer whale food web, and supporting a more reliable risk management paradigm.

Table 40: Hypothetical male resident killer whales (i.e., spending 100% time in seven habitat areas): the implications of PCB ambient concentration scenarios for the PCB burden of killer whales are modelled (i.e., average of seven habitat areas). Four scenarios were evaluated, including empirical sediment values modelled using the worst case data from US waters (Puget Sound East Harbor Island) and Canadian waters (Victoria Inner Harbour), as well as average values for Burrard Inlet and the Strait of Georgia. In addition, four test values were used in order to evaluate the efficacy of current decision-making tools: the Disposal at Sea regulatory screening limit for sediment PCBs (CEPA AL), the required minimum analytical Detection Limit for permit proponents (EC DL), the CCME Interim Sediment Quality Guideline to protect aquatic biota in the marine environment (CCME ISQG), and the Detection Limit for sediment PCBs at the DFO Laboratory for Expertise on Aquatic Analytical Chemistry (LEACA DL).

Disposal Scenario	PCB levels in dredge material or tested ($\mu g \cdot kg^{-1}$ DW)	Male KW PCB levels* ($\mu g \cdot kg^{-1}$ LW)	% exceedance of health threshold		
			1300 $\mu g \cdot kg^{-1}$	10000 $\mu g \cdot kg^{-1}$	17000 $\mu g \cdot kg^{-1}$
Dredge Scenarios					
Puget Sd-East Harbor Is.	935	72316386	100	100	100
Victoria Harbour	370	28617180	100	100	100
Burrard Inlet	31	2389921	100	100	100
Strait of Georgia	1	77344	100	99	95
Methods/analysis scenarios					
CEPA AL	100	7734373	100	100	100
EC-DL minimum	40	3093749	100	100	100
CCME ISQG	21.5	1662890	100	100	100
LEACA-DL	0.119	9204	99	30	20

Notes: KW, killer whale; PCBs, polychlorinated biphenyls; DW, dry weight; LW, lipid weigh; CEPA AL, Canadian Environmental Protection Act Action Level Low; EC DL, the current analytical Detection Limit for permit proponents in Pacific Region (EC DL); LEACA DL, Laboratory for Expertise in Aquatic Chemical Analysis (DFO) Detection Limit. *Average PCB concentration of habitat areas include the killer whale critical habitats (Northern resident killer whale critical habitat, Southern Resident killer whale critical habitat-Canada, Southern resident killer whale critical habitat-Juan de Fuca, and Southern resident killer whale critical habitat-Puget Sound), as well as Outer Coast, Queen Charlotte Strait and Strait of Georgia.

Table 41: Hypothetical female resident killer whales (i.e., spending 100% time in seven habitat areas): the implications of PCB ambient concentration scenarios for the PCB burden of killer whales are modelled (i.e., average of seven habitat areas). Four scenarios were evaluated, including empirical sediment values modelled using the worst case data from US waters (Puget Sound East Harbor Island) and Canadian waters (Victoria Inner Harbour), as well as average values for Burrard Inlet and the Strait of Georgia. In addition, four test values were used in order to evaluate the efficacy of current decision-making tools: the Disposal at Sea regulatory screening limit for sediment PCBs (CEPA AL), the current analytical Detection Limit for permit proponents in Pacific Region (EC DL), the CCME Interim Sediment Quality Guideline to protect aquatic biota in the marine environment (CCME ISQG), and the Detection Limit for sediment PCBs at the DFO Laboratory for Expertise on Aquatic Analytical Chemistry (LEACA DL).

Disposal Scenario	PCB levels in dredge material or tested ($\mu g \cdot kg^{-1}$ DW)	Female KW PCB levels* ($\mu g \cdot kg^{-1}$ LW)	% exceedance of health threshold		
			1300 $\mu g \cdot kg^{-1}$	10000 $\mu g \cdot kg^{-1}$	17000 $\mu g \cdot kg^{-1}$
Dredge Scenarios					
Puget Sd-East Harbor Is.	935	11059782	100	100	100
Victoria Harbour	370	4376598	100	100	100
Burrard Inlet	31	365505	100	100	100
Starit of Georgia	1	11829	99.8	30	25
Methods/analysis scenarios					
CEPA AL	100	1182864	100	100	100
EC-DL minimum	40	473146	100	100	100
CCME ISQG	21.5	254316	100	100	99.9
LEACA-DL	0.119	1408	71	1	0.10

Notes: KW, killer whale; PCBs, polychlorinated biphenyls; DW, dry weight; LW, lipid weigh; CEPA AL, Canadian Environmental Protection Act Action Level Low; EC DL, Environment Canada Detection Limit (minimum required for Disposal at Sea); LEACA DL, Laboratory for Expertise in Aquatic Chemical Analysis (DFO) Detection Limit. *Average PCB concentration of habitat areas include the killer whale critical habitats (Northern resident killer whale critical habitat, Southern Resident killer whale critical habitat-Canada, Southern resident killer whale critical habitat-Juan de Fuca, and Southern resident killer whale critical habitat-Puget Sound), as well as Outer Coast, Queen Charlotte Strait and Strait of Georgia.

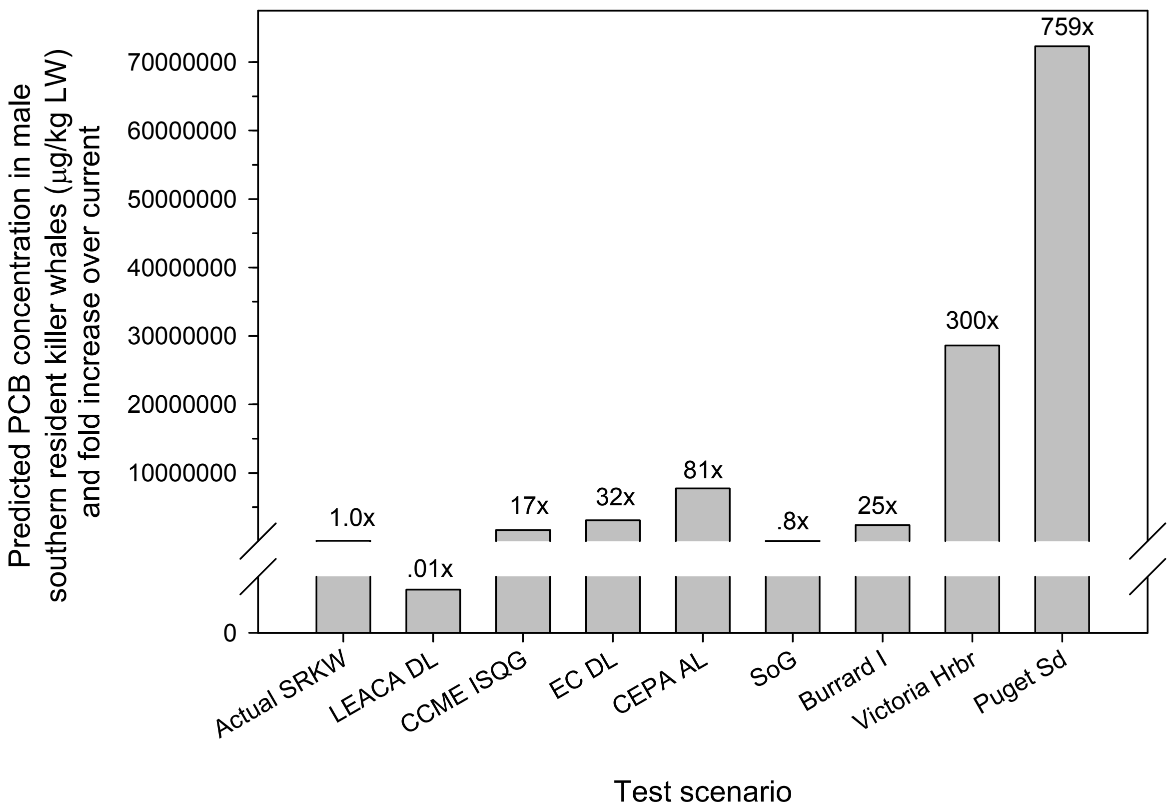


Figure 24: Evaluating guidelines, laboratory practices and habitat quality. We used the model to predict what the PCB burden in a hypothetical killer whale would become if it lived in an environment where the sediments had PCB concentrations equivalent to one of eight scenarios: either of two guidelines (CEPA Action Level Low; CCME Interim Sediment Quality Guideline to protect aquatic biota), two analytical detection limits (EC Detection Limit as per USEPA Method 8080), or measured sediment PCB values from four different sites (Strait of Georgia average, Burrard Inlet average, and Victoria Inner Harbour from Grant et al (2010), and Puget Sound (East Harbor Island)). The actual southern resident killer whale (SRKW) is the model-based predicted average for males. Numbers above each bar represent the fold-difference over the actual SRKW predicted value. See Table 40 for additional details.

4.1.5 Using Backward Calculations to Derive Target Sediment PCB Levels to Protect Killer Whales

In an effort to generate new Sediment Quality Guidelines that are protective of killer whales, the backward application of the BSAF model was carried out. Under this premise, proposed target sediment concentrations involving realistic scenarios of habitat distribution to protect 95% of resident killer whales (Table 42). The overall mean target sediment levels ranged from 0.012 to 0.20 μg·kg^{-1} dry weight. Similar sediment target values were observed under the assumption that resident killer whales spend 100% of their time in model areas (i.e., a hypothetical resident killer whale with 100% presence in model areas) (Table 43).

Table 42: Derivation of target Sediment Quality Guidelines (SQGs) to protect 95% of the population of northern and southern resident killer whales using realistic distribution in model areas and a diet of 96% Chinook salmon, 2% halibut, and 2% sable fish.

	Toxic Effect Thresholds used		
	Harbour seal PCB toxicity (17 ppm)[1]	**Bottlenose dolphin PCB toxicity (10 ppm)[2]**	**Revised Harbour seal PCB toxicity (1.3 ppm)[3]**
Resident killer whale population	**Target SQG (μg·kg^{-1} dry weight)**	**Target SQG (μg·kg^{-1} dry weight)**	**Target SQG (μg·kg^{-1} dry weight)**
NRKW male	0.050	0.030	0.004
NRKW female	0.320	0.200	0.024
SRKW male	0.040	0.020	0.003
SRKW female	0.250	0.150	0.020
Average	**0.200**	**0.100**	**0.012**
SD	**0.140**	**0.080**	**0.010**

1- (Ross et al. 1996c), 2- (Hall et al. 2006), 3- (Mos et al. 2010).

Table 43: Derivation of target Sediment Quality Guidelines (SQGs) to protect 95% of resident killer whales assuming 100% presence in model areas, with a diet of 96% Chinook salmon, 2% halibut, and 2% sable fish.

	Toxic Effect Thresholds used		
	Harbour seal PCB toxicity (17 ppm)[1]	**Bottlenose dolphin PCB toxicity (10 ppm)[2]**	**Revised Harbour seal PCB toxicity (1.3 ppm)[3]**
	Target SQG (μg·kg^{-1} dry weight)	**Target SQG (μg·kg^{-1} dry weight)**	**Target SQG (μg·kg^{-1} dry weight)**
Hypothetical male resident killer whale male (100% time in model areas)	0.050	0.030	0.004
Hypothetical female resident killer whale (100% time in model areas)	0.350	0.210	0.030
Average	**0.200**	**0.120**	**0.015**
SD	**0.150**	**0.090**	**0.010**

1- (Ross et al. 1996c), 2- (Hall et al. 2006), 3- (Mos et al. 2010).

4.2 MODEL PERFORMANCE

As described in Section 3.2, the ability of the model to estimate PCB congener concentrations in biota was tested by comparing predicted concentrations in biota (i.e., Johnstone Strait Chinook salmon and male northern resident killer whale) to available empirical values. Model predicted and empirical PCB congeners included are shown in Figures 23 and 24. The model bias (MB) geometric mean ± log MB (SD) was 1.30 ± 0.31 for Johnstone Strait Chinook salmon, and 1.23 ± 0.36 for male northern resident killer whales, which were close to one (MB ≈1), as in both cases the over prediction was small or negligible. These comparisons are an indication that the predicted concentrations of PCBs are similar to or within the range of observed PCB concentrations in Chinook salmon and resident killer whales (i.e., predicted mean concentration values are close to the observed values). Figures 25 and 26 also illustrate that congener patterns of PCBs in Chinook and killer whales are reasonably well reproduced by the model when compared against the empirical profiles found for both species. This is supported by the small uncertainty (i.e., error bias) of the model pointed out above (SD_{MB} = 0.31 for Chinook salmon; and SD_{MB} = 0.36 for male NRKW). Predicted BSAF values in Chinook salmon and male killer whales were also very similar to empirical data observed in both species in Northern resident killer whale Critical Habitat (Figure 27). The model, therefore, appears to produce little systematic over- or under-estimation of PCB congener concentrations.

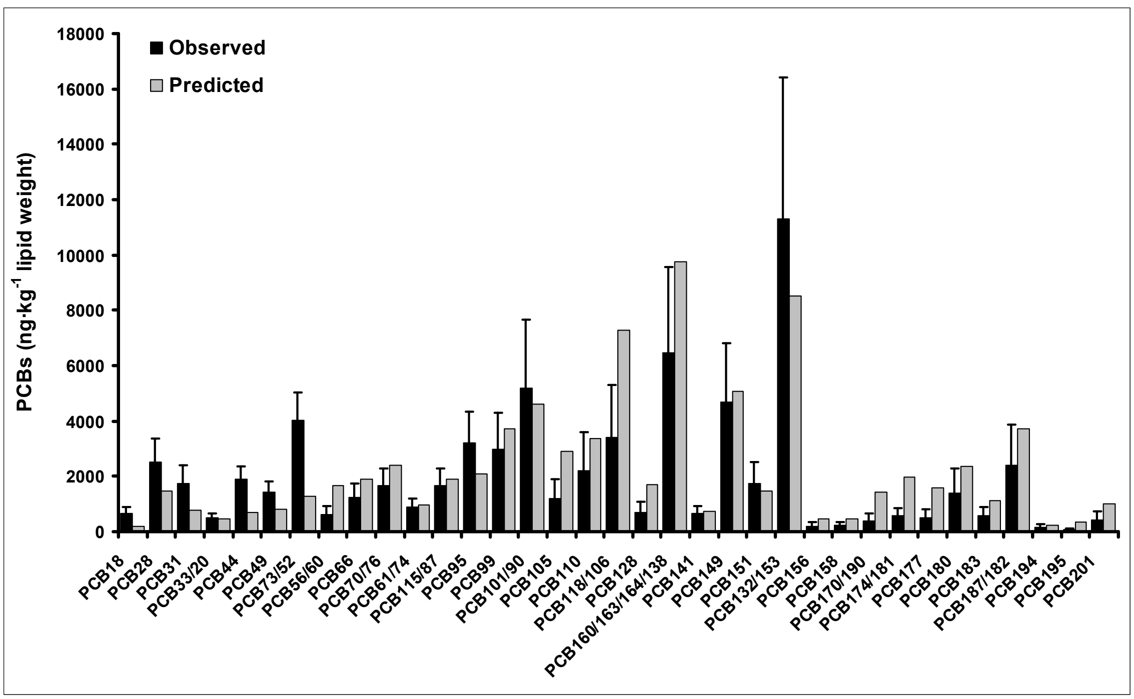


Figure 25: Model predicted and observed concentrations for specific PCB congeners (ng·kg^{-1} lipid weight organism) of approximately 35 PCB congeners in Chinook salmon in the northern resident killer whale Critical Habitat. Error bars is the standard deviation for observed concentrations.

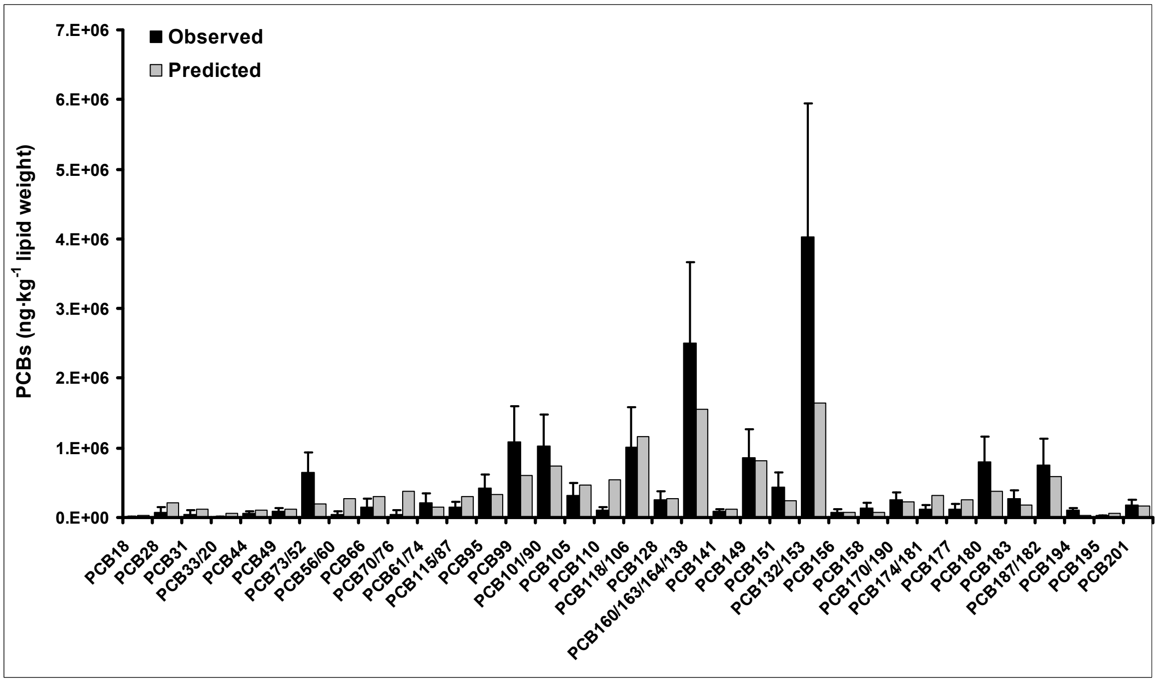


Figure 26: Model predicted and observed concentrations for specific PCB congeners (ng·kg^{-1} lipid weight organism) of approximately 35 PCB congeners in male NRKW in the northern resident killer whale Critical Habitat. Error bars is the standard deviation for observed concentrations.

