Lower Duwamish Waterway Group

Port of Seattle / City of Seattle / King County / The Boeing Company

Lower Duwamish Waterway Remedial Investigation

REMEDIAL INVESTIGATION REPORT

APPENDIX D. FOOD WEB MODEL

DRAFT

For submittal to:

The US Environmental Protection Agency
Region 10
Seattle, WA

The Washington State Department of Ecology
Northwest Regional Office
Bellevue, WA

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Acronyms

CSO	combined sewer overflow
CT	central tendency
DO	dissolved oxygen
DOC	dissolved organic carbon
dw	dry weight
Ecology	Washington State Department of Ecology
EFDC	Environmental Fluid Dynamics [Computer] Code
EPA	US Environmental Protection Agency
EPC	exposure point concentration
ERA	ecological risk assessment
FS	feasibility study
FWM	food web model
GIS	geographic information system
HHRA	human health risk assessment
IDW	inverse distance weighting
LDW	Lower Duwamish Waterway
NLOC	non-lipid organic carbon
NLOM	non-lipid organic matter
NOAA	National Oceanic and Atmospheric Administration
NRS	nominal range sensitivity
OC	organic carbon
PCB	polychlorinated biphenyl
POC	particulate organic carbon
RBTC	risk-based threshold concentration
RI	remedial investigation
RM	river mile
RME	reasonable maximum exposure
ROC	receptor of concern
SD	standard deviation
SE	standard error
SPAF	species predictive accuracy factor

SWAC	spatially weighted average concentration
TOC	total organic carbon
TSS	total suspended solids
ww	wet weight
Windward	Windward Environmental LLC

D.1 Introduction

This appendix describes the food web model (FWM) developed for the Lower Duwamish Waterway (LDW). A comprehensive dataset of chemical concentrations in sediment and tissue collected in the LDW has been compiled for the remedial investigation (RI) and to support the baseline risk assessments. These data were also used to support a FWM for total polychlorinated biphenyls (PCBs) in the LDW. Three draft memoranda describing the FWM have been submitted to the US Environmental Protection Agency (EPA) and the Washington State Department of Ecology (Ecology); these memoranda present the rationale for the specific model selected (Windward 2005f), describe the modeling approach (Windward 2005g), and present the results of preliminary modeling runs (Windward 2005h). The selection of initial parameter values and optimal methods for applying the FWM in the LDW were discussed in a series of meetings with EPA and the National Oceanic and Atmospheric Administration (NOAA). In addition, Jon Arnot, the co-author of the model (Arnot and Gobas 2004a), was consulted regarding technical details.

The FWM was developed to estimate the relationship between total PCB concentrations in tissue and sediment in order to estimate risk-based threshold concentrations (RBTCs) for total PCBs in sediment for the RI (see Section 8 of the RI and Section D.8). The FWM may also be used in the feasibility study (FS) to assess residual risks from PCBs that may remain following various sediment cleanup alternatives. Figure D.1-1 illustrates how the FWM will be used in the RI/FS process.

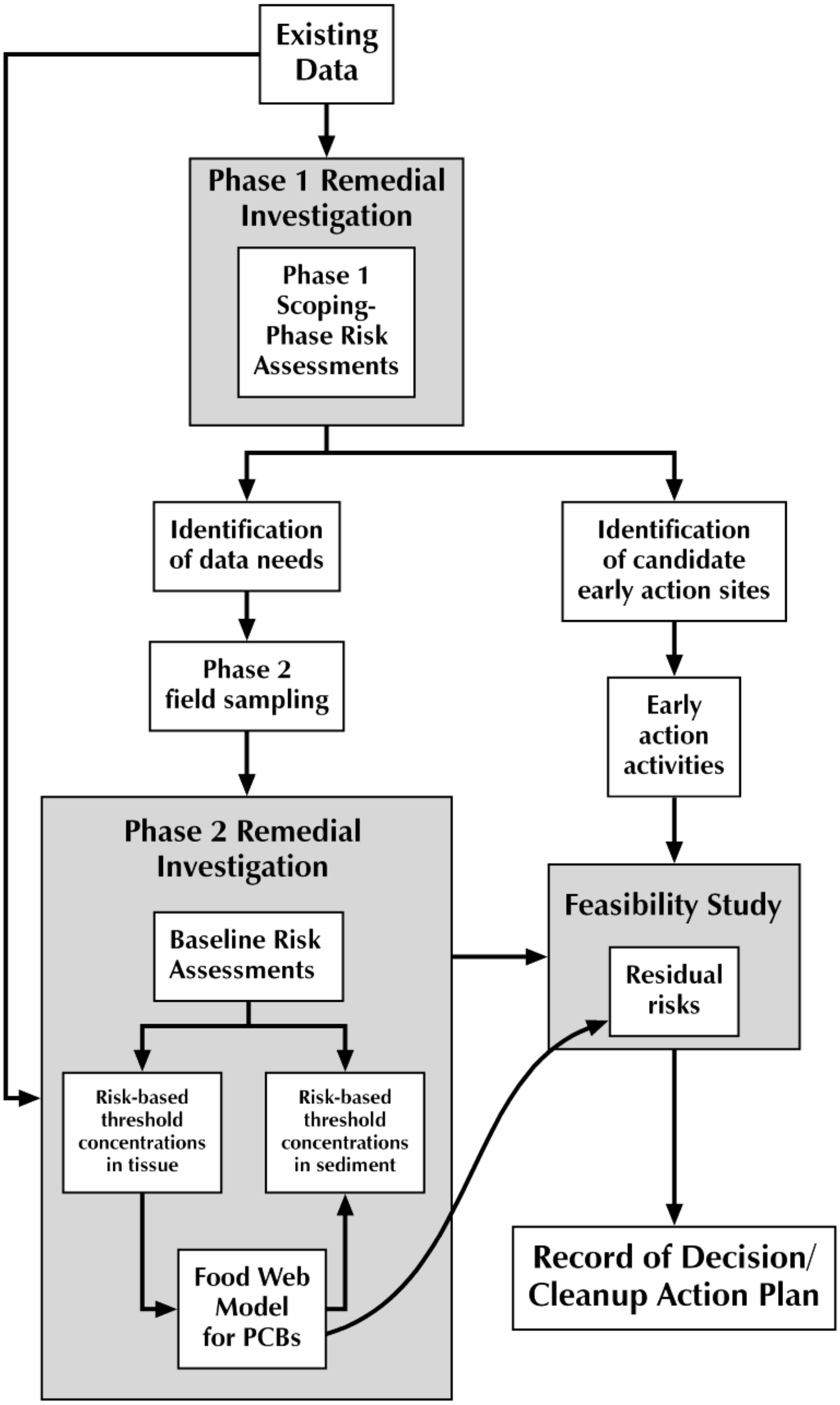

Figure D.1-1. Use of the FWM in the RI/FS process

The FWM was calibrated using literature-derived and site-specific environmental data. The purpose of the calibration process was to identify sets of parameter values that best estimated empirical data. The calibration process does not necessarily identify the "true" value for each FWM parameter, because numerous combinations of parameters can produce the same results, or offer mechanistic insights regarding the bioaccumulation of PCBs in the LDW food web. Nonetheless, the results of the calibrated FWM were used in the development of sediment RBTCs for PCBs, and may serve as a tool to support risk management decision making at the site.

The selected FWM and its application to the LDW are discussed in greater detail in the subsections that follow. Section D.2 describes the Arnot and Gobas FWM (Arnot and Gobas 2004a). Section D.3 describes the approach for applying the FWM to the LDW. Section D.4 presents the model input parameters and describes how values were selected. Section D.5 presents methods and results of the calibration process, and Section D.6 presents methods and results of sensitivity and uncertainty analyses. Tests of the model's performance at the modeling area scale and for clams at clam intertidal locations are presented in Section D.7. Use of the FWM in the calculation of sediment RBTCs is discussed in Section D.8. A summary is provided in Section D.9.

