DEVELOPMENT AND APPLICATION OF A METHODOLOGY FOR THE DERIVATION OF SEDIMENT QUALITY CRITERIA IN BRITISH COLUMBIA, CANADA

by

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ABSTRACT

Although bioaccumulation of polychlorinated biphenyls (PCBs) in food webs is well-recognized, this phenomenon is not currently incorporated into the methodology for developing sediment quality guidelines (SQGs) in British Columbia. The aim of this study is to develop and apply an empirical and modelling approach to the development of sediment quality criteria for the protection of marine mammals, using harbour seal pups as a proxy. An average empirical Biota-Sediment Accumulation Factor (BSAF) of 490 (lower SD 220, upper SD 1100) g-dry-weight sediment/ g-lipid-weight biota (n=6) was calculated for ΣPCBs, which was overestimated by the model predication (BSAF of 1000 g-dry-weight sediment/ g-lipid-weight biota). Using the empirical BSAF and a Threshold Effects Concentration (TEC) of 1.3 μg PCB/g lipid, a SQG for ΣPCBs of 0.82 ng/g dw is proposed in order to protect 95% of the pup seal population, which is not being achieved by the current SQG of 20 ng/g dw.

Keywords: bioaccumulation; PCBs; Burrard Inlet; food web; sediment quality guideline; risk assessment

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BSAF	Biota-Sediment Accumulation Factor
CAEL	Canadian Association for Environmental Analytical Laboratories Inc.
CEPA	Canadian Environmental Protection Act
DDT	dichlorodiphenyltrichloroethane
F&OC	Fisheries and Oceans Canada
GC/HRMS	Gas Chromatography/High-Resolution Mass Spectrometry
IOS	Institute of Ocean Sciences (Fisheries and Oceans Canada)
MB	Model Bias
MOE	Ministry of the Environment
NK	natural killer (cell)
NOAEL	No Observable Adverse Effects Level
OC	Organic Carbon
PBDE	polybrominated diphenyl ether
PCB	polychlorinated biphenyl
POP	Persistent Organic Pollutant
REM	Resource and Environmental Management, School of (Simon Fraser University)
SQG	Sediment Quality Guideline
TEC	Threshold Effects Concentration
UN	United Nations

INTRODUCTION

Persistent organic pollutants (POPs) have been recognized as a primary threat to marine life around the world (UNEP 1995), and are of special concern in highly productive and ecologically diverse near-shore waters. While the use of legacy POPs, including industrial chemicals such as polychlorinated biphenyls (PCBs) and organochlorine pesticides such as dichlorodiphenyltrichloroethane (DDT), has been phased out in most developed nations, it is concerning these chemicals continue to be used for a variety of industrial and agricultural applications in developing countries worldwide. Due to their impact on environment and human health, twelve POPs were targeted by the United Nations (UN) as a priority for global action in the reduction or elimination of their emission under the Stockholm Convention (UN 2004). A further nine POPs, including some polybrominated diphenyl ether (PBDE) congeners, were listed under the amended Convention in May 2009 (UN 2009). At the national level, the *Canadian Environmental Protection Act (CEPA)*,*1999,* requires that environmental quality criteria be developed for the purpose of pollution prevention or control of toxic substances. Despite these regulatory actions, legacy POPs are still frequently detected at concentrations of concern in marine biota, and many newly developed or 'emerging' POPs are largely unregulated on a global, national, or regional scale.

PCBs belong to the POP family due to their lipophilic nature, along with their resistance to metabolism and environmental degradation, and their ability to cause harm to the environment. Their prevalent global distribution is solely the result of anthropogenic activity. They have historically been used in a wide range of applications due to their chemical stability and miscibility with organic compounds. Industrial mixtures of specific congener compositions have been utilized in heat transfer and hydraulic systems, in the formation of cutting and lubricating oils, in pesticides, paints, plastics, adhesives, inks, as wax extenders, dedusting agents, flame retardants, and in the electrical industry as dielectric fluids in capacitors and transformers (Hutchinson 1994). Such widespread use, however, has allowed PCBs to become ubiquitous environmental contaminants.

PCBs were never manufactured in Canada. However, approximately 40 000 tonnes were imported into the country from the United States (CCREM 1986). Of this, only 60 percent are accounted for in disposal storage or in active use in electrical applications (CCREM 1987). This leaves a significant amount (nearly 16 000 tonnes) of PCBs that are unaccounted for and assumed to be present in the environment, and thus may potentially enter marine ecosystems and bioaccumulate through the aquatic food chain. Furthermore, PCBs are characterized as having the potential to undergo long-range atmospheric transport across international boundaries. This means that domestic sources are proliferated by use and distribution of these contaminants worldwide (Ross *et al.* 2004). Not surprisingly, *chronic* exposure of marine mammals to PCBs is regarded to be more of a problem than acute exposure, primarily because wildlife are more commonly exposed to low-levels of these contaminants for extended periods of time in their natural settings.

Organochlorine contaminants, such as PCBs, have been linked to adverse affects associated with endocrine disruption in marine mammals (Brouwer *et al.* 1989). Both field and laboratory experiments have linked hormonal modulation in organochlorine-exposed organisms to endocrine disruption, and thyroid hormones are more commonly being utilized as sensitive biomarkers (biochemical measurements that indicate toxic effect) for studies of toxic responses to chemical exposure (Chiba *et al.* 2001). Such hormones play important roles in DNA transcription, differentiation of tissues, regulation of growth, and metabolic processes (Chiba *et al.* 2001).

Chlorinated contaminants have been shown to induce deficiencies in both plasma thyroid hormones and vitamin A (retinol) levels in captive and free-ranging seal species (Brouwer *et al.* 1989; Jenssen *et al.* 1995), and increases in the receptor expression of circulatory vitamin A and thyroid hormones in seal blubber (Tabuchi *et al.* 2006; Mos *et al.* 2007). Hypothyroidism is associated with adverse affects such as decreased basic metabolic rate, impairment of growth, and higher sensitivity to cold (Chiba *et al.* 2001). Similarily, vitamin A levels are used as biomarkers to signify an adverse toxic effect in contaminant field studies. PCBs, in particular, are thought to interfere with vitamin A metabolism, transport, and storage, which in turn results in more rapid excretion of this highly-regulated lipophilic retinoid (Zile 1992) that is crucial in the growth, development, reproduction, and immune function of mammals. Significant linear correlations between PCB concentrations in blubber and plasma retinol, blubber retinol, retinoic acid receptor expression, total thyroxine, and thyroid receptor a expression have been documented in previous studies (Mos 2006; Mos *et al.* 2006, 2007; Tabuchi *et al.* 2006).

Organochlorines are also suspected to be the cause of reproductive and immunological disorders among marine mammals (Colborn and Smolen 1996). Lahvis *et al.* (1995) propose that PCBs may cause immunosuppressing effects which render marine mammals more susceptible to opportunistic bacterial, viral, and parasitic infection. An example of this is the onset of morbillivirus after organochlorine exposure, which then further degrades the immune system of its host, eventually leading to starvation and death. In general, an immunotoxic mechanism of action of dioxin-like PCBs (mono-*ortho* and non-*ortho* congeners) is attributed to interaction with the aryl hydrocarbon receptor (AhR) in the cytosol, which results in the transcription of cytochrome P450 enzymes, mediating the toxic effects of dioxin (Safe 1994). Biomarker-based research on harbour seal populations has revealed a series of correlative evidence of POP effects on immunological endpoints, such as PCB-related decreases in lymphocyte function, lymphocyte signalling, total lymphocyte counts and phagocytosis; increases in respiratory bursts (Mos *et al*. 2006); and increases in AhR expression on circulatory blood cells (Mos 2006).

In addition to these toxic properties, PCBs are known to be resistant to metabolism and environmental degradation. The persistent and lipophilic nature of PCBs makes it possible for them to bioaccumulate through the food web, and potentially reach elevated concentrations in higher trophic level organisms such as harbour seals (*Phoca vitulina*) (Cullon *et al.* 2005).

This combination of evidence has led to PCBs meeting the requirement for virtual elimination under *CEPA* (1999). They are defined as being “CEPA-toxic” (applied to substances that enter the environment in amounts that have or may have an immediate or long-term harmful effect on the environment or human health) and thus, are listed in the Toxic Substances List of the Act; they are persistent and bioaccumulative; their presence in the environment results primarily from human activity; and they are neither a naturally-occurring radionuclide nor an inorganic substance. A key role in the virtual elimination process is the development of appropriate environmental quality guidelines, which may provide guidance in targeting geographical locations that require further remediation. At present, it is the role of the Canadian Council of Ministers of the Environment (CCME) to develop chemical-specific water, soil and sediment guidelines in order to protect organisms from the harmful effects of excessive contaminant exposure. These guidelines are established using toxicological data (normally only available for lab test animals such as nymphs, mayflies, and sometimes rainbow trout) to derive a dose-response curve that relates the concentration of a chemical in an environmental media (ie. sediment) to an observed adverse biological effect. The “threshold effects level” is then calculated according to a standard formula, and represents the concentration below which adverse biological effects are expected to occur rarely. This value is then recommended as the sediment quality guideline (SQG) (CCME 1995), and may be subsequently adapted by the province of British Columbia.

As chemicals are released into the environment, they enter aquatic ecosystems and adsorb to suspended particles. These particles, in turn, are eventually deposited into bottom sediments, where contaminants may accumulate over time. Sediment quality guidelines (as opposed to water quality guidelines) are most relevant for PCBs due to their hydrophobicity and associated tendency to partition into sediment as opposed to water. Although many organisms, such as benthic invertebrates, interact directly with the sediment in which these contaminants may be found, the establishment of environmental guidelines based on toxicity testing that only considers these conventional 'lab organisms' is insufficient for biomagnifying substances. Concentrations of biomagnifying substances are greatest in high trophic level organisms, which are not typically included in laboratory toxicity testing. Thus most SQGs, including those developed by the CCME, are developed only to protect benthic invertebrates, and do not consider adverse effects to higher trophic level organisms through direct exposure to sediment or through the consumption of contaminated prey (CCME 1999, Word *et al.* 2002).

Sediment particles enter marine ecosystems through both natural (eg. discharge by rivers, erosion of the coast and seabed) and anthropogenic routes (eg. sewage disposal, dredging, and aquaculture). The process of sedimentation is characterized by the movement and dispersal of these particles as a result of gravity, surface waves, and currents (Hill *et al.* 2008), and provides context to the environmental fate of hydrophobic compounds such as PCBs that preferentially bind to organic particles in the aquatic environment. The current distribution of PCBs in the Strait of Georgia basin is controlled predominantly by sediment flux, bio-mixing, and geochemistry: PCB concentrations are highest where the flux of diluting, inorganic sediment is low, where the depth of benthic mixing is high (causing older, more contaminated sediment to surface), and the concentration of organic carbon is high (Johannessen *et al.* 2008). The surface sediment microlayer compartment of the marine environment receives the majority of the PCBs deposited into the ocean, with high concentrations of surface active agents such as organic compounds and lipids concentrating and stabilizing PCBs in this layer (Kalmaz and Kalmaz 1979). It is the process of bioturbation, in which sediment and solutes are reworked (burrowed in, ingested, used to infill abandoned dwellings) by benthic fauna and flora, which provides a direct entry pathway for organic pollutants such as PCBs to enter the aquatic food web.

Efforts to construct sediment and organic carbon budgets have recognized the important contribution of continental margins in estimating the amount of particulate matter (and associated contaminants) that is produced within, buried by, or exported for a region of interest (Johannessen *et al.* 2003). The larger Strait of Georgia presents a particularly complex basin in which to study sedimentation given its semi-enclosed nature, mix of natural and anthropogenic sediment sources, and the complexity of the sediment dispersal processes at play (Hill *et al.* 2008). The Fraser River is the dominant source of particles to the Strait, contributing 64% of particulate inputs annually, with the remainder of the dissolved and particulate organic carbon coming primarily from in situ primary production (Johannessen *et al.* 2003). While the fresh water that is flushed out with these particles is subsequently discharged through the Juan de Fuca Strait, sedimentation results in nearly complete trapping of particles within the greater Strait of Georgia, causing the burial or oxidation of organic carbon in the sediments of the Strait (Johannessen *et al.* 2003). Most particles tend to settle in the southern Strait, and may then be transported northward (Johannessen *et al.* 2005). This localized sediment accumulation has a direct effect on associated contaminant burdens in sediment, with inorganic particles being discharged at the Fraser River estuary effectively diluting organic contaminant concentrations in surface sediment (Johannessen *et al.* 2008).

Burrard Inlet, which is even more enclosed than the Strait of Georgia, is directly bordered by Greater Vancouver and its associated particulate and contaminant discharges (eg. sewage outfalls, combined sewer overflows, ballast water from ships docked at the port). Semi-enclosed coastal seas such as the Inlet are especially at risk to eutrophication or increased pollution due to the frequently high ratio of anthropogenic loading to ocean exchange (Johannessen *et al.* 2003). In general, sediments in the Inlet range from fine mud in areas of greater depositional activity such as Port Moody arm, to coarse cobble and gravel at the First and Second Narrows regions and estuarine areas such as the mouth of the Capilano River (Levings *et al.* 2003). While surface currents are not as dominant as in the more exposed Strait of Georgia, maximum tidal currents at the First and Second Narrows of the Inlet (the areas of greatest tidal mixings) are reported to be 11 km/h (Levings *et al.* 2003). Furthermore, net deposition of sediments at the First Narrows necessitates dredging activities to allow for channel navigation (Levings *et al.* 2003). Sedimentation rates in the Port Moody Arm have been estimated to range from 0.3 cm/yr (Yunker *et al.* 1999) to 1 cm/yr (Pederson and Waters 2003), and dredging of deep-sea berths (indicating net deposition) is also needed periodically in this region. Given that the highest concentrations of PCBs in a marine environment are likely to be found in areas where the contaminant flux is low (Johannessen *et al.* 2008), regions of the Inlet that are further removed from sediment influx sources may actually be the predominant points of contaminant entry into the food web from sediments.

The biomagnification theory describes the process whereby lipophilic compounds, such as PCBs, increase in concentration with increasing trophic level (Gobas and Morrison 2000). Thus, in order to protect all aquatic life, guidelines for biomagnifying substances should be developed taking into consideration the organisms at the highest trophic level of the marine food web in addition to the benthic invertebrates. In an effort to aid environmental professionals in managing the risks of POP exposure posed to all components of the marine ecosystem, steady-state mass balance models are being developed to improve the scientific understanding of POP pollution dynamics and to characterize the distribution of a contaminant among the various trophic levels of the food web (Mackay 1989). A steady-state approach, in which the net flux of parent chemical being absorbed or depurated by an organism is held constant, is justified in situations in which organisms have been exposed to contaminants over a long period of time, and for chemicals with fast exchange kinetics (Arnot and Gobas 2004). Two such models were developed by Colm Condon and Diego Natale as part of their Resource and Environmental Management (REM) 699 projects (Condon 2007, Natale 2007). These models simulate the flux of contaminants in the environmental compartments of the Strait of Georgia and Burrard Inlet, BC, respectively, and relate the concentration of PCBs in environmental media to those in various wildlife that inhabit the area with a ratio known as the Biota-Sediment Accumulation Factor (BSAF). This is simply the ratio of the chemical concentration in an organism (C_{biota}) to the chemical concentration in the sediment in which the organism resides ($C_{sediment}$), and can be expressed mathematically as follows:

$$BSAF = C_{biota} / C_{sediment} \qquad (1)$$

While the mandate of the British Columbia Ministry of the Environment (MOE) is to protect wildlife from contaminant exposure through the establishment of environmental quality guidelines, Condon (2007) found that BC's ambient sediment quality guidelines (SQGs) are perhaps inadequate for protecting top trophic level organisms. He determined that for the Strait of Georgia, the current SQG for ΣPCBs of 2.0 μg/g organic carbon is 20 times higher than the ΣPCB concentration in the sediment that is at or below the No Observable Adverse Effects Level (NOAEL) for all organisms for which toxicity data exist. Condon further concluded that the current approach used by the MOE to derive sediment quality guidelines is not accounting for the large degree of PCB biomagnification in the local food web. While these conclusions were subject to some key limitations, namely uncertainty in several parameters (ie. food web/trophic interactions) and lack of sufficient model validation, this is of relevance for the management of the thousands of POPs that are currently unregulated in Canada.

Objectives

The goals of the project are two-fold:

1. To further develop a model that relates PCB concentrations in sediment to PCB concentrations, and their health risks, in harbour seals by
    1. conducting field studies to measure the concentration of PCBs in sediment and harbour seals of Burrard Inlet with the goal of calculating an empirical BSAF value; and,
    2. testing the capacity of a previously-developed food-web bioaccumulation model to estimate the BSAF for harbour seals.
2. To apply the empirical and model-predicted BSAFs to derive a SQG for PCBs in Burrard Inlet that protects harbour seals from harmful exposure to PCBs in this region.

METHODS

This study was comprised of five complimentary components, described in detail below:

1. a field study to determine PCB concentrations in sediment and harbour seals;
2. the determination of the empirical harbour seal-sediment accumulation factors (BSAFs) using field concentration measurements;
3. the calculation of BSAFs using a food-web bioaccumulation model;
4. a performance analysis comparing model-generated BSAFs to those measured in the field; and,
5. the application of the model-predicted and field-measured BSAFs to determine a sediment quality guideline for the protection of harbour seals from PCBs.

Field Study

Sampling Site

This study was conducted in Burrard Inlet, an urbanized coastal fjord which extends for 20 kilometers between the Strait of Georgia and Port Moody, as shown in Figure 1. The area is subject to inputs of PCBs and other POPs from numerous point (ie. industrial and municipal effluent discharges, as shown in red in Figure 2) and non-point sources (ie. storm water runoff).

Sample Collection – Sediment

Surface sediment samples collected from nine locations throughout Burrard Inlet in the summer of 2004 (Figure 2) were used in this study. This allowed for greater confidence in the resultant BSAF values that were calculated, as the tissue samples were collected from seals that inhabit the same region of Burrard Inlet. Sediment samples were collected using a petit ponar grab sampler and placed in 250 mL pre-cleaned glass jars, as described by Natale (2007).

Sample Collection – Harbour Seal Tissue

Harbour seals were chosen as indicators of contamination at the highest trophic levels of the Burrard Inlet marine food web for several reasons. They are a non-migratory species which are relatively easy to capture, and are not endangered. They are also well-studied, widely distributed in temperate coastal waters in the Northern hemisphere, and have long life spans (typically 30-40 years).

Harbour seal tissue sampling took place in August 2006, aboard a Department of Fisheries and Oceans Canada (F&OC) research vessel. Sampling efforts focused on harbour seal pups that reside on the log booms off the coast of Port Moody, BC (49°17'N, 122°51'W). Field work was conducted in collaboration with members of the Institute of Ocean Sciences (IOS) in Sidney, British Columbia. A summer sampling time was chosen because of favourable weather, during which seals are more likely to be found relaxing and basking in the sunshine, and thus, easier to catch.

Harbour seal pups were live-captured from log booms by hand (as described in Simms and Ross 2000). Initial determination of sex and assessment of general body condition was followed by weight, length, and axillary girth measurements. A summary of these observations can be found in Table 1. Blubber biopsies were taken following shaving and cleansing of the site with Betadine. An Acu-Punch 3.5 mm and 6 mm biopsy sampler was used to extract three blubber cores (2 x 3.5 mm, 1 x 6 mm) from the organism, which were immediately wrapped in aluminium foil and kept cool (4ºC) and protected from light. After cleansing with Betadine, a topical anesthetic of Xylocaine was administered. Seals were returned to the exact location where they were captured within an hour of the initial capture time.

Field work was carried out under the auspices of F&OC, with approval of animal care committees and scientific research permits for researchers in British Columbia (Fisheries and Oceans Canada Animal Care Committee with guidelines from the Canadian Council of Animal Care).

Sample Analysis

Analysis for congener-specific PCB concentrations for both sediment and seal tissue samples took place at AXYS Analytical Services Ltd. (Sidney, BC), which is certified for PCB analysis by the Canadian Association for Environmental Analytical Laboratories Inc. (CAEL). Quantitative analysis was carried out using gas chromatography/high-resolution mass spectrometry (GC/HRMS) according to the standard operating procedures. Each sample batch included a procedural blank, a replicate, and a certified reference material for quality assurance/quality control purposes. The organic carbon content of sediment samples is reported on a dry-weight basis as g OC/g dry sediment. Each biota sample was analysed for lipid content using the gravimetric lipid determination by weight of extract method, which is reported as a fraction of the wet sample.

Empirical BSAF Calculations

In order to achieve a site-specific indication of the environmental quality guidelines necessary to protect top predators, an empirical BSAF value was calculated for harbour seals. This is the ratio of the chemical concentration in an organism ($C_{biota,}$ or C_b) to the chemical concentration in the sediment in the environment in which the organism resides ($C_{sediment}$, or C_s). The error of this value was calculated by taking the square root of the sum of squares of the standard deviations of the chemical biota and sediment concentrations, as shown in the following equation:

$$SD_{BSAF} = \sqrt{\left(SD^2{}_{C_s} + SD^2{}_{C_b}\right)} \quad (2)$$

For the BSAF calculation for total PCBs, individual congener concentrations were added to give a total PCB concentration.

Data Analysis Method Selection and Implications

Not all of the 209 PCB congeners were detected analytically in every tissue sample collected. To deal with the treatment of low or non-detectable concentrations in a systematic way during data analysis, a comparison of measured concentrations to the Method Detection Limit (MDL) is commonly used. Unfortunately this was not possible, as only one laboratory blank was analyzed instead of the three needed to derive an MDL. As a result, five different methods were considered to determine whether they produced significantly different output in the wet weight concentration of ΣPCBs in harbour seal pups. Table 4 provides a summary of the various methods, and the resultant concentration of ΣPCBs in harbour seals. Individual congener concentrations were compared to either the concentration of the lab blank, or an improvised MDL of three times the concentration of the blank; if the congener concentration was less than this value, it was either set to zero, or set to be half the concentration of the blank. Ultimately, Method 4 was chosen, in which individual congener values were set to zero if they were lower than the conservative MDL. Before using the contaminant concentrations for further analysis, the concentration of the lab blank was subtracted from each value and concentrations were lipid-normalised or used on a dry weight basis. Average PCB congener concentrations for all seal samples were calculated by taking an average of the concentration in individuals, not including zero values in the overall number of samples. Total PCB concentrations were calculated by summing these concentrations for all congeners. For the purpose of comparison to model predictions, only the subset of 29 congeners for which predicted concentrations were generated by the model (listed below) were used in the sum carried forward to BSAF calculations.