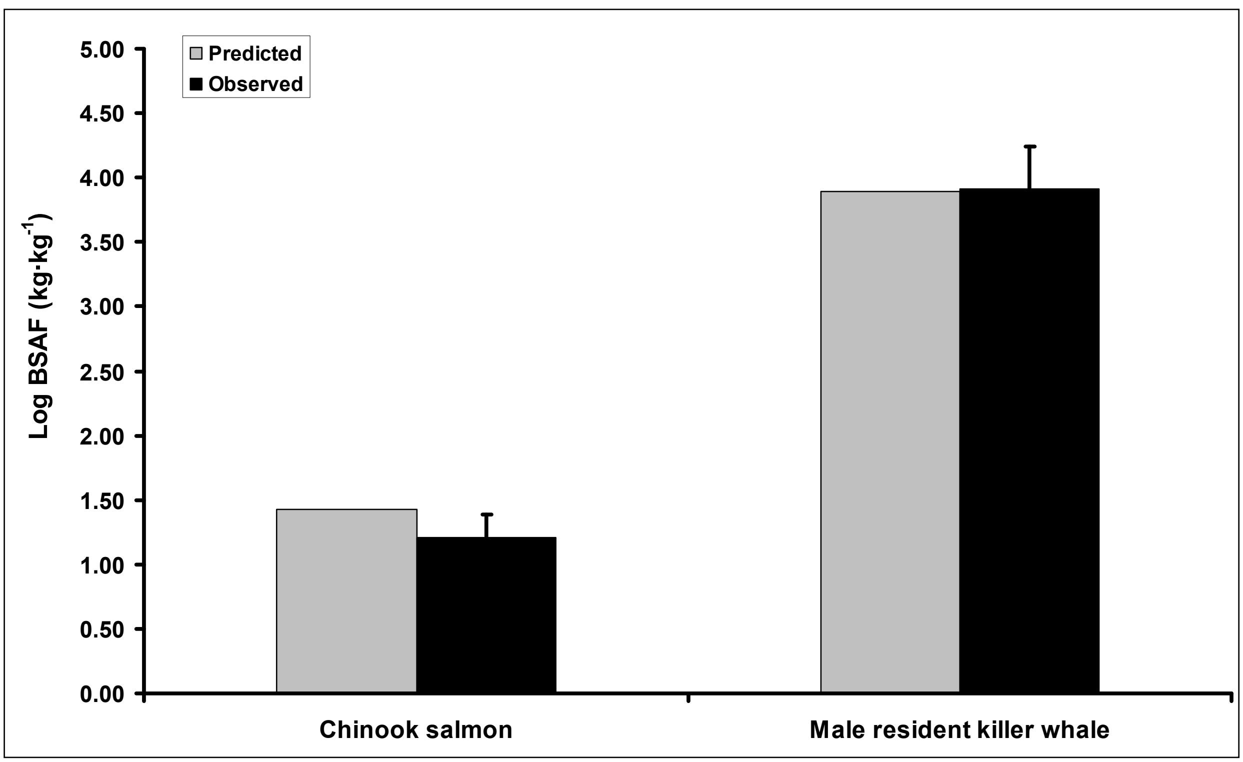


Figure 27: Predicted Biota-Sediment Accumulation Factor (BSAF) values in Chinook salmon and male killer whale were similar to empirical data observed in both species in Northern resident killer whale Critical Habitat. Error bars are standard deviations of observed values.

4.3 UNCERTAINTY ANALYSIS

The uncertainty analysis explores the spread or deviations from the mean concentration in biota (e.g., Chinook salmon and killer whales) calculated from the error model bias (BSAF) and standard deviations of the empirical sediment data to measure the spread of the data, and therefore, to look how far the observations are from the mean. The empirical PCB sediment data has a wide range of values (large spatial variation), as shown in Table 44. The mean of the log PCB sediment concentrations ranged from -3.62 ± 0.74 mg·kg^{-1} dry weight in NRKW Critical Habitat to 4.55 ± 1.17 mg·kg^{-1} dry weight for the SRKW Critical Habitat in the USA (Puget Sound). Table 44 shows that uncertainty values in the model in terms of the standard deviation SD_{CB} of log C_B (i.e., the geometric mean concentration of biota) ranged from 0.32 to 1.31 for Chinook salmon (i.e., SD of log C_{fish}), and from 0.42 to 1.34 for male resident killer whales (i.e., SD log $C_{KW\text{-}male}$). This portrays that over- and under-estimations of PCB congeners for specific species can be considerable even if the predicted mean concentration values are close to the observed values. The standard deviations can be viewed as the uncertainty in the BSAF model estimates.

Table 44: Uncertainty values showing the standard deviations of the Log PCB concentrations for Chinook salmon and male resident killer whales in the model areas.

	Mean log C_S (mg·kg^{-1} dry weight)	SD of log C_S	SD of log $BSAF_{fish}$	SD of log C_{fish} (mg·kg^{-1} wet weight)	SD of Log $BSAF_{KW\text{-}male}$	SD log $C_{KW\text{-}male}$ (mg·kg^{-1} wet weight)
NRKW Critical Habitat (CH)	-3.62	0.74	0.18	0.77	0.33	0.81
Queen Charlotte Strait	-3.63	1.08	0.18	1.09	0.33	1.13
Outer coast	-3.63	1.08	0.18	1.09	0.33	1.13
Strait of Georgia	-3.58	0.51	0.18	0.54	0.33	0.60
SRKW CH in Canada	-3.68	0.27	0.18	0.32	0.33	0.42
SRKW CH in USA (Puget Sound)	-1.85	1.23	0.18	1.24	0.33	1.30
SRKW CH in USA summer core & Juan de Fuca Strait)	-2.53	0.40	0.18	0.47	0.33	0.54

4.4 SENSITIVITY ANALYSIS

4.4.1 Evaluating the Effects of Changing the Resident Killer Whale Diet on Model Outcomes

It was assumed that resident killer whales eat 96% Chinook salmon, 2% halibut, and 2% sablefish. However, these percentages fluctuate during the year. We also assumed that the Chinook salmon that resident killer whales consume were composed only of South Thompson and Lower Fraser River stocks. Again, this is not the case in reality and further modeling efforts could attempt to include more of the stocks they eat. Studies that target these Chinook stocks and test for PCB concentrations would be very beneficial for a food web exercise such as this, as it would provide the PCB concentrations of the actually stocks that resident killer whales primarily consume, which could improve predictions of PCB concentrations in killer whales.

Further model scenarios were conducted that incorporated more species in the NRKW diet (i.e., the addition of chum and coho salmon, lingcod, squid, and dover sole) while the killer whales are in Critical Habitat. No significant differences were observed in the outcomes (i.e., biota concentrations) by changing the killer whale diet in the NRKW Critical Habitat. Under this premise, it was assumed that the predicted PCB concentrations in biota in the coastal food web models for all habitat areas are not significantly affected when making changes in diet composition as all the coastal models use the same coastal food web. On the contrary, when changing the diet in the oceanic food web (i.e., outer coast model), the PCB congener concentrations predicted in killer whales using the previous diet is significantly higher than the PCB congener concentrations in killer whales when using the new diet. These results are expanded upon below.

NRKW Critical Habitat

Males: No significant differences (t-test, t = 0.5682; $p = 0.5716$) were found between the PCB congener concentrations predicted in the coastal food web model for the NRKW Critical Habitat using the previous resident killer whale diet composition (96% Chinook salmon; 2% halibut; and 2% sablefish), and the concentration predicted using the new diet composition (i.e., 70% Chinook salmon; 10% chum salmon; 5% coho salmon; 3% halibut; 3% sablefish; 3% lingcod; 3% dover sole; and 3% gonatid squid). When comparing the PCB congener concentrations using the previous and new diet, the relative frequencies of the outcomes were similar as shown in Figure 28.

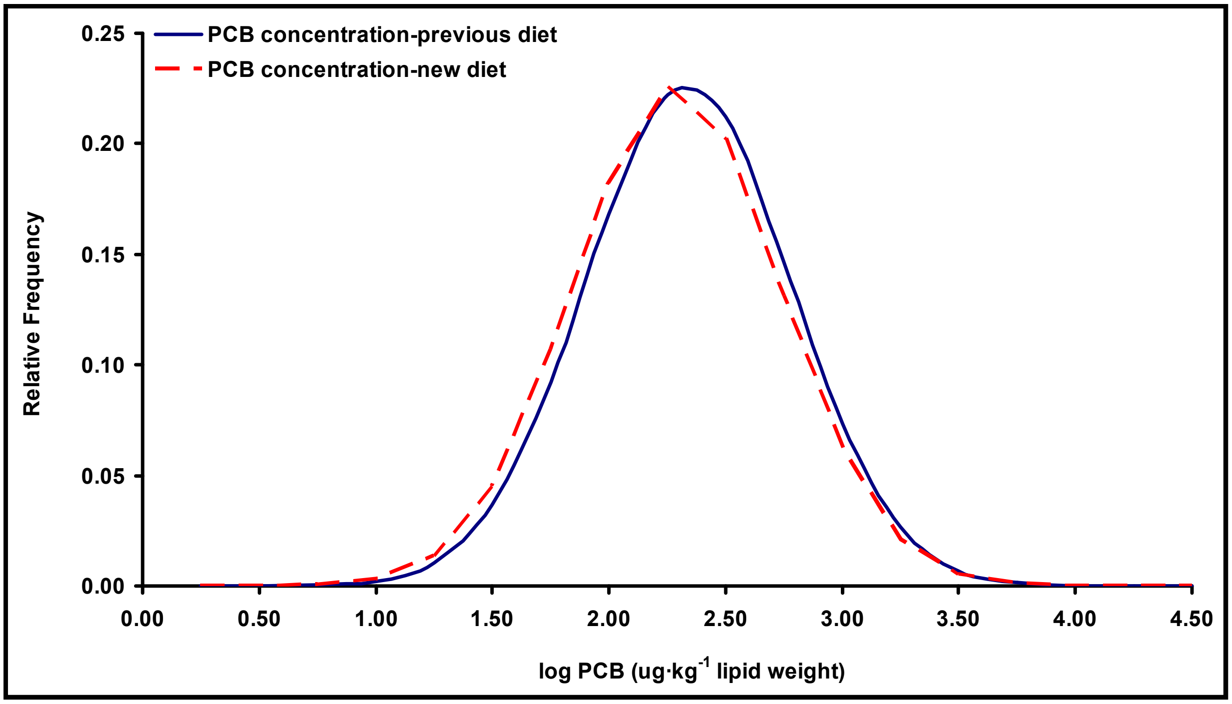


Figure 28: Normal probability density curves showing the comparisons of PCB concentrations predicted in male killer whale with the coastal PCB food web bioaccumulation model using initial diet composition (black line) versus new diet (red dashed line) in northern resident killer whale Critical Habitat.

Females: No significant differences (t-test, t = 0.5831; p = 0.5616) were found between the PCB congener concentrations predicted in the coastal food web model for the NRKW Critical Habitat using the previous resident killer whale diet composition (96% Chinook salmon; 2% halibut; and 2% sablefish), and the concentration predicted using the new diet composition (i.e., 70% Chinook salmon; 10% chum salmon; 5% coho salmon; 3% halibut; 3% sablefish; 3% lingcod; 3% dover sole; and 3% gonatid squid). When comparing the PCB congener concentrations using the previous and new diet, the relative frequencies of the outcomes were similar as shown in Figure 29.

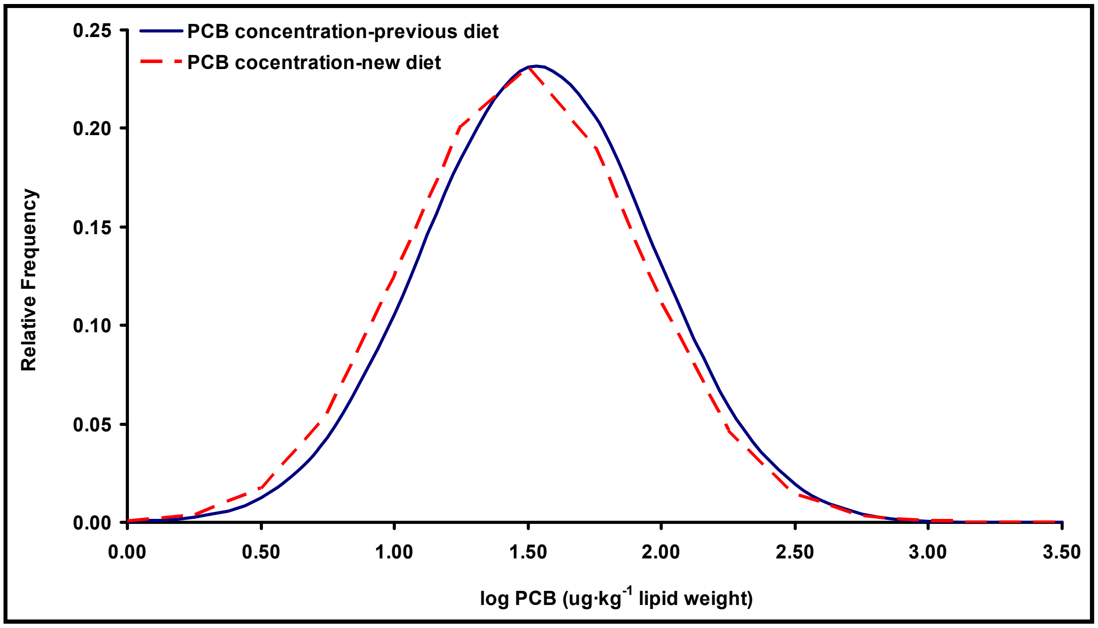


Figure 29: Normal probability density curves showing the comparisons of PCB concentrations predicted in female killer whale with the coastal PCB food web bioaccumulation model using initial diet composition (black line) versus new diet (red dashed line) in northern resident killer whale critical habitat.

Outer Coast Habitat

Males: A significant difference (t-test, t = 3.0781; p = 0.003) was found between the PCB congener concentrations predicted in the oceanic food web model for the Outer coast habitat using the previous resident killer whale diet composition (96% Chinook salmon; 2% halibut; and 2% sablefish), and the concentration predicted using the new diet composition (i.e., 70% Chinook salmon; 10% chum salmon; 5% coho salmon; 3% halibut; 3% sablefish; 3% lingcod; 3% dover sole; and 3% gonatid squid). When comparing the PCB congener concentrations of the two diets, the concentration of the previous diet was significantly higher than the new diet (Figure 30).