D.2 Description of the Arnot and Gobas Food Web Model

To estimate the relationship between total PCB concentrations in tissue and sediment in the LDW, an update of the original Gobas model (Arnot and Gobas 2004a) was applied to the LDW. The original Gobas model (Gobas 1993) is a steady-state,[1] mass-balance bioaccumulation model that was originally developed to describe the bioaccumulation of PCBs in the Great Lakes food web. The Gobas model was later refined (Arnot and Gobas 2004a) to reflect a clearer understanding of bioaccumulation processes based on subsequent field and laboratory studies (Arnot and Gobas 2004b; Gobas and MacLean 2003; Gobas et al. 1999; Nichols et al. 2001; Roditi and Fisher 1999). New elements added by Arnot and Gobas (2004a) to refine the model included:

- A new model for partitioning chemicals into organisms that separates the organism into three components: lipids, non-lipid organic matter or non-lipid organic carbon for phytoplankton, and water

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[1] A steady-state assumption means that concentrations of chemicals in tissues are assumed to not change over time or that concentrations of chemicals in tissues maintain a state of relative equilibrium even after undergoing fluctuations or transformations. The steady-state assumption is reasonable for applications to field situations in which organisms have been exposed to hydrophobic organic chemicals over a long period of time particularly at sites with contaminated sediment. Concentrations in tissues fluctuate slowly compared to exposures, so body burden – especially average body burden in a population of individuals – tends to reflect the average concentration to which the population is exposed over time.

- Kinetic models for predicting chemical concentrations in algae, phytoplankton, and zooplankton
- New allometric relationships for predicting gill ventilation rates in a wide range of aquatic species
- A mechanistic model for predicting changes in the concentration of organic chemicals in the gut contents of a range of species as it passes through the gastrointestinal tract

The Arnot and Gobas FWM (Arnot and Gobas 2004a) has five compartments: phytoplankton/algae, zooplankton, filter-feeding benthic invertebrates, scavenger/predator/detritivore benthic invertebrates, and fish. The FWM estimates concentrations of hydrophobic organic chemicals for each compartment using equations that represent the biological processes involved in the uptake and loss of hydrophobic organic chemicals (Figure D.2-1). Thus, each compartment (e.g., fish) has it own unique set of equations. The model has three physical media: sediment, water column water, and porewater.

The Arnot and Gobas model is based on several fundamental assumptions, including:

- Primary routes for the uptake of hydrophobic organic chemicals by zooplankton, benthic invertebrates, and fish are ventilation of porewater or water column water and ingestion of sediment or organisms.
- Primary routes for the loss of hydrophobic organic chemicals by zooplankton, benthic invertebrates, and fish are metabolism, growth dilution, ventilation of porewater or water column water, and fecal egestion.
- Chemicals are assumed to be homogeneously distributed within each tissue phase of the organism (i.e., lipids, water, and non-lipid organic matter [NLOM; e.g., proteins and carbohydrates] or non-lipid organic carbon [NLOC][2]).
- Organisms are assumed to be single compartments that exchange chemicals with their surrounding environments.
- Chemical losses via egg deposition or sperm ejection are assumed to be negligible.

Justification is provided for these assumptions in Arnot and Gobas (2004a).

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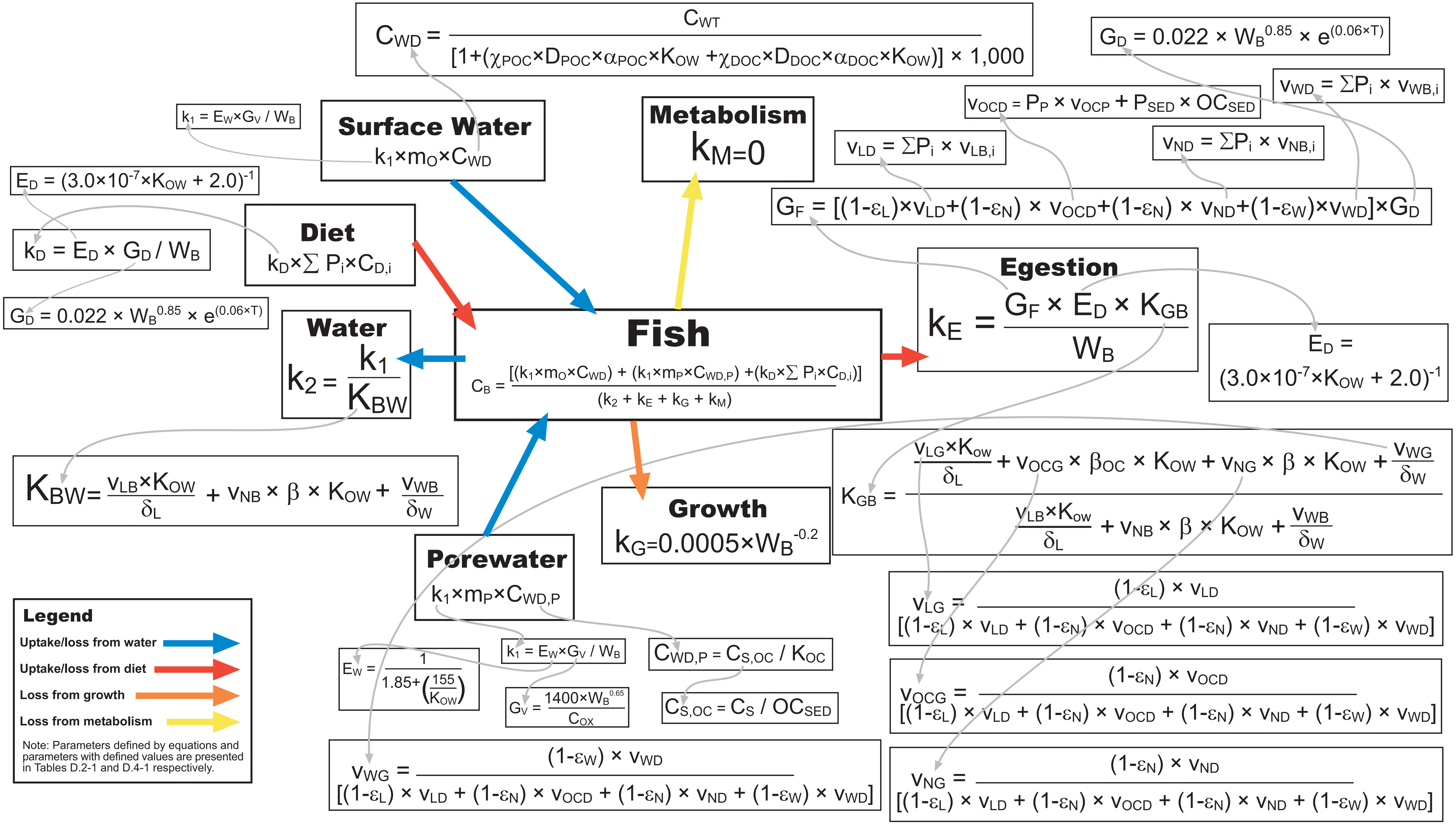
[2] NLOC was used as the third phase for chemical partitioning in phytoplankton instead of NLOM, as discussed in Section D.4.2.1. For sediment, PCBs were assumed to partition into organic carbon.