Selection of PCB Congeners

The following 29 groups of single or co-eluting PCB congeners were used to generate BSAFs: 18/30, 20/28, 44/47/65, 49/69, 52, 66, 61/70/74/76, 83/99, 90/101/113, 105, 110/115, 118, 128/166, 129/138/160/163, 146, 147/149, 135/151/154, 153/168, 156/157, 170, 177, 180/193, 183/185, 187, 194, 198/199, 203, 206, and 209. They are similar to the congeners selected by Condon (2007), except that PCB 31 was not treated as a co-eluting congener with PCB 20 and 28, and PCBs 8, 15, and 37 were not included as they were not detected in the blubber samples collected. Furthermore, the selected congeners represent 53 of the possible 209 congeners, and comprise the majority (87%) of the total PCB mass in the empirical performance analysis dataset for harbour seal pups. Thus, it is reasonable to assume that the chosen subset is representative of the entire suite of PCB congeners.

Selection of Sediment Data Subset

While a suite of PCB concentration data from nine sediment samples taken throughout Burrard Inlet were available for use, it was decided to proceed with a subset of 6 samples from only the Central Harbour and Port Moody regions of the inlet because these are areas frequented by harbour seals. Data provided by Natale (2007), shown in Figure 3, illustrate that sediment PCB concentrations throughout the Inlet are relatively constant. The exception are the concentrations found in the Inner Harbour sediment, which may be attributed to sample collection in close proximity to a sewer discharge point in that region. As the inclusion of the subset of sediment concentration data from this region would artificially inflate the calculation of the mean concentration for the whole inlet, data from the Inner Harbour was excluded from BSAF calculations. However, in some cases calculations that both include and exclude this data are provided in the results for comparison purposes.

Model BSAF Calculations

Calculation Tools

The existing steady-state bioaccumulation model developed by Condon (2007) in Microsoft Excel using Visual Basic was adapted for the Burrard Inlet ecosystem. For a more detailed description of model structure and input refer to this work, which is a generalized modeling framework based on the model theory of Gobas and Arnot (2010). The appendices provide additional information on model inputs and parameterization.

The model algorithms used for phytoplankton, zooplankton, benthic invertebrates, and fish are based on the presumption that the exchange of PCBs between an organism and its ambient environment can be described by the following equation for these classes of organisms:

$$C_B = \frac{\left\lfloor k_1 \left(m_O \times \phi \times C_{WT,O} + m_P \times C_{WD,S} \right) + k_D \times \sum P_i \times C_{D,i} \right\rfloor}{\left(k_2 + k_E + k_G + k_M \right)} \qquad (3)$$

where C_B is the wet weight concentration (g/kg wet wt) of the PCB congener in the organism, k_1 is the clearance rate constant ([L/kg wet wt]/d) for uptake via the respiratory area (ie. gills and skin), m_O is the fraction of the respiratory ventilation that involves sediment associated pore water, ϕ (unitless) is the fraction of total chemical concentration in the overlying water that is freely dissolved and can be absorbed via membrane diffusion, $C_{WT,O}$ is the total concentration of the PCB congener in the water column above the sediments (g/L), $C_{WD,S}$ is the freely dissolved PCB congener concentration in the sediment associated pore (or interstitial) water (g/L), k_D is the clearance rate constant ([kg/kg wet wt]/d) for the chemical uptake via ingestion of food and water, P_i is the fraction of the diet consisting of prey item i, $C_{D,i}$ is the concentration of PCB congener (g/kg) in prey item i, k_2 is the rate constant (1/d) for elimination of PCBs via the respiratory area (ie. gills and skin), k_E is the rate constant (1/d) for the elimination of the PCB congener via excretion into egested feces, k_G is the growth rate constant expressed as fixed annual proportional increases in the organism's wet weight W_B (kg) over time t, ie. $dW_B/(W_B*dt)$, and k_m is the rate constant (1/d) for metabolic transformation of the PCB congener.

The steady-state solution of the mass balance equation in the harbour seal is as follows:

$$C_{HS,l} = \frac{\left(k_A \times C_{AG} + k_D \sum \left[P_i \times C_{D,i}\right]\right)}{\left(k_O + k_E + k_U + k_G + k_P + k_L + k_M\right)} \qquad (4)$$

where $C_{HS,l}$ is the lipid-normalized concentration of the PCB congener in the seal, C_{AG} is the gaseous aerial concentration (g/L), k_A is the inhalation rate constant ([L/kg lipid]/d), k_D is the clearance rate constant ([kg/kg lipid]/d) for PCB uptake via ingestion of food and water, P_i is the fraction of the diet consisting of prey item i, $C_{D,i}$ is the concentration of the PCB congener (g/kglipid) in prey item i, k_O is the rate constant (1/d) for exhalation of PCB via the lungs, k_E is the rate constant (1/d) for the elimination of the PCB congener via excretion into egested feces, k_U is the rate constant for urinary excretion of PCBs, k_G is the rate constant for growth dilution (accounting for year-to-year increases in the net growth of the animals), k_P is the rate constant for transfer of PCBs into the pups (representing the increase in lipid mass [equivalent to the post-parturition lipid mass of the pup] over the duration of the gestation period), k_L is the rate constant for transfer of PCBs to the pups as a result of lactation (portraying the growth of lipid mass of the female seals over the year that is transferred to the pup during lactation), and k_G, k_P, and k_L are expressed as fixed annual-proportional increases in body lipid weight over time t, ie. $dW_{S,l}/(W_{S,l}*dt)$, where $W_{S,l}$ is the weight of the lipids in the seal and has units of $d^{-1}*k_M$, the rate constant for metabolic transformation of the PCB congener. PCB uptake in harbour seals is predominantly due to dietary uptake and inhalation of air, whereas elimination of PCBs from the seals can be attributed to exhalation of air and excretion in fecal matter and urine. Additional elimination routes include the metabolism of some PCB congeners (Boon *et al.*

1987, Boon *et al.*1997), the transfer of PCBs from female adults to their offspring through giving birth and lactating (which occurs for approximately four weeks in harbour seals), molting, and growth periods.

The chemical characteristics of PCBs can be used to describe the uptake and elimination pathways of these chemicals in organisms using a relatively simple model. As a result of their lipophilic nature, PCBs are primarily found in the lipid tissues (which constitute a large fraction of overall body mass) of organisms such as harbour seals. In addition, PCBs show a tendency to establish chemical equilibrium within an organism, reaching equivalent concentrations in the lipids of the various parts of the organism. When representing the transfer of PCBs from a mother to her pups for example, this characteristic is especially relevant, as it can be assumed that the PCBs in the mother and pup achieve an internal equilibrium. Thus, even through the processes of parturition, and lactation, the lipid-normalized concentration of PCBs in both animals remains constant in the short-term. Similarly, during the molting process, the shedding of a layer of skin with a lipid PCB burden that is equivalent to that in the lipids of the remainder of the body does not cause a change in lipid-normalized concentration, as proportional declines in PCB mass and lipid mass occur throughout the process. Over the long term, however, a continuous growth-induced decline in PCB concentration in harbour seals is experienced as a result of fetal development, milk production, and skin formation, as these are processes that add mass in addition to year-to-year increases in body mass. This growth-induced decline in seal PCB body burdens is compensated through the intake of PCB-contaminated prey that makes this growth possible.

Burrard Inlet Food Web Structure

The organisms and feeding relationships used to construct the food web for the study area centred on those used by Condon (2007), in which harbour seals (*Phoca vitulina)*, double-crested cormorants (*Phalacrocorax auritus)*, and great blue herons (*Ardea herodias)* are included as top trophic level predators. The feeding relationships which link these top predators to the sediment in Burrard Inlet through their prey are illustrated in Figure 4, and in more detail in Table 2.

Of primary interest for this study was the food web of the harbour seal, which was modified from that used by Condon (2007) to incorporate information on prey consumption estimates from Cullon *et al.* (2005). In general, the diet matrix assumptions made by Condon (2007) were:

- juvenile seals eat the same prey as adults;
- seal pups consume only mother's milk;
- diet composition values selected are annual averages;
- seals eat primarily mature fish; and,
- salmon and herring are migratory and feed primarily outside Burrard Inlet.

Table 2 outlines the matrix of diet compositions for select organisms of Burrard Inlet. Table 2 shows that Pacific hake and Pacific herring dominate the diet of harbour seals in the area. Due to uncertainty regarding the assumption of herring being an immigrant species to the inlet, three distinct food web assumptions were evaluated in a sensitivity analysis:

1) <u>Herring Immigrant Scenario</u>
   - In this scenario, herring reside outside Burrard Inlet, and thus obtain most of their PCB load from feeding and growing *outside* the inlet.
2) <u>Herring Native Scenario</u>
   - This scenario assumes that herring spend the majority of their adult life feeding and growing *inside* Burrard Inlet. The diet composition of herring, adapted from Foy and Norcross (1999) as shown in Table 2, was included as a model input. Feeding preferences shown are estimated annual averages, which fall within the seasonal ranges of diet composition by prey category as described in Foy and Norcross (1999).
3) <u>Herring Semi-native Scenario</u>
   - This scenario assumes that herring spend part of their adult life in Burrard Inlet and part in the Strait of Georgia or Pacific Ocean. This was achieved by inputting this same diet composition as in the herring native scenario, and lowering the lipid content of the herring by half.

Unless otherwise described, the third scenario of the herring being a semi-native species is the one that has been investigated most thoroughly.

Model Performance Analysis

Comparison of Predicted and Observed BSAFs

The BSAFs predicted by the bioaccumulation model were compared to empirically-derived BSAFs for the harbour seal pup. In order to do this in a quantitative manner, an equation for the mean model bias (MB*) for total PCBS was utilized, as shown below:

$$MB^{*} = 10^{\left(\sum_{i=1}^{m} \frac{\left[\log\left(BSAF_{P,\sum PCB} / \sum BSAF_{O,\sum PCB}\right)\right]}{m}\right)} \qquad (5)$$

In this equation MB* is the geometric mean of the ratio of the predicted BSAF for total PCBs ($BSAF_{P,\sum PCB}$) and the observed BSAF of total PCBs ($BSAF_{O,\sum PCB}$) for all observations, *m*. Assuming a log-normal distribution of the ratio of predicted and observed BSAFs ($BSAF_{P,i}$ / $BSAF_{O,i}$), the mean model bias value (MB*) is the geometric mean of the ratio $BSAF_P$/ $BSAF_O$ for total PCBs in each species for which empirical data were available.

The model bias is a measure of the systematic overprediction (MB* > 1) or underprediction (MB* < 1) of the model. This type of model performance analysis is especially useful when the model is being applied to make practical estimations of BSAFs for exposure assessment or for the derivation of environmental quality guidelines when total PCBs are of primary interest. This is because the model bias value tracks the central tendency of the ability of the model to predict individual PCB congener concentrations.

The variability of over- and underestimation of measured BSAF values by the model is quantified as the 95% confidence interval of MB*, calculated as shown below:

$$95\%\ CI = \text{antilog}(\log MB^* \pm [t_{v,0.05} * \log SD]) \quad (6)$$

The 95% confidence interval of the model bias value is a measure of the uncertainty of model predictions; it gives the range of BSAFs that includes 95% of the observed BSAFs. The performance of the model improves when MB* approaches 1, and the 95% confidence intervals on this value decreases. Error from model parameterization, model structure, analytical errors in the empirical data, and natural, spatial, and temporal variability in the data used for model performance analysis are all represented by the MB* and 95% CI values.

Model Application

As shown in Figure 5, the outputs of the model can be used to both 'forwards' calculate PCB concentrations in wildlife based on measured PCB concentrations in sediment, or 'backwards' calculate target sediment concentrations given an ecological criteria (biota concentration) to be met. The former results in an ecological risk assessment for top predators, while the latter can be used to establish a sediment quality guideline.

In the forward application of the model, the observed distribution of PCB concentrations in sediment (C_S) is used to calculate the concentration distribution in harbour seals (C_B) according to the following equation:

$$\log C_B = \log BSAF + \log C_s \qquad (7)$$

A comparison of concentration distributions in target organisms (harbour seals in this case) to toxic threshold concentrations can be used to facilitate the risk assessment process.

In the backward application of the model, target PCB concentrations in biota (harbour seals) are used to calculate the PCB concentration in the sediment that is expected to meet ecological criteria according to the following equation:

$$\log C_S = \log C_B - \log BSAF \qquad (8)$$

Ecological Risk Assessment for Top Predators

Ecological risk assessment is a process used to evaluate the likelihood of adverse ecological effects occurring as the result of exposure to one or more stressors (US EPA 1992). In this case, a single line of evidence was used to assess ecological risk: the comparison of observed total PCB concentrations in harbour seal pups to concentrations suspected to cause adverse effects. When the concentrations in biota are depicted as a normal probability density function, comparison to established toxic effects concentrations allows for an assessment of the proportion of the population expected to have PCB concentrations exceeding this threshold concentration, and thus expected to be adversely affected by PCB exposure. Because the threshold total PCB concentration that causes toxic effects in harbour seals is not well established, two toxic threshold concentrations were considered in this study: 17 mg PCB/kg lipid, proposed by Kannan *et al.* (2000), and 1.3 mg PCB/kg lipid proposed more recently by Mos *et al.* (in press 2010).

Because of the lack of data on the concentration of PCBs that are associated with affects on the immune system of marine mammals, Kannan *et al.* (2000) carried out a synthesis study to derive threshold effects concentrations from toxicity studies published in the literature. Results of semi-field (ie. experiments in which laboratory raised animals are fed food items collected from the field) and field toxicity studies, in which endpoints of not only hepatic vitamin A and thyroid hormone concentrations, but also suppression of natural killer (NK) cell activity and proliferative response of lymphocytes to mitogens, were compiled. The authors extrapolated a threshold concentration for PCBs in the blubber of marine mammals of 17 μg PCBs/g, lipid weight, after applying a safety factor of 2 to observed threshold *blood* PCB levels. The use of this safety factor was justified as 1) congener patterns of PCBs in blood were similar to those in blubber (Boon *et al.* 1987), and 2) the lipid-normalized concentrations of total PCBs in the blubber of clinically healthy bottlenose dolphins was found to be 2-fold greater than the concentrations in their blood (Reddy *et al.* 1998).

More recently, however, a harbour seal biomarker study was carried out by Mos *et al.* (in press 2010) in which a novel benchmark of 1.3 μg PCBs/g lipid weight was derived for the protection of marine mammal health. This threshold concentration is based on the 95% lower confidence limit of the mean PCB concentration in the tissue of the study seals, and is meant to provide a higher level of sensitivity to PCB-associated developmental effects during the sensitive nursing stage of the seal's life. During this period of immune system development, marine mammals may be especially susceptible to experiencing life-long damaging effects, even at low exposure doses (Mos 2006).

Derivation of Sediment Quality Guideline

Following the determination of a BSAF value for a particular contaminant in a given ecosystem, it is possible to relate this value back to a sediment quality guideline using a back calculation. The threshold toxic effects concentration, C_{toxic} (or TEC), may be used as follows to establish a new SQG ($C_{sediment}$) which is protective of harbour seals:

$$\log C_{sediment} = \log C_{toxic} - \log BSAF - 1.96*(SD_{MB}) \qquad (9)$$

The final term in the equation is included in order to ensure with 95% certainty that the seal population will be protected by the calculated SQG. The standard deviation used is the error associated with the model bias value, as described in the model performance analysis section. Again, both toxic threshold concentrations of 17 mg PCB/kg lipid and 1.3 mg PCB/kg lipid were considered in the calculation of a target geometric mean sediment ∑PCB concentration.

RESULTS AND DISCUSSION

PCB Concentrations in the Marine Food Web

Sediment

Figure 6 and Table 3 show geometric mean PCB concentrations detected in surface sediment samples collected from the Central Harbour and Port Moody regions of Burrard Inlet (n=6). Concentrations range from 0.032 [95% confidence interval of 0.014-0.075] ng/g dry weight for PCB 209 to 0.80 [95% confidence interval of 0.44-1.50] ng/g dry weight for the co-eluting group of PCBs 129/138/160/163. The concentration of total PCBs measured in the sediment was 8.1 [95% confidence interval of 4.7-14] ng/g dry weight.

Biota

Unfortunately it was not possible for low or non-detectable PCB congener concentrations in biota to be compared to a conventional MDL in this study, as only one laboratory blank was analyzed instead of the three needed to derive a MDL. As a result, the data from the analysis of the laboratory blank sample was used to derive a detection limit for the concentration data (ie. three times the concentration of the blank).

Five alternate data analysis methods (described in more detail in the methods section) were considered for the calculation of total PCBs from individual congener concentrations to determine if they produced a significantly different output in the corrected, lipid weight concentration of total PCBs. These ranged from correcting the concentration to zero if the individual congener concentration was less than the concentration of the laboratory blank, to setting a conservative MDL of three times the concentration of the blank.

As shown in Table 4, there was no significant effect on total PCB concentrations as a result of four of the different methods considered (Methods 1-2 and 4-5). The concentration calculated by these four of the methods was 1.3×10^4 ng/g lipid. However, Method 3, in which congener concentrations that were less than the blank concentration were ignored so as not to artificially lower the total PCB value too substantially, resulted in a concentration of 2.2×10^3 ng/g lipid (one order of magnitude lower). The method in which the conservative MDL approach was used and all congener concentrations less than three times the concentration of the blank were replaced by zero, with the average being calculated using only non-zero values in n (Method 4), was ultimately chosen.

Mean tissue PCB concentrations in Burrard Inlet harbour seal pups were calculated on a lipid-normalized basis (Figure 7 and Table 5), and range from 0.57 to 810 ng/g lipid for PCB 209 and co-eluting PCBs 153/168 respectively. Total lipid-normalized PCBs were measured to be 4000 ng/g lipid [95% confidence interval 2200-7300 ng/g lipid]. In comparison, total lipid-normalized PCB concentrations in harbour seals inhabiting the northern Strait of Georgia were reported as 2317 ng/g lipid [95% confidence interval 1398-3841 ng/g lipid, n=10] in Condon (2007). Total PCB concentrations in harbour seals inhabiting various regions of British Columbia and Washington were also reported with standard errors in Ross *et al.* (2004) as 1143 ± 262 ng/g lipid (Queen Charlotte

Strait), 2475 ± 174 ng/g lipid (Strait of Georgia), and 18 135 ± 3082 ng/g lipid (Puget Sound). PCB concentrations in harbour seals inhabiting the urbanized Burrard Inlet reported in this study fall between those inhabiting more remote areas (northern Strait of Georgia and Queen Charlotte Strait) and the highly urbanized Puget Sound.

Empirical BSAF Calculations

Concentrations of PCBs in sediment and biota were used to calculate empirical BSAF values on a g-dry-weight sediment/ g-lipid-weight biota basis for comparison with model-generated values. The empirical BSAF for ΣPCBs in harbour seal pups using only sediment data from the Central Harbour and Port Moody was calculated to be 490 (lower SD 220, upper SD 1100) g-dry-weight sediment/ g-lipid-weight biota (n=6). Empirical BSAFs by congener, displayed with model predictions in Figure 10 and Table 6, ranged from 18 g-dry-weight sediment/ g-lipid-weight biota for PCB 209 to 1300 g-dry-weight sediment/ g-lipid-weight biota for PCB 153/168. Organic carbon and lipid normalized empirical BSAFs are also shown in Table 6 for comparison purposes, and range from 0.46 kg OC/kg lipid for PCB 209 to 32 kg OC/kg lipid for PCB 153/168, with the BSAF of ΣPCBs being 13 (lower SD 3.7, upper SD 43) kg OC/kg lipid.

BSAF Model Calculations

Model-predicted BSAFs for ΣPCBs for all included species are shown in Figure 11. Figure 11 shows that BSAFs increase with increasing trophic level, ranging from 0.0083 g-dry-weight sediment/ g-lipid-weight biota for chum to 420 g-dry-weight sediment/ g-lipid-weight biota for harbour seal pups. This represents an increase in the BSAF of approximately 50 000 times (on a lipid weight basis) from chum to seal pups, and an increase in concentration of approximately 420 times from sediment (dry weight) to seal pups (lipid weight).

Figure 8, Figure 9, Figure 10 and Table 6 show both empirical and model-predicted BSAFs of PCB congeners for each of the three food web scenarios modelled (ie. scenario 1 - herring as an immigrant species; scenario 2 - herring as a native species; and scenario 3 - herring as a semi-native species). Only empirical BSAFs in units of g-dry-weight sediment/ g-lipid-weight biota are discussed here for purposes of model comparison. If assuming herring is an *immigrant* species (Figure 8), predicted BSAF values range from 0.15 g-dry-weight sediment/ g-lipid-weight biota for PCB 44/47/65 to 2 100 g-dry-weight sediment/ g-lipid-weight biota for PCB 147/149 and PCB 187, with a BSAF for total PCBs of 790 g-dry-weight sediment/ g-lipid-weight biota. Assuming herring is a *native* species (Figure 9), predicted BSAF values are slightly higher than scenario 1, ranging from 0.20 g-dry-weight sediment/ g-lipid-weight biota for PCB 44/47/65 to 3 200 g-dry-weight sediment/ g-lipid-weight biota for PCB 147/149 and PCB 156/157, with a BSAF for total PCBs of 1 200 g-dry-weight sediment/ g-lipid-weight biota. Under the third assumption (ie. herring is a *semi-native* species) (Figure 10), predicted BSAF values range from 0.17 g-dry-weight sediment/ g-lipid-weight biota for PCB 44/47/65 to 2 800 g-dry-weight sediment/ g-lipid-weight biota for PCB 147/149, PCB 156/157, and PCB 187. The BSAF for total PCBs with this scenario falls between the BSAFs calculated in the other two scenarios at 1000 g-dry-weight sediment/ g-lipid-weight biota.

Model Performance Analysis

The performance of the model was analyzed in order to investigate the potential of the model as a decision-making tool.

Figure 8, Figure 9, and Figure 10 illustrate the difference between model-predicted and observed BSAFs on a congener-specific basis. It seems there is a general trend of model *over*-prediction for PCB congeners 156/157 and higher ($\log K_{ow} > 7.2$), which is possibly a systematic trend rather than being solely attributable to various sources of error that could be introduced in the empirical data collection and modelling phases. This observation may be attributed to both environmental and physiological factors.

One explanation is that heavier PCB congeners adhere to sediment particles more readily, and thus may not be as easily taken up through food web transfer. Ross *et al.* (2004) found that more heavily chlorinated PCB congeners are more rapidly deposited into and retained by sediments in areas adjacent to sources given that they are less volatile than lighter congeners and more likely to be scavenged by particles during the process of sedimentation. Furthermore, the metabolic removal of specific congeners (eg. those associated with the induction of detoxifying enzymes, such as the P450 group of enzymes, or planar 2,3,7,8-tetrachlorodibenzo-*p*-dioxin and related congeners) in the liver of harbour seal pups may also contribute to a change in the composition of retained PCBs (Boon *et al.* 1997, De Swart *et al.* 1995, Ross *et al*. 2004).