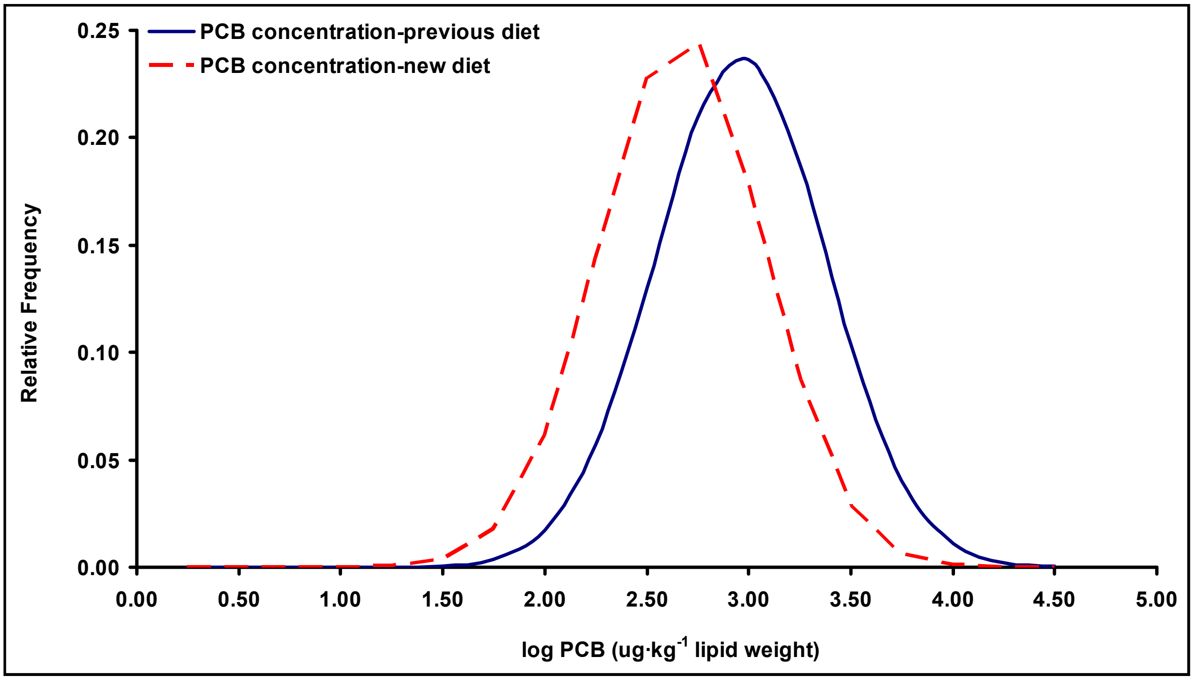


Figure 30: Normal probability density curves showing the comparisons of PCB concentrations predicted in male killer whale with the coastal PCB food web bioaccumulation model using initial diet composition versus new diet in Outer coast habitat.

Females: A significant difference (t-test, t = 3.1311; *p* = 0.0025) was found between the PCB congener concentrations predicted in the oceanic food web model for the Outer coast habitat using the previous resident killer whale diet composition (96% Chinook salmon; 2% halibut; and, 2% sablefish), and the concentration predicted using the new diet composition (i.e., 70% Chinook salmon; 10% chum salmon; 5% coho salmon; 3% halibut; 3% sablefish; 3% lingcod; 3% dover sole; and 3% gonatid squid). When comparing the PCB congener concentrations of the two diets, the concentration of the previous diet was significantly higher than the new diet (Figure 31).

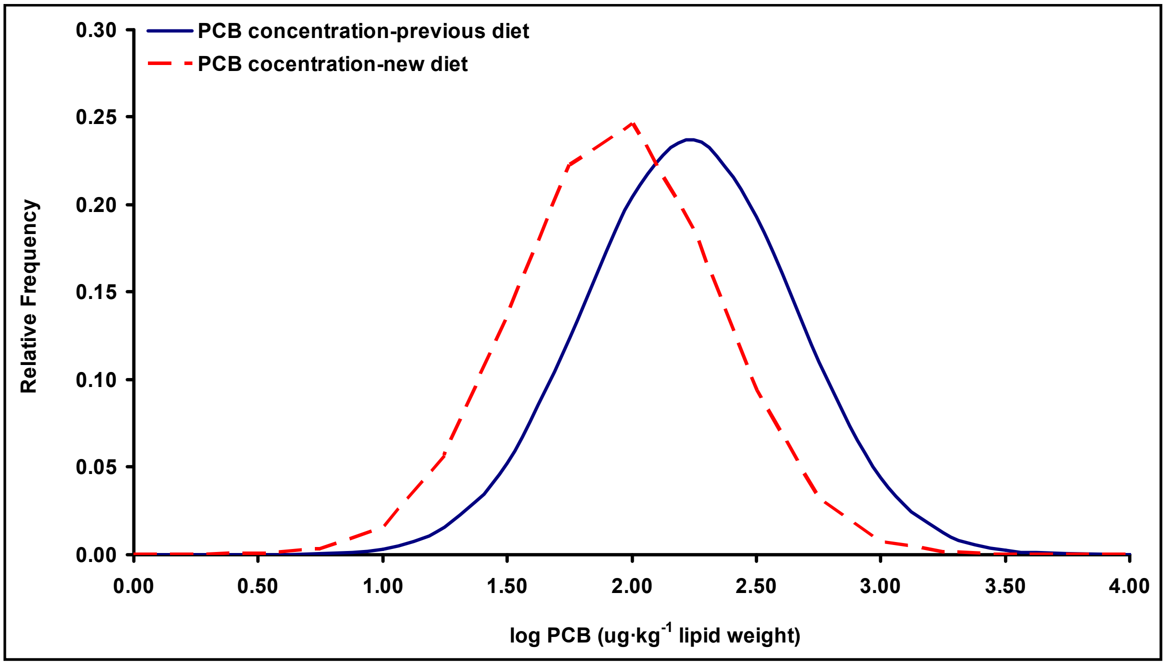


Figure 31: Normal probability density curves showing the comparisons of PCB concentrations predicted in female killer whale with the coastal PCB food web bioaccumulation model using initial diet composition versus new diet in Outer coast habitat.

While no differences was observed in predicted PCB concentration in resident killer whales for the coastal food web (NRKW Critical Habitat) when changing the diet, the differences found in PCB concentrations in the oceanic food web may be well attributed to the reliance of the killer whales' major prey item (i.e., Chinook salmon) on Gonatid squid, which makes up most of the Chinook diet composition (70% use in the model using previous diet) during its oceanic-offshore life stage. In addition, the new diet produced changes in the diet composition for killer whales and Chinook salmon in the outer coast by redistributing diet proportions (in the new diet, squid comprised 3% in the killer whale diet, and 10% in the Chinook diet items composition). This implies that squid might be a major prey delivering significant levels of PCBs to oceanic Chinook salmon.

4.4.2 Evaluating the Effects of Water PCB Concentrations on Model Outcomes

In an effort to assess and test the implications of water as one of the sources delivering PCBs to the aquatic food web, a sensitivity analysis on the water and sediment concentrations was conducted to determine whether changes in the concentrations of PCBs in water are associated with substantial changes in the PCB concentration predicted in Chinook salmon and killer whales in the NRKW critical habitat (coastal food web) and outer coast (oceanic food web) models. The mean of water column concentrations of PCBs (32 $pg \cdot L^{-1}$) in XAD resin (dissolved) measured over three seasons in the Strait of Georgia (Dangerfield *et al.* 2007) was used to calculate an empirical sediment:water PCB concentration ratio. The observation in this study that the PCB concentration in the water does not vary with depth indicates that there is no significant PCB concentration gradient in the water column. This implies that all organisms including phytoplankton are exposed to approximately the same PCB concentration and that the thermocline and halocline do not appear to have a major impact on the PCB concentration in the water column.The sensitivity analysis showed that a 10-fold increase in PCB water concentration caused the predicted PCB concentration in biota to increase by 9.5 times. The results of the sensitivity analysis are shown in Table 45 and indicate that PCBs in the water column are the main source of PCBs in killer whales. This means that the main pathway of killer whale exposure to PCBs after ocean disposal is through a release of PCBs from sediments to the water column. PCBs from contaminated sediments will enter the water column and become absorbed by phytoplankton, zooplankton and fish directly from the water and indirectly from the water as a result of dietary exposure. PCB concentrations in sediment dredgeate in excess of those current present can be expected to increase PCB concentrations in the water column and the food web.

For the specific case of the outer coast model, the results of the sensitivity analysis may indicate that the bioaccumulation of PCB in the ocean food web is likely to be driven by PCB water concentration. Similar scenario was found for the NRKW critical habitat model relying on a coastal food web. Recent studies in the Strait of Georgia showed that the net flux of PCBs appears to be from atmosphere to seawater (Noël *et al.* 2009) and from seawater into the sediments (Johannessen *et al.* 2008a), implying that local atmosphere has the highest PCB input. This supports the notion that air and water may be delivering a major portion of PCBs to the aquatic food web, notably in more remote areas. Within the aquatic ecosystem, the concentrations in water and sediments are related. However, these relationships are complex and dependent on the sediment diagenesis and organic carbon cycling in the system, sorption and desorption rates and the source materials (e.g. aerial particles versus water borne particles), and other processes controlling water-sediment concentration (fugacity) relationships may indeed have some role to play (Gobas and Maclean 2003).

Table 45: Effect of PCB water concentration in predicted PCB concentration in biota.

Habitat/food web	PCB water concentration used in the model ($ng{\cdot}L^{-1}$)	10 times increase in PCB water concentration ($ng{\cdot}L^{-1}$)	Previous PCB concentration predicted in chinook salmon ($ng{\cdot}kg^{-1}$ wwt)	PCB concentration ($ng{\cdot}kg^{-1}$ ww) predicted in chinook after 10x increase in water concentration	Previous PCB concentration predicted in male killer whale ($ng{\cdot}kg^{-1}$ ww)	PCB concentration predicted in male killer whale ($ng{\cdot}kg^{-1}$ ww) after increase in water concentration
NRKW critical habitat (coastal food web)	0.003	0.034	11700	111620	3.44.E+06	3.28E+07
Outer coast (oceanic food web)	0.009	0.090	47160	459545	1.36E+07	1.33E+08

Ratio PCB concentration after 10 times increase in PCB water concentration/Previous PCB concentration in biota

	Chinook salmon	Resident killer whale
NRKW critical habitat	9.5	9.5
Outer coast	9.7	9.7

4.5 MODEL VALIDATION: COMPARISON TO INDIVIDUAL-BASED KILLER WHALE MODEL

Hickie *et al.* (2007) developed an individual-based (IB) bioaccumulation model for killer whales and used it to characterize the history of PCB accumulation by the SRKW and NRKW populations. The dynamic IB model (briefly described below) provides a detailed time-course of the uptake, distribution and elimination of PCBs by representative male and female whales over their entire life span. It focuses specifically on the link between contaminant concentrations in killer whale tissues and their prey. In contrast, the Gobas food web model (FWM), used throughout most of this report, uses a steady-state approach to estimate contaminant concentrations in four specific age/sex classes of killer whales. It is able to relate concentrations in killer whales with those in their prey and with sediment and water concentrations. The two models have similar basic approaches for estimating concentrations in killer whales relative to their prey, but are markedly different in the focus and level of detail in calculations and output. In this section we compare the performance of the two models to evaluate whether the steady-state treatment of killer whales in the FWM adequately captures the effects of their complex life history on PCB accumulation as described in the IB model.

Description of the individual-based (IB) model

The IB model simulates the growth, bioenergetics, dietary contaminant accumulation, distribution two internal compartments (blubber and core), and elimination (via feces and biotransformation) in individual male and female killer whales on a daily basis from weaning till death (approximately 50 and 80 years respectively). Equilibrium distribution of contaminants is assumed between the blubber (about 30% of total mass and 40% lipid) and the remaining “core” of the animal (about 5% lipid on average) which includes skin, bone, blood, muscle and organs. The set of calculations performed for each day for a male or non-reproductive female are summarized in Figure 32 and parameter values are summarized in Table 46. Additional subroutines account for contaminant accumulation and losses in reproductively active females (gestation, birth, and nursing) and their progeny until they are weaned. Dietary contaminant levels are defined for each year the model runs as annual average concentration. In this application the model was run to *pseudo* steady-state by using a constant diet concentration throughout each model simulation and by running the model recursively until predicted concentrations for all ages stabilized. This required three to four recursions (or whale generations) to reach stability, illustrating the fact that the time to reach steady-state for killer whales may require several decades.

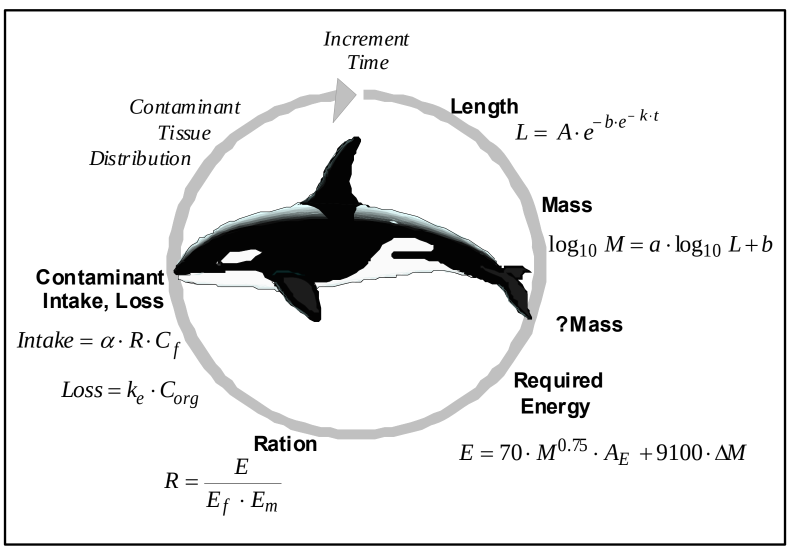


Figure 32: Overview of the structure of the IB model calculations for a male or non-reproductively active female killer whale. They are also used to calculate the energy requirements for nurslings, which are then converted into a demand for milk from the mother.

Table 46: Summary of parameter values used in the individual-based models (Hickie et al. 2007).

Parameter	Standard Value	References
Life History		
gestation (days)	510	1, 2
age at weaning (days)	365	
age at maturity (years)	♀: 14, ♂: 18	
Fetal Growth		
mass(g) at time t (days)	$(0.1053 \cdot t)^3$	3
Gompertz Growth Curve		
A (cm)	♀: 584 ♂: 683	4
b	♀: 1.08 ♂: 0.885	
k (d^{-1})	♀: 0.00048 ♂: 0.00074	
Length-Mass Regression		
m	3.229	4
B	-5.452	
Body Compartments		
volume fraction	Blubber: 0.3 Core: 0.7	estimated
lipid (%)	Blubber: 40 Core: 0.1	
Metabolic Rate Activity Factor (A_E)	3.9	5
Energy Cost of Growth ($kcal \cdot kg^{-1}$)	9100	6
Food		
energy density ($kcal \cdot kg^{-1}$ ww) (E_f)	Fish: 2100 Milk: 3000	Milk: 7
lipid (%)	Fish: 10 Milk: 27.0	Fish: 8
fraction metabolizable energy (E_m)	Fish: 0.82 Milk: 0.9	
ΣPCB Kinetics		
assimilation efficiency from diet (%)	Food: 90 Milk: 90	9
transfer efficiency (lipid normalized):		
- mother to fetus (K_{FB})	0.60	10, 11
- mother to milk (K_{MB})	0.60	12,13
elimination rate constant k_e (d^{-1})	0.000055 (0.02 yr^{-1})	Estimated

1- Olesiuk et al. 1990; 2- Walker et al. 1988; 3- Lockyer, 1981; 4- Clark et al. 2000; 5- Kreite. 1995; 6- Markussen et al. 1990; 7- Lauer et al. 1969; 8- Ronald et al. 1984; 9- Marsili et al. 1995; 10- Subramanian et al. 1988; 11- Salata et al. 1995; 12- Kawai et al. 1988; 13- Addison and Brodie, 1987.

The model was written in Excel VBA for use on a personal computer. Output from the model includes a summary of input variables and the time course of growth, energetics, food intake, and contaminant intake, total burden, concentrations and losses. For females, the output also includes the same information for all progeny from conception to weaning. Example output for males and females from a simulation for NRKWs are shown in Figure 33 in relation to measured concentrations from that population (from Hickie *et al.* 2007).

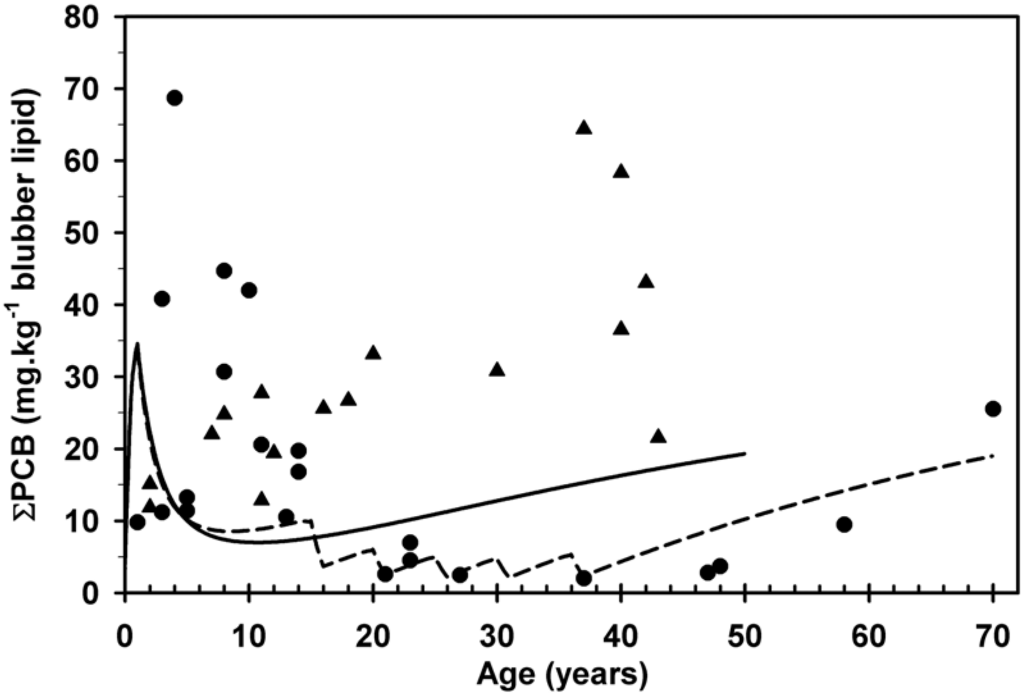


Figure 33: Measured ΣPCB in male (triangles) and female (circles) NRKWs from 1994-2000 and estimated concentrations (males solid line; females dashed line) derived from the individual-based model using a fixed diet concentration of 13 μg·kg^{-1} reflecting current levels in Johnstone Strait Chinook salmon. Modelled concentrations show a rapid rise associated with nursing, followed by a growth dilution phase up to ~10 years of age. Males and females diverge following the onset of reproduction at about ~14 years. In this simulation, the female has five calves with a five year interval between each birth. Concentrations rise with age in the female after reproduction ceases (Hickie et al. 2007 - supplementary information).