$$C_{WD} = \frac{C_{WT}}{[1+(\chi_{POC}\times D_{POC}\times \alpha_{POC}\times K_{OW} + \chi_{DOC}\times D_{DOC}\times \alpha_{DOC}\times K_{OW})]\times 1{,}000}$$

$$G_D = 0.022 \times W_B^{0.85} \times e^{(0.06\times T)}$$

$$v_{WD} = \Sigma P_i \times v_{WB,i}$$

$$v_{OCD} = P_P \times v_{OCP} + P_{SED} \times OC_{SED}$$

$$v_{LD} = \Sigma P_i \times v_{LB,i}$$

$$v_{ND} = \Sigma P_i \times v_{NB,i}$$

$$G_F = [(1-\varepsilon_L)\times v_{LD} + (1-\varepsilon_N)\times v_{OCD} + (1-\varepsilon_N)\times v_{ND} + (1-\varepsilon_W)\times v_{WD}]\times G_D$$

$$k_1 = E_W \times G_V / W_B$$

Surface Water

$$k_1 \times m_O \times C_{WD}$$

Metabolism

$$k_M = 0$$

$$E_D = (3.0\times 10^{-7}\times K_{OW} + 2.0)^{-1}$$

$$k_D = E_D \times G_D / W_B$$

Diet

$$k_D \times \sum P_i \times C_{D,i}$$

$$G_D = 0.022 \times W_B^{0.85} \times e^{(0.06\times T)}$$

Water

$$k_2 = \frac{k_1}{K_{BW}}$$

Fish

$$C_B = \frac{[(k_1\times m_O\times C_{WD}) + (k_1\times m_P\times C_{WD,P}) + (k_D\times \sum P_i\times C_{D,i})]}{(k_2 + k_E + k_G + k_M)}$$

Egestion

$$k_E = \frac{G_F \times E_D \times K_{GB}}{W_B}$$

$$E_D = (3.0\times 10^{-7}\times K_{OW} + 2.0)^{-1}$$

$$K_{BW} = \frac{v_{LB}\times K_{OW}}{\delta_L} + v_{NB}\times \beta \times K_{OW} + \frac{v_{WB}}{\delta_W}$$

$$K_{GB} = \frac{\frac{v_{LG}\times K_{ow}}{\delta_L} + v_{OCG}\times \beta_{OC}\times K_{OW} + v_{NG}\times \beta\times K_{OW} + \frac{v_{WG}}{\delta_W}}{\frac{v_{LB}\times K_{ow}}{\delta_L} + v_{NB}\times \beta\times K_{OW} + \frac{v_{WB}}{\delta_W}}$$

Growth

$$k_G = 0.0005 \times W_B^{-0.2}$$

Porewater

$$k_1 \times m_P \times C_{WD,P}$$

$$v_{LG} = \frac{(1-\varepsilon_L)\times v_{LD}}{[(1-\varepsilon_L)\times v_{LD} + (1-\varepsilon_N)\times v_{OCD} + (1-\varepsilon_N)\times v_{ND} + (1-\varepsilon_W)\times v_{WD}]}$$

$$E_W = \frac{1}{1.85 + \left(\frac{155}{K_{OW}}\right)}$$

$$k_1 = E_W \times G_V / W_B$$

$$C_{WD,P} = C_{S,OC} / K_{OC}$$

$$v_{OCG} = \frac{(1-\varepsilon_N)\times v_{OCD}}{[(1-\varepsilon_L)\times v_{LD} + (1-\varepsilon_N)\times v_{OCD} + (1-\varepsilon_N)\times v_{ND} + (1-\varepsilon_W)\times v_{WD}]}$$

$$G_V = \frac{1400\times W_B^{0.65}}{C_{OX}}$$

$$C_{S,OC} = C_S / OC_{SED}$$

$$v_{WG} = \frac{(1-\varepsilon_W)\times v_{WD}}{[(1-\varepsilon_L)\times v_{LD} + (1-\varepsilon_N)\times v_{OCD} + (1-\varepsilon_N)\times v_{ND} + (1-\varepsilon_W)\times v_{WD}]}$$

$$v_{NG} = \frac{(1-\varepsilon_N)\times v_{ND}}{[(1-\varepsilon_L)\times v_{LD} + (1-\varepsilon_N)\times v_{OCD} + (1-\varepsilon_N)\times v_{ND} + (1-\varepsilon_W)\times v_{WD}]}$$

Legend

Uptake/loss from water

Uptake/loss from diet

Loss from growth

Loss from metabolism

Note: Parameters defined by equations and parameters with defined values are presented in Tables D.2-1 and D.4-1 respectively.

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Figure D.2-1. Equations and parameters used to estimate total PCB concentrations for fish in the Arnot and Gobas model

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Model equations are separated into biological equations that simulate the biological processes leading to uptake and loss of chemicals by organisms (Figure D.2-1), environmental equations that simulate the partitioning of the chemical in the environment, and a single chemical equation that derives a log K_{OC} value from log K_{OW} (Table D.2-1). Each species in the model has a master equation that combines chemical uptake and loss for that species (C_B). The master equation has two potential chemical uptake mechanisms and four potential chemical loss mechanisms. Chemical concentrations in phytoplankton are calculated assuming aqueous uptake across the cell wall ($k_1 \times m_O \times C_{WD}$), loss across the cell wall (k_2), and loss via growth dilution (k_G). Chemical concentrations in zooplankton, invertebrates, and fish are calculated assuming uptake from water (i.e., water column water and porewater) via the respiratory surface ($k_1 \times (m_O \times C_{WD} + m_P \times C_{WD,P})$) and uptake from the diet ($k_D \times \sum P_i \times C_{D,i}$). Chemical loss mechanisms for zooplankton, invertebrates, and fish include metabolism (k_M), growth dilution (k_G), loss to water via the respiratory surface (k_2), and fecal egestion (k_E). Because the Arnot and Gobas model assumes steady state conditions, it does not recognize short-term changes in rates of uptake or loss from short-term changes in biological or environmental conditions. For each model run, one value was calculated for each uptake or loss mechanism.

Water column water, porewater, and sediment are the three environmental media included in the FWM. Total PCB concentrations in the water column are entered as whole water total PCB concentrations. The dissolved fraction is calculated in the model by estimating the relative partitioning of PCBs to particulate organic carbon (POC), dissolved organic carbon (DOC), and the freely dissolved phase (Table D.2-1). Total PCB concentrations in porewater are estimated assuming equilibrium partitioning with the sediment (Table D.2-1). The equilibrium partitioning equation does not account for partitioning to colloidal carbon within the sediment matrix. Total PCB concentrations in sediment are entered as total dry weight concentrations and converted to organic carbon (OC)-normalized concentrations for uptake and loss calculations. One sediment compartment represents both bottom sediments and suspended sediments. Exposure through direct sediment contact via the dermis or integument is not explicitly modeled in the FWM.

Exposure routes for chemicals in sediment include diffusion to porewater and the ingestion of sediment. The exposure route for chemicals in water column water and porewater is ventilation across the respiratory surface (e.g., gills) or cell wall.