An alternate explanation is that the lower-molecular-weight PCB congeners are taken up by pups more readily through placental transfer and lactation. Debier *et al.* (2003) reported higher concentrations of heavier (more highly chlorinated, and thus more persistent) PCBs in the blubber of adult female grey seals, while their milk was found to contain more lower-chlorinated PCB congeners; thus, a high contribution of lower-chlorinated PCBs in the blubber of their offspring was assumed. A similar trend in relationship between partition ratio, degree of chlorination, and $\log K_{ow}$ was observed by Greig *et al.* (2007), in which lower-molecular-weight, lipid soluble chemicals accumulated more readily in the fetus compared with higher-molecular-weight, lipophilic compounds. It is possible that this variability may be explained by a difference in individual congener blood transport mechanisms or distribution in serum, erythrocytes, and lipoproteins, with heavier congeners preferentially binding to lipoproteins and albumin rather than being dissolved in lipid (Matthews *et al.* 1984). It may also be the result of a physical barrier for larger PCB congeners in the placenta or mammary glands, or the differential mobilization of specific PCB congeners from the mother’s blubber during milk production (Ross *et al.* 2004).

Finally, the overprediction of the model may be attributed to an imbalance in the PCB signatures of harbour seal prey assumed by the model. West *et al.* (2008) examined the geographic distribution and magnitude of PCBs in three populations of Pacific herring (harbour seal prey) from both the Strait of Georgia and Puget Sound water bodies. They determined that the Strait of Georgia populations were isolated from one another as a result of differential exposure to contaminants related to where the populations reside and feed, rather than differences in age, size, trophic level, or lipid content. Differences in migration behaviour between these isolated populations may correspond with different contaminant fingerprints as a result of the differing amounts of time spent in contaminated habitats (West *et al.* 2008). Similarly, Cullon *et al.* (2005) observed that the PCB signature in Puget Sound (more urbanized) is far heavier than that of the Georgia Basin. Harbour seals consuming prey that spends a greater proportion of time in close proximity to urbanized environments, such as Burrard Inlet (as opposed to prey herring from the west coast of Vancouver Island), could be exposed to a heavier PCB fingerprint, and vice versa. As the feeding ecology assumptions used in the model are subject to a considerable amount of uncertainty, it may be that the imbalance in the assumption of a higher proportion of local (more contaminated, heavier PCB signature) herring versus those from remote areas in the harbour seal diet is resulting in an overprediction of the model.

Mean MB* values for each food web scenario, along with their 95% confidence intervals, are shown in Table 8 and Figure 12. An MB* greater than one indicates an over-prediction of the BSAF by the model, while an MB* less than one represents an under-prediction of the model (eg. MB* = 2 means the model over-predicts the empirical data by a factor of 2; MB* = 0.5 means the model under-predicts the empirical data by a factor of 2). Table 7 illustrates the model bias values for the calculation of the BSAF of total PCBs for the food web scenario where herring is a native species. These MB values were then used to calculate overall mean model bias values for total PCBs (MB*) for each of the three food web scenarios with both sediment datasets in order to get a better understanding of the model's strengths and weaknesses. In general, it can be seen that the model has a tendency to over-predict the measured BSAFs by a factor of 1.5 to 2.5 for harbour seal pups when considering the results from using the subset of sediment data (rationalized earlier).

The 95% confidence intervals of the mean MB* (calculated as CI = antilog (geometric mean ± ($t_{\nu,\,0.05}$ × standard deviation)), represent the uncertainty in the BSAF model estimates. These confidence intervals play an integral role in assessing probabilities that ΣPCB concentrations in organisms of the inlet exceed ecological health criteria because they represent the range of BSAFs that includes 95% of the observed BSAFs.

Using the subset of sediment data, the most plausible scenario of herring being a semi-native species in the inlet results in a MB* of 2.1 (95% confidence interval 1.2-3.8). While this represents a slight under-prediction of the model, the range of the 95% confidence interval is indicative of the errors in model parameterization and model structure, analytical errors in the empirical biota and sediment data, and the natural, spatial, and temporal variability in empirical sediment data that could result in such a discrepancy.

In comparison, the MB* value for seal pups in Condon's model (2007) was 3.18. Condon highlighted the fact that the performance dataset he used for seals was limited spatially, temporally, chemically, and statistically. The dataset available for this research resolved one of these issues, as the sediment and seal concentration data are spatially matched, however there remains concerns surrounding other sources of error, primarily in model food-web parameterization.

Model Application

Ecological Risk Assessment for Top Predators

Figure 13 shows observed and model-predicted total PCB concentrations in Burrard Inlet harbour seal pups in relation to two toxicity reference values. It can be seen that the toxicity reference value of 17 000 ng/g lipid is exceeded by 4% of the modelled harbour seal population (total PCB concentration is $10^{3.92 \pm 0.26}$ ng/g lipid in biota), while the 1 300 ng/g lipid effects threshold is exceeded by nearly 100% of the harbour seal population. Figure 14 shows only model-predicted total PCB concentrations in Burrard Inlet adult male harbour seals in relation to the same two toxicity reference values. The toxicity reference value of 17 000 ng/g lipid is exceeded by 7% of the modelled harbour seal population (total PCB concentration is $10^{4.00 \pm 0.26}$ ng/g lipid in biota), while the 1 300 ng/g lipid effects threshold is exceeded by 100% of the harbour seal population. A similar result is observed for the empirical harbour seal population (total PCB concentration is $10^{3.6 \pm 0.26}$ ng/g lipid in biota), and for the adult male harbour seal population. This indicates that harbour seals in the Burrard Inlet food web are not currently being protected from adverse effects of current PCB concentrations in Burrard Inlet. Furthermore, this provides a rationale for the derivation of a more ecologically-relevant sediment quality guideline to remedy this situation.

Derivation of a Proposed Sediment Quality Guideline

The ‘back calculation’ conducted to examine the appropriateness of the current sediment quality guideline was carried out under two scenarios: using the toxic threshold value proposed by Kannan *et al*. (2000), and using the value proposed by Mos *et al.* (in press 2010). The calculation was carried out according to equation (8), as previously discussed in the methods section, for both pup and adult male harbour seals. Target sediment concentrations for both age groups are summarized in Table 9.

The threshold effects concentration of 17 mg PCBs/kg lipid weight (Kannan *et al*. 2000) was used in the back-calculations for target ΣPCB sediment concentrations shown in Figure 15 (pup seals). The calculation was performed using the model-predicted BSAF under all three food web scenarios and the empirical BSAF. Resultant target ΣPCB sediment concentrations are graphed along with the actual distribution of ΣPCB sediment concentrations in the Central Harbour and Port Moody regions of the inlet and the current provincial ΣPCB sediment quality guideline of 20 ng/g dw in Figure 15. Figure 15 shows that both the current sediment concentrations in the inlet and the SQG are higher than the back calculated proposed sediment quality guideline concentration that ensures the protection of 95% of the pup seal population under all food web scenarios. Thus, assuming a threshold toxic ΣPCB concentration of 17 µg PCBs/g lipid weight in the blubber of marine mammals results in a target sediment concentration of 5.0 ng/g dw (in the case of the herring semi-native food web scenario), which is 1) lower than the current sediment quality guideline of 20 ng/g dry weight; and 2) currently exceeded by as much as 97% of the Burrard Inlet sediments, where the mean ΣPCB concentration in sediment is 8.1 ng/g dw [95% confidence interval 4.7-14 ng/g dw]. The target sediment concentration of 11 ng/g dw (calculated using the empirical BSAF of 490 g-dry-weight sediment/ g-lipid-weight biota), is lower (approximately half of) the current sediment quality guideline.

The backcalculation for adult male seals was only carried out using the model-predicted BSAFs, as empirical data for this age group was not collected. Table 9 illustrates the target sediment concentration for adult male seal pups (4.2 ng/g dw for the semi-native herring scenario) is very similar to that for pups.

Figure 16 shows the back-calculations for target ΣPCB sediment concentrations calculated using the more conservative toxic reference value of 1.3 µg PCBs/g lipid weight (Mos *et al.* in press 2010), as summarized in Table 9. As before, the calculation was performed using the model-predicted BSAF value for all three proposed food web scenarios and the empirical BSAF, and resultant target ΣPCB sediment concentrations are graphed in relation to the actual ΣPCB sediment concentration distribution in the inlet. Figure 16 shows that the current SQG again exceeds the back calculated ‘safe’ sediment concentration that ensures the protection of 95% of the seal population under all food web scenarios. Furthermore, 100% of the actual ΣPCB sediment concentration distribution exceeds the recommended target concentration in the case of all the food web scenarios. Thus, assuming a threshold toxic ΣPCB concentration of 1.3 µg PCBs/g lipid weight in the blubber of marine mammals results in a target sediment concentration of 0.38 ng/g dw (in the case of the herring semi-native food web scenario), which is 1) two orders of magnitude *lower* than the current sediment quality guideline; and 2) currently *exceeded* by all sediments in Burrard Inlet. The target sediment concentration of 0.32 ng/g dw (calculated using the empirical BSAF of 490 g-dry-weight sediment/ g-lipid-weight biota), is also significantly lower than the current sediment quality guideline.

The backcalculation for adult male seals was again carried out using only the model-predicted BSAF for lack of empirical data. Table 9 illustrates the target sediment concentration for adult male seal pups (0.32 ng/g dw in the case of the semi-native herring scenario) is very similar to that for pups.

Ideally, the fundamental goal in the creation of any environmental quality guideline is to maintain an environmental concentration of the contaminant which is not expected to pose a risk of chronic adverse effect to any member of the ecosystem. Adverse effects are classified by Health and Welfare Canada (1990, now known as Health Canada) as “functional impairments or pathological lesions that may affect the performance of the organism or reduce its ability to respond to additional stressors”. While maintaining contaminant levels which are not expected to result in such effects may be practically difficult to achieve, it is nevertheless important to strive for guidelines which accomplish this ultimate objective in taking a conservative approach to management. Ultimately, organisms must be able to maintain viable populations over generations.

As such, the back-calculations and resultant threshold sediment concentrations for ΣPCBs conducted using the more conservative tissue toxicity threshold suggested by Mos *et al.* (in press 2010) are perhaps most suitable for achieving the objective of protecting all species, regardless of trophic level or life stage. Although it may not be practical or feasible to implement for PCBs as a legacy contaminant, a new sediment quality guideline for ΣPCBs that is orders of magnitude lower than the current guideline warrants further consideration. While this may be lower than necessary to protect all receptors given the uncertainty in the model-generated BSAF values, PCBs are only one contaminant of the many that these organisms are exposed to, and contaminant exposure represents only one of many stressors that marine species are subjected to.

This work highlights the importance of considering and accounting for the phenomenon of bioaccumulation and biomagnification in the process of establishing environmental quality guidelines for emerging contaminants, such as PBDEs, and other POPs that are currently unregulated in Canada. The mechanistic bioaccumulation model provides a useful tool with which site-specific information, such as organic carbon content of sediment and food web structure, can be taken in to account when deriving guidelines for regions throughout the province and beyond. Recognizing that modelling is an iterative process, continuous refinement and improvement of the model with increased understanding of the natural system it represents will improve its usefulness as a management tool.

CONCLUSION AND KEY FINDINGS

1. A field study was conducted to measure the concentration of PCBs in sediment and harbour seals of Burrard Inlet. From this, an empirical BSAF of 490 (lower SD 220, upper SD 1100) g-dry-weight sediment/ g-lipid-weight biota (n=6) was derived for ΣPCBs.

2. The capacity of a previously-developed food-web bioaccumulation model to estimate the BSAF for harbour seals was tested. A systematic over-prediction of the model (MB* ranging between 1.6 and 2.4) under all three food web scenarios tested was observed for ΣPCBs (MB* under the scenario of herring as a semi-native species was 2.1 [95% confidence interval 1.2-3.8], MB* under the scenario of herring as an immigrant species was 1.6 [95% confidence interval 0.90-2.9], and MB* under the scenario of herring as a native species was 2.4 [95% confidence interval 1.3-4.3]). This overprediction was largely explained by the failure of the model to adequately explain the behavior of the higher-chlorinated PCB congeners (ie. $\log K_{ow} > 7.2$).

3. The empirical and model-predicted BSAFs were then applied to derive a proposed SQG for ΣPCBs that protects harbour seal pups and adult males

from harmful exposure to these contaminants in Burrard Inlet. Two effects thresholds were investigated in this calculation, both of which resulted from affects to immune function and the endocrine system of the harbour seal: the first based on a controlled captive feeding study, and the second a field study. Depending on the TEC that is assumed, and using model-predicted BSAFs, a SQG for ΣPCBs of 5.0 ng/g dw (TEC of 17 μg PCB/g lipid) or 0.38 ng/g dw (TEC of 1.3 μg PCB/g lipid) is proposed in order to protect 95% of the pup seal population. The target sediment ΣPCB concentration increases to 11 ng/g dw (TEC of 17 μg PCB/g lipid) or 0.82 ng/g dw (TEC of 1.3 μg PCB/g lipid) for the same level of protection using the empirical BSAF. Findings for the same level of protection in the adult male harbour seal population indicate that target sediment concentrations for ΣPCBs would be in the same order of magnitude as for seal pups (4.2 ng/g dw using a TEC of 17 μg PCB/g lipid or 0.32 ng/g dw using a TEC of 1.3 μg PCB/g lipid). In all cases, SQG values are more conservative than the current SQG for ΣPCBs. As a result, the current SQG, which does not account for bioaccumulation or biomagnification processes, may not be protective of top trophic level organisms.

RECOMMENDATIONS

This research has focused on establishing an empirical BSAF, and comparing this to a model-predicted BSAF value, for PCBs in the harbour seal (*Phoca vitulina*) due to its increasingly accepted status as an indicator species for providing an integrated measure of the contamination of coastal environments. In an effort to further improve scientific understanding in this field, I make three recommendations for future research.

While there is significant uncertainty associated with the modelling process, this uncertainty has been quantified in such a way as to allow for the subsequent derivation of a proposed, site-specific sediment quality guideline that is protective of 95% of the harbour seal population. However, the performance analysis of a mechanistic bioaccumulation model that can be used to predict BSAFs is in and of itself a useful endeavour. With further refinement, such a model could be adapted and used as a management tool at other locations facing similar stressors from pollution input to derive site-specific environmental quality guidelines. As modelling is an iterative process, it is recommended that a next step in refining this model be to conduct a more thorough analysis of the food web structure in the location where it is applied (perhaps through scat analysis), as the food web characterization has been shown to affect the model output considerably.

The second recommendation I make is to adopt and test this model for other POPs. While PCBs are well-studied and perhaps better understood than many other POPs, there is great predictive potential of the model for emerging contaminants of concern (eg. PBDEs).

Finally, as suggested by Condon (2007), I recommend that the MOE revise its methodology for deriving SQGs for POPs. The incorporation of bioaccumulation in the derivation of environmental quality guidelines by regulatory authorities would have significant impact on the risk assessment and management of persistent organic pollutants in our environment, and ensure that the protection goal for *all* of the valued components of an ecosystem is met.

REFERENCE LIST

Arnot J.A., and F.A.P.C. Gobas. 2004. A food web bioaccumulation model for organic chemicals in aquatic ecosystems. *Environ Tox Chem* **23**(10): 2343-2355.

Boon, J.P., J. Van der Meer, C.R. Allchin, R.J. Law, J. Klungsoyr, P.E.G. Leonards, H. Spliid, E. Storr-Hansen, C. Mckenzie, and D.E. Wells. 1997. Concentration-dependent changes of PCB patterns in fish-eating mammals: Structural evidence for induction of cytochrome p450. *Arch Environ Contam Toxicol* **33**: 298-311.

Boon, J.P., P.J.H. Reijnders, J. Dols, P. Wensvoort, and M.T.J. Hillebrand. 1987. The kinetics of individual polychlorinated biphenyl congeners in female harbour seals (*Phoca vitulina*), with evidence for structure-related metabolism. *Aquat Toxicol* **10**: 307-324.

Brouwer, A., P.J.H. Reijinders, and J.H. Koeman. 1989. Polychlorinated biphenyl (PCB)-contaminated fish induces vitamin A and thyroid hormone deficiency in the common seal (*Phoca vitulina*). *Aquat Toxicol* **15**: 99-106.

Canadian Council of Ministers of the Environment (CCME). 1995. Protocol for the derivation of Canadian sediment quality guidelines for the protection of aquatic life. Retrieved March 20, 2008, from http://www.ccme.ca/assets/pdf/sedqg_protocol.pdf

CCME. 1999. Canadian tissue residue guidelines for the protection of wildlife consumers of aquatic biota: Polychlorinated biphenyls (PBCs). Canadian Environmental Quality Guidelines. Winnipeg, MB, pp 1-9.

CCREM (Canadian Council of Resource and Environmental Ministers). 1986. The PCB story. [in CCME 1999].

CCREM (Canadian Council of Resource and Environmental Ministers). 1987. PCB action plan: Final report. [in CCME 1999].

Chiba, I., A. Sakakibara, Y. Goto, T. Isono, Y. Yamamoto, H. Iwata, S. Tanabe, K. Shimazaki, F. Akahori, A. Kazusaka, and S. Fujita. 2001. Negative correlation between plasma thyroid hormone levels and chlorinated hydrocarbon levels accumulated in seals from the coast of Hokkaido, Japan. *Environ Tox Chem* **20(5)**: 1092-1097.

Colborn, T. and M.J. Smolen. 1996. Epidemiological analysis of persistent organochlorine contaminants in cetaceans. *Rev Environ Contam Toxicol* **146**: 91-172.

Condon, C. 2007. Development, evaluation, and application of a food web bioaccumulation model for PCBs in the Strait of Georgia, British Columbia. Report No. 413, School of Resource and Environmental Management. Burnaby, BC: Simon Fraser University.

Cottrell, P.E., S.J. Jeffries, B. Beck, and P.S. Ross. 1999. Birth mass, growth rates and weaning age of free-ranging harbor seal (*Phoca vitulina*) pups in southern British Columbia. *Proceedings,* 13th Biennial Conference on the Biology of Marine Mammals,Wailea, HI, USA, November 29–December 3, p 38.

Cullon, D.L., S.J. Jeffries, and P.S. Ross. 2005. Persistent organic pollutants in the diet of harbour seals (*Phoca vitulina*) inhabiting Puget Sound, Washington (USA), and the Strait of Georgia, British Columbia (Canada): A food basket approach. *Environ Tox Chem* **24**: 2562-2572.

Debier, C., P.P. Pomeroy, C. Dupont, C. Joiris, V. Comblin, E. Le Boulengé, Y. Larondelle, and J.P. Thomé. 2003. Dynamics of PCB transfer from mother to pup during lactation in UK grey seals *Halichoerus grypus:* Differences in PCB profile between compartments of transfer and changes during the lactation period. *Marine Ecology Progress Series* **247:** 249-256.

De Swart, R.L., P.S. Ross, H.H. Timmerman, W.C. Hijman, E. De Ruiter, A.k.D. Liem, A. Brouwer, H. Van Loveren, P.J.H. Reijnders, J.G. Vos, and A.D.M.E. Osterhaus. 1995. Short-term fasting does not aggravate immunosuppression in harbour seals (*Phoca vitulina*) with high body burdens of organoshlorines. *Chemosphere* **31:** 4289-4306.

Foy, R.J., and B.L. Norcross. 1999. Spatial and temporal variability in the diet of juvenile Pacific herring (*Clupea pallasi*) in Prince William Sound, Alaska. *Can J Zool* **77(5):** 697-706.

Gobas, F.A.P.C, and J.A. Arnot. 2010. Food web bioaccumulation model for polychlorinated biphenyls in San Francisco Bay, California, USA. *Environ Tox Chem **29**: 1385-1395.*

Gobas, F.A.P.C., and H.A. Morrison. 2000. Bioconcentration and biomagnification in the aquatic environment. Handbook of Property Estimation Methods for Chemicals: Environmental and Health Sciences. Eds. R.S. Boethling and D. MacKay. Boca Raton, Lewis Publishers.

Government of Canada. 1999. *Canadian Environmental Protection Act, 1999*. Available online at: http://www.ec.gc.ca/CEPARegistry/the_act/

Greig, D.J., G.M. Ylitalo, A.J. Hall, D.A. Fauquier, and F.M.D. Gulland. 2007. Transplacental transfer of organochlorines in California Sea Lions (*Zalophus californianus). Environ Tox Chem.* **26:** 37-44.

Health and Welfare Canada. 1990. Biological safety factors in toxicological risk assessment. Report 90-EHD-154. Environmental Health Directorate, Health Protection Branch, Ottawa.

Hill, P.R., K. Conway, D.G. Lintern, S. Meulé, K. Picard, and J.V. Barrie. 2008. Sedimentary processes and sediment dispersal in the southern Strait of Georgia, BC, Canada. *Marine Environmental Research* **66**: S39-S48.

Hutchinson, J.D. and M.P. Simmonds. 1994. Organochlorine contamination in pinnipeds. Reviews of Environmental Contamination and Toxicology. Volume 136. Springer-Verlag, New York Inc.

Jenssen, B.M., J.U. Skaare, S. Wolstad, A.T. Nastad, O. Haugen, B. Kloven, and E.G. Sormo. 1995. Biomarkers in blood to assess effects of polychlorinated biphenyls in free-living grey seal pups. In Blix, A.S., L. Walloe, O. Ulltang, eds, *Whales, Seals, Fish and Man: Developments in Marine Biology,* Vol 4. Elsevier, Amsterdam, The Netherlands, pp 607-615.

Johannessen, S.C., R.W. Macdonald, and D.W. Paton. 2003. A sediment and organic carbon budget for the greater Strait of Georgia. *Estuarine, Coastal and Shelf Science* **56**: 845-860.

Johannessen, S.C., R.W. Macdonald, C.A. Wright, B. Burd, D.P. Shaw, and A. van Roodselaar. 2008. Joined by geochemistry, divided by history: PCBs and PBDEs in Strait of Georgia sediments. *Marine Environmental Research* **66**: S112-S120.

Kalmaz, E.V. and G.D. Kalmaz. 1979. Transport, distribution, and toxic effects of polychlorinated biphenyls in ecosystems: Review. *Ecological Modelling* **6:** 223-251.

Kannan, K., A.L. Blankenship, P.D. Jones, and J.P. Giesy. 2000. Toxicity reference values for the toxic effects of polychlorinated biphenyls to aquatic mammals. *Human and Ecological Risk Assessment* **6**(1): 181-201.

Lahvis, G.P., R.S. Wells, D.W. Kuehl, J.L. Stewart, H.L. Rhinehart, and C.S. Via. 1995. Decreased lymphocyte responses in free-ranging bottlenose dolphins (*Tursiops truncatus*) are associated with increased concentrations of PCBs and DDT in peripheral blood. *Environ Health Perspect* **103**: 67-72.