Comparison of the Gobas FWM and Hickie IB Models

The Gobas FWM estimates contaminant concentrations in four specific age/sex classes at steady-state to provide a view of the extent of contamination in a population of whales. The four classes are:

- 160 kg nursing calf, 26% body lipid;
- 1000 kg growing juvenile, 26% body lipid;
- 2700 kg reproductively active female, 26% body lipid;
- 5000 kg adult male, 26% body lipid.

The ages of these animals are not specified in the model but would likely fall in the ranges of 0 to 2 years for calves, 2 to 15 years for juveniles and 15 to ~50 years for adults. In contrast, the IB model calculates age-specific contaminant concentrations in male and female whales over the duration of their life span and for each of the females' progeny until they are weaned. Predicted concentrations from the IB model are thus associated with a lifetime of contaminant exposure and are subject to the effects of growth dilution as juveniles and losses associated with birth and lactation in females. Predicted concentrations in calves vary depending on a number of factors including birth order, length of nursing and time interval between calves. Because of these factors, concentrations of highly bioaccumulative chemicals, such as PCBs, do not usually reach steady-state in individuals but they may approach the appearance of steady-state in a population of animals.

The differences between the two models thus limit comparisons to PCB concentrations for the four age/sex classes from the food web model and ones that most closely match from the IB model. To make these comparisons between the two models we used the diet concentrations for killer whale prey (primarily Chinook salmon) calculated by the Gobas FWM to define the long-term diet concentration used in modeling each scenario with the IB model. Therefore, any differences between the predicted concentrations between the models are limited to how the models characterize PCB bioaccumulation by killer whales from their prey. Overall, predicted concentrations of PCBs from the two models showed good agreement across the four age/sex classes and modeling scenarios as shown by the regression:

$$\log [PCB]_{IBM} = 0.95 * \log [PCB]_{FWM} + 0.27 \quad (r^2 = 0.94, n = 32, p < 0.001)$$

Within this regression there was excellent agreement between the models for adult males and females as shown by mean $[PCB]_{IBM} : [PCB]_{FWM}$ ratios of 0.98 and 1.13, respectively. These ratios were calculated using concentrations estimated for 35 year-old males and 32 year-old females from the IB model output. Concentrations in reproductively active females are quite stable over time, meaning that the concentrations derived from the food web model for adult females are reflective of a wide age range. In the case of adult males, the IB model predicts a near linear increase in concentration with age once they reach full size at about 20 years of age. The concentration ratios between the models would, however, only reach a value of 1.30 if the comparison was based on 50 year-old males. Values within a factor of two are generally considered to reflect good agreement with these types of models.

Comparisons between results for calves are also quite good as shown by the mean mean $[PCB]_{IBM} : [PCB]_{FWM}$ ratio of 1.37. For this we used the average predicted concentration across all calves produced by a female in the IB modeling scenarios. As noted above, there are many factors that can affect predicted concentrations in calves, including number of calves produced, time interval between calves, length of nursing and calf growth rate. The difference between the two models is within the range of uncertainty that is generated across multiple calves from Monte Carlo uncertainty analysis for the IB model.

Comparisons between results for juveniles is not as good as for the other three age/sex classes as shown by the mean $[PCB]_{IBM} : [PCB]_{FWM}$ ratio of 3.61. For this we used concentrations predicted for 5 year-old males in order to match the body mass used in the food web model. At this age, however, concentrations in juveniles are still affected by the enhanced intake from nursing and are subject to substantial growth dilution of their contaminant burden which can last into their mid to late teens when growth slows. The ratio falls to 2.20 if concentrations for 12 year-old juveniles (~3000 kg) are used. Predictions for PCB concentrations for animals classified as juveniles are highly dependent on age, rate of growth as well as contaminant loading acquired from both nursing and prey.

5. CONCLUSIONS

5.1 RESPONSES TO REQUEST FOR ADVICE

Are current Ocean Disposal Rejection/Screening Limits for environmental contaminants (including PCBs, mercury and PAHs) under CEPA 1999 adequate to prevent northern and southern resident killer whale Critical Habitat from destruction, as required by SARA Section 58?

Section 58 of SARA states that "*no person shall destroy any part of the critical habitat of any listed endangered species or of any listed threatened species*". This effort evaluated the possible impacts of disposal operations at sites located *within* Critical Habitat of Northern Resident killer whales (n=2) and Southern Resident killer whales (n=3). The current Disposal at Sea Action Level Low for total PCBs under CEPA 1999 does not provide a sufficiently sensitive means of assessing risks to SARA-listed killer whales, and therefore does not permit a precautionary evaluation of the destruction of Critical Habitat of northern and southern resident killer whales, as required by SARA Section 58. A PCB concentration equivalent to this screening limit in sediments at disposal sites within Critical Habitat is predicted to increase PCB concentrations in killer whales by 0.62 – 3.12 %.

Modeling efforts indicate that disposal of sediments with PCB levels that exceed those measured in the ambient receiving environment has the potential to increase the exposure of killer whales to PCBs. Current disposal practices are unlikely to increase PCB delivery to killer whales in many cases, since much of the disposal material comes from the lower Fraser River and has a lower PCB concentration than that the sediments at the disposal sites. However, the disposal of sediments from some of the more contaminated sites for which data are available (including Victoria Harbour) are predicted to increase the PCB concentrations in killer whales by 5.8 – 29.3%, causing an increase in the proportion of the population exceeding health thresholds of up to 10.8%. Clearly, disposal of dredge materials has the potential to increase the availability of PCBs to killer whales.

A broader evaluation of the adequacy of the CEPA screening limit for PCBs was carried out using hypothetical killer whales and prey that reside in an environment where the CEPA ALL is substituted for the PCB concentration in all the underlying sediments. In such a scenario, predicted PCB concentrations in male killer whales would be 7,734 mg/kg (lipid; males) or 7,734 ppm. This is 81 times higher than the current PCB levels in killer whales, and underscores both the vulnerability of killer whales to persistent contaminants that biomagnify, and the insensitivity of the CEPA screening limit to support risk evaluation of disposal materials for high trophic level animals.

Results provide a basic risk management tool to evaluate disposal permit applications for sites within Critical Habitat based on dredge material PCB measurements.

Further research will evaluate the strength of some of our underlying assumptions for these values (diet selection for killer whales, estimates for killer whale and Chinook salmon distribution over space and time, and the sometimes limited sediment PCB data used to explore scenarios. While uncertainty analyses of the model found under- and over-estimations of different PCB congeners in biota, the overall model bias was very low as there was little systematic over- or under-prediction of total PCB concentrations.

Do PCBs in materials deposited outside of killer whale habitat increase the risk of harm or mortality of northern and southern resident killer whales, as required by SARA Section 32?

Section 32 of SARA states that "*No person shall kill, harm, harass, capture or take an individual of a wildlife species that is listed as an extirpated species, an endangered species or a threatened species."* This effort evaluated the possible impacts of disposal operations at sites found *outside* of designated Critical Habitat of killer whales, but within their habitat in general. Both disposal sites for northern resident killer whales are situated within Critical Habitat, while disposal sites in southern resident killer whale habitat are situated both within Critical Habitat (3) and outside of Critical Habitat (8). Modeling efforts predict that disposal at all disposal sites within southern resident killer whale habitat (both inside and outside of Critical Habitat) of sediment containing a higher than ambient concentration of PCBs would increase the delivery of PCBs to killer whales. Since this increase is not proportionate to the increase in surface area contaminated by PCBs, this underscores the sensitivity to contamination of the sites within Critical Habitat. The results indicate that disposal of sediments that have PCB levels that exceed the ambient levels is most harmful when it takes place within Critical Habitat, but is also harmful, albeit less so, when it takes place at sites outside of Critical Habitat.

Within the assumptions of the modeling approaches employed, there was little systematic over- or under-prediction of PCB concentrations. Further research to evaluate such assumptions or improve the confidence in the outcomes could include more information on the concentrations of contaminants present in materials disposed at sea, by evaluating more specific disposal activities on a site-specific basis, and a characterization of sediment /biota bioavailability in typical dredged sites and materials (e.g. due to variation organic carbon content).

What contaminants, or levels thereof, in disposal materials might represent destruction of killer whale Critical Habitat as per SARA Section 58, or present a risk of harm or mortality to individuals as per SARA Section 32?

Killer whales are long-lived and occupy a very high trophic level, and are therefore at particular risk to accumulating high concentrations of biomagnifiying substances, such as persistent organic pollutants (POPs). These contaminants can also affect killer whale prey, which compromises the function of the Critical Habitat. Currently, PCBs are a dominant toxicological concern in killer whales, and represent the only POP evaluated in disposal materials under CEPA, hence our evaluation of this contaminant in the context of killer whales. The expansion of the current Disposal at Sea list of targeted contaminants for screening (from the present five: PCBs, PAHs, Hg, Cd and plastic content) would better enable a more ecosystem-based assessment of risks to all biota in the marine environment, and help to leverage outcomes that are consistent with SARA, CEPA and the Oceans Act.

Polycyclic aromatic hydrocarbons (PAHs) have been implicated in liver tumours and other health effects in fish inhabiting industrialized harbours, but this complex class of contaminants is readily metabolized at lower levels of the food web, and killer whales are unlikely to be at significant risk of health impacts at current levels of dietary exposure.

Organic forms of mercury biomagnify in food webs, and do represent a concern to upper trophic levels. Marine mammals are thought to be largely protected from the effects of mercury as a result of long-term exposure to natural background concentrations of this metal in the environment (Ikemoto *et al.* 2004). However, anthropogenic contributions to mercury emissions following the industrial revolution, both globally and locally, have raised concerns that the detoxification abilities of marine mammals (including killer whales) may not be sufficient.

Dioxins and furans represent a significant environmental concern in coastal BC, but risks to killer whales have been limited by the 1989 implementation of source control and regulations (Hagen *et al.* 1997), and by the relatively rapid apparent metabolism of dioxin-like contaminants by killer whales and their prey (Ross *et al.* 2000a). Nevertheless, dioxins and furans are important contaminants at lower trophic levels (e.g. invertebrates) and can present a significant risk to humans. The presence of high levels of dioxins and furans in sediments has been noted near pulp and paper mills and urban centers, leading to localized commercial invertebrate fisheries closures in British Columbia.

There are contaminants of increasing concern in the environment that warrant consideration under the auspices of either SARA or CEPA. These include such persistent contaminants as PBDEs, documented to be doubling in the BC coastal environment every 3.5 years, which enter the Strait of Georgia in significant quantities through municipal outfalls (Johannessen *et al.* 2008a;Ross *et al.* 2008;Ross *et al.* 2009). In addition, pharmaceuticals and personal care products, including artificial fragrances, are increasingly used, and many are specifically designed to have biological effects. Since no Sediment Quality Guidelines exist for many of these contaminants, more work is needed to assess risks and environmental behaviour of these contaminants.

Are current analytical standards as applied under Disposal at Sea operations in the Pacific Region sufficient to enable science-based advice under the terms of SARA protection orders and/or permitting?

Given the extraordinary vulnerability of killer whales to PCB contamination, it is important that any study designs, monitoring efforts and analytical approaches deliver accurate information to scientists and managers tasked with the conservation and/or recovery of killer whales under SARA. This will facilitate a rigorous assessment of disposal permit applications.

Current Disposal at Sea decision parameters based on CEPA screening limits and low resolution analytical methods are too high to permit robust science-based advice regarding killer whales under the terms of SARA protection orders and/or permitting. By applying the sediment Detection Limits associated with CEPA Disposal at Sea practices in the Pacific Region (USEPA Method 8080 for PCBs: 40 $\mu g \cdot kg^{-1}$ dry weight) for permit applicants to the model, PCB concentrations in killer whales are predicted to increase by 0.25 – 1.24% under scenarios that have realistic foodwebs for killer whales and their prey.

A broader evaluation of dredgeate PCBs was carried out using killer whales and prey that reside in a hypothetical environment where the current Detection Limits using Low Resolution methods is substituted for the PCB concentration in underlying sediments, and the PCB concentrations within the various compartments all come to steady state with those sediments. In such a scenario, predicted PCB concentrations in male killer whales would be 3,094 mg/kg (lipid; males) or 3,094 ppm. This is 32 times higher than the current PCB levels in male southern resident killer whales, highlighting the failure of current analytical protocols to adequately evaluate dredgeate quality in support of disposal permitting.

Detection Limits give the highest concentration of PCB in sediment that would be classified analytically as containing zero PCB. When our modeling is applied to sediments at the DL, the result is a very contaminated killer whale population. This highlights the importance of using

High Resolution Gas Chromatography/ Mass Spectrometry, which has much lower DLs (as much as 350 times lower) and captures pattern-specific data for the 209 congeners, as the preferred method for PCB determination. Lower DLs are particularly relevant for high trophic level wildlife, where biomagnification of low PCB concentrations in water or sediments leads to high concentrations in killer whales.

Wider geographical coverage of sediment contaminant concentration data would provide increased confidence about the relative risks associated with ocean disposal vs. ambient contamination. For example, the addition of new reference sites would be of value. The food web bioaccumulation model is very sensitive to sediment concentration inputs, such that increased sampling would decrease model bias and would reduce the uncertainty associated with determining sediment concentration averages for large areas. In addition, approaches to the monitoring of disposal sites could be revisited to ensure that frequency, spatial coverage, ancillary analyses (e.g. organic carbon, sedimentation rates, substrate characteristics), and contaminant analyses are conducted in a manner that support an adequate science-based evaluation of risks to killer whales, particularly in Critical Habitat.

What are the estimated threshold concentrations of contaminants (notably PCBs) in sediments or disposal materials that would be considered as adequate to prevent negative effects on killer whale health and destruction of Critical Habitat?

The model was used to back-calculate sediment PCB levels that would be considered as protective of killer whales. Under this premise, proposed target sediment concentrations involving realistic scenarios of habitat distribution to protect the majority (95%) of resident killer whales in Critical Habitats ranged from 0.012 to 0.200 $\mu g \cdot kg^{-1}$ PCBs dry weight. Similar sediment target values were observed under the assumption that a hypothetical resident killer whale had a 100% presence in each model area. We consider these target PCB concentrations in sediments to be protective of killer whale health in the context of toxicity from PCBs.

Although it would be ideal to require that the concentration of PCBs in all disposed material met this target, in reality only 4/61 (6.6%) sediment sites for which we have PCB measurements in coastal British Columbia and Washington State fall below the most liberal effects threshold of 0.200 $\mu g \cdot kg^{-1}$. Since most killer whale habitat has sediment PCB concentrations well above these recommended levels, heightened scrutiny of disposal criteria for PCBs may not improve killer whale habitat quality. However, a revised approach to the assessment of ocean disposal could ensure that disposal at least not further degrade habitat quality for killer whales (i.e. PCB content in disposal materials not exceeding ambient sediment concentrations measured in Critical Habitat), or, even better, result in reduced PCB concentrations in Critical Habitat sediments (i.e. when PCB content in disposal materials is less than that measured in ambient Critical Habitat sediments and may effectively result in sediment PCB ‘dilution’).

Can we detect a contribution of ambient sediment-associated and/or disposal-associated PCBs in killer whale Critical Habitat using a food web bioaccumulation modeling approach?

The modeling effort predicts that the disposal of any dredge materials containing PCB levels that exceed those measured in the ambient receiving disposal site would increase the delivery of PCBs to killer whales. Using four disposal scenarios in Critical Habitat, PCB concentrations in northern resident killer whales would increase by up to 29.3%, and in southern residents would increase by up to 5.9%. The effects of any of a number of scenarios can be tested using formulae generated for each of the resident killer whale populations, thereby characterizing the net model impact of a proposed disposal operation. The caveats here include the variation in site-specific sedimentation rates (burial), the quantity of dredge material being deposited, oceanographic circumstances including stratification and exchange, and the frequency of disposal.

Can we attribute PCBs in northern and southern resident killer whale habitat areas used by killer whales and their prey?