Table D.2-1. Equations for the Arnot and Gobas Model

PARAMETER	SYMBOL	UNIT	EQUATION	NOTES	SOURCE
Biological					
Chemical concentration in the modeled species	C_B	µg/kg ww	$C_B = \{k_1 \times (m_O \times C_{WD} + m_P \times C_{WD,P}) + k_D \times \sum P_i \times C_{D,i}\}/(k_2 + k_E + k_G + k_M)$		Arnot and Gobas (2004a)
Chemical concentration in prey item i	$C_{D,i}$	µg/kg ww	$C_{D,I} = C_B$ or $C_{D,I} = C_S$ (depending on diet)	Concentration of prey items are represented by the equation for chemical concentration in the modeled species (C_B) for any organisms consumed or by the input value for concentration of total PCBs in sediment C_S for sediment consumed	Arnot and Gobas (2004a)
Fraction of water column water ventilated	m_O	fraction	$m_O = 1 - m_p$	fraction of total water ventilated from water column water (water not directly in association with the sediment)	Arnot and Gobas (2004a)
Rate constant for aqueous uptake by fish, invertebrates, and zooplankton	k_1	L/kg·day	$k_1 = E_W \times G_V/W_B$	chemical uptake via the respiratory area (e.g., gills or other respiratory surface)	Gobas (1993); Gobas and MacKay (1987) as cited in Arnot and Gobas (2004a)
Rate constant for aqueous uptake by phytoplankton /algae	k_1	L/kg·day	$k_1 = (A + (B/K_{OW}))^{-1}$	chemical uptake across the cell wall	Arnot and Gobas (2004a)
Rate constant for chemical elimination via the respiratory area	k_2	day^{-1}	$k_2 = k_1/K_{BW}$	chemical loss via the respiratory surface (e.g., gills or cell wall)	Gobas (1993) as cited in Arnot and Gobas (2004a)
Rate constant for chemical uptake via the diet	k_D	kg food/kg organism·day	$k_D = E_D \times G_D/W_B$	For phytoplankton/algae, k_D is zero.	Gobas (1993) as cited in Arnot and Gobas (2004a)
Rate constant for chemical elimination via excretion into egested feces	k_E	day^{-1}	$k_E = G_F \times E_D \times K_{GB}/W_B$	For phytoplankton/algae, k_E is zero.	Gobas et al. (1993) as cited in Arnot and Gobas (2004a)
Rate constant for growth of aquatic organisms	k_G	day^{-1}	$k_G = 0.000502 \times W_B^{-0.2}$	This regression relationship was established at temperatures around 10°C. (Mean water column temperatures in the LDW were 11°C.)	Thomann et al. (1992) as cited in Arnot and Gobas (2004a)
Dietary chemical transfer efficiency	E_D	%	$E_D = (3.0 \times 10^{-7} \times K_{OW} + 2.0)^{-1}$		Arnot and Gobas (2004a)
Respiratory surface chemical uptake efficiency	E_W	%	$E_W = (1.85 + (155/K_{OW}))^{-1}$		Gobas (1988) as cited in Arnot and Gobas (2004a)

PARAMETER	SYMBOL	UNIT	EQUATION	NOTES	SOURCE
Feeding rate – filter feeders	G_D	kg/d	$G_D = G_V \times Css \times \sigma$		Morrison et al. (1996) as cited in Arnot and Gobas (2004a)
Feeding rate – other species	G_D	kg/d	$G_D = 0.022 \times W_B^{0.85} \times e^{(0.06 \times T)}$	based on studies of feeding rates in cold-water fish (being used for zooplankton and aquatic invertebrate species as well).	Weiniger (1978) as cited in Arnot and Gobas (2004a)
Fecal egestion rate	G_F	kg/d	$G_F = \{(1 - \varepsilon_L) \times v_{LD}) + (1 - \varepsilon_N) \times v_{OCD} + (1 - \varepsilon_N) \times v_{ND} + (1 - \varepsilon_W) \times v_{WD}\} \times G_D$		Arnot and Gobas (2004a)
Gill ventilation rate	G_V	L/d	$G_V = 1{,}400 \times W_B^{0.65}/C_{OX}$		Arnot and Gobas (2004a)
Organism-water partition coefficient on a wet weight basis	K_{BW}	L water/kg biota	$K_{BW} = k_1/k_2 = v_{LB} \times K_{OW}/\delta_L + v_{NB} \times \beta \times K_{OW} + v_{WB}/\delta_W$		Arnot and Gobas (2004a)
NLOM content of organism	v_{NB}	%	$v_{NB} = 1 - (v_{LB} + v_{WB})$		Arnot and Gobas (2004a)
NLOC content of phytoplankton	v_{NP}	%	$v_{NP} = 1 - (v_{LP} + v_{WP})$		Arnot and Gobas (2004a)
Phytoplankton/algae-water partition coefficient on a wet weight basis	K_{PW}	L water/kg phytoplankton / algae	$K_{PW} = v_{LP} \times K_{OW}/\delta_L + \beta_{OC} \times v_{NP} \times K_{OW} + v_{WP}/\delta_W$		Arnot and Gobas (2004a)
Chemical partition coefficient between the contents of the gastrointestinal tract and the organism	K_{GB}	kg biota/kg digesta	$K_{GB} = (v_{LG} \times K_{OW}/\delta_L + v_{OCG} \times \beta_{OC} \times K_{OW} + v_{NG} \times \beta \times K_{OW} + v_{WG}/\delta_W)/(v_{LB} \times K_{OW}/\delta_L + v_{NB} \times \beta \times K_{OW} + v_{WB}/\delta_W)$		Arnot and Gobas (2004a)
Lipid fraction of gut contents	v_{LG}	kg lipid/kg digesta ww	$v_{LG} = (1 - \varepsilon_L) \times v_{LD}/[(1 - \varepsilon_L) \times v_{LD} + (1 - \varepsilon_N) \times v_{OCD} + (1 - \varepsilon_N) \times v_{ND} + (1 - \varepsilon_W) \times v_{WD}]$		Arnot and Gobas (2004a)
NLOC fraction of gut contents	v_{OCG}	kg lipid/kg digesta ww	$v_{OCG} = [(1 - \varepsilon_N) \times v_{OCD}]/[(1 - \varepsilon_L) \times v_{LD} + (1 - \varepsilon_N) \times v_{OCD} + (1 - \varepsilon_N) \times v_{ND} + (1 - \varepsilon_W) \times v_{WD}]$	NLOC was added to the model to account for higher affinity of PCBs for NLOC compared to NLOM	January 2006 update to Arnot and Gobas model (Arnot and Gobas 2004a). Updated model, AQUAWEB, can be found on Environmental Toxicology Research Group website (Gobas 2006)
NLOM fraction of gut contents	v_{NG}	kg NLOM/kg digesta ww	$v_{NG} = (1 - \varepsilon_N) \times v_{ND}/[(1 - \varepsilon_L) \times v_{LD} + (1 - \varepsilon_N) \times v_{OCD} + (1 - \varepsilon_N) \times v_{ND} + (1 - \varepsilon_W) \times v_{WD}]$		Arnot and Gobas (2004a)

PARAMETER	SYMBOL	UNIT	EQUATION	NOTES	SOURCE
Water fraction of gut contents	v_{WG}	kg water/kg digesta ww	$v_{WG} = (1 - \varepsilon_W) \times v_{WD} / [(1 - \varepsilon_L) \times v_{LD} + (1 - \varepsilon_N) \times v_{OCD} + (1 - \varepsilon_N) \times v_{ND} + (1 - \varepsilon_W) \times v_{WD}]$		Arnot and Gobas (2004a)
Overall lipid content of the diet	v_{LD}	kg lipid/kg food ww	$v_{LD} = \Sigma P_i \times v_{LB,i}$		Arnot and Gobas model spreadsheet (Gobas 2006)
Overall NLOC content of the diet	v_{OCD}	kg NLOC/kg food ww	$v_{OCD} = P_P \times v_{OCP} + P_{sed} \times OC_{sed}$	Phytoplankton/algae and sediment are the only dietary items with non-lipid organic carbon content.	January 2006 (Gobas 2006) update to Arnot and Gobas model (Arnot and Gobas 2004a)
Overall NLOM content of the diet	v_{ND}	kg NLOM/kg food ww	$v_{ND} = \Sigma P_i \times v_{NB,i}$		Arnot and Gobas model spreadsheet (Gobas 2006)
Overall water content of the diet	v_{WD}	kg water/kg food ww	$v_{WD} = \Sigma P_i \times v_{WB,i}$		Arnot and Gobas model spreadsheet (Gobas 2006)
Non-lipid organic carbon (NLOC) content of phytoplankton	v_{OCP}	kg NLOC/kg phytoplankton	$v_{OCP} = 1 - (v_{LP} + v_{WP})$		
Fraction of non-lipid organic matter (NLOM) in organism *i*	$v_{NB,i}$	kg NLOM/kg organism	$v_{NB,i} = 1 - (v_{LB,i} + v_{WB,i})$	B = biota	
Environmental					
Freely dissolved chemical concentration in the porewater	$C_{WD,P}$	µg/L	$C_{WD,P} = C_{S,OC}/K_{OC}$		Kraaij et al. (2002), as cited in Arnot and Gobas (2004a)
Chemical concentration in the sediment, organic carbon normalized	$C_{S,OC}$	µg/kg	$C_{S,OC} = C_S/OC_{sed}$		Calculated using Phase 1 and Phase 2 sediment data
Freely dissolved chemical concentration in the water	C_{WD}	µg/L	$C_{WD} = (C_{WT} \times \phi)/1{,}000$	Simulates sequestering of chemical by DOC and POC in the water.	Arnot and Gobas (2004a)
Bioavailable solute fraction	ϕ	unitless	$\phi = 1/(1 + \chi_{POC} \times D_{POC} \times \alpha_{POC} \times K_{OW} + \chi_{DOC} \times D_{DOC} \times \alpha_{DOC} \times K_{OW})$	Simulates sequestering of chemical by DOC and POC in the water.	Arnot and Gobas (2004a)