Levings, C.D., J.E. Stein, C.M. Stehr, and S.C. Samis. 2003. Introduction to the PICES practical workshop; Objectives, overview of the study area, and projects conducted by the participants. *Marine Environmental Research* **57**: 3-18.

Mackay, D. 1989. Modeling the long-term behaviour of an organic contaminant in a large lake: Application to PCBs in Lake Ontario. *Journal of Great Lakes Research.* **15:** 283-297.

Matthews, H.B., J.R. Surles, J.G. Carver, and M.W. Anderson. 1984. Halogentated biphenyl transport by blood components. *Fundam Apply Toxicol.* **4:** 420-428.

Mos, L. 2006. <u>Physiological and molecular biomarkers of environmental contaminant-associated immunotoxicity in harbour seals (*Phoca Vitulina*).</u> Department of Biology. Victoria, BC: University of Victoria.

Mos, L. B. Morsey, S.J. Jeffries, M.B. Yunker, S. Raverty, S. De Guise, and P.S. Ross. 2006. Chemical and biological pollution contribute to the immunological profiles of free-ranging harbour seals. *Environ Toxicol Chem* **25:** 3110-3117.

Mos, L. M. Cameron, S.J. Jeffries, B.F. Koop, and P.S. Ross. In press 2010. Risk-based analysis of polychlorinated biphenyl toxicity in harbour seals. *Integrated Environmental Assessment and Management.*

Mos, L., M. Tabuchi, N. Dangerfield, S.J. Jeffries, B.F. Koop, and P.S. Ross. 2007. Contaminant-associated disruption of vitamin A and its receptor (RARα) in free-ranging harbour seals (*Phoca vitulina*). *Aquat Toxicol* **81**: 319-328.

Natale, D. 2007. <u>Development of a food web model to develop sediment target levels for selected persistent organic pollutants in Burrard Inlet.</u> Report No. 424, School of Resource and Environmental Management. Burnaby, BC: Simon Fraser University.

Pedersen, T. F., and R.D. Waters. 1989. Assessment of the enrichment of cadmium and other metals in the sediments of Port Moody and Indian Arm, British Columbia. Ocean Dumping Research Fund Report No. 15.2.2. Vancouver, BC: Environment Canada.

Reddy, M., S. Echols, B. Finklea, D. Busbee, J. Reif, and S. Ridgway. 1998. PCBs and chlorinated pesticides in clinically healthy *Tursiops truncates*: Relationships between levels in blubber and blood. *Mar Pollut Bull* **36**: 892-903.

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

Simms, W., and P.S. Ross. 2000. Developmental changes in circulatory vitamin A (retinol) and its transport proteins in free-ranging harbour seal (*Phoca vitulina*) pups. *Can J Zool* **78(10)**: 1862-1868.

Safe, S.H. 1994. Polychlorinated biphenyls (PCBs): Environmental impact, biochemical and toxic responses, and implications for risk assessment. *Crit Rev Toxicol* **24**: 87-149.

Tabuchi, M., N. Veldhoen, N. Dangerfield, C.C. Helbing, and P.S. Ross. 2006. PCB-related alteration of thyroid hormones and thyroid hormone receptor gene expression in free-ranging harbour seals (*Phoca vitulina*). *Environ Health Perspect* **114**: 1024-1031.

United Nations (UN). 2004. *Stockholm Convention on Persistent Organic Pollutants*. Accessed May 24, 2009. Available online at: http://www.pops.int/documents/convtext/convtext_en.pdf

United Nations (UN). 2009. *Stockholm Convention on Persistent Organic Pollutants: Adoption of Amendments to Annexes A, B, and C.* Accessed June 6, 2010. Available online at: *http://chm.pops.int/Convention/tabid/54/language/en-US/Default.aspx#convtext*

United Nations Environment Programme (UNEP). 1995. *Global Programme of Action Washington Declaration on Protection of the Marine Environment from Land-Based Activities*. Accessed June 6, 2010. Available online at: http://www.gpa.unep.org/documents/washington_declaration_english.pdf

US Environmental Protection Agency. 1992. Framework for ecological risk assessment. Risk Assessment Forum, Washington, DC. EPA/630/R-02/011.

West, J.E., S.M. O'Neill, and G.M. Ylitalo. 2008. Spatial extent, magnitude, and patterns of persistent organochlorines pollutants in Pacific herring (*Clupea pallasi*) populations in the Puget Sound (USA) and Strait of Georgia (Canada). *Science of the Total Environment.* **394:** 369-378.

Word, J.Q., B.B. Albrecht, M.L. Anghera, R. Baudo, M.S. Bay, D.M. Di Toro, J.L. Hyland, C.G. Ingersoll, P.F. Landrum, E.R. Long, J.P. Meador, D.W. Moore, T.P. O'Connor, and J.P. Shine. 2002. Predictive ability of sediment quality guidelines. In: Wenning, R.J., G.E. Batley, C.G. Ingersoll, and D.W. Moore, editors. Use of sediment quality guidelines and related tools for the assessment of contaminated sediments. Pensacola, USA: Society of Environmental Toxicology and Chemistry, pp 121-162.

Yunker, M. B., R.W. Macdonald, D. Goyette, D.W. Paton, B.R. Fowler, D. Sullivan, and J. Boyd. 1999. Natural and anthropogenic inputs of hydrocarbons to the Strait of Georgia. *Science of the Total Environment* **225:** 181–209.

Zile, M.H. 1992. Vitamin-A homeostasis endangered by environmental pollutants. *Proc Soc Exp Biol Med* **201:** 141-153.

FIGURES AND TABLES

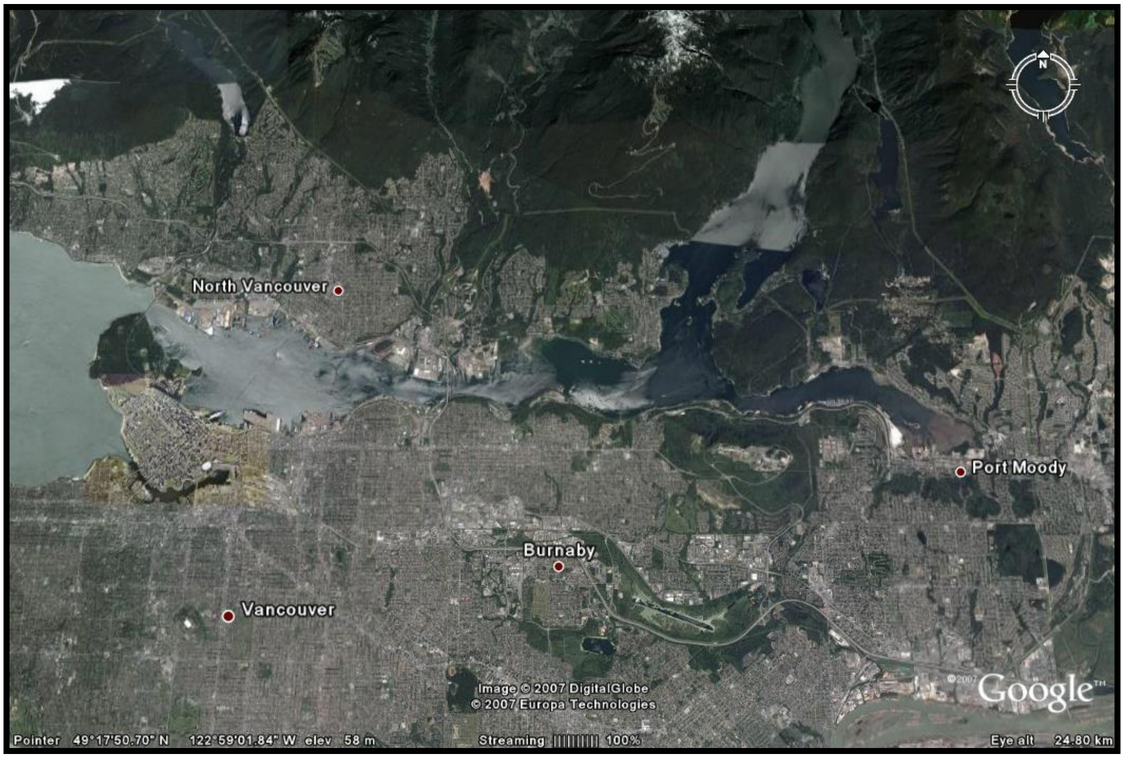

Figure 1: Map of the study site - Burrard Inlet, BC

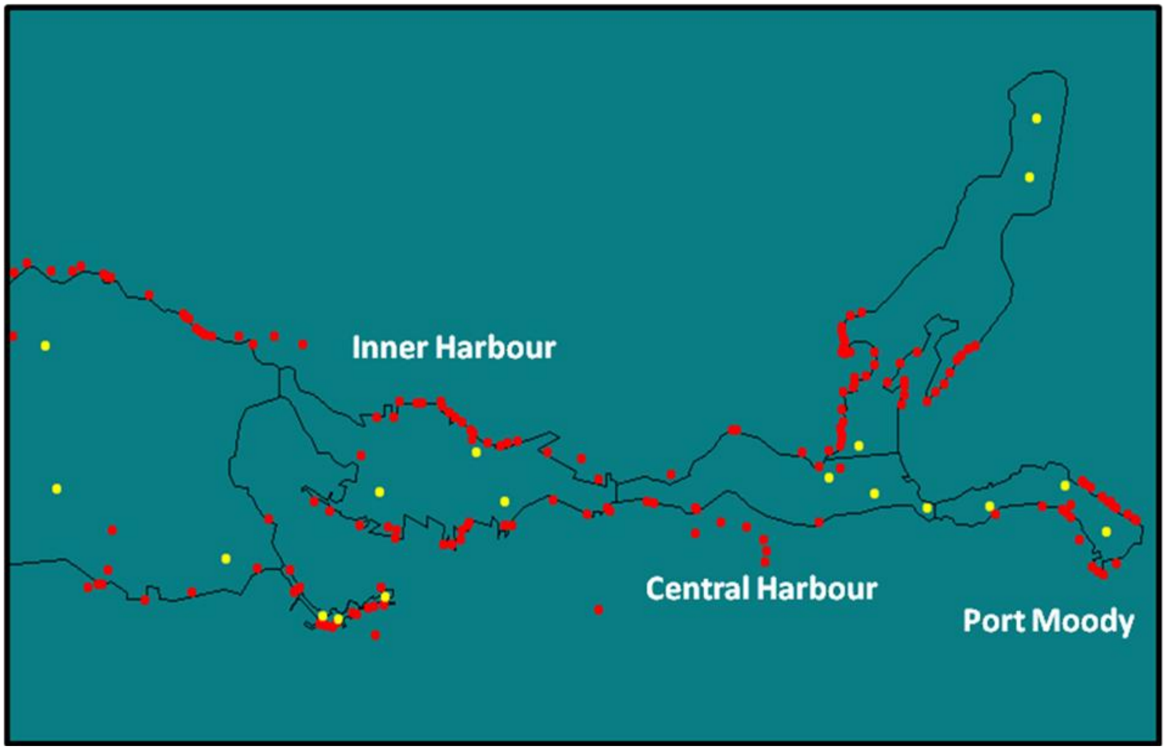

Figure 2: Location of point source pollution (red) and sediment sampling locations (yellow) within the study area. Surface sediment samples used in this study were collected from nine locations in Burrard Inlet.

Table 1: Field observations and description of captured harbour seals

Seal	Sex	Umbilicus	Age (d)	Weight (kg)	Length (cm)	Girth (cm)	Lipid Content (%)	Body condition	Teeth
PV06-28	F	dry/white, healed 0.5cm	25	20.1	89	68	37	eyes good, few sores on flippers	fully erupted
PV06-29	M	dry/white, healed 0cm	21	12.3	86	49	76	sore on hind flippers	fully erupted
PV06-30	F	dry/white, healed 0.2cm	21	13.1	83	57	72	eyes good, flippers good	fully erupted
PV06-31	F	dry/white, healed 0.3cm	21	14.7	79	63	63	eyes good, flippers good	fully erupted
PV06-33	M	dry/white, healed 0cm	17	12.6	76	58	70	eyes good, flippers good	almost fully erupted
PV06-34	M	dry/white, healed 0.2cm	28	21.3	88	72	60	eyes good, flippers good	fully erupted
				Mean lipid content:			**63**		

Note that a seventh seal, PV06-32, was sampled but not analyzed due to poor body condition.

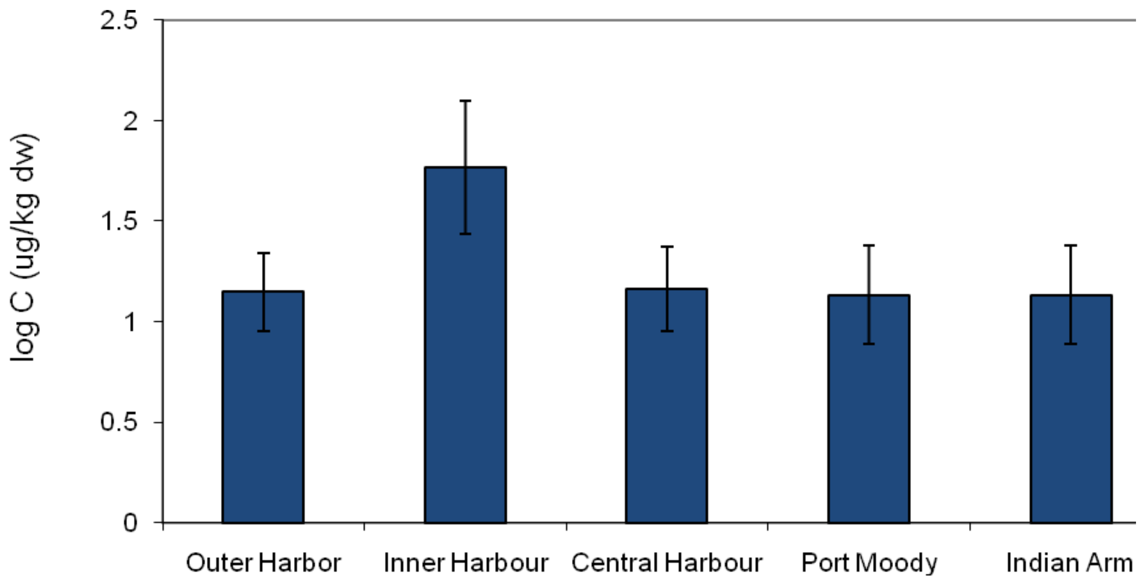

Figure 3: Comparison of observed geometric mean ∑PCB concentrations and their standard deviations in sediments from the five regions of Burrard Inlet. *Source: Natale (2007)*

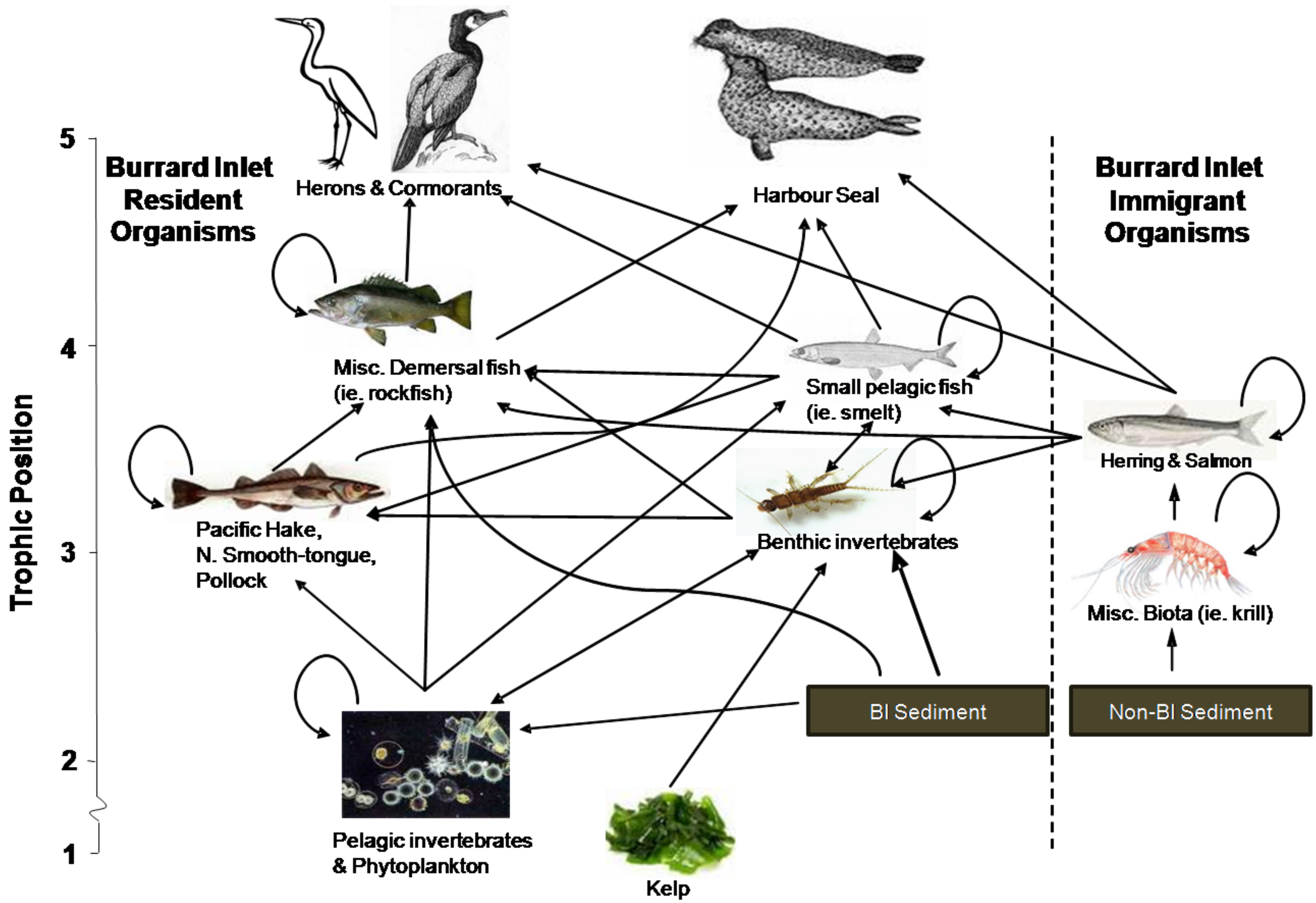

Figure 4: Schematic illustration of trophic linkages for major feeding groups of concern in Burrard Inlet. Arrows point from prey to predators. The trophic position scale (left) is based on the feeding relationships depicted in Table 2. *Adapted from Figure 3-1 in Condon (2007).*

Table 2: A matrix of diet compositions (percentage wet weight) for select organisms of Burrard Inlet. Values represent annual averages. The herring and harbour seal diets are in bold as they were modified from the food web used by Condon (2007).

Predator \ Prey	Sediment / Detritus	Phytoplankton	Kelp / Seagrass	Herbivorous zooplankton	N. plumchrus	P. minutus	Shellfish	Crab	Grazing invertebrates	Carnivorous zooplankton	Euphausia pacifica (krill)	Predatory invertebrates	Herring	Small pelagic fish (seal prey)	Small pelagic fish (bird prey)	River lamprey	Misc. demersal fish (seal prey)	Misc. demersal fish (bird prey)	Chum	Coho	Chinook	Pacific hake	Spiny dogfish	Pollock	Northern smooth-tongue	English sole	TOTALS (%)
Herbivorous zooplankton	30	70																									100
N. plumchrus	30	70																									100
P. minutus	30	70																									100
Shellfish	21	57.9	10	3	5	2	1	0.1																			100
Crab	43.8	0.2	10	6	2	2	15	1	20																		100
Grazing invertebrates	37.4	17.6	30	5	5	5																					100
Carnivorous zooplankton	5			35.9	40.4	10.2	3	0.2	0.3	5																	100
Euphausia pacifica (krill)	14	80.9		5			0.1																				100
Predatory invertebrates	50			6.5	5	5	5.5	1	10		11.1	3.6	2.2	0.1													100
Herring				**40**	**3**				**48**		**5**	**4**															**100**
Small pelagic fish (seal prey)		0.5		10	15	7	3	1	5	26.4	15	5	2	10					0.1								100
Small pelagic fish (bird prey)		0.5		10	15.1	7	3	1	5	26.4	17	5			10												100
River lamprey													64	19.9					5.3	5.1	5.1	0.1		0.5			100
Misc. demersal fish (seal prey)	10	0.5		4	5.1	5	17.2		9	15.4	10	4	3	5			10		0.1	0.4	0.3	1					100
Misc. demersal fish (bird prey)	10	0.5		5.8	5.1	5	17.2		9	15.4	13	4			5			10									100
Salmonids																											0
Pacific hake		0.5		2	2	1	3	0.1		16.3	70	0.5		4					0.1	0.1	0.1			0.2	0.1		100
Spiny dogfish				1	1	1	5	1	15	11	8	23.5	2	4.2		0.5	1		5.5	7	7	5.6	0.5	0.1		0.1	100
Pollock		1		1	3	2	0.1			9	66.8	5	5	1								1		0.1	5		100
Northern smooth-tongue		2		31	30	15	0.1		5.6	5	10.2			1								0.1					100
English sole				1.1	1.1		3		13			10		25			41.8					5					100
Double crested Cormorant (adult)													2.7		5.7			91.6									100
Great Blue Heron (adult)															10.9			89.1									100
Harbor Seal (adult/juvenile)													**32.4**	**4.3**			**15.5**		**1.4**	**1.4**	**1.4**	**42.6**		**1**			**100**