Since several habitat areas are used by the two populations of resident killer whales, we employed the model to attribute the proportion of accumulated PCBs in killer whales to each of the areas used. In this manner, we estimate that southern resident killer whales acquire via their prey 49% of their PCBs in US inland waters, 18% in Canadian inland waters, and 33% in outer coast waters (Canada, USA and international waters). Northern resident killer whales acquire 14% of their PCBs in Canadian inland waters and 86% in outer coast waters (Canadian, international and USA). PCBs attributable to Critical Habitat were estimated to be 3% for northern residents and 15% for southern residents. It appears that resident killer whales receive the majority of their PCB burden from areas outside of Critical Habitat. Thus Canadian Critical Habitat is providing a relatively small portion of PCBs to resident killer whales.

This exercise underscores the relatively limited extent to which management actions could exert a net benefit on the quality of Critical Habitat, the risks associated with global release of megatonne quantities of persistent contaminants, and the need to work nationally and internationally to control or eliminate such sources. The modeling effort presented here provides a means of predicting the added impact of proposed disposal operations on killer whale PCB burdens, and can therefore be used as a risk management tool.

Are new thresholds for sediment quality and/or disposal screening necessary to protect resident killer whale Critical Habitat (SARA Section 58) and/or health of individuals (SARA Section 32)?

Revised Sediment Quality Guidelines and/or Disposal at Sea Lower Action Levels could better protect resident killer whale Critical Habitat and their habitat from chemical degradation. Proposed target sediment concentrations that would protect 95% of resident killer whales ranged from 0.012 to 0.200 $\mu g \cdot kg^{-1}$ dry weight. This number should be regarded as a target, rather than a potential new guideline or regulated limit, since it is below most current ambient environmental levels. Actions that would lead to the reduction of sediment PCBs in killer whale habitat would ultimately reduce PCB delivery to killer whales via the food web.

In this light, the adoption of the ambient sediment PCB concentration in Critical Habitat as a benchmark below which disposal might proceed would presumably lead to lower net PCB sediment concentrations over time. This benchmark will decline with the turndown in environmental cycling of PCBs over time, with the consequence that, eventually, ambient PCB concentrations in Critical Habitat, and elsewhere, will drop below the 0.012 to 0.200 $\mu g \cdot kg^{-1}$ target range. Disposal decisions could help in a small way to achieve this outcome by not increasing the sediment PCB concentrations or inventory in Critical Habitat, by rendering sediment PCBs unavailable through the addition of cleaner disposal materials (capping), by not dredging in highly contaminated areas, and/or by disposal of more contaminated sediments in suitable locations (e.g. sites with high burial rates and high stability) outside of Critical Habitat.

Is there adequate information to develop a set of basic guiding principles for disposal practices and/or disposal site selection that would reduce contaminant risks to killer whale Critical Habitat to avoid Section 58 destruction and/or killer whale health to avoid Section 32 harm or mortality?

We suggest that the ambient sediment PCB concentration becomes an important benchmark for a management-based evaluation of risks to killer whales and to killer whale Critical Habitat. Disposal of materials with PCB concentrations lower than ambient in Critical Habitat with active natural sedimentation will not increase sediment PCB concentrations, might help to bury contaminated sediments, and should not to lead to increased PCB concentrations in killer whales.

Should PCB content in disposal materials exceed that of ambient sediment concentrations at disposal sites, particularly in Critical Habitat, a more difficult decision will have to be made regarding alternative disposal options that will have to consider cost vs the net benefit both specifically to killer whales, and generally to the environment. Options include 1) land-based storage (cost-effective but limited net environmental gain), 2) destruction at a suitable facility on land (effective but very costly and would have other consequences including greenhouse gas production, and 3) disposal at an ocean disposal site outside of killer whale Critical Habitat. The latter option might be preferable, but site-specific characteristics should be considered, as virtually all coastal waters outside of Critical Habitat for resident killer whales also comprise general habitat for killer whales.

Basic guiding principles for the choice of existing or new disposal sites outside of Critical Habitat for disposal of materials that exceed ambient PCB concentrations are outside of the scope of this effort but could include: 1) areas with high natural sedimentation rates, hence relatively rapid burial, and/or 2) areas infrequently visited by killer whales or Chinook salmon and their prey.

5.2 DATA GAPS

The food web bioaccumulation model requires sediment PCB concentrations and organic carbon content as critical inputs to calculate PCB concentrations in biota. These data were gathered from the literature and from unpublished data collected by Fisheries & Oceans Canada. Certain areas included in the model had plenty of sediment PCB data to go into the model (i.e., the Strait of Georgia); however, other areas lacked data (i.e., Queen Charlotte Strait) or had very few samples (i.e., outer coast and northern resident killer whale Critical Habitat). This highlights the need for more sampling so that the accuracy of the model can be improved. Furthermore, sediment sampling needs to occur in background areas to provide more precise PCB distributions, because most sampling studies focus on hot spots known to be contaminated with PCBs, which results in overestimations in biota. Further work is required to address resident killer whale habitat impacts.

Due to limited time we were unable to include the results of a Puget Sound sediment study conducted by the United States Army Corps of Engineers in 2009. This study focused on measuring PCB concentrations in surface sediments in locations in Puget Sound that are not considered contaminant hotspots. The studies included from Puget Sound in this report focused on sediment hot spots, thus the average Puget Sound concentration utilized is likely an overestimate. We attempted to account for this by calculating a geometric mean for total PCBs to reduce the bias from some of the very high PCB concentrations in the samples. Future work could include the additional samples from the Army Corps of Engineers to obtain a more normal average PCB level for Puget Sound.

A weakness of the model is that the sediments were assumed equally important in the outer coast area even though they are much deeper and potentially less connected to the food web as in the coastal areas. There is not much information about how connected the sediments are in these regions let alone relative to one another. This would require empirical measurements of PCBs in surface waters and sediments of the outer coast, and these data are currently lacking. However, these adjustments may have very little influence on the model predictions, especially because the observed and predicted concentrations were very close and model bias was very low.

The accuracy of the food web bioaccumulation model was tested by comparing the model predictions of PCB concentrations in biota to empirical data. This was only possible for Chinook salmon and northern resident killer whales. More sampling of other important species included in the food web is required to ensure the model makes accurate predictions at all trophic levels. However, this was not the goal of this exercise. In addition, metabolic rates for specific PCB congeners in killer whales were assumed to be zero in the PCB food web bioaccumulation models since current biotransformation rate values and our understanding on metabolic capacity of PCB congeners is limited. This requires research and assessment of PCB metabolism in toothed cetacean species to improve further applications of the model. Accurate empirical measurements of lipid content for secondary prey diet items (i.e., halibut and sablefish) in Critical Habitats of resident killer whales are also needed to further improve the prediction of PCB concentrations. However, there was good agreement between observed and predicted PCB concentrations in Chinook salmon and NRKWs, which indicates that this component of the model worked well even though metabolism was neglected in the Gobas model. Furthermore, the Gobas model results were consistent with the Hickie (2007) killer whale model, which did address metabolism. Thus further adjustments are likely unnecessary.

The model attempted to predict the consequences of disposal of dredgeate in Critical Habitat and the general habitat of resident killer whales. However, actual PCB concentrations of the dredged material were not available, thus we were only able to use the detection limits, the Action Levels CEPA sets for disposal at sea that are currently in use, and sediment PCB concentrations from other studies that sampled in the areas where dredging occurs. The model prediction of the effects of dredging could have been put on a firmer basis with empirical data for disposal at sea materials, as measured PCB data would remove uncertainty associated with this component. Existing analytical methodology to screen for PCBs in dredgeate is based on measuring Aroclor. This is an outdated and inappropriate method of assessing PCB concentrations in sediments (Muir and Sverko 2006). Currently a limited number of PCB congeners are measured, and the detection limit of the test is too high (40 $\mu g \cdot kg^{-1}$), to provide numerical accuracy for most samples. Improvement of this methodology would ensure greater accuracy in information on sediment contaminant levels.

PCBs are no longer used in Canada and the US, but PBDEs are increasingly being used. However, there are very limited data on PBDEs in sediments or biota and currently there are no Sediment Quality Guidelines or dredging Action Levels for them. Toxicity testing of PBDEs in a variety of species is required. Given they are persistent, bioaccumulative, and toxic to certain species, guidelines need to be established and empirical studies need to be conducted so that their fate in biota can also be modeled. However, modeling of PBDEs is hampered for several reasons: they are less stable than PCBs, many congeners are metabolized, and they are in disequilibrium in the environment as they are doubling every 3.5 years. Other newly emerging contaminants, including pharmaceuticals, personal care products and other household products, including artificial fragrances, also need to be examined for their potential to bioaccumulate and biomagnify. This is a difficult task due to the different degree of metabolism between PBDE congeners and because analytical methods for many of the pharmaceuticals are still in development.

Despite the data gaps mentioned above, there were many strengths of this exercise. The model was science-based and relied upon empirical data, published research and validated models. It reflects the work of a multidisciplinary team with expertise in toxicology, contaminant transport and fate, sediment behaviour, food web models, and killer whale and Chinook salmon biology and ecology. Resident killer whales in BC are among the best studied cetacean populations in the world, and Chinook salmon are among the best studied salmonids in BC. Furthermore, this study was designed to specifically address management needs, and the outcomes i) provide guidance (answers) and ii) identify data gaps and research needs.

5.3 ASSUMPTIONS

As with any model, many assumptions were required for simplicity and transparency. Some of the major assumptions are discussed below. The seven geographic compartments (areas) were based on resident killer whale and Chinook salmon distributions, but had to be constrained due to data limitations (especially sediment data). If more time was available these areas (especially the outer coast area) could be reduced in size to capture important aspects, such as high use areas by NRKWs.

A major assumption of the model is that contaminant-sediment partitioning is at steady-state, and this may not be the case either, especially when including long-lived species such as killer whales. However, again the comparison between observed and predicted PCB concentrations in Chinook salmon and killer whales was very similar with very little over- or under-predictions, so assuming steady-state conditions may not be problematic.

We based Chinook distributions on data from fishing mortality and these data may not accurately represent the time spent in an area. We also lacked data on Chinook feeding ecology, especially when in the outer coast area. This would be improved with data on stomach contents; however, those data were unavailable in time to be included in this effort. It was assumed that all SRKWs and NRKWs (respectively) behave the same and spend the same amount of time in all areas; however, in reality this is not the case and the output reflects an "average" rather than being specific for certain pods or individuals.

5.4 OTHER CONSIDERATIONS

Studies on other bioaccumulative contaminants present in sediments and dredged materials should be conducted to determine their impact on resident killer whales and their Critical Habitat. This is especially important for PBDEs as their concentrations are increasing very rapidly in the environment.

Natural siltation/smothering of sediments potentially buries disposal at sea sites (especially near the mouth of the Fraser River), and may essentially reduce exposure of organisms in the food web to PCBs present in disposal materials. This is an important consideration for disposal practices, in terms of frequency of disposal and the site selection process.

The model predicted that a considerable proportion of the resident killer whale PCB burden comes from the outer coast area, thus ocean disposal in Critical Habitat is likely a small contributor to the killer whale PCB burden. PCBs are distributed throughout the ecosystem and are not restricted to urban areas, but cleanup on that scale is impractical. This highlights the issue of global contaminants (i.e., ocean and atmospheric currents delivering contaminants from Asia to North America), and the importance of source control. The Stockholm Convention on Persistent Organic Pollutants is a global treaty to eliminate or reduce the release of POPs into the environment, and Canada is a participant.

5.5 RECOMMENDATIONS

1. Increase sediment sampling for the entire suite of PCB congeners (rather than Aroclor-based determinations), especially in background/reference areas (e.g. outside of the Strait of Georgia).
2. Increase sampling of PCBs in marine surface water and air to improve model predictions and accuracy.
3. Increase sampling of PCBs in biota (especially organisms included in the modelled food web, e.g., South Thompson Chinook salmon) to verify model output and determine model bias/error.
4. Obtain PCB concentrations from material disposed at sea to determine implications for killer whales using these food web bioaccumulation models.
5. Establish Sediment Quality Guidelines, Disposal at Sea Action Levels, and regulated limits for other marine pollution prevention programs to address other contaminants that bioaccumulate, such as PBDEs.
6. Include new chemicals in research that will ultimately inform ocean disposal and/or SARA deliberation, including PBDEs and pharmaceuticals.
7. Design field studies to better capture Chinook salmon annual distributions and feeding ecology.
8. Generate revised or new Sediment Quality Guidelines and Action Levels that account for biomagnification and explicitly consider high trophic level biota.
9. Conduct field research with appropriate agencies and researchers to better understand the fate of dredgeate plumes (particles and contaminants) during disposal operations at existing disposal sites, or sedimentation processes and particle fate at proposed (new) sites.

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APPENDIX I

SPECIES AT RISK ACT
MEASURES TO PROTECT LISTED WILDLIFE SPECIES
General Prohibitions
Section 32
(1) No person shall kill, harm, harass, capture or take an individual of a wildlife species that is listed as an extirpated species, an endangered species or a threatened species.

(2) No person shall possess, collect, buy, sell or trade an individual of a wildlife species that is listed as an extirpated species, an endangered species or a threatened species, or any part or derivative of such an individual.

(3) For the purposes of subsection (2), any animal, plant or thing that is represented to be an individual, or a part or derivative of an individual, of a wildlife species that is listed as an extirpated species, an endangered species or a threatened species is deemed, in the absence of evidence to the contrary, to be such an individual or a part or derivative of such an individual.

Protection of Critical Habitat
Section 58
(1) Subject to this section, no person shall destroy any part of the critical habitat of any listed endangered species or of any listed threatened species—or of any listed extirpated species if a recovery strategy has recommended the reintroduction of the species into the wild in Canada—if
(*a*) the critical habitat is on federal land, in the exclusive economic zone of Canada or on the continental shelf of Canada;
(*b*) the listed species is an aquatic species; or
(*c*) the listed species is a species of migratory birds protected by the *Migratory Birds Convention Act, 1994*.

(2) If the critical habitat or a portion of the critical habitat is in a national park of Canada named and described in Schedule 1 to the *Canada National Parks Act*, a marine protected area under the *Oceans Act*, a migratory bird sanctuary under the *Migratory Birds Convention Act, 1994* or a national wildlife area under the *Canada Wildlife Act*, the competent Minister must, within 90 days after the recovery strategy or action plan that identified the critical habitat is included in the public registry, publish in the *Canada Gazette* a description of the critical habitat or portion that is in that park, area or sanctuary.

(3) If subsection (2) applies, subsection (1) applies to the critical habitat or the portion of the critical habitat described in the *Canada Gazette* under subsection (2) 90 days after the description is published in the *Canada Gazette*.

(4) If all of the critical habitat or any portion of the critical habitat is not in a place referred to in subsection (2), subsection (1) applies in respect of the critical habitat or portion of the critical habitat, as the case may be, specified in an order made by the competent minister.

(5) Within 180 days after the recovery strategy or action plan that identified the critical habitat is included in the public registry, the competent minister must, after consultation with every other competent minister, with respect to all of the critical habitat or any portion of the critical habitat that is not in a place referred to in subsection (2), (*a*) make the order referred to in subsection

(4) if the critical habitat or any portion of the critical habitat is not legally protected by provisions in, or measures under, this or any other Act of Parliament, including agreements under section 11; or (*b*) if the competent minister does not make the order, he or she must include in the public registry a statement setting out how the critical habitat or portions of it, as the case may be, are legally protected.

(5.1) Despite subsection (4), with respect to the critical habitat of a species of bird that is a migratory bird protected by the *Migratory Birds Convention Act, 1994* that is not on federal land, in the exclusive economic zone of Canada, on the continental shelf of Canada or in a migratory bird sanctuary referred to in subsection (2), subsection (1) applies only to those portions of the critical habitat that are habitat to which that Act applies and that the Governor in Council may, by order, specify on the recommendation of the competent minister.

(5.2) The competent minister must, within 180 days after the recovery strategy or action plan that identified the critical habitat that includes habitat to which the *Migratory Birds Convention Act, 1994* applies is included in the public registry, and after consultation with every other competent minister, (*a*) make the recommendation if he or she is of the opinion there are no provisions in, or other measures under, this or any other Act of Parliament, including agreements under section 11, that legally protect any portion or portions of the habitat to which that Act applies; or (*b*) if the competent minister does not make the recommendation, he or she must include in the public registry a statement setting out how the critical habitat that is habitat to which that Act applies, or portions of it, as the case may be, are legally protected.

(6) If the competent minister is of the opinion that an order under subsection (4) or (5.1) would affect land in a territory that is not under the authority of the Minister or the Parks Canada Agency, he or she must consult the territorial minister before making the order under subsection (4) or the recommendation under subsection (5.2).

(7) If the competent minister is of the opinion that an order under subsection (4) or (5.1) would affect a reserve or any other lands that are set apart for the use and benefit of a band under the *Indian Act*, he or she must consult the Minister of Indian Affairs and Northern Development and the band before making the order under subsection (4) or the recommendation under subsection (5.2).

(8) If the competent minister is of the opinion that an order under subsection (4) or (5.1) would affect an area in respect of which a wildlife management board is authorized by a land claims agreement to perform functions in respect of wildlife species, he or she must consult the wildlife management board before making the order under subsection (4) or the recommendation under subsection (5.2).

(9) If the competent minister is of the opinion that an order under subsection (4) or (5.1) would affect land that is under the authority of another federal minister, other than a competent minister, he or she must consult the other federal minister before making the order under subsection (4) or the recommendation under subsection (5.2).