PARAMETER	SYMBOL	UNIT	EQUATION	NOTES	SOURCE
Chemical					
Organic carbon-water partition coefficient	K_{OC}	L/kg	$K_{OC} = 0.35 \times K_{OW}$	There are many different relationships established between K_{OW} and K_{OC}. This relationship was based on the analysis of a wide range of analytes (including PCB congeners) and soil/sediment matrices. The authors excluded data that may not have represented equilibrium conditions that can be very influential for high-molecular-weight PCBs. It is consistent with the commonly used approximation of $K_{OC} = 0.4\ K_{OW}$.	Seth et al. (1999)

Source: Arnot and Gobas (2004a).

C – centigrade
DOC – dissolved organic carbon
LDW – Lower Duwamish Waterway
NLOC – non-lipid organic carbon
NLOM – non-lipid organic matter
PCB – polychlorinated biphenyl
POC – particulate organic carbon
ww – wet weight

D.3 Approach for Applying the Food Web Model in the Lower Duwamish Waterway

Numerous simplifications and assumptions are required to apply a steady-state bioaccumulation model to the dynamic estuarine environment in the LDW. This section presents the species that were modeled and spatial aspects of applying the FWM in the LDW. Parameter-specific assumptions are discussed in Section D.4 and general model uncertainties are discussed in Section D.6.

D.3.1 SPECIES MODELED

In order to apply the Arnot and Gobas model to the LDW, each species or species assemblage to be modeled was assigned to a compartment (i.e., phytoplankton/algae, zooplankton, filter-feeding benthic invertebrates, scavenger/predator/detritivore benthic invertebrates, and fish). Even though all compartments share a master equation (see equation for C_B in Table D.2-1), they have different sub-models (e.g., equations for rate constants) and different parameters defining those sub-models. Thus, selection of a compartment determines the parameters that need to be defined for each species or species assemblage.

Three species of adult fish, two species of adult crabs, and soft-shell clam species were modeled in the LDW. These species are referred to as target species because they were either receptors of concern (ROCs) in the ecological risk assessment (ERA) or served as key prey species for other receptors in the ERA or in the human health risk assessment (HHRA). Target species modeled included:

- English sole as: 1) an ROC in the ERA representing benthic fish that primarily consume invertebrates, 2) prey for wildlife ROCs, and 3) seafood consumed by people
- Pacific staghorn sculpin as: 1) an ROC in the ERA representing fish that consume both invertebrates and small fish, and 2) prey for wildlife ROCs
- Shiner surfperch as: 1) prey for wildlife ROCs, and 2) seafood consumed by people
- Dungeness crabs as: 1) an ROC in the ERA representing larger and more mobile invertebrates, 2) prey for wildlife ROCs, and 3) seafood consumed by people
- Slender crabs as: 1) prey for wildlife ROCs, and 2) seafood consumed by people
- Clams as: 1) prey for wildlife ROCs, and 2) seafood consumed by people

Fish and crabs were each modeled using a fish compartment.[3] Large clams[4] (*Mya arenaria*) were modeled using for a filter-feeding benthic invertebrate compartment.

Other prey species modeled included phytoplankton, zooplankton, benthic invertebrates, and juvenile fish. Phytoplankton, zooplankton, and juvenile fish were modeled using phytoplankton/algae, zooplankton, and fish compartments respectively. Benthic invertebrates, which make up a large portion of fish diets (see Section D.4.2.2), were modeled as a single assemblage using a scavenger/predator/ detritivore benthic invertebrate compartment. These species were modeled to serve as prey, approximating the transfer of chemicals from environmental media through the food web.

D.3.2 SPATIAL CONSIDERATIONS

The FWM was calibrated at the LDW-wide spatial scale (River Mile [RM] 0.0 to RM 5.25) (Map D.3-1). This assumes that the factors affecting a species' average bioaccumulation LDW-wide, and the factors affecting that species' average bioaccumulation at other spatial scales where the model is to be used, are similar. EPA/Ecology expressed an interest in applying the FWM at both the LDW-wide scale and smaller scales. Four subsections of the LDW (modeling areas M1, M2, M3, and M4) were defined, based on the four fish and crab tissue sampling areas (Map D.3-1). The performance of the FWM was tested for each modeling area (Section D.7.1).

Statistical analyses were conducted at the tissue sampling areas scale (ANOVAs) to explore absolute differences in total PCB concentrations in tissue among areas and at the tissue sampling subareas scale (regressions) in order to explore relationships between total PCB concentrations in tissue vs. sediment. This information was used to draw conclusions about how well the FWM is expected to perform at the scale of the modeling areas.

D.3.2.1 Summary of the literature on spatial scale of exposure

Little is known about the foraging ranges of the modeled species; thus, the spatial extent of their PCB exposure was unknown. Literature data suggest that English sole have home ranges on the order of 4 to 10 km^2 (Day 1976; Stern et al. 2003; Lassuy 1989), but these studies do not clearly indicate what the English sole foraging range

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[3] Crabs are large mobile invertebrates that eat shrimp, juvenile crabs, and fish. Crabs were modeled using fish equations instead of scavenger/predator/ detritivore benthic invertebrate equations because the majority of the species used to develop the scavenger/predator/ detritivore benthic invertebrate equations and constants were filter feeders or detritivores. In addition, it was determined early in the modeling process that using fish equations resulted in estimates that were more similar to empirical data for crabs.

[4] The average length of *Mya arenaria* collected in the LDW for the 14 composite clam samples was 7.0 cm. *Macoma nasuta*, a smaller species, was collected at three locations in the LDW and included with *Mya arenaria* in three composite samples. Average length of the *Macoma nasuta* collected was 2.2 cm.