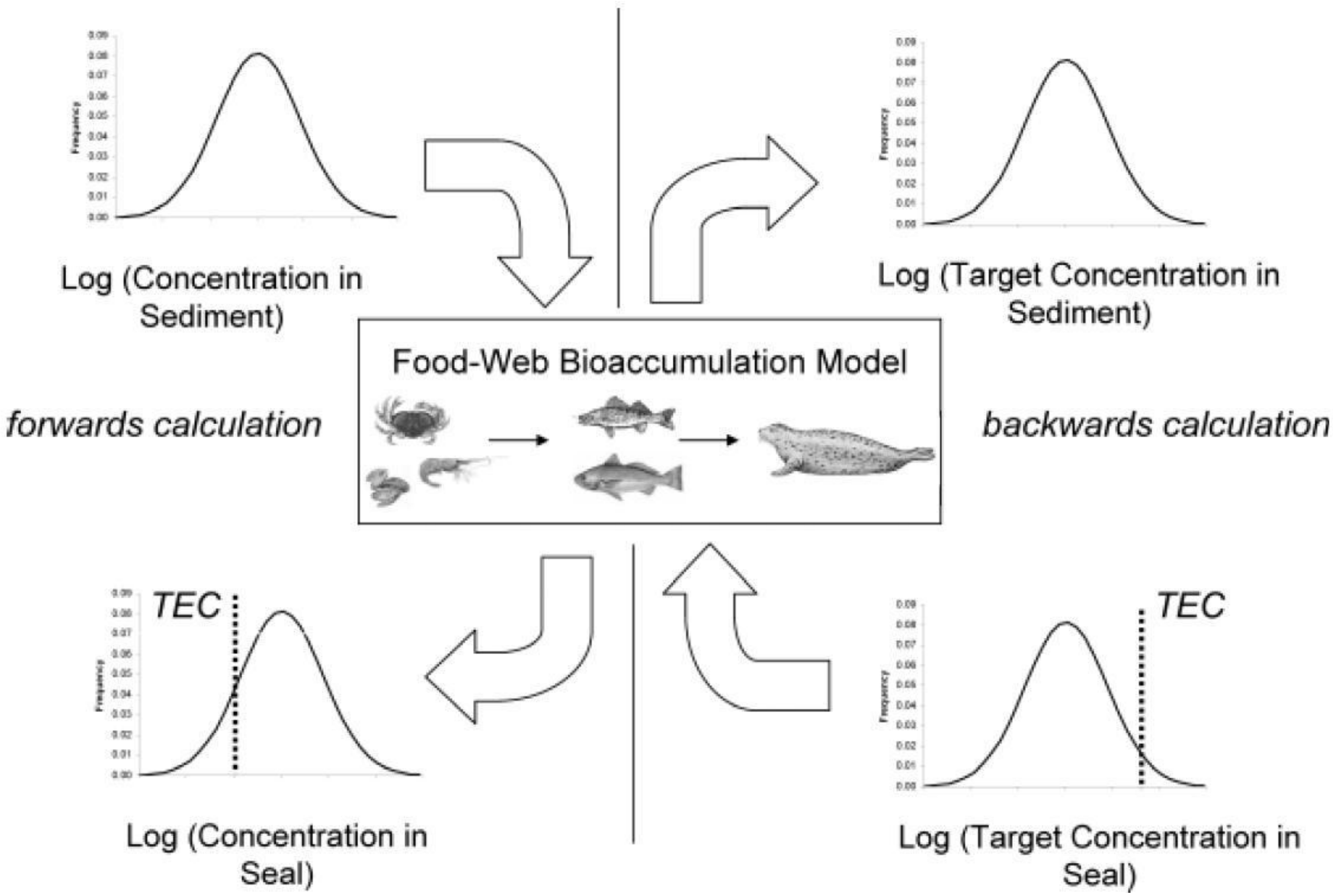

Figure 5: Conceptual diagram illustrating how the outputs of the model can be used both 'forwards' calculate PCB concentrations in wildlife based on measured PCB concentrations in sediment for a risk assessment (to calculate the fraction of the population with concentrations exceed threshold effects concentrations [TEC]), or 'backwards' calculate target sediment concentrations given an ecological criteria (biota concentration) to be met. ***Adapted from Figure 2 in Gobas and Arnot (2010).***

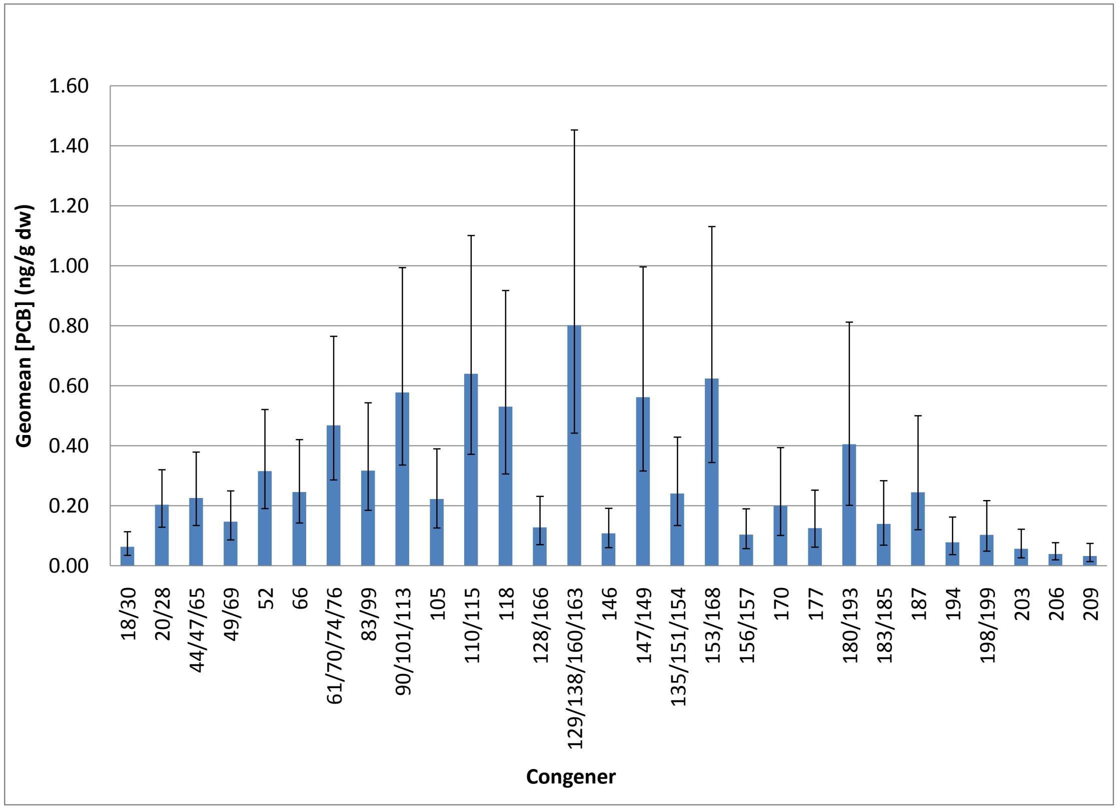

Figure 6: Geometric mean PCB concentrations (ng/g dry weight) in Burrard Inlet sediment (Central Harbour and Port Moody samples only) by congener. Only concentrations for model-input congeners are shown. Error bars represent the upper and lower 95% confidence intervals (n=6).

Table 3: Geometric mean PCB concentrations (ng/g dry weight) in Burrard Inlet sediment (Central Harbour and Port Moody samples only) by congener.

Congener	Geometric Mean (ng/g dw)	Lower 95% CI	Upper 95% CI
18/30	0.063	0.035	0.11
20/28	0.20	0.13	0.32
44/47/65	0.23	0.14	0.38
49/69	0.15	0.086	0.25
52	0.32	0.19	0.52
66	0.25	0.14	0.42
61/70/74/76	0.47	0.29	0.77
83/99	0.32	0.19	0.54
90/101/113	0.58	0.34	0.99
105	0.22	0.13	0.39
110/115	0.64	0.37	1.1
118	0.53	0.31	0.92
128/166	0.13	0.070	0.23
129/138/160/163	0.80	0.44	1.5
146	0.11	0.061	0.19
147/149	0.56	0.32	1.0
135/151/154	0.24	0.14	0.43
153/168	0.62	0.34	1.1
156/157	0.10	0.057	0.19
170	0.20	0.10	0.39
177	0.13	0.062	0.25
180/193	0.41	0.20	0.81
183/185	0.14	0.069	0.28
187	0.25	0.12	0.50
194	0.078	0.038	0.16
198/199	0.10	0.049	0.22
203	0.057	0.026	0.12
206	0.039	0.020	0.077
209	0.032	0.014	0.075
TOTAL PCBS	8.1	4.7	14

Table 4: Comparison of data analysis methods considered due to a lack of a conventional method detection limit

	Method Description	Concentration of ΣPCBs in seals (ng/g lipid)
1	If $C_{PCB\ congener} < C_{blank} \rightarrow$ use $C_{PCB\ congener}=0$ (include value in *n*)	1.3×10^4
2	If $C_{PCB\ congener} < C_{blank} \rightarrow$ use $C_{PCB\ congener}=1/2*C_{blank}$ (include value in *n*)	1.3×10^4
3	If $C_{PCB\ congener} < C_{blank} \rightarrow$ ignore $C_{PCB\ congener}$ (do not include value in *n*)	2.2×10^3
4	If $C_{PCB\ congener} <$ MDL of $3*C_{blank} \rightarrow$ use $C_{PCB\ congener}=0$ (do not include value in *n*)	1.3×10^4
5	If $C_{PCB\ congener} <$ MDL of $3*C_{blank} \rightarrow$ use $C_{PCB\ congener}=1/2*C_{blank}$ (include value in *n*)	1.3×10^4

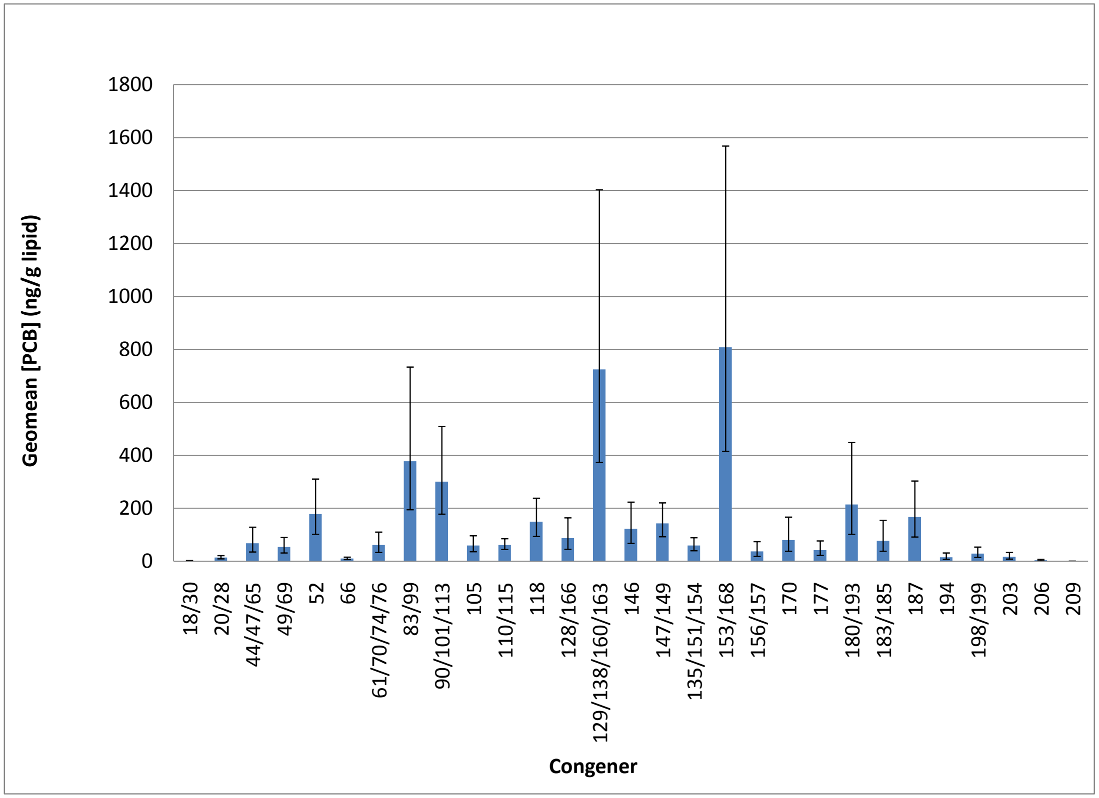

Figure 7: Geometric mean PCB congener concentrations (ng/g lipid) in harbour seal pups inhabiting Burrard Inlet. Error bars represent 95% confidence intervals (n=6).

Table 5: Geometric mean of observed PCB concentrations (ng/g lipid) in Burrard Inlet harbour seal pups by congener.

Congener	Geometric Mean (ng/g lipid)	Lower 95% CI	Upper 95% CI
18/30	1.4	0.81	2.5
20/28	14	9.2	21
44/47/65	68	36	130
49/69	53	32	90
52	180	100	310
66	10	6.5	15
61/70/74/76	61	33	110
83/99	380	190	730
90/101/113	300	180	510
105	59	36	96
110/115	61	44	85
118	150	93	240
128/166	87	46	164
129/138/160/163	720	370	1400
146	120	67	220
147/149	140	92	220
135/151/154	59	39	89
153/168	810	420	1600
156/157	37	18	74
170	80	38	170
177	42	22	77
180/193	210	100	450
183/185	77	38	160
187	170	92	300
194	15	7.1	32
198/199	28	15	54
203	16	8.1	33
206	3.7	1.9	7.2
209	0.57	0.33	0.98
TOTAL PCBS	4000	2200	7300

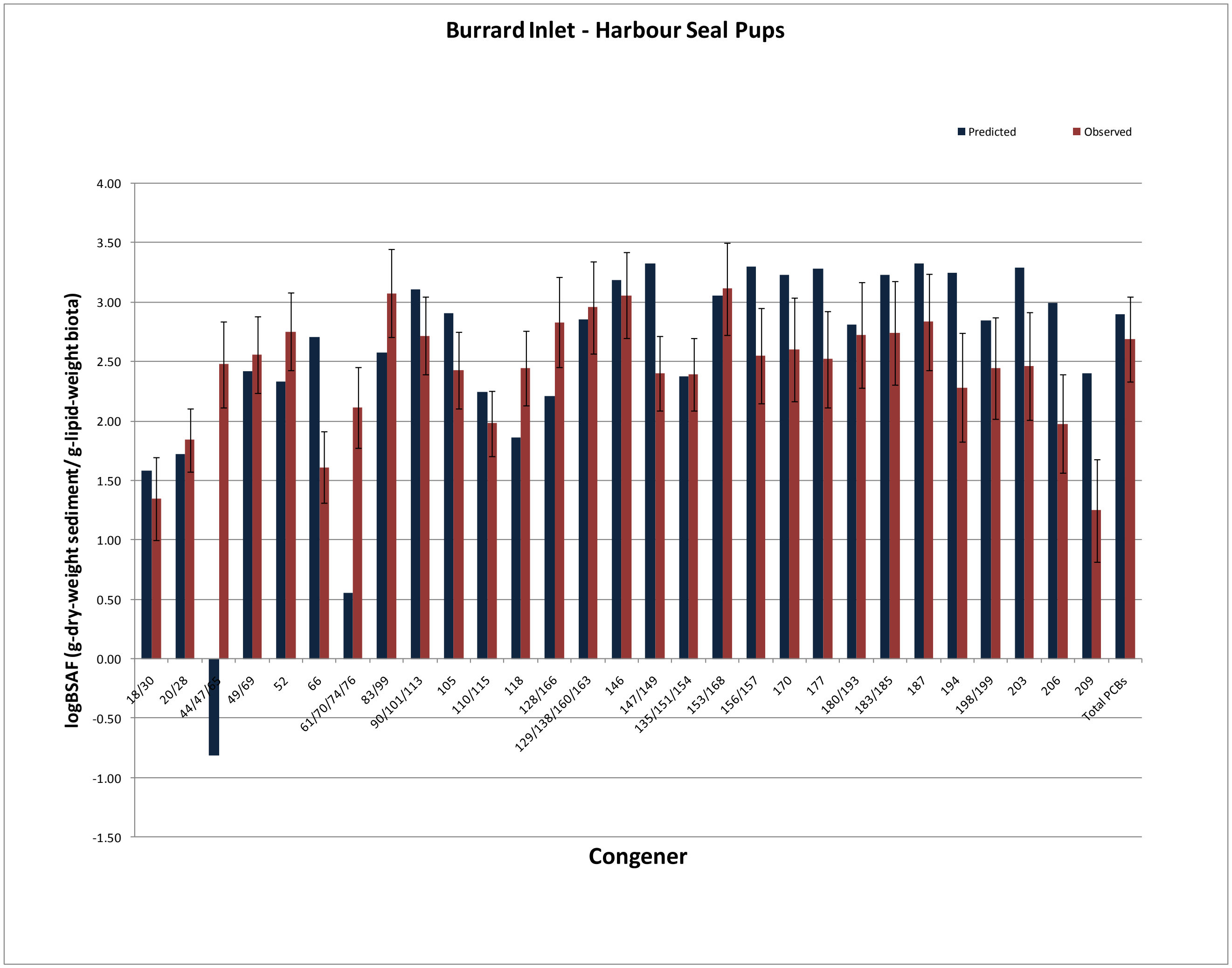

Figure 8: Model-predicted versus empirical BSAFs (g-dry-weight sediment/ g-lipid-weight biota) by congener for harbour seal pups. Model predictions are shown for the food web scenario in which herring is an immigrant species to the Inlet. Error bars for the empirical values represent one standard deviation of the observed mean (n=6).

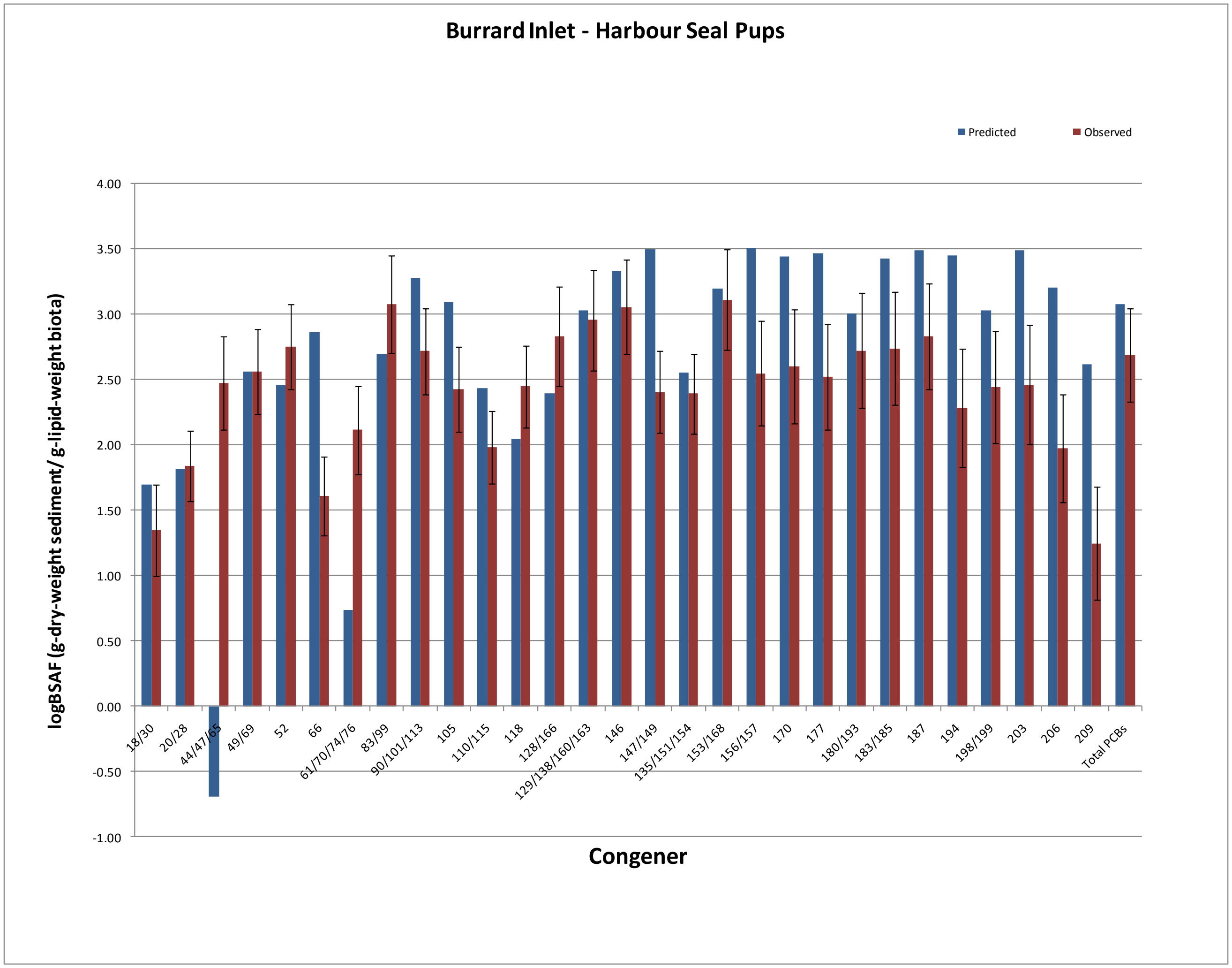

Figure 9: Model-predicted versus empirical BSAFs (g-dry-weight sediment/ g-lipid-weight biota) by congener for harbour seal pups. Model predictions are shown for the food web scenario in which herring is a native species to the Inlet. Error bars for the empirical values represent one standard deviation of the observed mean (n=6).

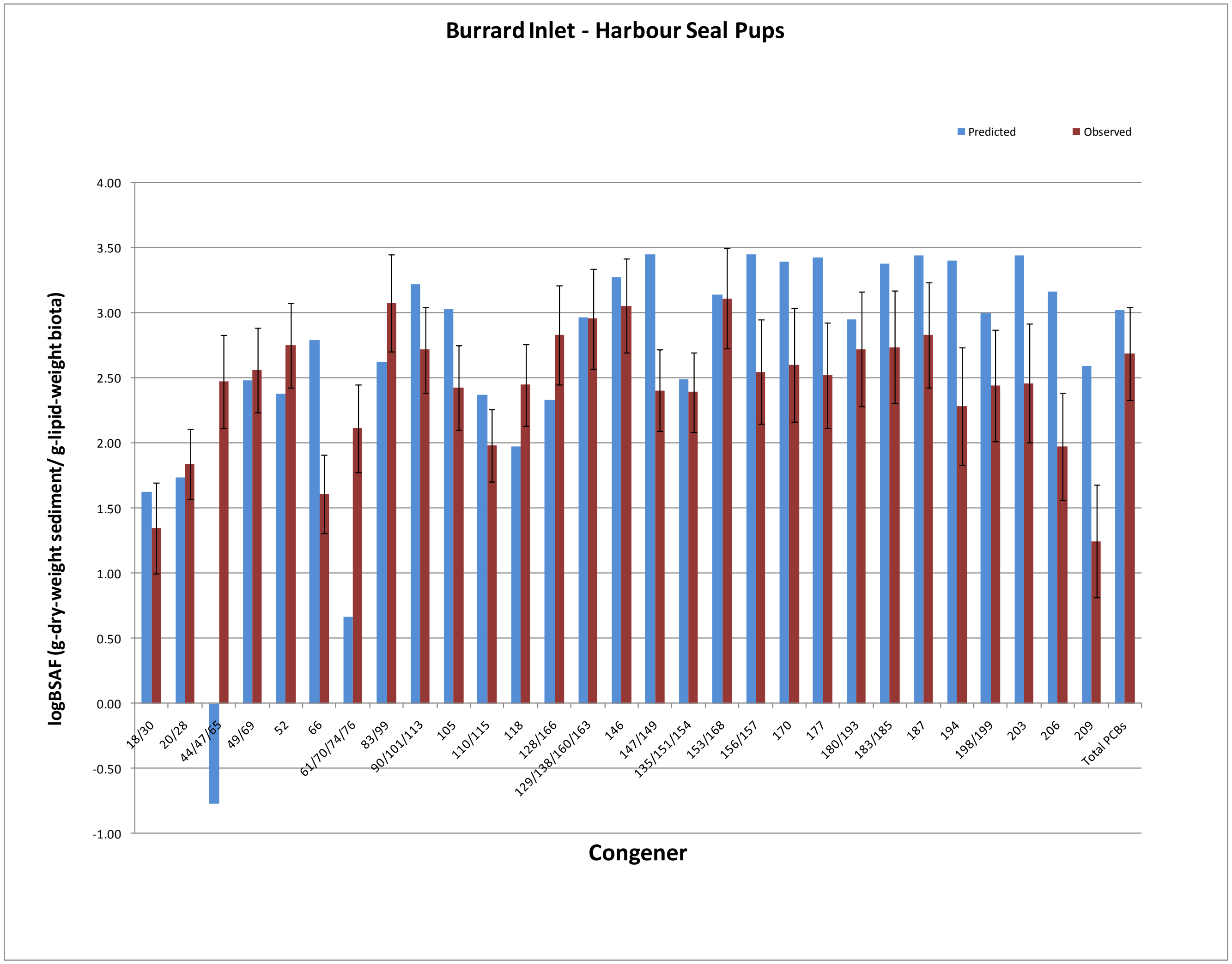

Figure 10: Model-predicted versus empirical BSAFs (g-dry-weight sediment/ g-lipid-weight biota) by congener for harbour seal pups. Model predictions are shown for the food web scenario in which herring is a semi-native species to the Inlet. Error bars for the empirical values represent one standard deviation of the observed mean (n=6).