APPENDIX II

Table 1: *Disposal activity at Sand Heads disposal site from 2000 to present (Sean Standing, Environment Canada, Disposal at Sea Program, #201-401 Burrard St., Vancouver, BC V6C 3S5, pers. comm., 2010).*

Substance Disposed	Load Site	Year	Volume Disposed (m^3)
Dredged material	Fraser River-Maintenance	1999	230,135
Dredged material	Fraser River-Seaspan Coastal Intermodal-Tilbury Island	2000	2,500
Dredged material	Fraser River North Arm-Mainland sand and gravel ramp	2000	1,200
Dredged material	Fraser River-Interfor-ACORN	2000	6,500
Dredged material	Fraser River-Ritchie Brothers	2000	2,805
Dredged material	Lower Fraser River-Crescent Beach Marina	2000	4,310
Dredged material	Fraser River-Noble Custom Cut Ltd.	2000	3,973
Dredged material	Fraser River-Toews Bros	2000	3,000
Dredged material	Fraser River-North West Hardwoods	2000	2,600
Dredged material	Fraser River-Ladner-PWC	2000	11,000
Dredged material	Fraser River-Deas Slough	2000	6,000
Dredged material	Fraser River-Maintenance	2000	27,000
Dredged material	Fraser River-Maintenance	2000	143,120
Dredged material	Fraser River-Maintenance	2000	1,250
Dredged material	Fraser River-Maintenance	2000	27,400
Dredged material	Fraser River-Maintenance	2000	22,210
Dredged material	Fraser River-Maintenance	2000	20,120
Dredged material	Fraser River-Rivtow-Barge Ramp	2000	3,994
Dredged material	Fraser River-Barnston Island-GVRD	2000	3,000
Dredged material	Fraser River-Conag-Surrey	2000	4,000
Dredged material	Fraser River-Crescent Beach	2001	14,907
Dredged material	Fraser River-Conag-Delta	2001	2,800
Dredged material	Fraser River-Seaspan Coastal Intermodal-Tilbury Island	2001	1,200
Dredged material	Fraser River-Canadian Fishing Co.	2001	4,000
Dredged material	Fraser River-Four Seas Industries Ltd.	2001	1,500
Dredged material	Fraser River-Maintenance	2001	250,000
Dredged material	Fraser River-Maintenance	2001	500,000
Dredged material	Fraser River-RIVTOW-Alaska Way	2001	10,299
Dredged material	Fraser River-Tilbury Cement	2002	4,000
Dredged material	Fraser River-Cannery Channel	2002	59,881
Dredged material	Fraser River-Seaspan Coastal Intermodal-Tilbury Island	2002	2,000
Dredged material	Fraser River-Bridgeview Marina	2002	4,000
Dredged material	Fraser River-North West Hardwoods	2002	2,600
Dredged material	Fraser River-Cannery Channel	2002	43,698
Dredged material	Fraser River-Maintenance	2002	961,660
Dredged material	Fraser River-Gulf Site	2002	34,634
Dredged material	Fraser River-Cannery Channel	2002	3,896
Dredged material	Fraser River-CONAG-Surrey	2002	2,500

Dredged material	Fraser River-Ladner-River West Marina	2002	2,800
Dredged material	Fraser River-Annacis Channel	2002	1,000
Dredged material	Fraser River-Anbrook Industries	2002	500
Dredged material	Fraser River-Tilbury Cement	2002	4,000
Dredged material	Fraser River North Arm-Mainland sand & gravel ramp	2002	700
Dredged material	Fraser River-Westview Dredging	2002	800
Dredged material	Lower Fraser-Crescent Beach	2003	20,000
Dredged material	Fraser River-Mckenzie Mills	2003	1,300
Dredged material	Fraser River-Maintenance	2003	474,360
Dredged material	Fraser River-North Fort Marina	2003	1,500
Dredged material	Fraser River-Scotch Pond	2004	12,300
Dredged material	Fraser River-Tilbury Cement	2004	4,000
Dredged material	Fraser River-CONAG-Delta	2004	2,800
Dredged material	Fraser River-CONAG-Surrey	2004	2,800
Dredged material	Fraser River-River House Marina	2004	4,000
Dredged material	Various approved sites on Fraser River Estuary	2004	1,000,000
Dredged material	Various approved sites in the channels of Fraser River	2004	700,000
Dredged material	Fraser River-Granite Is Holdings	2004	750
Dredged material	Fraser River-Tilbury Cement	2004	1,000
Dredged material	Fraser River North Arm-FRPD Yard	2004	1,000
Dredged material	Fraser River-Ocean Fish-Rice Mill	2005	1,550
Dredged material	Fraser River-Tilbury Cement	2005	3,000
Dredged material	Fraser River-Nelson Pond	2005	19,105
Dredged material	Fraser River-Maintenance	2005	1,200,000
Dredged material	Fraser River-Maintenance	2006	600,000
Dredged material	Fraser River-Cannery Channel	2006	23,522
Dredged material	Fraser River-Husby Forest Products	2006	4,000
Dredged material	Fraser River-Lehigh-Surrey	2007	1,500
Dredged material	Fraser River-Lafarge-COQ	2007	2,400
Dredged material	Fraser River-Lehigh-Tilbury Cement	2007	4,000
Dredged material	Fraser River-CONAG-Delta	2007	2,500
Dredged material	Fraser River-North West Hardwoods	2007	2,000
Dredged material	Fraser River North Arm-No. 8 Road Ramp	2007	1,000
Dredged material	Fraser River-Captains Cove Marina	2007	35,000
Dredged material	Fraser River North Arm-FRPD Yard	2007	700
Dredged material	Fraser River-Deas Pacific Marine	2007	61,900
Dredged material	Fraser River North Arm-Remple Bros	2007	2,000
Dredged material	Fraser River-Tilbury Cement	2007	4,000
Dredged material	Fraser River North Arm-FRPD Yard	2007	300
Dredged material	Fraser River-De Wall's Marina	2008	3,850
Dredged material	Fraser River-Seaspan Coastal Intermodal-Tilbury Island	2008	3,200
Dredged material	Fraser River-Lehigh-Surrey	2008	1,200
Dredged material	Fraser River-Larfarge Canada Pitt River Quarries	2008	1,200
Dredged material	Fraser River-Samson V Moorage Berth	2008	1,200

Dredged material	Fraser River North Arm -No. 8 Road Ramp	2008	1,200
Dredged material	Fraser River-Vancouver Fraser Port Authority	2008	1,199,104
Dredged material	Fraser River-Tilbury Island-Seaspan Coastal Intermodal	2008	1,000
Dredged material	Fraser River-Lehigh-Surrey	2008	2,400
Dredged material	Fraser River North Arm -Shelter Island Marina	2008	4,000
Dredged material	Fraser River-Catalyst Paper-Surrey	2009	2,200
Dredged material	Fraser River-Lafarge Canada Pitt River Quarries	2009	1,200
		Total Volume Disposed	**7,855,533**

Table 2: *Disposal activity at Johnstone Strait-Hanson Island disposal site from 2000 to present (Sean Standing, Environment Canada, Disposal at Sea Program, #201-401 Burrard St., Vancouver, BC V6C 3S5, pers. comm., 2010).*

Substance Disposed	Load Site	Year	Volume Disposed (m^3)
Dredged material	Vancouver Island-Port McNeill-Western-A Frame Booming	2000	3,886
Dredged material	Vancouver Island-Port McNeill-MB-Lower Dry Land	2000	1,237
Dredged material	Vancouver Island-Beaver Cove-CANFOR	2000	1,114
Dredged material	Vancouver Island-Beaver Cove-CANFOR	2001	1,000
Dredged material	Vancouver Island-Beaver Cove-CANFOR	2003	1,680
Dredged material	Vancouver Island-Port McNeill-MB-Lower Dry Land	2003	1,120
Dredged material	Vancouver Island-Port McNeill-Western-A Frame Booming GD	2003	3,360
Dredged material	Vancouver Island-Port McNeill-Western-A Frame Booming GD	2004	2,600
Dredged material	Vancouver Island-Beaver Cove-CANFOR	2005	1,800
Dredged material	Vancouver Island-Beaver Cove-CANFOR-ENGL	2005	1,200
Dredged material	Vancouver Island-Port McNeill-MB-Lower Dry Land	2005	1,200
Dredged material	Vancouver Island-Port McNeill-Western-A Frame Booming GD	2005	3,000
Dredged material	Vancouver Island-Port McNeill-MB-Lower Dry Land	2007	2,700
Dredged material	Vancouver Island-Port McNeill-Western-A Frame Booming GD	2007	1,800
Dredged material	Vancouver Island-Port McNeill-Float plane dock	2007	4,000
Dredged material	Vancouver Island-Beaver Cove-CANFOR-LOG	2008	3,600
Dredged material	Vancouver Island-Port McNeill-MB-Lower Dry Land	2008	1,200
Dredged material	Vancouver Island-Port McNeill-Western-A Frame Booming GD	2008	600
		Total Volume Disposed	**37,097**

Table 3: *Disposal activity at Johnstone Strait-Hickey Point disposal site from 2000 to present (Sean Standing, Environment Canada, Disposal at Sea Program, #201-401 Burrard St., Vancouver, BC V6C 3S5, pers. comm., 2010).*

Substance Disposed	Load Site	Year	Volume Disposed (m^3)
Dredged material	Vancouver Island -Eve River	2000	3,524
Dredged material	Vancouver Island-Kelsey Bay-MB	2000	9,733
Dredged material	Vancouver Island-Kelsey Bay-MB	2001	2,500
Dredged material	Vancouver Island-Port McNeill-Western-A Frame Booming	2001	3,500
Dredged material	Vancouver Island-Port McNeill-MB-Lower Dry Land	2001	500
Dredged material	Vancouver Island-Beaver Cove-FC	2001	500
Dredged material	Vancouver Island-Kelsey Bay-MB	2005	1,600
Dredged material	Vancouver Island-Menzies Bay-Weyerhaeuser	2007	600
Dredged material	Vancouver Island-Kelsey Bay-MB	2007	1,200
Dredged material	Vancouver Island -Eve River	2008	1,800
Dredged material	Vancouver Island-Kelsey Bay-Harbour sort north	2008	1,800
		Total Volume Disposed	**27,257**

APPENDIX III

Table 1: *Sediment and water PCB congener concentrations included in the outer coast area included in the model.*

PCB congener		Sediment concentration ($ng \cdot kg^{-1}$ dry weight)	Variability (SD)	Total water concentration ($ng \cdot L^{-1}$)
PCB	**8**	30.67	43.37	1.49E-03
PCB	**18**	15.10	21.35	5.74E-04
PCB	**28**	35.04	49.55	8.53E-04
PCB	**31**	20.14	28.48	4.26E-04
PCB	**33**	16.60	23.47	4.10E-04
PCB	**44**	15.33	21.68	3.04E-04
PCB	**49**	16.38	23.16	2.77E-04
PCB	**52**	25.12	35.53	4.42E-04
PCB	**56**	20.13	28.46	3.06E-04
PCB	**60**	5.75	8.13	7.75E-05
PCB	**66**	19.56	27.66	3.06E-04
PCB	**70**	25.16	35.58	3.51E-04
PCB	**74**	9.96	14.09	1.38E-04
PCB	**87**	12.32	16.72	1.26E-04
PCB	**95**	26.15	36.97	3.79E-04
PCB	**99**	24.84	32.24	2.49E-04
PCB	**101**	39.38	51.26	4.23E-04
PCB	**105**	14.50	18.67	8.36E-05
PCB	**110**	27.11	36.74	2.90E-04
PCB	**118**	37.95	49.67	2.88E-04
PCB	**128**	7.62	9.38	4.49E-05
PCB	**132**	11.84	15.29	9.68E-05
PCB	**138**	53.96	67.90	1.94E-04
PCB	**141**	4.34	5.51	2.69E-05
PCB	**149**	33.32	44.48	2.46E-04
PCB	**151**	10.46	14.27	7.96E-05
PCB	**153**	48.48	60.18	2.54E-04
PCB	**156**	2.69	3.44	1.27E-05
PCB	**158**	2.60	3.46	1.45E-05
PCB	**170**	9.80	10.85	3.64E-05
PCB	**174**	9.11	10.66	3.96E-05
PCB	**177**	9.13	11.08	4.09E-05
PCB	**180**	14.55	18.13	5.37E-05
PCB	**183**	4.95	5.46	1.94E-05
PCB	**187**	18.91	23.71	7.65E-05
PCB	**194**	4.56	5.20	8.12E-06
PCB	**195**	2.10	2.47	5.41E-06
PCB	**201**	9.26	11.05	2.24E-05

Table 2: *Sediment and water PCB congener concentrations included in the Queen Charlotte Strait area included in the model.*

PCB congener		Sediment concentration ($ng \cdot kg^{-1}$ dry weight)	Variability (SD)	Total water concentration ($ng \cdot L^{-1}$)
PCB	**8**	30.67	43.37	1.49E-03
PCB	**18**	15.10	21.35	5.74E-04
PCB	**28**	35.04	49.55	8.53E-04
PCB	**31**	20.14	28.48	4.26E-04
PCB	**33**	16.60	23.47	4.10E-04
PCB	**44**	15.33	21.68	3.04E-04
PCB	**49**	16.38	23.16	2.77E-04
PCB	**52**	25.12	35.53	4.42E-04
PCB	**56**	20.13	28.46	3.06E-04
PCB	**60**	5.75	8.13	7.75E-05
PCB	**66**	19.56	27.66	3.06E-04
PCB	**70**	25.16	35.58	3.51E-04
PCB	**74**	9.96	14.09	1.38E-04
PCB	**87**	12.32	16.72	1.26E-04
PCB	**95**	26.15	36.97	3.79E-04
PCB	**99**	24.84	32.24	2.49E-04
PCB	**101**	39.38	51.26	4.23E-04
PCB	**105**	14.50	18.67	8.36E-05
PCB	**110**	27.11	36.74	2.90E-04
PCB	**118**	37.95	49.67	2.88E-04
PCB	**128**	7.62	9.38	4.49E-05
PCB	**132**	11.84	15.29	9.68E-05
PCB	**138**	53.96	67.90	1.94E-04
PCB	**141**	4.34	5.51	2.69E-05
PCB	**149**	33.32	44.48	2.46E-04
PCB	**151**	10.46	14.27	7.96E-05
PCB	**153**	48.48	60.18	2.54E-04
PCB	**156**	2.69	3.44	1.27E-05
PCB	**158**	2.60	3.46	1.45E-05
PCB	**170**	9.80	10.85	3.64E-05
PCB	**174**	9.11	10.66	3.96E-05
PCB	**177**	9.13	11.08	4.09E-05
PCB	**180**	14.55	18.13	5.37E-05
PCB	**183**	4.95	5.46	1.94E-05
PCB	**187**	18.91	23.71	7.65E-05
PCB	**194**	4.56	5.20	8.12E-06
PCB	**195**	2.10	2.47	5.41E-06
PCB	**201**	9.26	11.05	2.24E-05

Table 3: *Sediment and water PCB congener concentrations included in the northern resident killer whale Critical Habitat area included in the model.*

PCB congener	Sediment concentration (ng·kg^{-1} dry weight)	Variability (SD)	Total water concentration (ng·L^{-1})
PCB 8	9.76	13.80	3.33E-04
PCB 18	4.83	6.83	1.29E-04
PCB 28	20.03	28.33	3.43E-04
PCB 31	8.80	12.45	1.31E-04
PCB 33	6.18	8.74	1.07E-04
PCB 44	7.39	10.45	1.03E-04
PCB 49	6.83	9.66	8.10E-05
PCB 52	11.29	15.97	1.40E-04
PCB 56	12.66	17.90	1.35E-04
PCB 60	4.48	5.46	4.24E-05
PCB 66	14.72	18.38	1.62E-04
PCB 70	16.58	21.40	1.62E-04
PCB 74	6.74	8.96	6.56E-05
PCB 87	9.60	12.58	6.88E-05
PCB 95	14.85	19.98	1.51E-04
PCB 99	18.83	19.66	1.33E-04
PCB 101	24.51	28.76	1.85E-04
PCB 105	11.15	11.73	4.52E-05
PCB 110	17.96	21.13	1.35E-04
PCB 118	30.53	33.56	1.63E-04
PCB 128	6.55	6.21	2.71E-05
PCB 132	7.74	8.16	4.45E-05
PCB 138	42.99	45.86	1.09E-04
PCB 141	2.86	4.05	1.25E-05
PCB 149	20.85	25.80	1.08E-04
PCB 151	6.16	8.56	3.29E-05
PCB 153	32.25	33.23	1.19E-04
PCB 156	1.83	2.52	6.06E-06
PCB 158	1.71	1.99	6.66E-06
PCB 170	6.07	5.48	1.58E-05
PCB 174	7.68	8.40	2.34E-05
PCB 177	6.12	6.73	1.93E-05
PCB 180	10.01	10.06	2.60E-05
PCB 183	4.56	3.75	1.26E-05
PCB 187	14.85	15.95	4.22E-05
PCB 194	2.89	2.78	3.61E-06
PCB 195	2.21	1.14	4.01E-06
PCB 201	6.79	6.66	1.15E-05