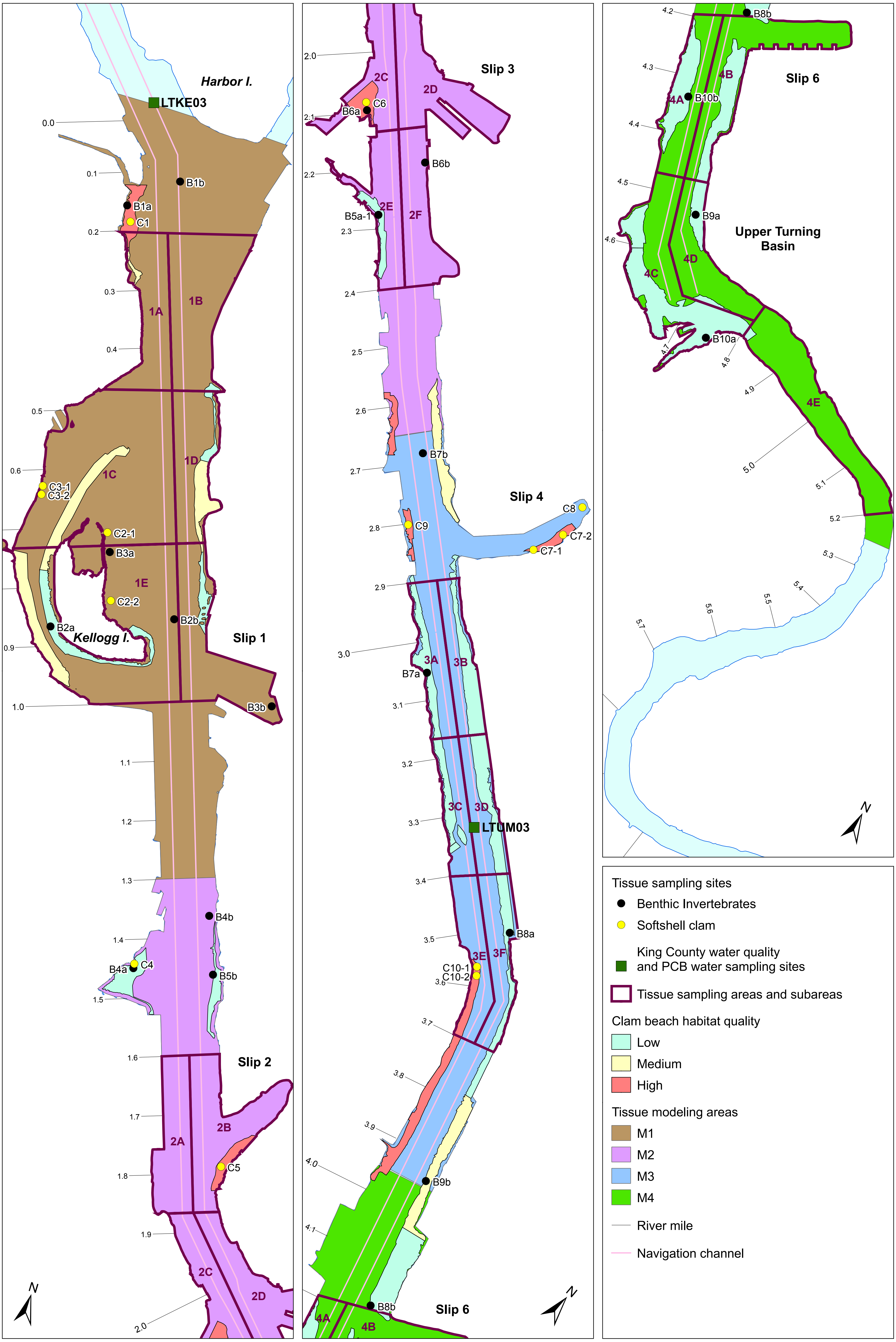
may be in the LDW. No studies were available to estimate the foraging ranges of the other fish and crab species modeled. Discussion among regional experts has suggested that foraging ranges of English sole and Dungeness crab are likely to be larger than half the LDW and as large as or larger than the entire LDW. In fact, English sole are known to migrate from the LDW into the deeper waters of Puget Sound as part of the spawning cycle (Angell et al. 1975); English sole are believed to return to similar areas after spawning (Day 1976). Because they generally have smaller body sizes and thus lower nutritional requirements, Pacific staghorn sculpin and shiner surfperch foraging ranges may be smaller.

D.3.2.2 Summary of statistical findings on spatial scale of exposure in the LDW

Data from 190 composite tissue samples collected between 1997 and 2005 for seven species (English sole, shiner surfperch, Pacific staghorn sculpin, Dungeness crab, slender crab, clams, and benthic invertebrates) were used to develop FWM input parameter values (e.g., lipid content and water content) and to test model performance (e.g., total PCB concentrations in tissue). Data from 1,264 surface sediment samples (baseline sediment database) collected since 1990 were used to calculate total PCB concentrations in sediment and percent sediment organic carbon. Statistical analyses were conducted on the co-located sediment and tissue data for PCBs. These analyses were helpful in assessing whether average total PCB concentrations in tissues varied by tissue sampling area, and if tissue concentrations in samples collected from specific subareas were correlated with subarea spatially weighted average concentrations (SWACs).

The statistical analyses provided modest support for the assumption that English sole and crab species in the LDW integrate exposure over areas larger than the modeling areas and that Pacific staghorn sculpin, and to a lesser extent shiner surfperch, may integrate exposures over areas smaller than the modeling areas. A summary of the analyses is provided below.

ANOVAs were conducted to evaluate whether there were differences among the four sampling areas in either 2004 or in 2005. Crabs were not evaluated because of insufficient sample sizes in some tissue sampling areas. The highest average sediment total PCB concentrations were in Area T3; whereas those in Areas T1, T2, and T4 were just below or just above the SWAC for the entire LDW (Figure D.3-1).



Wind Ward environmental LLC

0 0.1 0.2 Miles

0 0.1 0.2 Kilometers

Scale is the same for each inset map

Map D.3-1. Lower Duwamish Waterway modeling areas and tissue and water sampling locations

DRAFT

Prepared by CEH, 10/02/07; MAP 2756; W:\Projects\00-08-06_Duwamish_RI\data\gis\Phase2 RI\Food Web Model

For both English sole and shiner surfperch, the relative magnitudes (rank ordering) of mean $\log_{10}$-transformed total PCB concentrations in all four sampling areas were consistent in 2005 and 2004[5] (Figure D.3-2). Both species had their lowest mean tissueconcentrations in Area T4 in both years. In 2004, the mean of $\log_{10}$-transformed concentrations in Area T4 was significantly lower than mean of $\log_{10}$-transformed concentrations from the two areas with the highest mean concentrations (Areas T1 and T2 for English sole; Areas T2 and T3 for shiner surfperch)[6]. Also in that year, the mean of $\log_{10}$-transformed concentrations in tissues from the two areas with the highest mean concentrations did not differ significantly[7] and the mean of $\log_{10}$-transformed concentrations in tissue samples from the two areas with the lowest concentrations did not differ significantly[8] (Areas T3 and T4 for English sole; Areas T1 and T4 for shiner surfperch). Statistical differences between concentrations in areas with intermediate concentrations were marginally significant.[9]

In 2005, fewer statistically significant differences existed between the modeling areas. For English sole, concentrations in Area T4 were lower than concentrations in T2[10] but for shiner surfperch, there were no significant differences among areas.[11]

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[5] Interaction effect in two-way ANOVA not significant. See methods and results of two-way ANOVA discussed in Section 4.2.1.4.1 of the RI.

[6] Based on post hoc multiple pairwise ANOVA comparisons run after finding a significant effect of area in a one-way ANOVA testing for effects of year ($p < 0.0005$ for both species; see methods and results of two-way ANOVA discussed in Section 4.2.1.4.1 of the RI. For log-transformed tissue concentrations in English sole, T4 < T1 ($p = 0.003$); and T4 < T2 ($p = 0.008$). For log-transformed tissue concentrations in shiner surfperch, T4 < T3 ($p = 0.001$); and T4 < T2 ($p = 0.009$).