Table 6: Model-predicted and empirical BSAFs of PCB congeners in harbour seal pups. The subset of sediment PCB concentrations from the Central Harbour and Port Moody were used in the calculation of the observed BSAFs.

	Model-predicted logBSAFs (g-dry-weight sediment/ g-lipid-weight biota)			Observed logBSAF			
Congener	FOOD WEB SCENARIO 1: Herring Immigrant	FOOD WEB SCENARIO 2: Herring Native	FOOD WEB SCENARIO 3: Herring Semi-native	Empirical (g dw sed/ g lipid biota)	SD (n=6)	Empirical (kg OC sed/ kg lipid biota)	SD (n=6)
18/30	1.6	1.7	1.6	1.3	0.35	-0.24	0.58
20/28	1.7	1.8	1.7	1.8	0.27	0.25	0.49
44/47/65	-0.8	-0.7	-0.8	2.5	0.36	0.89	0.55
49/69	2.4	2.6	2.5	2.6	0.32	0.98	0.51
52	2.3	2.5	2.4	2.8	0.32	1.2	0.57
66	2.7	2.9	2.8	1.6	0.30	0.025	0.46
61/70/74/76	0.55	0.74	0.66	2.1	0.34	0.53	0.55
83/99	2.6	2.7	2.6	3.1	0.37	1.5	0.56
90/101/113	3.1	3.3	3.2	2.7	0.33	1.1	0.56
105	2.9	3.1	3.0	2.4	0.32	0.84	0.54
110/115	2.2	2.4	2.4	2.0	0.28	0.40	0.53
118	1.9	2.0	2.0	2.4	0.31	0.86	0.54
128/166	2.2	2.4	2.3	2.8	0.38	1.3	0.59
129/138/160/163	2.9	3.0	3.0	3.0	0.39	1.4	0.56
146	3.2	3.3	3.3	3.1	0.36	1.5	0.53
147/149	3.3	3.5	3.4	2.4	0.31	0.82	0.49
135/151/154	2.4	2.6	2.5	2.4	0.31	0.81	0.47
153/168	3.1	3.2	3.1	3.1	0.39	1.5	0.53
156/157	3.3	3.5	3.5	2.5	0.40	0.96	0.60
170	3.2	3.4	3.4	2.6	0.43	1.0	0.51
177	3.3	3.5	3.4	2.5	0.41	0.94	0.46

	Model-predicted logBSAFs (g-dry-weight sediment/ g-lipid-weight biota)			Observed logBSAF			
Congener	**FOOD WEB SCENARIO 1: Herring Immigrant**	**FOOD WEB SCENARIO 2: Herring Native**	**FOOD WEB SCENARIO 3: Herring Semi-native**	**Empirical (g dw sed/ g lipid biota)**	**SD (n=6)**	**Empirical (kg OC sed/ kg lipid biota)**	**SD (n=6)**
180/193	2.8	3.0	3.0	2.7	0.44	1.1	0.49
183/185	3.2	3.4	3.4	2.7	0.43	1.2	0.47
187	3.3	3.5	3.4	2.8	0.40	1.3	0.42
194	3.2	3.4	3.4	2.3	0.45	0.70	0.45
198/199	2.8	3.0	3.0	2.4	0.43	0.86	0.42
203	3.3	3.5	3.4	2.5	0.45	0.88	0.45
206	3.0	3.2	3.2	2.0	0.41	0.39	0.46
209	2.4	2.6	2.6	1.2	0.43	-0.34	0.43
Total PCBs	2.9	3.1	3.0	2.7	0.26	1.1	0.53

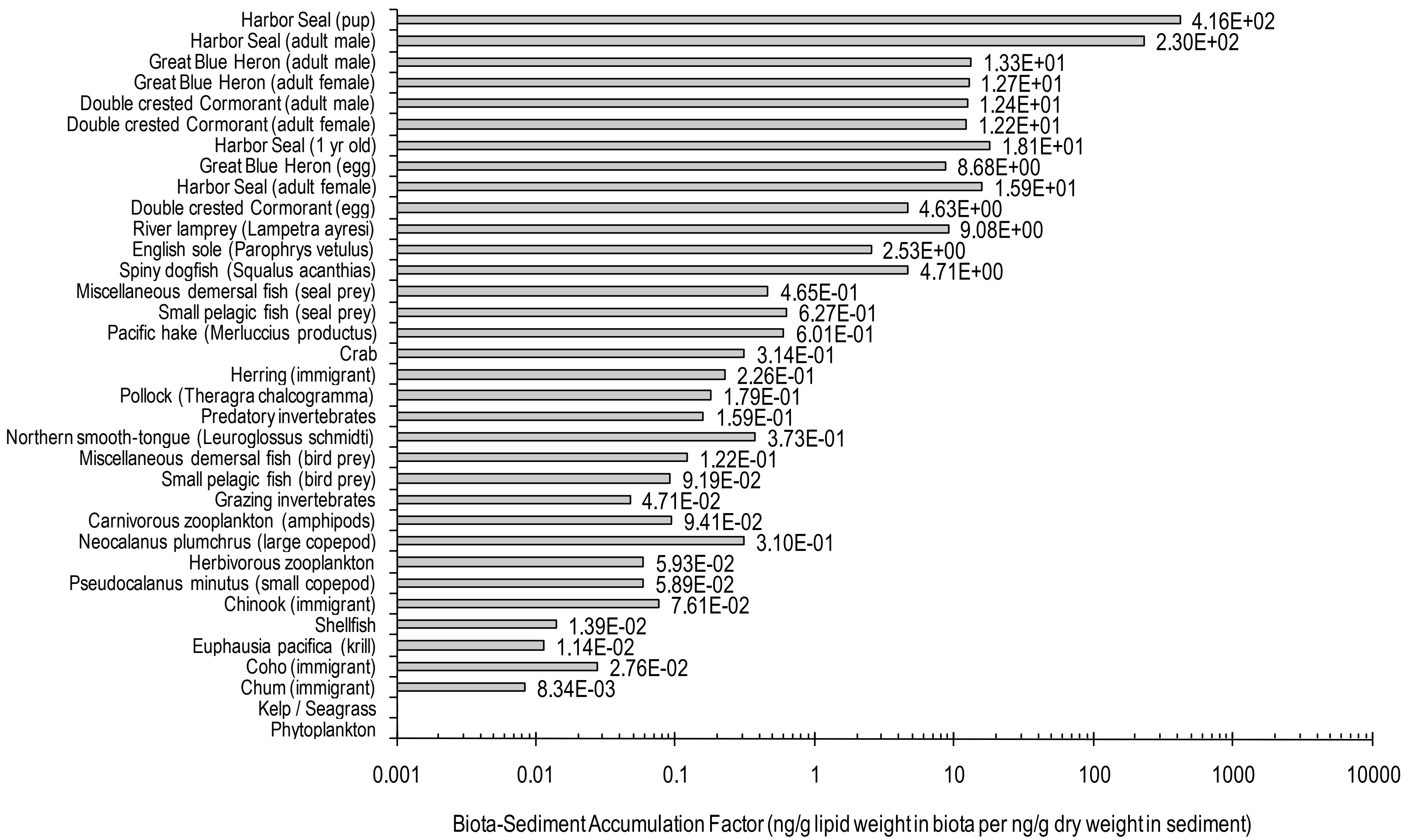

Figure 11: Model-predicted BSAF values for total PCBs by species for the scenario where herring is a semi-native species.

Table 7: Example of values used to calculate mean MB* and associated 95% confidence interval, in this case for food web scenario 2 (assuming herring is a native species) using only the subset of sediment data from the Central Harbour and Port Moody regions.

Seal Number	$BSAF_P$ (g-dry-wt sed/ g-lipid-wt biota)	$BSAF_O$ (g-dry-wt sed/ g-lipid-wt biota)	log (BSAFp/BSAFo)
PV-06-028	1200	230	0.71
PV-06-029	1200	940	0.10
PV-06-030	1200	350	0.53
PV-06-031	1200	340	0.54
PV-06-033	1200	490	0.38
PV-06-034	1200	1100	0.035

Table 8: The mean model bias (MB*) comparing observed and model-predicted BSAF values and their 95% confidence intervals for harbour seal pups of Burrard Inlet. Values are shown by food web scenario using the entire sediment dataset, and a subset from the Central Harbour and Port Moody region only.

	Inner Harbour, Central Harbour, and Port Moody Sediment Data			Central Harbour and Port Moody Sediment Data Only		
	Mean MB* (n=6)	**Lower 95% Confidence Interval**	**Upper 95% Confidence Interval**	**Mean MB* (n=6)**	**Lower 95% Confidence Interval**	**Upper 95% Confidence Interval**
FOOD WEB SCENARIO 1: Herring Immigrant	2.6	1.5	4.7	1.6	0.90	2.9
FOOD WEB SCENARIO 2: Herring Native	4.1	2.3	7.4	2.4	1.3	4.3
FOOD WEB SCENARIO 3: Herring Semi-native	3.7	2.0	6.6	2.1	1.2	3.8

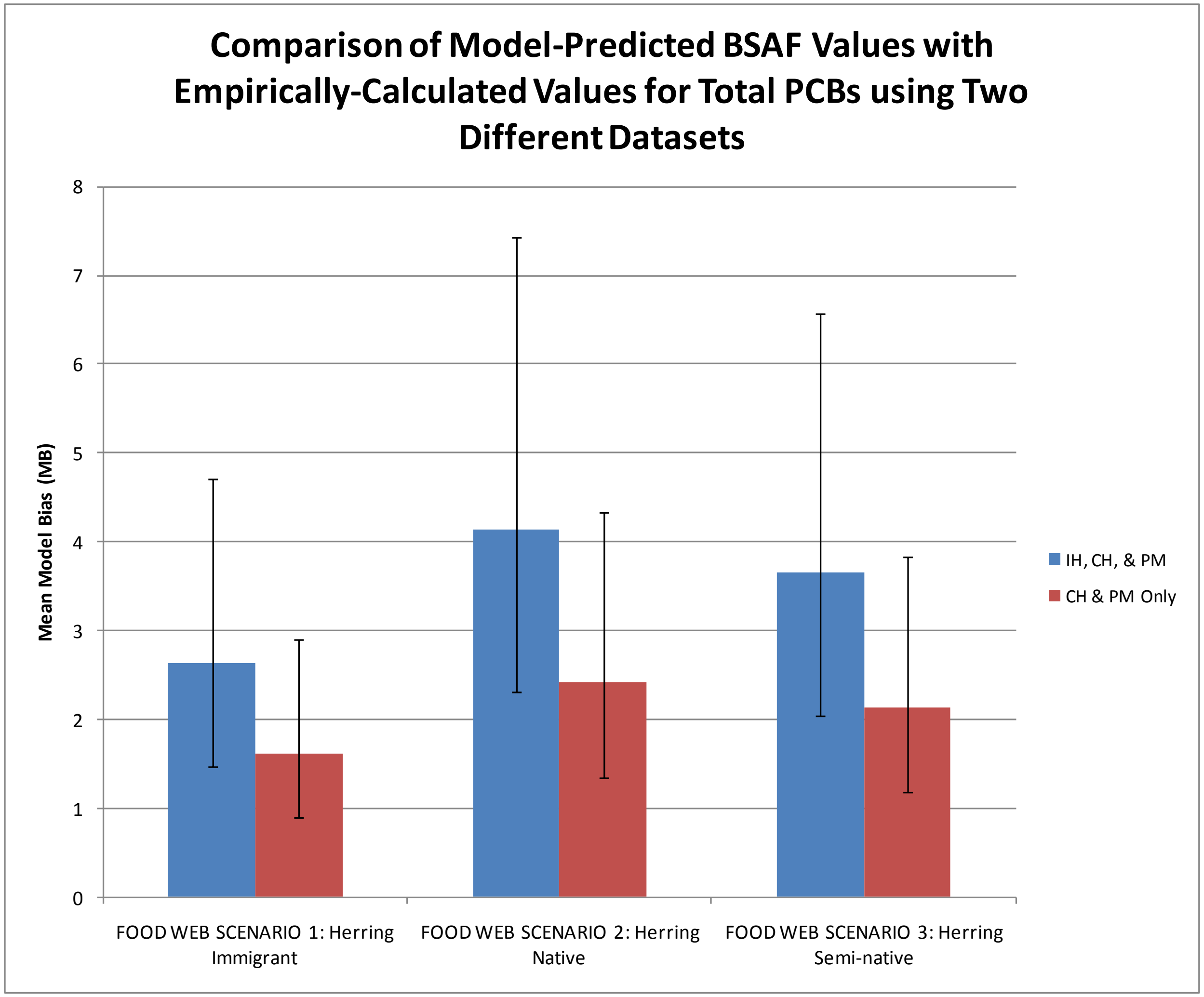

Figure 12: Comparison of model-predicted and empirically-calculated BSAFs for total PCBs using two different sediment datasets. Errors bars represent the 95% confidence intervals of the mean. IH = Inner Harbour, CH = Central Harbour, PM = Port Moody.

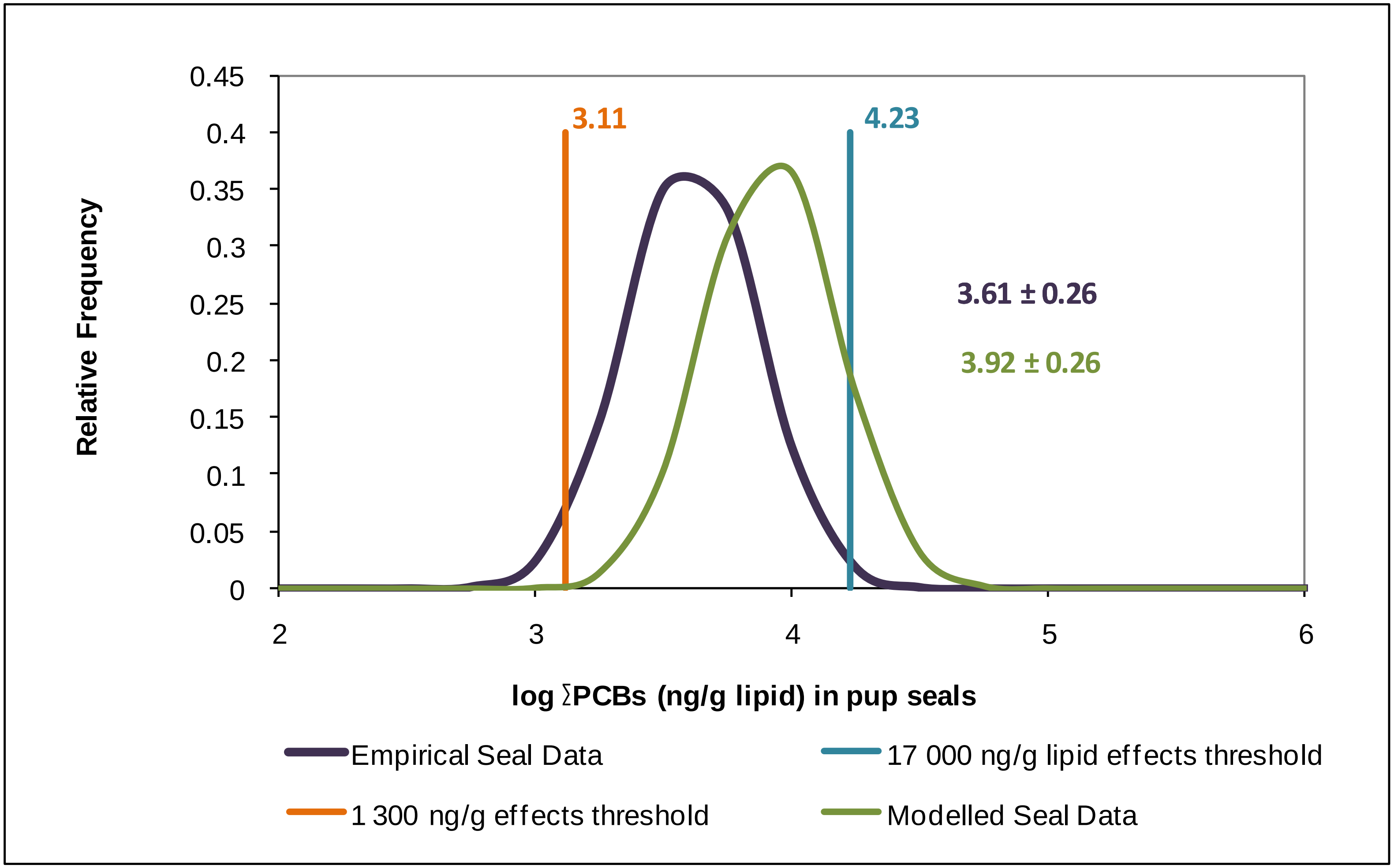

Figure 13: Distribution of observed ∑PCB concentrations (congener subset used in model only) in harbour seal pups inhabiting Burrard Inlet plotted along with both the conventional value for harbour seal toxicity ($\log C_{toxic}$ = 4.2 ng/g lipid; *Kannan et al.* 2000) and the harbour seal PCB toxicity threshold ($\log C_{toxic}$ = 3.1 ng/g lipid) of Mos *et al.* (in press 2010).

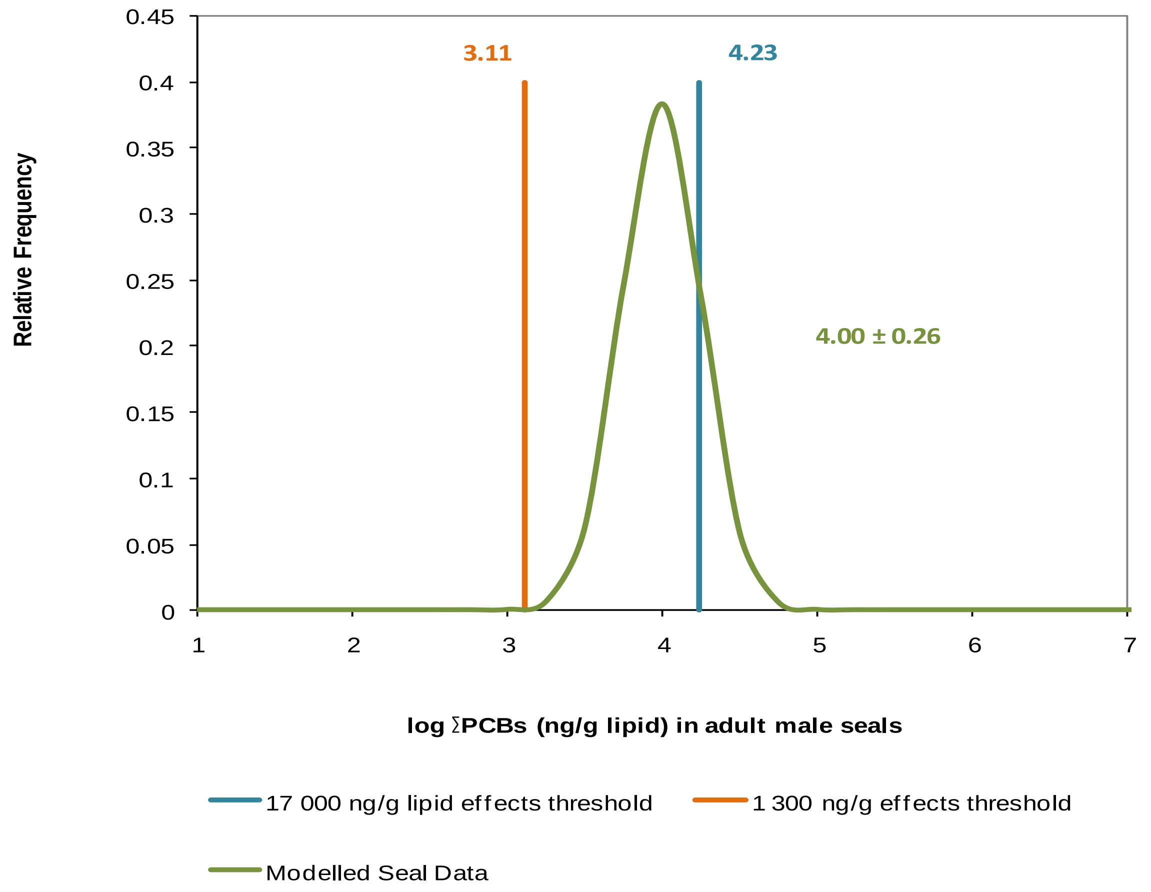

Figure 14: **Distribution of observed ∑PCB concentrations (congener subset used in model only) in adult male seals inhabiting Burrard Inlet plotted along with both the conventional value for harbour seal toxicity (logCtoxic = 4.2 ng/g lipid; Kannan et al. 2000) and the harbour seal PCB toxicity threshold (log C_{toxic} = 3.1 ng/g lipid) of Mos *et al.* (in press 2010).**

Table 9: Target SQGs back calculated using the empirical and model-predicted BSAF value (calculated with the Central Harbour and Port Moody sediment data subset under the three food web assumptions) and two Ctoxic values, compared to the current PCB SQG. Calculations are shown for both pup seals (empirical and model-predicted calculation) and adult male seals (empirical calculation only).