Table 4: *Sediment and water PCB congener concentrations included in the Strait of Georgia area included in the model.*

PCB congener	**Sediment concentration (ng·kg⁻¹ dry weight)**	**Variability (SD)**	**Total water concentration (ng·L⁻¹)**
PCB 8	6.88	217.12	6.68E-04
PCB 18	6.88	75.99	5.24E-04
PCB 28	61.92	54.28	3.01E-03
PCB 44	37.84	108.56	1.50E-03
PCB 49	32.68	130.27	1.10E-03
PCB 52	34.40	173.69	1.21E-03
PCB 66	89.45	54.28	2.80E-03
PCB 74	22.36	43.42	6.20E-04
PCB 95	51.92	74.64	1.51E-03
PCB 99	29.24	119.41	5.87E-04
PCB 101	76.15	111.33	1.63E-03
PCB 105	29.24	119.41	3.37E-04
PCB 110	74.19	98.98	1.59E-03
PCB 118	86.52	124.39	1.31E-03
PCB 128	18.92	119.41	2.23E-04
PCB 138	110.81	134.66	7.98E-04
PCB 149	61.92	162.84	9.15E-04
PCB 151	6.88	303.97	1.05E-04
PCB 153	84.29	119.41	8.83E-04
PCB 156	3.44	54.28	3.23E-05
PCB 170	15.48	271.40	1.15E-04
PCB 177	15.48	173.69	1.39E-04
PCB 180	24.08	97.70	1.78E-04
PCB 183	13.76	0.00	1.08E-04
PCB 187	43.00	86.85	3.48E-04
PCB 194	5.16	0.000	1.84E-05
PCB 203	5.16	0.000	2.42E-05

Table 5: *Sediment and water PCB congener concentrations included in the southern resident killer whale Critical Habitat in Canada area included in the model.*

PCB congener	Sediment concentration (ng·kg^{-1} dry weight)	Variability (SD)	Total water concentration (ng·L^{-1})
PCB 8	3.32	46.02	3.22E-04
PCB 18	3.32	16.11	2.53E-04
PCB 28	29.88	11.51	1.45E-03
PCB 44	18.26	23.01	7.25E-04
PCB 49	15.77	27.61	5.32E-04
PCB 52	16.60	36.82	5.84E-04
PCB 66	43.16	11.51	1.35E-03
PCB 74	10.79	9.20	2.99E-04
PCB 95	26.00	17.93	7.54E-04
PCB 99	14.11	25.31	2.83E-04
PCB 101	38.50	27.35	8.27E-04
PCB 105	14.11	25.31	1.63E-04
PCB 110	38.83	23.54	8.30E-04
PCB 118	45.00	33.36	6.84E-04
PCB 128	9.13	25.31	1.08E-04
PCB 138	56.67	36.90	4.08E-04
PCB 149	29.88	34.52	4.42E-04
PCB 151	3.32	64.43	5.05E-05
PCB 153	40.67	25.31	4.26E-04
PCB 156	1.66	11.51	1.56E-05
PCB 170	7.47	57.53	5.55E-05
PCB 177	7.47	36.82	6.69E-05
PCB 180	11.62	20.71	8.58E-05
PCB 183	6.64	0.00	5.21E-05
PCB 187	20.75	18.41	1.68E-04
PCB 194	2.49	0.00	8.88E-06
PCB 203	2.49	0.00	1.17E-05

Table 6: *Sediment and water PCB congener concentrations included in the southern resident killer whale Critical Habitat in the USA (summer core & Juan de Fuca Strait) area included in the model.*

PCB congener		Sediment concentration (ng·kg^{-1} dry weight)	Variability (SD)	Total water concentration (ng·L^{-1})
PCB	**8**	1.60E+02		2.04E-02
PCB	**18**	1.60E+02		1.60E-02
PCB	**28**	1.70E+02		1.09E-02
PCB	**44**	2.30E+02		1.20E-02
PCB	**49**	1.42E+02	1.42E+02	6.32E-03
PCB	**52**	3.20E+02	1.41E+01	1.48E-02
PCB	**66**	1.60E+02		6.59E-03
PCB	**74**	1.33E+02	4.75E+01	4.87E-03
PCB	**95**			0.00E+00
PCB	**99**	1.27E+02		3.36E-03
PCB	**101**	3.35E+02	3.54E+01	9.46E-03
PCB	**105**	2.90E+02		4.40E-03
PCB	**110**	3.10E+02		8.72E-03
PCB	**118**	3.55E+02	2.12E+01	7.10E-03
PCB	**128**	2.40E+02		3.72E-03
PCB	**138**	3.63E+02	1.16E+02	3.44E-03
PCB	**149**	2.69E+02	1.78E+02	5.24E-03
PCB	**151**	2.99E+01		6.00E-04
PCB	**153**	3.67E+02	1.31E+02	5.05E-03
PCB	**156**	1.50E+01	5.93E+01	1.85E-04
PCB	**170**	6.75E+02	1.34E+02	6.60E-03
PCB	**177**	6.73E+01		7.94E-04
PCB	**180**	6.50E+02		6.31E-03
PCB	**183**	5.99E+01		6.19E-04
PCB	**187**	3.85E+02	9.19E+01	4.10E-03
PCB	**194**	2.24E+01		1.05E-04
PCB	**203**	2.24E+01		1.39E-04

Table 7: *Sediment and water PCB congener concentrations included in the southern resident killer whale Critical Habitat in the USA (Puget Sound) area included in the model.*

PCB congener		Sediment concentration (ng·kg^{-1} dry weight)	Variability (SD)	Total water concentration (ng·L^{-1})
PCB	**8**	2.90E+02	1.00E+00	2.43E-02
PCB	**18**	5.52E+02	1.15E+00	3.62E-02
PCB	**28**	3.36E+03	5.52E+00	1.41E-01
PCB	**44**	6.09E+03	2.13E+01	2.09E-01
PCB	**49**	1.54E+03	1.60E+01	4.48E-02
PCB	**52**	3.29E+03	4.55E+00	1.00E-01
PCB	**66**	3.53E+03	2.23E+01	9.54E-02
PCB	**74**	1.05E+03	5.33E+00	2.52E-02
PCB	**95**	0.00E+00	0.00E+00	0.00E+00
PCB	**99**	1.38E+03	1.47E+01	2.39E-02
PCB	**101**	4.30E+03	1.14E+01	7.96E-02
PCB	**105**	8.34E+03	1.33E+01	8.29E-02
PCB	**110**	6.70E+02	1.00E+00	1.24E-02
PCB	**118**	2.76E+03	1.02E+01	3.61E-02
PCB	**128**	4.17E+03	1.82E+01	4.24E-02
PCB	**138**	4.15E+03	1.13E+01	2.58E-02
PCB	**149**	2.92E+03	2.00E+01	3.72E-02
PCB	**151**	3.24E+02	3.73E+01	4.26E-03
PCB	**153**	3.97E+03	1.47E+01	3.59E-02
PCB	**156**	1.62E+02	6.66E+00	1.31E-03
PCB	**170**	4.88E+03	1.35E+01	3.12E-02
PCB	**177**	7.29E+02	2.13E+01	5.63E-03
PCB	**180**	4.66E+03	2.48E+01	2.96E-02
PCB	**183**	6.48E+02	0.00E+00	4.39E-03
PCB	**187**	1.01E+04	1.61E+01	7.07E-02
PCB	**194**	2.43E+02	0.00E+00	7.47E-04
PCB	**203**	2.43E+02	0.00E+00	9.84E-04

APPENDIX IV

Table 1: *Predicted PCB congener concentrations (ng·kg^{-1} wet weight) of fish-diet items for resident killer whales included in the model to represent the outer coast area included in the model.*

PCB congener	Chinook salmon	Halibut	Sablefish
	Predicted Concentration (ng·kg^{-1} wet weight)	Predicted Concentration (ng·kg^{-1} wet weight)	Predicted Concentration (ng·kg^{-1} wet weight)
PCB 8	256.59	163.17	170.93
PCB 18	194.36	118.61	120.95
PCB 28	949.46	528.87	511.08
PCB 31	676.46	366.15	347.89
PCB 33	439.85	245.73	237.87
PCB 44	565.69	302.35	285.15
PCB 49	757.08	393.01	364.08
PCB 52	1097.84	573.97	534.09
PCB 56	1063.70	543.22	497.98
PCB 60	351.34	176.51	160.02
PCB 66	996.86	511.28	470.02
PCB 70	1476.98	745.24	677.66
PCB 74	588.84	296.88	269.82
PCB 87	997.74	488.63	433.98
PCB 95	1471.27	745.65	680.35
PCB 99	2039.47	997.99	885.54
PCB 101	3051.29	1498.84	1335.48
PCB 105	1485.20	742.18	645.98
PCB 110	2108.34	1035.39	922.30
PCB 118	3714.63	1814.14	1588.99
PCB 128	793.65	394.16	343.06
PCB 132	1128.72	549.19	482.28
PCB 138	4182.10	2306.99	2017.09
PCB 141	451.48	222.95	194.21
PCB 149	3332.52	1625.64	1422.28
PCB 151	1033.33	503.52	440.97
PCB 153	4956.14	2502.96	2176.19
PCB 156	263.19	135.58	117.90
PCB 158	269.40	134.83	117.27
PCB 170	808.86	439.62	383.69
PCB 174	866.40	451.94	393.09
PCB 177	885.78	458.96	399.06
PCB 180	1193.27	649.73	567.17
PCB 183	432.47	231.40	201.65
PCB 187	1699.95	902.43	785.87
PCB 194	87.65	63.59	59.00
PCB 195	101.40	62.27	55.38
PCB 201	389.85	246.03	220.13
Total	47159.14	24169.61	21516.45

Table 2: *Predicted PCB congener concentrations (ng·kg^{-1} wet weight) of fish-diet items for resident killer whales included in the model to represent the Queen Charlotte Strait area included in the model.*

PCB congener	Chinook salmon Predicted Concentration (ng·kg^{-1} wet weight)	Halibut Predicted Concentration (ng·kg^{-1} wet weight)	Sablefish Predicted Concentration (ng·kg^{-1} wet weight)
PCB 8	182.59	159.35	170.93
PCB 18	130.98	114.85	120.95
PCB 28	575.30	503.23	511.08
PCB 31	397.59	346.53	347.89
PCB 33	267.37	233.95	237.87
PCB 44	328.22	285.47	285.15
PCB 49	426.72	369.03	364.08
PCB 52	623.10	539.69	534.09
PCB 56	590.26	508.46	497.98
PCB 60	192.05	164.68	160.02
PCB 66	555.42	478.97	470.02
PCB 70	810.50	695.90	677.66
PCB 74	322.90	277.19	269.82
PCB 87	533.69	453.37	433.98
PCB 95	810.69	696.93	680.35
PCB 99	1090.26	925.79	885.54
PCB 101	1635.94	1391.77	1335.48
PCB 105	814.92	688.16	645.98
PCB 110	1130.15	961.36	922.30
PCB 118	1988.27	1679.58	1588.99
PCB 128	432.88	365.24	343.06
PCB 132	601.59	508.43	482.28
PCB 138	2534.03	2159.74	2017.09
PCB 141	244.78	206.49	194.21
PCB 149	1782.56	1504.65	1422.28
PCB 151	551.96	466.05	440.97
PCB 153	2749.95	2322.84	2176.19
PCB 156	148.99	126.07	117.90
PCB 158	148.10	125.02	117.27
PCB 170	483.11	410.94	383.69
PCB 174	496.78	420.72	393.09
PCB 177	504.50	426.97	399.06
PCB 180	714.00	607.46	567.17
PCB 183	254.33	215.97	201.65
PCB 187	991.89	841.57	785.87
PCB 194	69.42	61.01	59.00
PCB 195	68.33	58.89	55.38
PCB 201	269.91	233.28	220.13
Total	26454.03	22535.61	21516.45

Table 3: *Predicted PCB congener concentrations (ng·kg^{-1} wet weight) of fish-diet items for resident killer whales included in the model to represent the northern resident killer whale Critical Habitat area included in the model.*

PCB congener	Chinook salmon	Halibut	Sablefish
	Predicted Concentration (ng·kg^{-1} wet weight)	Predicted Concentration (ng·kg^{-1} wet weight)	Predicted Concentration (ng·kg^{-1} wet weight)
PCB 8	36.13	31.57	34.71
PCB 18	26.17	23.06	24.85
PCB 28	207.14	183.43	190.13
PCB 31	109.71	96.98	99.25
PCB 33	62.66	55.50	57.60
PCB 44	99.94	88.22	89.77
PCB 49	112.57	98.95	99.26
PCB 52	177.15	155.90	156.96
PCB 56	235.08	205.92	204.79
PCB 60	94.84	82.70	81.48
PCB 66	264.80	232.18	231.46
PCB 70	338.38	295.47	291.85
PCB 74	138.37	120.80	119.26
PCB 87	264.23	228.01	220.53
PCB 95	291.60	254.93	252.55
PCB 99	524.87	452.68	437.43
PCB 101	646.28	558.70	541.99
PCB 105	406.58	346.58	327.49
PCB 110	475.28	410.81	398.43
PCB 118	1022.47	874.88	834.13
PCB 128	240.87	205.21	194.02
PCB 132	250.68	214.77	205.41
PCB 138	1373.83	1174.67	1104.90
PCB 141	104.33	88.92	84.20
PCB 149	713.62	609.92	580.90
PCB 151	207.56	177.52	169.27
PCB 153	1194.92	1017.52	959.47
PCB 156	66.97	57.05	53.70
PCB 158	63.15	53.78	50.77
PCB 170	202.94	173.22	162.87
PCB 174	278.57	237.22	223.11
PCB 177	224.36	191.00	179.69
PCB 180	333.23	284.46	267.47
PCB 183	157.87	134.62	126.56
PCB 187	522.45	445.30	418.65
PCB 194	32.95	28.85	28.19
PCB 195	51.37	44.24	41.95
PCB 201	142.33	122.85	116.93
Total	11696.22	10058.40	9661.99

Table 4: *Predicted PCB congener concentrations (ng·kg^{-1} wet weight) of fish-diet items for resident killer whales included in the model to represent the Strait of Georgia area included in the model.*

PCB congener	Chinook salmon Predicted Concentration (ng·kg^{-1} wet weight)	Halibut Predicted Concentration (ng·kg^{-1} wet weight)	Sablefish Predicted Concentration (ng·kg^{-1} wet weight)
PCB 8	41.39	56.23	61.54
PCB 18	62.79	83.45	89.31
PCB 28	1190.00	1476.59	1519.86
PCB 44	1000.14	1196.96	1210.71
PCB 49	1095.11	1274.27	1271.69
PCB 52	1086.22	1272.91	1274.66
PCB 66	3328.87	3824.96	3794.30
PCB 74	978.24	1102.41	1083.38
PCB 95	2148.53	2437.96	2403.95
PCB 99	1858.47	2004.18	1926.83
PCB 101	4518.64	4912.57	4742.78
PCB 105	2649.00	2717.52	2539.73
PCB 110	4421.59	4804.56	4637.33
PCB 118	6935.58	7268.20	6879.67
PCB 128	1723.50	1771.03	1656.85
PCB 138	9150.42	9146.44	8428.93
PCB 149	5085.64	5311.84	5023.48
PCB 151	554.28	580.48	549.78
PCB 153	7833.47	7987.70	7437.34
PCB 156	318.23	322.44	299.07
PCB 170	1333.72	1335.18	1230.89
PCB 177	1442.24	1457.63	1349.98
PCB 180	2066.68	2068.32	1906.56
PCB 183	1222.85	1227.50	1132.88
PCB 187	3874.90	3895.47	3597.67
PCB 194	161.51	157.86	148.64
PCB 203	274.00	269.39	249.14
Total	66355.99	69964.02	66446.94

Table 5: *Predicted PCB congener concentrations (ng·kg⁻¹ wet weight) of fish-diet items for resident killer whales included in the model to represent the southern resident killer whale Critical Habitat in Canada area included in the model.*

PCB congener	Chinook salmon	Halibut	Sablefish
	Predicted Concentration ($ng \cdot kg^{-1}$ wet weight)	Predicted Concentration ($ng \cdot kg^{-1}$ wet weight)	Predicted Concentration ($ng \cdot kg^{-1}$ wet weight)
PCB 8	19.97	27.13	29.69
PCB 18	30.30	40.26	43.09
PCB 28	574.16	712.44	733.32
PCB 44	482.55	577.52	584.15
PCB 49	528.38	614.82	613.58
PCB 52	524.09	614.16	615.01
PCB 66	1606.14	1845.50	1830.70
PCB 74	471.99	531.90	522.72
PCB 95	1076.03	1220.98	1203.94
PCB 99	896.69	966.99	929.67
PCB 101	2284.49	2483.65	2397.81
PCB 105	1278.11	1311.17	1225.39
PCB 110	2314.31	2514.76	2427.23
PCB 118	3607.09	3780.08	3578.01
PCB 128	831.57	854.50	799.41
PCB 138	4679.34	4677.31	4310.39
PCB 149	2453.79	2562.96	2423.82
PCB 151	267.48	280.16	265.32
PCB 153	3779.55	3853.97	3588.43
PCB 156	153.54	155.57	144.30
PCB 170	643.50	644.21	593.89
PCB 177	695.86	703.29	651.35
PCB 180	997.15	997.94	919.89
PCB 183	590.01	592.25	546.60
PCB 187	1869.59	1879.51	1735.83
PCB 194	77.93	76.17	71.72
PCB 203	132.20	129.98	120.21
Total	32865.80	34649.15	32905.44