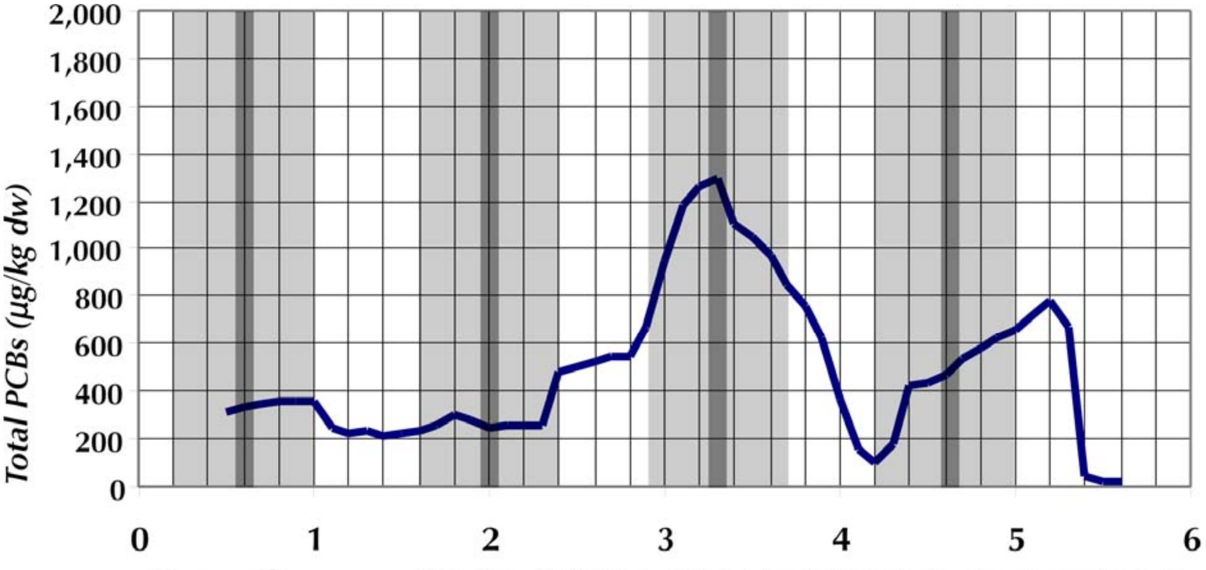
[7] $p > 0.92$ for both species.

[8] $p > 0.43$ for both species.

[9] For English sole mean log-transformed tissue concentrations were lower in T3 than T2 ($p = 0.049$); and for shiner surfperch they were lower in T1 than T3 ($p = 0.075$).

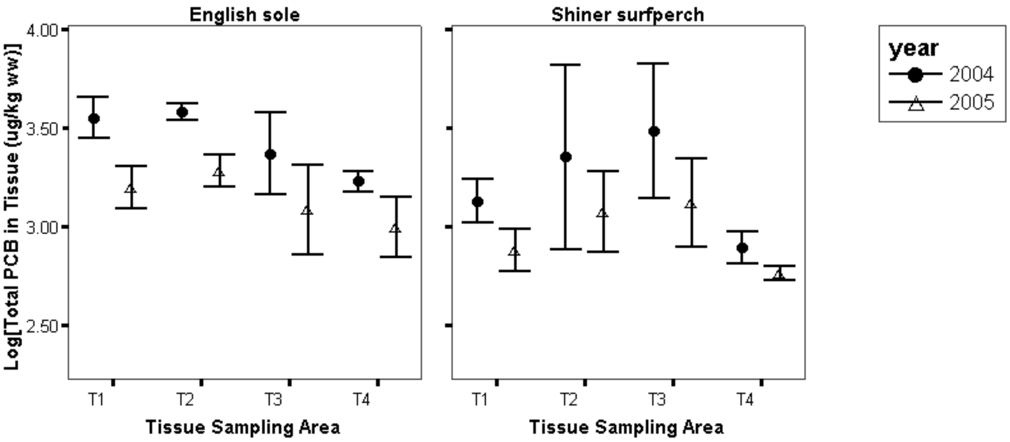
[10] For English sole, mean log-transformed tissue concentrations were lower in T4 than T2 ($p = 0.025$) and in T3 than T2 ($p = 0.080$). The lowest p-value for all other pairwise comparisons was 0.20.

[11] For shiner surfperch, the lowest pairwise comparison p-value was 0.13.



Note: the wide light gray vertical stripes indicate the four discrete sampling areas; the narrow darker gray stripes mark the centers of the sampling areas.

Figure D.3-1. One-mile rolling average total PCB concentration in LDW surface sediment



Note: Lines connecting area means were added for pattern assessment.

Figure D.3-2. Mean and 95% confidence intervals for total PCBs in English sole and shiner surfperch tissues by tissue sampling area

Regression analyses of total PCB concentrations in shiner surfperch and Pacific staghorn sculpin composite samples relative to average total PCB concentrations in sediment were performed to determine if there was a relationship at the spatial scale of a subarea (defined as one-sixth of the associated modeling area, roughly 0.3 mi in length and half the width of the waterway). Tissue data were available from 22 of 24 subareas for shiner surfperch and from 23 of 24 subareas for Pacific staghorn sculpin. Other species were sampled on an area-wide basis (see Map 4-9 in the RI). Significant positive linear relationships were identified using 2004 data for both Pacific staghorn sculpin (Figure D.3-3, $R^2 = 0.51$) and shiner surfperch (Figure D.3-4, $R^2 = 0.64$), in which sediment concentrations explained more than 50% of the variance in tissue concentrations. In 2005, the relationship for shiner surfperch was significant but not as strong (Figure D.3-5, $R^2 = 0.29$). A regression analysis was not conducted in 2005 for Pacific staghorn sculpin because fewer data were available for 2005. These results demonstrate that total PCB concentrations in sediment do not explain all the variability in total PCB concentrations in tissue at a subarea scale for these species.

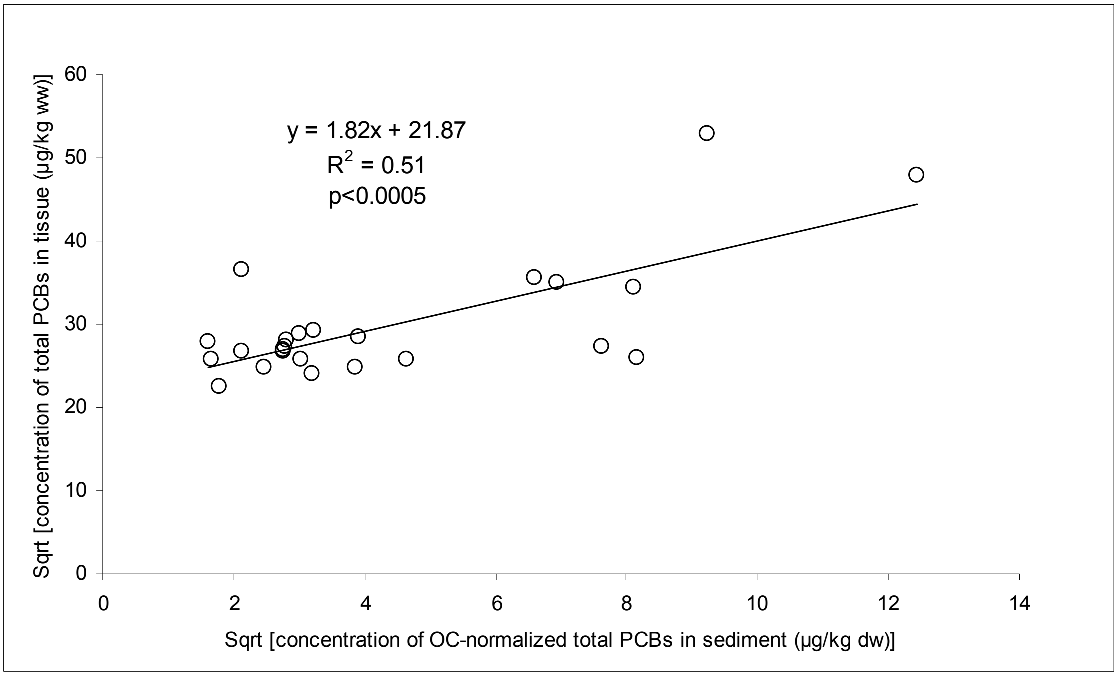


Figure D.3-3. **Regression between total PCB concentrations in sediment and 2004 Pacific staghorn sculpin tissue on a subarea basis**