		Ctoxic used	
PUP SEALS	**Food web scenario (for predicted BSAF)**	**17 mg/kg lipid (Kannan *et al.* 2000)**	**1.3 mg/kg lipid (Mos *et al.* in press 2010)**
Target SQG (ng/g dry wt) (using model-predicted BSAF)	herring semi-native	5.0	0.38
	herring native	4.4	0.34
	herring immigrant	6.6	0.51
Target SQG (ng/g dry wt) (using empirical BSAF)		11	0.82
ADULT MALE SEALS			
Target SQG (ng/g dry wt) (using model-predicted BSAF)	herring semi-native	4.2	0.32
	herring native	5.9	0.30
	herring immigrant	3.9	0.45

Current SQG	**(ng/g dry wt)**	20

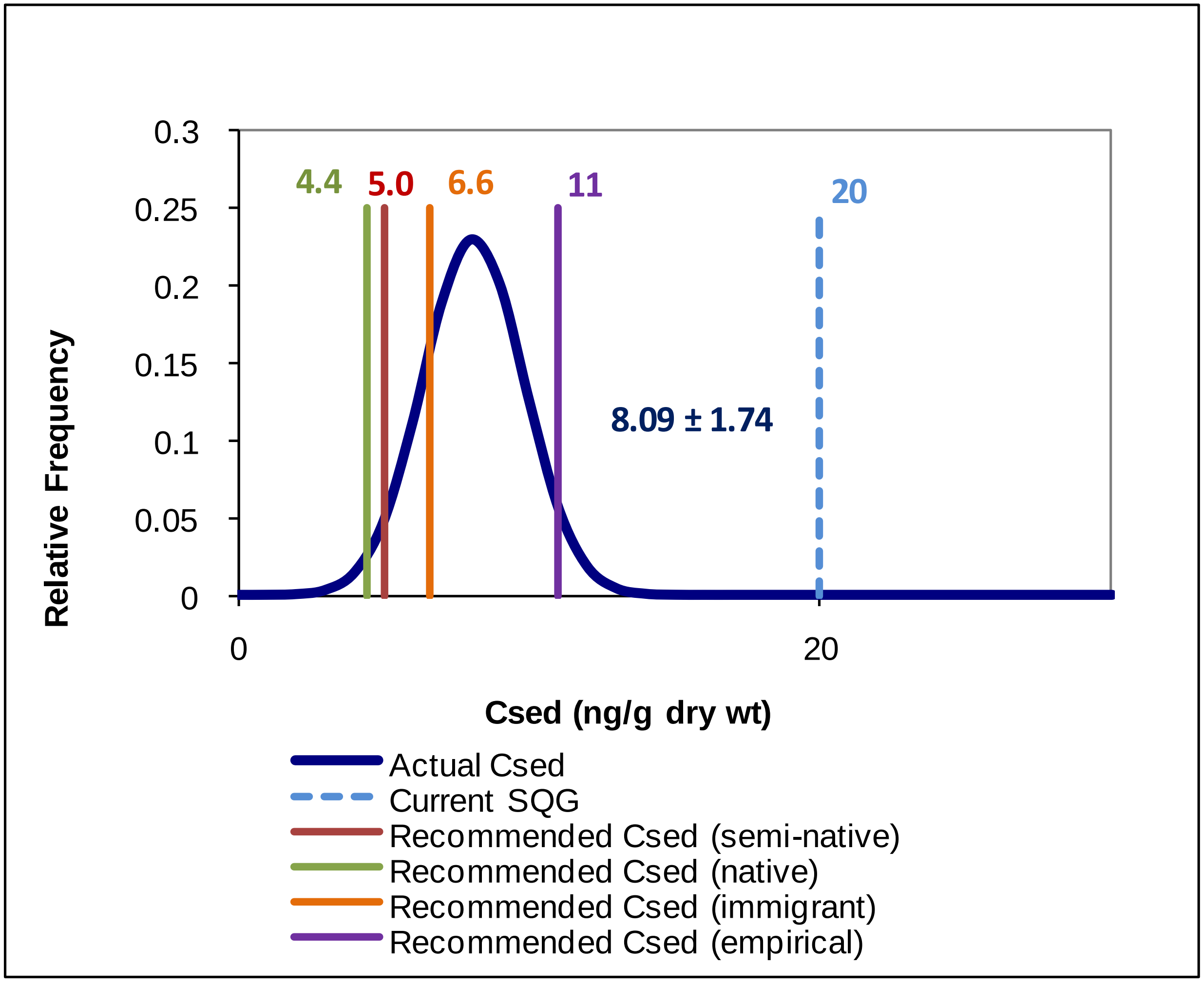

Figure 15: **Back calculated target sediment concentrations for the three proposed food web scenarios (herring as a native, semi-native, and immigrant species to the inlet) use the toxic threshold concentration of 17 mg/kg lipid. The current sediment distribution in the Central Harbour and Port Moody regions of the inlet, along with the current provincial SQG, are also shown for comparison.**

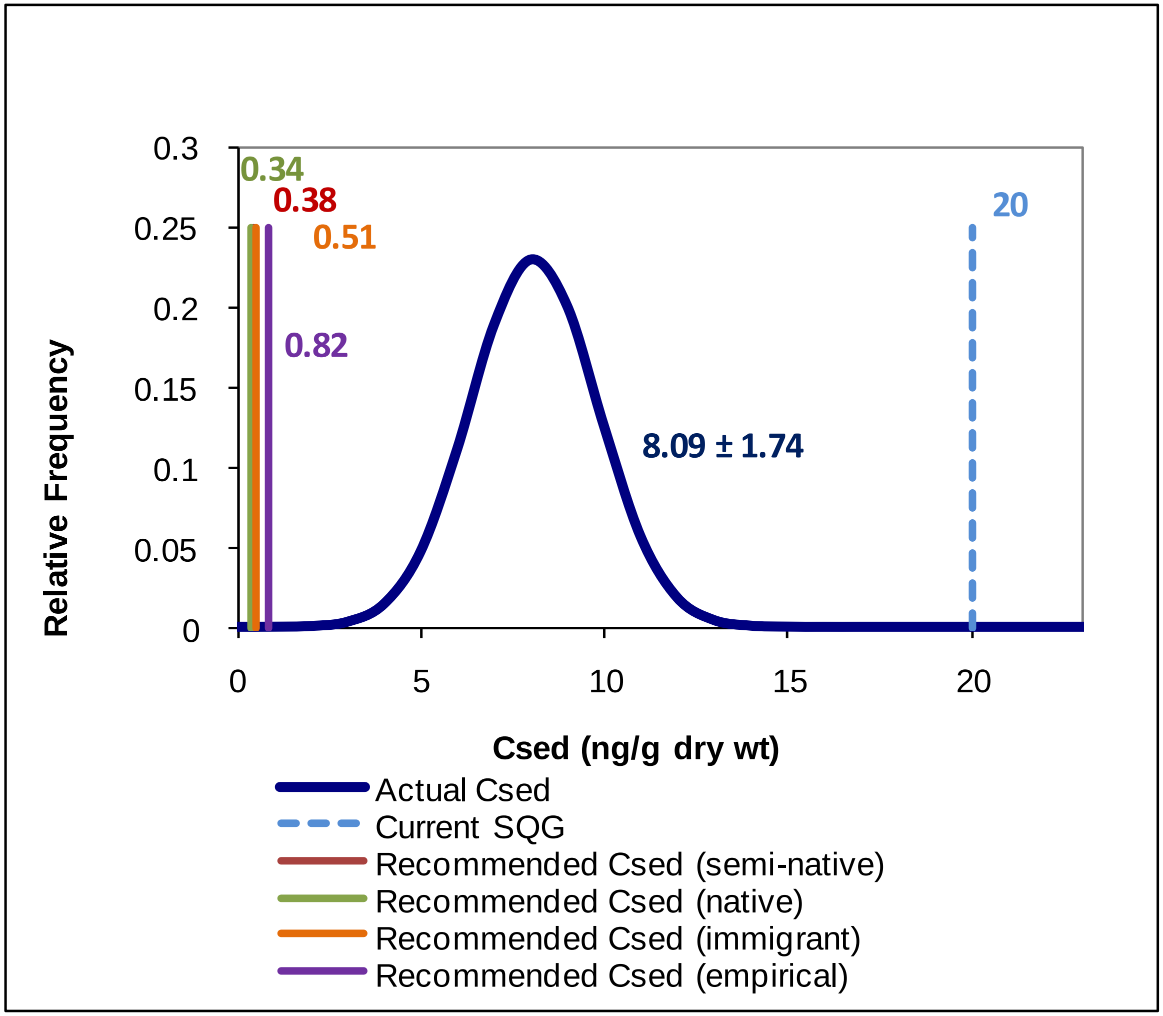

Figure 16: **Back calculated target sediment concentrations for the three proposed food web scenarios (herring as a native, semi-native, and immigrant species to the inlet) use the toxic threshold concentration of 1.3 mg/kg lipid. The current sediment distribution in the Central Harbour and Port Moody regions of the inlet, along with the current provincial SQG, are also shown for comparison.**

APPENDICES

Seal k_M Values

Table 10: Estimated seal metabolic rate constants

Congener or homolog name	Metabolic rate (male seals)	Metabolic rate (female seals)
	d^{-1}	d^{-1}
18/30	1.47E-02	1.47E-02
20/28	2.29E-02	2.29E-02
44/47/65	7.80E-01	7.80E-01
49/69	3.65E-03	3.65E-03
52	0.00E+00	0.00E+00
66	2.00E-01	2.00E-01
61/70/74/76	8.05E-03	8.05E-03
83/99	0.00E+00	0.00E+00
90/101/113	2.66E-03	2.66E-03
105	2.66E-02	2.66E-02
110/115	4.43E-02	4.43E-02
118	3.03E-02	3.03E-02
128/166	7.80E-03	7.80E-03
129/138/160/163	2.25E-03	2.25E-03
146	0.00E+00	0.00E+00
147/149	2.23E-02	2.23E-02
135/151/154	3.46E-03	3.46E-03
153/168	0.00E+00	0.00E+00
156/157	6.38E-04	6.38E-04
170	0.00E+00	0.00E+00
177	9.59E-03	9.59E-03
180/193	0.00E+00	0.00E+00
183/185	0.00E+00	0.00E+00
187	6.61E-04	6.61E-04
194	0.00E+00	0.00E+00
198/199	1.65E-04	1.65E-04
203	0.00E+00	0.00E+00
206	0.00E+00	0.00E+00
209	0.00E+00	0.00E+00

Empirical Model Input Data

Table 11: Empirical herring data used as model input (n = 2)

Congener or homolog rank order	Congener or homolog name	Herring	
		geomean	geo-SD
		ng/g (ww)	ng/g (ww)
1	18/30	4.22E-02	2.84E-02
2	20/28	3.96E-01	2.40E-02
3	44/47/65	4.87E-01	8.18E-02
4	49/69	2.29E-01	2.68E-03
5	52	7.10E-01	5.49E-02
6	66	3.92E-01	1.96E-02
7	61/70/74/76	8.38E-01	1.28E-02
8	83/99	1.34E+00	1.39E-01
9	90/101/113	1.77E+00	1.34E-01
10	105	5.41E-01	8.60E-02
11	110/115	1.43E+00	1.55E-01
12	118	1.43E+00	9.64E-02
13	128/166	3.35E-01	9.42E-02
14	129/138/160/163	2.82E+00	1.10E-01
15	146	5.80E-01	1.18E-01
16	147/149	1.88E+00	1.43E-01
17	135/151/154	7.69E-01	1.61E-01
18	153/168	3.30E+00	1.18E-01
19	156/157	1.56E-01	7.87E-02
20	170	2.80E-01	6.14E-02
21	177	3.16E-01	8.92E-02
22	180/193	8.43E-01	5.89E-02
23	183/185	3.32E-01	8.57E-02
24	187	9.88E-01	9.36E-02
25	194	1.12E-01	9.45E-02
26	198/199	2.12E-01	1.62E-01
27	203	1.07E-01	1.60E-01
28	206	5.09E-02	9.67E-02
29	209	2.48E-02	7.55E-02

Reference: data supplied by Dr. Jim West
Year collected: 2004

Table 12: Empirical salmon data used as model input (n = 3 for all salmon species)

Congener or homolog name	Chum		Coho		Chinook	
	geomean	**geo-SD**	**geomean**	**geo-SD**	**geomean**	**geo-SD**
	ng/g (ww)	ng/g (ww)	ng/g (ww)	ng/g (ww)	ng/g (ww)	ng/g (ww)
18/30	1.58E-02	5.85E-02	3.74E-02	1.17E-01	3.26E-02	1.42E-01
20/28	7.65E-02	3.83E-02	1.84E-01	6.36E-02	2.41E-01	1.43E-01
44/47/65	5.12E-02	4.08E-02	1.25E-01	5.72E-02	2.56E-01	1.31E-01
49/69	3.11E-02	4.23E-02	7.20E-02	7.19E-02	1.53E-01	1.37E-01
52	8.53E-02	3.82E-02	2.34E-01	6.61E-02	4.17E-01	1.32E-01
66	3.77E-02	3.26E-02	9.27E-02	4.94E-02	2.40E-01	1.40E-01
61/70/74/76	9.81E-02	4.06E-02	2.59E-01	6.44E-02	5.35E-01	1.43E-01
83/99	7.68E-02	1.68E-02	2.18E-01	7.74E-02	6.92E-01	1.24E-01
90/101/113	1.20E-01	1.30E-02	3.27E-01	6.92E-02	9.73E-01	1.31E-01
105	2.28E-02	2.11E-02	6.51E-02	1.04E-01	2.21E-01	9.58E-02
110/115	5.88E-02	3.13E-02	1.71E-01	9.38E-02	6.23E-01	1.08E-01
118	6.52E-02	2.16E-02	1.94E-01	8.01E-02	6.08E-01	1.08E-01
128/166	1.38E-02	1.74E-02	2.88E-02	1.06E-01	1.51E-01	1.20E-01
129/138/160/163	1.26E-01	1.03E-02	3.14E-01	8.64E-02	1.28E+00	1.08E-01
146	2.86E-02	3.34E-02	7.48E-02	8.74E-02	3.00E-01	1.27E-01
147/149	1.05E-01	3.66E-02	2.65E-01	7.80E-02	8.48E-01	1.10E-01
135/151/154	6.37E-02	2.52E-02	1.30E-01	8.02E-02	3.88E-01	1.24E-01
153/168	1.57E-01	3.15E-02	4.11E-01	9.12E-02	1.57E+00	1.12E-01
156/157	4.09E-03	4.35E-02	1.11E-02	6.31E-02	5.61E-02	9.04E-02
170	1.12E-02	3.49E-02	1.46E-02	1.09E-01	1.31E-01	8.47E-02
177	1.32E-02	2.58E-02	2.23E-02	8.73E-02	1.45E-01	1.09E-01
180/193	3.36E-02	2.35E-02	5.47E-02	9.70E-02	4.08E-01	1.03E-01
183/185	1.47E-02	3.52E-02	2.63E-02	9.38E-02	1.64E-01	1.10E-01
187	4.42E-02	2.42E-02	8.00E-02	1.11E-01	4.68E-01	1.21E-01
194	3.41E-03	3.58E-02	5.13E-03	1.37E-01	5.66E-02	2.79E-02
198/199	7.18E-03	2.67E-02	9.94E-03	1.35E-01	9.61E-02	9.84E-02
203	3.38E-03	4.71E-02	4.45E-03	1.45E-01	5.15E-02	8.88E-02
206	1.44E-03	6.01E-02	2.34E-03	1.54E-01	2.40E-02	5.76E-02
209	9.18E-04	8.08E-02	2.28E-03	1.34E-01	1.30E-02	7.68E-02

Reference: data supplied by Dr. David O. Carpenter
Year collected: 2003

Model Parameter Values

Table 13: Molecular weight, LeBas molar volume, $\log K_{ow}$, and $\log K_{oa}$ values of PCB congeners as used in the model

Congener or homolog name	Molecular weight	LeBas molar volume	Log Kow fw @ 9.5°C	Log Kow fw @ 37.5°C	Log Koa @ 10.5°C	Log Koa @ 37.5°C
	g/mol	cm^3/mol	Unitless	Unitless	Unitless	Unitless
18/30	257.5	247.4	5.37	5.27	7.93	6.82
20/28	257.5	247.4	5.80	5.70	8.61	7.44
44/47/65	257.5	247.4	5.96	5.86	9.47	8.22
49/69	292.0	268.4	5.88	5.78	9.18	7.96
52	292.0	268.4	5.97	5.87	8.81	7.62
66	292.0	268.4	6.33	6.23	9.87	8.58
61/70/74/76	292.0	268.4	6.33	6.23	9.68	8.41
83/99	326.4	289.4	6.53	6.42	9.87	8.58
90/101/113	326.4	289.4	6.52	6.41	9.85	8.56
105	326.4	289.4	6.79	6.68	10.7	9.36
110/115	326.4	289.4	6.62	6.51	9.76	8.48
118	326.4	289.4	6.88	6.77	10.4	9.09
128/166	360.9	310.4	6.87	6.77	10.5	9.16
129/138/160/163	360.9	310.4	6.96	6.86	10.6	9.26
146	360.9	310.4	7.02	6.92	10.6	9.22
147/149	360.9	310.4	6.80	6.70	10.3	8.94
135/151/154	360.9	310.4	6.77	6.67	10.3	8.99
153/168	360.9	310.4	7.05	6.95	10.6	9.20
156/157	360.9	310.4	7.31	7.21	11.1	9.74
170	395.3	331.4	7.40	7.30	11.3	9.89
177	395.3	331.4	7.21	7.11	11.1	9.73
180/193	395.3	331.4	7.49	7.39	11.6	10.14
183/185	395.3	331.4	7.33	7.23	11.3	9.88
187	395.3	331.4	7.30	7.20	11.1	9.71
194	429.8	352.4	7.92	7.83	11.9	10.45
198/199	429.8	352.4	7.32	7.23	11.7	10.22
203	429.8	352.4	7.77	7.68	11.9	10.43
206	464.2	373.4	8.20	8.11	12.5	10.98
209	498.7	394.4	8.27	8.20	13.8	12.16

References:
-$\log K_{ow}$ values derived from Li *et al.*, 2003
-$\log K_{oa}$ values derived from Chen *et al.*, 2003

Table 14: Environmental parameter definitions, values, and references

Model parameter	Name	Units	Mean	SD	Source
Concentration of particulate organic carbon in water	*Xpoc*	kg/L	5.66E-07	4.7143E-08	Estimated from Johannessen et al. 2003
Concentration of dissolved organic carbon in water	*Xdoc*	kg/L	1.32E-06	0.00000011	Johannessen et al. 2003
Concentration of suspended solids	Vss	kg/L	1.55E-05	1.5723E-06	Based on Arnot SFB data (same ratio of DOC:Vss as for SFB)
Mean annual water temperature	Tw	^oC	9.5	1.5	Estimated from: Davenne and Masson 2001
Mean annual air temperature	Ta	^oC	10.5	5.7	Estimated from monthly air temp for Port Moody Glenayre (1997): http://www.climate.weatheroffice.ec.gc.ca/climateData/monthlydata_e.html
Salinity	PSU	g/kg	30	2	Estimated from: http://www-sci.pac.dfo-mpo.gc.ca/osap/data/SearchTools/Searchlighthouse_e.htm
Density of organic carbon in sediment	δ_{OCS}	kg/L	0.9	0	Gobas and Arnot, 2005
Organic carbon content of sediment	OCS	unitless	0.011	0.007	From data
Dissolved oxygen concentration @ 90% saturation	Cox	mg O_2/L	7.5	1	Estimated from: Pawlowicz et al., 2003
Setschenow proportionality constant	S_PC	L/cm^3	0.0018	0	Xie, WH, Shiu, WY, Mackay, D. 1997.
Ideal gas law constant (Rgaslaw)	RGL	Pa.m^3/mol.K	8.314	0	known constant
Absolute temperature	Tabs	K	273.16	0	known constant
Molar concentration of seawater @ 35 ppt	MCS	mol/L	0.5	0	Xie, WH, Shiu, WY, Mackay, D. 1997.
Organic carbon burial rate	OCBR	gC/cm^2/yr	0.011	0.009	Johannessen et al. 2003
Primary production rate of organic carbon	PPR	gC/cm^2/yr	0.552	0	Pauly et al., 1996
Disequilibrium factor for POC partitioning in water column	Dpoc	unitless	1		Arnot and Gobas 2004 (eqn 4)
Disequilibrium factor for DOC partitioning in water column	Ddoc	unitless	1		Arnot and Gobas 2004 (eqn 4)
Proportionality constant for phase partitioning of POC	alphaPOC	unitless	0.35		Arnot and Gobas 2004 (eqn 4)
Proportionality constant for phase partitioning of POC	alphaDOC	unitless	0.08		Arnot and Gobas 2004 (eqn 4)

Table 15: General biological parameter definitions, values, and references

Group of organisms	Model parameter	Name	Units	Mean	SD	Source
All	Non-lipid organic matter – octanol proportionality constant	β	Unitless	0.035	0.0035	Gobas et al., 1999 (SD estimated)
Fish	Growth rate factor	GRF_F	Unitless	0.0007	0.00007	Thomann et al., 1992
Invertebrates	Growth rate factor	GRF_I	Unitless	0.00035	0.000035	Thomann et al., 1992
Scavengers	Particle scavenging efficiency	σ	Unitless	1	0	Default value
Poikilotherms/Homeotherms	Metabolic transformation rate	k_{Mp}	d^{-1}	0	0	Arnot and Gobas, 2004
Homeotherms	Mean homeothermic biota temperature	T_B	°C	37.5	1	Gobas and Arnot, 2005
Homeotherms	Density of lipids	δ_L	kg/L	0.9	0	Gobas and Arnot, 2005
Poikilotherms	Ew constant A	E_WA	Unitless	1.85	0.13	Arnot and Gobas, 2004
Zooplankton	Dietary absorption efficiency of lipid	ε_L	Unitless	0.72	0.02	Arnot and Gobas, 2004
Zooplankton	Dietary absorption efficiency of non-lipid organic matter	ε_N	Unitless	0.72	0.02	Arnot and Gobas, 2004
Zooplankton	Dietary absorption efficiency of water	ε_W	Unitless	0.55	0	Arnot and Gobas, 2004
Crabs	Dietary absorption efficiency of lipid	ε_L	Unitless	0.75	0.02	Arnot and Gobas, 2004
Crabs	Dietary absorption efficiency of non-lipid organic matter	ε_N	Unitless	0.75	0.02	Arnot and Gobas, 2004
Crabs	Dietary absorption efficiency of water	ε_W	Unitless	0.55	0	Arnot and Gobas, 2004
Invertebrates (except crabs and zooplankton)	Dietary absorption efficiency of lipid	ε_L	Unitless	0.75	0.02	Arnot and Gobas, 2004
Invertebrates (except crabs and zooplankton)	Dietary absorption efficiency of non-lipid organic matter	ε_N	Unitless	0.75	0.02	Arnot and Gobas, 2004
Invertebrates (except crabs and zooplankton)	Dietary absorption efficiency of water	ε_W	Unitless	0.55	0	Arnot and Gobas, 2004
Fish	Dietary absorption efficiency of lipid	ε_L	Unitless	0.9	0.02	Arnot and Gobas, 2004; Kelly et al., 2004
Fish	Dietary absorption efficiency of non-lipid organic matter	εN	Unitless	0.5	0.02	Arnot and Gobas, 2004
Fish	Dietary absorption efficiency of water	$\varepsilon\Omega$	Unitless	0.55	0	Arnot and Gobas, 2004