Table 6: *Predicted PCB congener concentrations (ng·kg^{-1} wet weight) of fish-diet items for resident killer whales included in the model to represent the southern resident killer whale Critical Habitat in the USA (summer core & Juan de Fuca Strait) area included in the model.*

PCB congener	**Chinook salmon**	**Halibut**	**Sablefish**
	Predicted Concentration (ng·kg^{-1} wet weight)	**Predicted Concentration (ng·kg^{-1} wet weight)**	**Predicted Concentration (ng·kg^{-1} wet weight)**
PCB 8	1.01.E+03	2.07E+03	2.24E+03
PCB 18	1.61.E+03	3.12E+03	3.30E+03
PCB 28	3.87.E+03	6.67E+03	6.81E+03
PCB 44	7.41.E+03	1.21E+04	1.22E+04
PCB 49	5.93.E+03	9.36E+03	9.29E+03
PCB 52	1.25.E+04	1.99E+04	1.98E+04
PCB 66	7.48.E+03	1.16E+04	1.15E+04
PCB 74	7.44.E+03	1.13E+04	1.10E+04
PCB 95	0.00.E+00	0.00E+00	0.00E+00
PCB 99	1.07.E+04	1.54E+04	1.47E+04
PCB 101	2.61.E+04	3.79E+04	3.65E+04
PCB 105	3.59.E+04	4.82E+04	4.48E+04
PCB 110	2.42.E+04	3.52E+04	3.39E+04
PCB 118	3.84.E+04	5.33E+04	5.02E+04
PCB 128	2.98.E+04	4.01E+04	3.73E+04
PCB 138	4.14.E+04	5.20E+04	4.74E+04
PCB 149	2.99.E+04	4.13E+04	3.89E+04
PCB 151	3.25.E+03	4.51E+03	4.26E+03
PCB 153	4.66.E+04	6.19E+04	5.72E+04
PCB 156	1.90.E+03	2.48E+03	2.28E+03
PCB 170	7.99.E+04	1.01E+05	9.22E+04
PCB 177	8.60.E+03	1.12E+04	1.03E+04
PCB 180	7.67.E+04	9.71E+04	8.84E+04
PCB 183	7.31.E+03	9.33E+03	8.52E+03
PCB 187	4.76.E+04	6.11E+04	5.58E+04
PCB 194	1.01.E+03	1.14E+03	1.04E+03
PCB 203	1.67.E+03	1.97E+03	1.78E+03
Total	5.58.E+05	7.51.E+05	7.02.E+05

Table 7: *Predicted PCB congener concentrations (ng·kg^{-1} wet weight) of fish-diet items for resident killer whales included in the model to represent the southern resident killer whale Critical Habitat in the USA (Puget Sound) area included in the model.*

PCB congener	Chinook salmon	Halibut	Sablefish
	Predicted Concentration (ng·kg^{-1} wet weight)	**Predicted Concentration (ng·kg^{-1} wet weight)**	**Predicted Concentration (ng·kg^{-1} wet weight)**
PCB 8	1.41E+03	2.74E+03	2.96E+03
PCB 18	4.18E+03	7.64E+03	8.04E+03
PCB 28	5.35E+04	8.72E+04	8.82E+04
PCB 44	1.32E+05	2.05E+05	2.03E+05
PCB 49	4.15E+04	6.26E+04	6.14E+04
PCB 52	8.41E+04	1.28E+05	1.26E+05
PCB 66	1.05E+05	1.56E+05	1.52E+05
PCB 74	3.64E+04	5.28E+04	5.11E+04
PCB 95	0.00E+00	0.00E+00	0.00E+00
PCB 99	6.68E+04	9.24E+04	8.79E+04
PCB 101	1.96E+05	2.74E+05	2.61E+05
PCB 105	5.52E+05	7.10E+05	6.64E+05
PCB 110	3.06E+04	4.28E+04	4.08E+04
PCB 118	1.64E+05	2.19E+05	2.06E+05
PCB 128	2.77E+05	3.58E+05	3.35E+05
PCB 138	2.46E+05	2.95E+05	2.74E+05
PCB 149	1.77E+05	2.35E+05	2.22E+05
PCB 151	1.94E+04	2.58E+04	2.43E+04
PCB 153	2.68E+05	3.41E+05	3.18E+05
PCB 156	1.08E+04	1.36E+04	1.26E+04
PCB 170	3.01E+05	3.64E+05	3.38E+05
PCB 177	4.89E+04	6.09E+04	5.67E+04
PCB 180	2.86E+05	3.45E+05	3.21E+05
PCB 183	4.13E+04	5.04E+04	4.69E+04
PCB 187	6.56E+05	8.04E+05	7.48E+05
PCB 194	5.29E+03	5.63E+03	5.40E+03
PCB 203	9.12E+03	1.02E+04	9.56E+03
Total	3.81E+06	4.95E+06	4.67E+06

Table 8: *Data for PCB congener concentrations predicted in male killer whale with the PCB food web bioaccumulation model using initial diet and new diet compositions for the northern resident killer whale Critical Habitat.*

	NRKW Critical Habitat (male)			
PCB congeners	**Previous Diet**	**New Diet**	**Previous Diet**	**New Diet**
	PCB μg·kg^{-1} lipid	**PCB μg·kg^{-1} lipid**	**Log PCB μg·kg^{-1} lipid**	**Log PCB μg·kg^{-1} lipid**
8	3.19.E+01	2.91.E+01	1.50	1.46
18	2.29.E+01	2.10.E+01	1.36	1.32
28	2.15.E+02	1.98.E+02	2.33	2.30
31	1.16.E+02	1.07.E+02	2.07	2.03
33	6.66.E+01	6.13.E+01	1.82	1.79
44	1.12.E+02	1.03.E+02	2.05	2.01
49	1.23.E+02	1.13.E+02	2.09	2.05
52	1.94.E+02	1.78.E+02	2.29	2.25
56	2.66.E+02	2.43.E+02	2.42	2.39
60	1.08.E+02	9.83.E+01	2.03	1.99
66	3.02.E+02	2.76.E+02	2.48	2.44
70	3.84.E+02	3.49.E+02	2.58	2.54
74	1.57.E+02	1.43.E+02	2.20	2.16
87	3.01.E+02	2.70.E+02	2.48	2.43
95	3.31.E+02	3.02.E+02	2.52	2.48
99	5.98.E+02	5.36.E+02	2.78	2.73
101	7.32.E+02	6.59.E+02	2.86	2.82
105	4.63.E+02	3.99.E+02	2.67	2.60
110	5.41.E+02	4.87.E+02	2.73	2.69
118	1.17.E+03	1.03.E+03	3.07	3.01
128	2.75.E+02	2.38.E+02	2.44	2.38
132	2.86.E+02	2.53.E+02	2.46	2.40
138	1.56.E+03	1.29.E+03	3.19	3.11
141	1.19.E+02	1.03.E+02	2.08	2.01
149	8.14.E+02	7.16.E+02	2.91	2.86
151	2.37.E+02	2.09.E+02	2.37	2.32
153	1.36.E+03	1.16.E+03	3.13	3.07
156	7.63.E+01	6.47.E+01	1.88	1.81
158	7.20.E+01	6.20.E+01	1.86	1.79
170	2.31.E+02	1.92.E+02	2.36	2.28
174	3.17.E+02	2.67.E+02	2.50	2.43
177	2.56.E+02	2.16.E+02	2.41	2.33
180	3.79.E+02	3.14.E+02	2.58	2.50
183	1.80.E+02	1.50.E+02	2.25	2.18
187	5.95.E+02	4.98.E+02	2.77	2.70
194	3.68.E+01	2.92.E+01	1.57	1.47
195	5.81.E+01	4.68.E+01	1.76	1.67
201	1.61.E+02	1.29.E+02	2.21	2.11
Mean	**3.48E+02**	**3.04E+02**	**2.34**	**2.29**
SD	**3.59E+02**	**3.07E+02**	**0.44**	**0.44**

Table 9: *Data for PCB congener concentrations predicted in female killer whale with the PCB food web bioaccumulation model using initial diet and new diet compositions for the northern resident killer whale Critical Habitat.*

PCB congeners	NRKW Critical Habitat (female)			
	Previous Diet	New Diet	Previous Diet	New Diet
	PCB μg·kg^{-1} lipid	PCB μg·kg^{-1} lipid	Log PCB μg·kg^{-1} lipid	Log PCB μg·kg^{-1} lipid
8	6.07E+00	5.53E+00	0.78	0.74
18	4.38E+00	4.02E+00	0.64	0.60
28	3.60E+01	3.32E+01	1.56	1.52
31	1.91E+01	1.76E+01	1.28	1.25
33	1.09E+01	1.01E+01	1.04	1.00
44	1.76E+01	1.62E+01	1.25	1.21
49	1.97E+01	1.81E+01	1.30	1.26
52	3.11E+01	2.85E+01	1.49	1.46
56	4.15E+01	3.79E+01	1.62	1.58
60	1.68E+01	1.52E+01	1.22	1.18
66	4.68E+01	4.28E+01	1.67	1.63
70	5.97E+01	5.44E+01	1.78	1.74
74	2.44E+01	2.23E+01	1.39	1.35
87	4.66E+01	4.19E+01	1.67	1.62
95	5.15E+01	4.70E+01	1.71	1.67
99	9.27E+01	8.31E+01	1.97	1.92
101	1.14E+02	1.03E+02	2.06	2.01
105	7.16E+01	6.17E+01	1.85	1.79
110	8.39E+01	7.55E+01	1.92	1.88
118	1.80E+02	1.59E+02	2.26	2.20
128	4.24E+01	3.67E+01	1.63	1.56
132	4.42E+01	3.92E+01	1.65	1.59
138	2.40E+02	1.99E+02	2.38	2.30
141	1.84E+01	1.60E+01	1.26	1.20
149	1.26E+02	1.11E+02	2.10	2.04
151	3.66E+01	3.23E+01	1.56	1.51
153	2.10E+02	1.80E+02	2.32	2.26
156	1.18E+01	9.98E+00	1.07	1.00
158	1.11E+01	9.56E+00	1.05	0.98
170	3.55E+01	2.95E+01	1.55	1.47
174	4.89E+01	4.12E+01	1.69	1.62
177	3.94E+01	3.33E+01	1.60	1.52
180	5.84E+01	4.85E+01	1.77	1.69
183	2.77E+01	2.31E+01	1.44	1.36
187	9.17E+01	7.68E+01	1.96	1.89
194	5.64E+00	4.49E+00	0.75	0.65
195	8.94E+00	7.20E+00	0.95	0.86
201	2.47E+01	1.98E+01	1.39	1.30
Mean	**5.41E+01**	**4.72E+01**	**1.54**	**1.48**
SD	**5.54E+01**	**4.73E+01**	**0.43**	**0.43**

Table 10: *Data for PCB congener concentrations predicted in male killer whale with the PCB food web bioaccumulation model using initial diet and new diet compositions for the Outer coast habitat.*

	Outer Coast (male)			
PCB congeners	**Previous Diet**	**New Diet**	**Previous Diet**	**New Diet**
	PCB µg·kg^{-1} lipid	**PCB µg·kg^{-1} lipid**	**Log PCB µg·kg^{-1} lipid**	**Log PCB µg·kg^{-1} lipid**
8	2.24E+02	1.47E+02	2.351	2.168
18	1.68E+02	1.05E+02	2.226	2.022
28	9.74E+02	5.49E+02	2.989	2.739
31	7.07E+02	3.85E+02	2.849	2.586
33	4.61E+02	2.60E+02	2.664	2.416
44	6.26E+02	3.36E+02	2.797	2.527
49	8.15E+02	4.23E+02	2.911	2.627
52	1.19E+03	6.21E+02	3.074	2.793
56	1.19E+03	6.04E+02	3.074	2.781
60	3.95E+02	1.97E+02	2.596	2.294
66	1.12E+03	5.74E+02	3.050	2.759
70	1.65E+03	8.28E+02	3.217	2.918
74	6.60E+02	3.31E+02	2.820	2.520
87	1.12E+03	5.39E+02	3.049	2.732
95	1.65E+03	8.30E+02	3.216	2.919
99	2.29E+03	1.10E+03	3.360	3.042
101	3.40E+03	1.65E+03	3.532	3.217
105	1.67E+03	7.96E+02	3.223	2.901
110	2.36E+03	1.14E+03	3.374	3.058
118	4.17E+03	1.98E+03	3.620	3.296
128	8.93E+02	4.24E+02	2.951	2.628
132	1.27E+03	6.02E+02	3.104	2.780
138	4.69E+03	2.38E+03	3.671	3.376
141	5.08E+02	2.41E+02	2.706	2.382
149	3.75E+03	1.77E+03	3.574	3.249
151	1.16E+03	5.50E+02	3.065	2.741
153	5.57E+03	2.67E+03	3.746	3.426
156	2.96E+02	1.43E+02	2.471	2.157
158	3.03E+02	1.45E+02	2.482	2.160
170	9.09E+02	4.56E+02	2.959	2.659
174	9.75E+02	4.76E+02	2.989	2.678
177	9.97E+02	4.85E+02	2.999	2.685
180	1.34E+03	6.74E+02	3.128	2.829
183	4.87E+02	2.41E+02	2.687	2.383
187	1.91E+03	9.44E+02	3.282	2.975
194	9.74E+01	6.20E+01	1.989	1.792
195	1.14E+02	6.25E+01	2.056	1.796
201	4.37E+02	2.45E+02	2.640	2.390
Mean	**1.38E+03**	**6.83E+02**	**2.96E+00**	**2.67E+00**
SD	**1.32.E+03**	**6.32.E+02**	**4.24.E-01**	**4.02.E-01**

Table 11: *Data for PCB congener concentrations predicted in female killer whale with the PCB food web bioaccumulation model using initial diet and new diet compositions for the Outer coast habitat.*

	Outer Coast (Female)			
PCB congeners	**Previous Diet**	**New Diet**	**Previous Diet**	**New Diet**
	PCB μg·kg^{-1} lipid	**PCB μg·kg^{-1} lipid**	**Log PCB μg·kg^{-1} lipid**	**Log PCB μg·kg^{-1} lipid**
8	5.01E+01	3.29E+01	1.700	1.517
18	3.78E+01	2.36E+01	1.577	1.373
28	1.91E+02	1.08E+02	2.282	2.032
31	1.37E+02	7.45E+01	2.136	1.872
33	8.90E+01	5.03E+01	1.949	1.701
44	1.16E+02	6.21E+01	2.063	1.793
49	1.54E+02	7.98E+01	2.187	1.902
52	2.23E+02	1.17E+02	2.348	2.067
56	2.17E+02	1.11E+02	2.337	2.044
60	7.18E+01	3.58E+01	1.856	1.554
66	2.04E+02	1.04E+02	2.310	2.019
70	3.02E+02	1.51E+02	2.480	2.180
74	1.20E+02	6.03E+01	2.080	1.781
87	2.04E+02	9.81E+01	2.309	1.992
95	3.01E+02	1.52E+02	2.478	2.181
99	4.16E+02	2.00E+02	2.620	2.302
101	6.22E+02	3.01E+02	2.794	2.479
105	3.03E+02	1.44E+02	2.481	2.159
110	4.31E+02	2.08E+02	2.634	2.319
118	7.58E+02	3.59E+02	2.880	2.555
128	1.62E+02	7.69E+01	2.209	1.886
132	2.30E+02	1.09E+02	2.363	2.039
138	8.50E+02	4.31E+02	2.929	2.634
141	9.21E+01	4.36E+01	1.964	1.640
149	6.80E+02	3.22E+02	2.833	2.507
151	2.11E+02	9.98E+01	2.324	1.999
153	1.01E+03	4.84E+02	3.005	2.685
156	5.36E+01	2.60E+01	1.729	1.414
158	5.49E+01	2.62E+01	1.740	1.418
170	1.65E+02	8.25E+01	2.216	1.916
174	1.77E+02	8.62E+01	2.247	1.935
177	1.81E+02	8.77E+01	2.257	1.943
180	2.43E+02	1.22E+02	2.386	2.086
183	8.81E+01	4.37E+01	1.945	1.640
187	3.46E+02	1.71E+02	2.539	2.233
194	1.75E+01	1.12E+01	1.244	1.048
195	2.05E+01	1.13E+01	1.313	1.053
201	7.89E+01	4.43E+01	1.897	1.647
Mean	**2.53E+02**	**1.25E+02**	**2.23E+00**	**1.94E+00**
SD	**2.38.E+02**	**1.14.E+02**	**4.17.E-01**	**3.96.E-01**