Group of organisms	Model parameter	Name	Units	Mean	SD	Source
Birds	Dietary absorption efficiency of lipid	$\varepsilon\Lambda$	Unitless	0.95	0.02	Derived from Drouillard and Norstrom, 2000
Birds	Dietary absorption efficiency of non-lipid organic matter	εN	Unitless	0.75	0.02	Gobas and Arnot, 2005
Birds	Dietary absorption efficiency of water	$\varepsilon\Omega$	Unitless	0.85	0	Gobas and Arnot, 2005
Seals	Dietary absorption efficiency of lipid	$\varepsilon\Lambda$	Unitless	0.97	0.02	Derived from Kelly et al, 2004; Trumble et al., 2003; Muelbert et al., 2003
Seals	Dietary absorption efficiency of non-lipid organic matter	εN	Unitless	0.75	0.02	Gobas and Arnot, 2005
Seals	Dietary absorption efficiency of water	$\varepsilon\Omega$	Unitless	0.85	0	Gobas and Arnot, 2005
Fish, birds, adult seals	Non-lipid organic matter fraction in biota	ϖNB	Unitless	0.2	0.01	Gobas and Arnot, 2005
All feeding Poikilotherms	ED constant A	$E\Delta A$	Unitless	8.5E-08	1.40E-08	Gobas and Arnot, 2005
All feeding Poikilotherms	ED constant B	$E\Delta B$	Unitless	2	0.6	Gobas and Arnot, 2005
Birds	ED constant A	$E\Delta A$	Unitless	3E-09	4.90E-10	Gobas and Arnot, 2005
Birds	ED constant B	$E\Delta B$	Unitless	1.04	2.00E-03	Gobas and Arnot, 2005
Seals	ED constant A	$E\Delta A$	Unitless	1E-09	1.70E-10	Gobas and Arnot, 2005
Seals	ED constant B	$E\Delta B$	Unitless	1.025	1.25E-03	Gobas and Arnot, 2005
Homeotherms	Lung uptake efficiency	$E\alpha$	Unitless	0.7	0	Gobas and Arnot, 2005

Table 16: Plant parameter definitions, values, and references

PHYTOPLANKTON

Parameter	Symbol	Units	Mean	SD	n	Reference
Lipid fraction in organism	v_{LB}	-	9.00E-04	2.00E-04	9	Mackintosh *et al.* , 2004
Non-lipid OC fraction in organism	v_{NB}	-	6.00E-04	2.00E-04	9	Mackintosh *et al.* , 2004
Water fraction in organism	v_{WB}	-	9.99E-01	-	-	Deduced
Growth rate constant	k_G	d^{-1}	1.25E-01	4.50E-02	-	Gobas & Arnot, 2005
Aqueous phase resistance constant	A_P	d^{-1}	6.00E-05	2.00E-05	-	Arnot & Gobas, 2004
Organic phase resistance constant	B_P	d^{-1}	5.50E+00	3.70E+00	-	Arnot & Gobas, 2004

KELP / SEAGRASS

Parameter	Symbol	Units	Mean	SD	n	Reference
Lipid fraction in organism	v_{LB}	-	8.00E-04	2.00E-04	9	Mackintosh *et al.* , 2004
Non-lipid OC fraction in organism	v_{NB}	-	6.20E-02	5.30E-02	9	Mackintosh *et al.* , 2004
Water fraction in organism	v_{WB}	-	9.37E-01	-	-	Deduced
Growth rate constant	k_G	d^{-1}	1.25E-01	4.50E-02	-	Gobas & Arnot, 2005
Aqueous phase resistance constant	A_P	d^{-1}	6.00E-05	2.00E-05	-	Arnot & Gobas, 2004
Organic phase resistance constant	B_P	d^{-1}	5.50E+00	3.70E+00	-	Arnot & Gobas, 2004

"-" = not available

Table 17: Invertebrate parameter definitions, values, and references

HERBIVOROUS ZOOPLANKTON

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	7.10E-08	-	-	Gobas and Arnot, 2005
Lipid fraction in biota	v_{LB}	-	3.96E-02	2.23E-01 *	12	Derived from Lee, 1974
NLOM fraction in biota	v_{NB}	-	1.46E-01	-	-	Deduced
Water fraction in biota	v_{WB}	-	8.14E-01	9.00E-03	-	Mauchline, 1998
Fraction of respiration involving pore water	m_P	-	0.00E+00	-	-	Estimated

* Geometric mean and SD

NEOCALANUS PLUMCHRUS

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	4.54E-06	7.55E-02 *	10	Derived from Evanson *et al.* , 2000
Lipid fraction in biota	v_{LB}	-	1.22E-01	2.33E-02 *	10	Derived from Evanson *et al.* , 2000
NLOM fraction in biota	v_{NB}	-	6.36E-02	-	-	Deduced
Water fraction in biota	v_{WB}	-	8.14E-01	9.00E-03	-	Mauchline, 1998
Fraction of respiration involving pore water	m_P	-	0.00E+00	-	-	Estimated

* Geometric mean and SD

PSEUDOCALANUS MINUTUS

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	8.84E-08	2.13E-01 *	2	Derived from Huntley, 2004
Lipid fraction in biota	v_{LB}	-	3.96E-02	2.23E-01 *	12	Derived from Lee, 1974
NLOM fraction in biota	v_{NB}	-	1.46E-01	-	-	Deduced
Water fraction in biota	v_{WB}	-	8.14E-01	9.00E-03	-	Mauchline, 1998
Fraction of respiration involving pore water	m_P	-	0.00E+00	-	-	Estimated

* Geometric mean and SD

SHELLFISH

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	8.06E-03	3.31E-01 *	10	Derived from Stout and Beezhold, 1981
Lipid fraction in biota	v_{LB}	-	1.20E-02	1.53E-01 *	11	Derived from Stout and Beezhold, 1981 and Mackintosh et al. 2004
NLOM fraction in biota	v_{NB}	-	1.88E-01	-	-	Deduced
Water fraction in biota	v_{WB}	-	8.00E-01	-	-	Estimated
Fraction of respiration involving pore water	m_P	-	2.00E-01	-	-	Estimated

* Geometric mean and SD

CRABS

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	5.37E-01	1.20E-01 *	7	Derived from Ikonomou *et al* ., 2004; Swain & Walton, 1994
Lipid fraction in biota	v_{LB}	-	3.00E-02	-	-	Stevenson, 2003
NLOM fraction in biota	v_{NB}	-	1.70E-01	-	-	Deduced
Water fraction in biota	v_{WB}	-	8.00E-01	-	-	Estimated
Fraction of respiration involving pore water	m_P	-	2.00E-01	-	-	Estimated

* Geometric mean and SD

GRAZING INVERTEBRATES

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	5.00E-02	- *	-	Estimated
Lipid fraction in biota	v_{LB}	-	1.50E-02	- *	-	Estimated
NLOM fraction in biota	v_{NB}	-	1.85E-01	-	-	Deduced
Water fraction in biota	v_{WB}	-	8.00E-01	-	-	Estimated
Fraction of respiration involving pore water	m_P	-	2.00E-01	-	-	Estimated

* Geometric mean and SD

CARNIVOROUS ZOOPLANKTON (AMPHIPODS)

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	3.23E-07	6.50E-02 *	8	Derived from EPA, 1999
Lipid fraction in biota	v_{LB}	-	3.68E-02	1.92E-01 *	7	Derived from Lee, 1974; Sargent & Lee, 1975
NLOM fraction in biota	v_{NB}	-	1.33E-01	-	-	Deduced
Water fraction in biota	v_{WB}	-	8.30E-01	1.41E-02	2	Derived from Lee, 1974; Sargent & Lee, 1975
Fraction of respiration involving pore water	m_P	-	5.00E-02	-	-	Estimated

* Geometric mean and SD

EUPHAUSIA PACIFICA (KRILL)

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	4.03E-05	1.83E-01 *	3	Derived from Huntley & Zhou, 2004; NMFS, 2005
Lipid fraction in biota	v_{LB}	-	1.59E-02	1.00E-02 *	3	Derived from Mauchline & Fischer, 1969
NLOM fraction in biota	v_{NB}	-	1.56E-01	-	-	Deduced
Water fraction in biota	v_{WB}	-	8.28E-01	3.80E-02	3	Derived from Mauchline & Fischer, 1969
Fraction of respiration involving pore water	m_P	-	5.00E-02	-	-	Estimated

* Geometric mean and SD

PREDATORY INVERTEBRATES

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	1.00E+00	- *	-	Estimated
Lipid fraction in biota	v_{LB}	-	2.00E-02	- *	-	Estimated
NLOM fraction in biota	v_{NB}	-	1.80E-01	-	-	Deduced
Water fraction in biota	v_{WB}	-	8.00E-01	-	-	Estimated
Fraction of respiration involving pore water	m_P	-	2.00E-01	-	-	Estimated

* Geometric mean and SD

Table 18: Fish parameter definitions, values, and references

HERRING (IMMMIGRANT)

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	Kg	5.95E-02	5.15E-02	-	Derived from data provided by Jim West
Lipid fraction in biota	v_{LB}	-	4.99E-02	1.1E-01	-	Derived from data provided by Jim West
NLOM fraction in biota	v_{NB}	-	2.00E-01	-	-	Estimated
Water fraction in biota	v_{WB}	-	7.5E-01	-	-	Deduced
Fraction of respiration involving pore water	m_P	-	0.00E+00	-	-	Estimated using fishbase.org

SMALL PELAGIC FISH (SEAL PREY)

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	4.49E-02	6.42E-01 *	4	Derived from Iverson *et al* ., 2002
Lipid fraction in biota	v_{LB}	-	3.86E-02	5.44E-01 *	4	Derived from Iverson *et al* ., 2002
NLOM fraction in biota	v_{NB}	-	2.00E-01	-	-	Estimated
Water fraction in biota	v_{WB}	-	7.61E-01	-	-	Deduced
Fraction of respiration involving pore water	m_P	-	0.00E+00	-	-	Estimated from Froese & Pauly, 2002

* Geometric mean and SD

SMALL PELAGIC FISH (BIRD PREY)

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	4.92E-03	3.63E-01 *	2	Derived from Butler, 1995; Iverson *et al* ., 2002
Lipid fraction in biota	v_{LB}	-	1.53E-02	4.85E-01 *	4	Derived from Harfenist *et al* ., 1995; Iverson *et al* ., 2002
NLOM fraction in biota	v_{NB}	-	2.00E-01	-	-	Estimated
Water fraction in biota	v_{WB}	-	7.85E-01	-	-	Deduced
Fraction of respiration involving pore water	m_P	-	0.00E+00	-	-	Estimated from Froese & Pauly, 2002

* Geometric mean and SD

RIVER LAMPREY

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	1.43E-02	5.00E-03	-	Derived from Beamish, 1980
Lipid fraction in biota	v_{LB}	-	1.25E-01	3.00E-02	-	Derived from Larsen, 1980
NLOM fraction in biota	v_{NB}	-	2.00E-01	-	-	Estimated
Water fraction in biota	v_{WB}	-	6.75E-01	-	-	Deduced
Fraction of respiration involving pore water	m_P	-	0.00E+00	-	-	Estimated from Froese & Pauly, 2002

MISCELLANEOUS DEMERSAL FISH (SEAL PREY)

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	1.81E-01	4.13E-01 *	5	Estimated from Iverson *et al* ., 2002; Gobas & Arnot, 2005
Lipid fraction in biota	v_{LB}	-	2.51E-02	1.28E-01 *	5	Estimated from Iverson *et al* ., 2002; Gobas & Arnot, 2005
NLOM fraction in biota	v_{NB}	-	2.00E-01	-	-	Estimated
Water fraction in biota	v_{WB}	-	7.75E-01	-	-	Deduced
Fraction of respiration involving pore water	m_P	-	5.00E-02	-	-	Estimated from Froese & Pauly, 2002

* Geometric mean and SD

MISCELLANEOUS DEMERSAL FISH (BIRD PREY)

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	4.72E-03	5.34E-01 *	3	Derived from Butler, 1995
Lipid fraction in biota	v_{LB}	-	1.63E-02	1.40E-01 *	2	Derived from Harfenist *et al* ., 1995
NLOM fraction in biota	v_{NB}	-	2.00E-01	-	-	Estimated
Water fraction in biota	v_{WB}	-	7.84E-01	-	-	Deduced
Fraction of respiration involving pore water	m_P	-	5.00E-02	-	-	Estimated from Froese & Pauly, 2002

* Geometric mean and SD

PACIFIC HAKE

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	3.74E-01	-	-	Derived from Saunders & McFarlane, 1999
Lipid fraction in biota	v_{LB}	-	5.20E-02	-	-	Stout and Beezhold, 1981
NLOM fraction in biota	v_{NB}	-	2.00E-01	-	-	Estimated
Water fraction in biota	v_{WB}	-	7.48E-01	-	-	Deduced
Fraction of respiration involving pore water	m_P	-	0.00E+00	-	-	Estimated from Froese & Pauly, 2002

CHUM (IMMMIGRANT)

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	Kg	3.96E-00	-	-	Derived from Hamilton et al., 2005 supporting information (data supplied by DO Carpenter)
Lipid fraction in biota	v_{LB}	-	4.83E-02	-	-	Derived from Hamilton et al., 2005 supporting information (data supplied by DO Carpenter)
NLOM fraction in biota	v_{NB}	-	2.00E-01	-	-	Estimated
Water fraction in biota	v_{WB}	-	7.5E-01	-	-	Deduced
Fraction of respiration involving pore water	m_P	-	0.00E+00	-	-	Estimated using fishbase.org

COHO (IMMMIGRANT)

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	Kg	3.50E-00	-	-	Derived from Hamilton et al., 2005 supporting information (data supplied by DO Carpenter)
Lipid fraction in biota	v_{LB}	-	6.39E-02	-	-	Derived from Hamilton et al., 2005 supporting information (data supplied by DO Carpenter)
NLOM fraction in biota	v_{NB}	-	2.00E-01	-	-	Estimated
Water fraction in biota	v_{WB}	-	7.4E-01	-	-	Deduced
Fraction of respiration involving pore water	m_P	-	0.00E+00	-	-	Estimated using fishbase.org

CHINOOK (IMMMIGRANT)

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	Kg	3.63E-00	-	-	Derived from Hamilton et al., 2005 supporting information (data supplied by DO Carpenter)
Lipid fraction in biota	v_{LB}	-	5.43E-02	-	-	Derived from Hamilton et al., 2005 supporting information (data supplied by DO Carpenter)
NLOM fraction in biota	v_{NB}	-	2.00E-01	-	-	Estimated
Water fraction in biota	v_{WB}	-	7.5E-01	-	-	Deduced
Fraction of respiration involving pore water	m_P	-	0.00E+00	-	-	Estimated using fishbase.org

SPINY DOGFISH

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	2.00E+00	2.00E-01	9	Mackintosh *et al.* , 2004
Lipid fraction in biota	v_{LB}	-	1.00E-01	5.00E-02	9	Mackintosh *et al.* , 2004
NLOM fraction in biota	v_{NB}	-	2.00E-01	-	-	Estimated
Water fraction in biota	v_{WB}	-	7.00E-01	-	-	Deduced
Fraction of respiration involving pore water	m_P	-	0.00E+00	-	-	Estimated from Froese & Pauly, 2002

POLLOCK

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	7.97E-02	1.19E-02	7	Derived from Iverson et al., 2002
Lipid fraction in biota	v_{LB}	-	2.16E-02	1.75E-01 *	36	Derived from Iverson et al., 2002
NLOM fraction in biota	v_{NB}	-	2.00E-01	-	-	Estimated
Water fraction in biota	v_{WB}	-	7.78E-01	-	-	Deduced
Fraction of respiration involving pore water	m_P	-	0.00E+00	-	-	Estimated from Froese & Pauly, 2002

* Geometric mean and SD

NORTHERN SMOOTH-TONGUE

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	7.50E-04	-	-	Estimated from Froese & Pauly, 2002
Lipid fraction in biota	v_{LB}	-	4.99E-02	-	-	Estimated
NLOM fraction in biota	v_{NB}	-	2.00E-01	-	-	Estimated
Water fraction in biota	v_{WB}	-	7.50E-01	-	-	Deduced
Fraction of respiration involving pore water	m_P	-	0.00E+00	-	-	Estimated from Froese & Pauly, 2002

ENGLISH SOLE

Parameter	Symbol	Units	Mean	SD	n	Reference
Wet weight of organism	W_B	kg	7.40E-02	-	-	Mackintosh et al., 2004
Lipid fraction in biota	v_{LB}	-	4.00E-02	-	-	Stout and Beezhold, 1981
NLOM fraction in biota	v_{NB}	-	2.00E-01	-	-	Estimated
Water fraction in biota	v_{WB}	-	7.60E-01	-	-	Deduced
Fraction of respiration involving pore water	m_P	-	5.00E-02	-	-	Estimated from Froese & Pauly, 2002

Table 19: Harbour seal parameter definitions, values, and references

			Harbour Seal (adult male)		
Model parameter	**Name**	**Units**	**Mean**	**SD**	**Reference**
Wet weight of the organism	W_B	kg	8.70E+01	6.60E+00	Bigg, 1969
Lipid fraction in biota	v_{LB}	Unitless	0.43	0.07	Arnot and Gobas, 2004
Non-lipid organic matter fraction in biota	v_{NB}	Unitless	0.20	0.01	
Water fraction in biota	v_{WB}	Unitless	0.37	-	
Dietary absorption efficiency of lipid	ε_L	Unitless	0.97	0.02	
Dietary absorption efficiency of non-lipid organic matter	ε_N	Unitless	0.75	0.02	
Dietary absorption efficiency of water	ε_W	Unitless	0.85	-	
Lung uptake efficiency	Ea	Unitless	0.7	-	
Growth rate constant	k_G	d^{-1}	7.50E-05	7.50E-06	Gobas and Arnot, 2005
Activity Factor	AF	Unitless	2.5	-	Gobas and Arnot, 2005
E_D constant A	E_DA	Unitless	1E-09	1.7E-10	
E_D constant B	E_DB	Unitless	1.03E+00	0.00125	

			Harbour Seal (adult female)		
Model parameter	**Name**	**Units**	**Mean**	**SD**	**Reference**
Wet weight of the organism	W_B	kg	6.48E+01	4.40E+00	Bigg, 1969
Lipid fraction in biota	v_{LB}	Unitless	0.15	0.07	P. Ross, pers. Comm.
Non-lipid organic matter fraction in biota	v_{NB}	Unitless	0.20	0.01	
Water fraction in biota	v_{WB}	Unitless	0.65	-	
Dietary absorption efficiency of lipid	ε_L	Unitless	0.97	0.02	
Dietary absorption efficiency of non-lipid organic matter	ε_N	Unitless	0.75	0.02	
Dietary absorption efficiency of water	ε_W	Unitless	0.85	-	
Lung uptake efficiency	Ea	Unitless	0.7	-	
Growth rate constant	k_G	d^{-1}	1.00E-05	1.00E-05	Gobas and Arnot, 2005
Activity Factor	AF	Unitless	2.50E+00	-	Gobas and Arnot, 2005
E_D constant A	E_DA	Unitless	1E-09	1.7E-10	
E_D constant B	E_DB	Unitless	1.03E+00	0.00125	

			Harbour Seal (1 yr old)		
Model parameter	**Name**	**Units**	**Mean**	**SD**	**Reference**
Wet weight of the organism	W_B	kg	3.33E+01	2.86E+00	Muelbert et al., 2003
Lipid fraction in biota	v_{LB}	Unitless	0.12	0.08	Muelbert et al., 2003
Non-lipid organic matter fraction in biota	v_{NB}	Unitless	0.25	-	
Water fraction in biota	v_{WB}	Unitless	0.64	0.05	Muelbert et al., 2003
Dietary absorption efficiency of lipid	ε_L	Unitless	0.97	0.02	
Dietary absorption efficiency of non-lipid organic matter	ε_N	Unitless	0.75	0.02	
Dietary absorption efficiency of water	ε_W	Unitless	0.85	-	
Lung uptake efficiency	Ea	Unitless	0.7	-	
Growth rate constant	k_G	d^{-1}	1.00E-03	1.00E-03	Gobas and Arnot, 2005
Activity Factor	AF	Unitless	2.5	-	Gobas and Arnot, 2005
E_D constant A	E_DA	Unitless	1E-09	1.7E-10	
E_D constant B	E_DB	Unitless	1.03E+00	0.00125	

			Harbour Seal (pup)		
Model parameter	**Name**	**Units**	**Mean**	**SD**	**Reference**
Wet weight of the organism	W_B	kg	2.39E+01	5.66E+00	Derived from data provided by PS Ross.
Lipid fraction in biota	v_{LB}	Unitless	0.63	0.14	Derived from data provided by PS Ross.
Non-lipid organic matter fraction in biota	v_{NB}	Unitless	0.00	0.00	
Water fraction in biota	v_{WB}	Unitless	0.37	0.05	Derived from Muelbert et al., 2003
Dietary absorption efficiency of lipid	ε_L	Unitless	0.97	0.02	
Dietary absorption efficiency of non-lipid organic matter	ε_N	Unitless	0.75	0.02	
Dietary absorption efficiency of water	ε_W	Unitless	0.85	-	
Lung uptake efficiency	Ea	Unitless	0.7	-	
Growth rate constant	k_G	d^{-1}	2.50E-02	-	Gobas and Arnot, 2005
Activity Factor	AF	Unitless	1.5	-	Gobas and Arnot, 2005
E_D constant A	E_DA	Unitless	1E-09	1.7E-10	
E_D constant B	E_DB	Unitless	1.03E+00	0.00125	