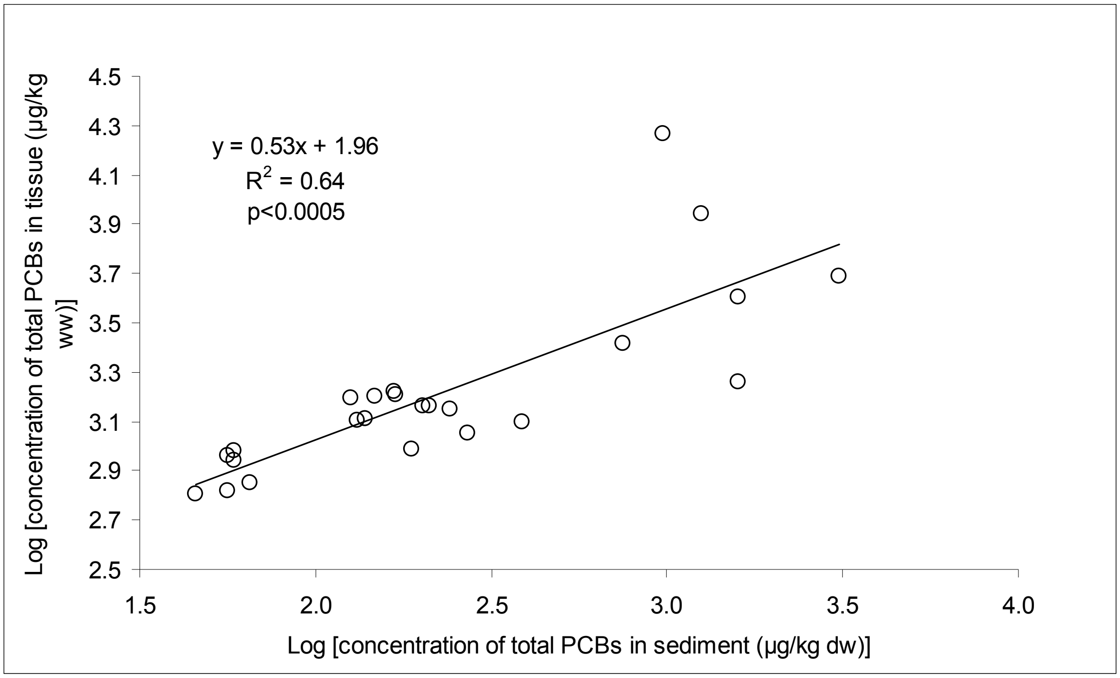


Figure D.3-4. **Regression between total PCB concentrations in sediment and 2004 shiner surfperch tissue on a subarea basis**

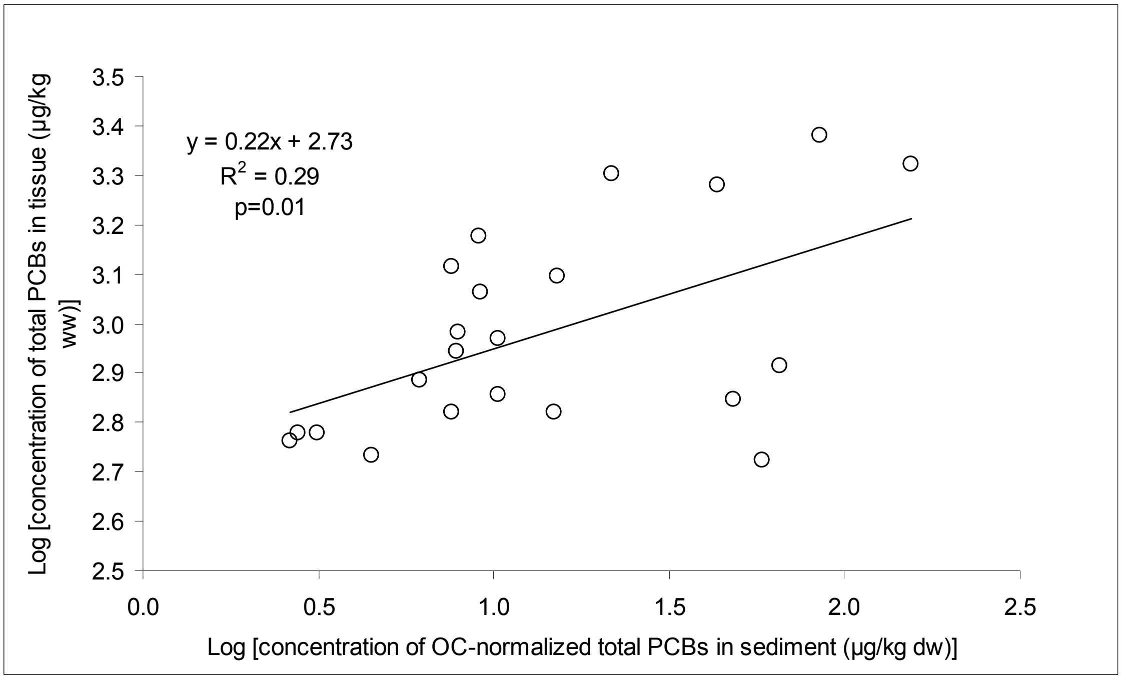


Figure D.3-5. **Regression between total PCB concentrations in sediment and 2005 shiner surfperch tissue on a subarea basis**

Results from the ANOVAs indicate that applying the FWM at areas smaller than the LDW could be appropriate for shiner surfperch because tissue concentrations varied among tissue sampling areas and patterns of tissue concentrations roughly corresponded to patterns of total PCB concentrations in sediment. Although the ANOVAs indicated differences in tissue concentrations among tissue sampling areas for English sole, the patterns of tissue concentrations did not correspond with patterns of sediment concentrations. English sole and the crab species appear to be wide-ranging species relative to the spatial scale of the modeling areas, thus the FWM should not be applied at that spatial scale for English sole and crabs.

Regression analyses at the subarea scale for 2005 data revealed that for Pacific staghorn sculpin, 51% of the variance in total PCB tissue concentrations was explained by concentrations of total PCBs in sediment, indicating that it may be appropriate to apply the FWM for Pacific staghorn sculpin at areas smaller than the LDW. Regression analyses at the subarea scale for 2005 and 2004 data for shiner surfperch, revealed that 29% and 64% respectively, of the variance in total PCB tissue concentration of shiner surfperch is explained by concentrations of total PCBs in sediment, indicating that other factors accounted for significant amounts of the variance in tissue concentration in 2005 and that spatial trends in tissue concentrations were not consistent between years. Thus, in combination with the results of the ANOVAs as described above, the FWM may be applied at smaller scales for shiner surfperch but it application is somewhat uncertain. In addition, it is likely that species do not use all areas of the LDW equally, and some species may leave the LDW for part of the year. Nonetheless, the performance of the FWM at the modeling area was tested for all species. Methods and results of this test are presented in Section D.7.1.

D.4 Model Parameters

Application of the Arnot and Gobas (2004a) FWM to the LDW required the selection of values for 114 input parameters (including dietary fractions). Because the Arnot and Gobas model was applied in the LDW assuming steady-state conditions, it was most appropriate for parameter values to represent means of populations (as opposed to individuals) and means over several years (as opposed to shorter periods [e.g., 1 month]). Uncertainty regarding the estimates of mean values for parameters was represented quantitatively through the use of probability distributions. The model was run and calibrated probabilistically in order to systematically explore all plausible parameter sets and their corresponding estimated total PCB concentrations in tissue. Probability distributions were developed for 95 parameters, and point estimates were used to characterize 19 parameters with limited data, low variability, and/or low sensitivity.

To characterize a parameter distribution, several statistical descriptors (e.g., mean, mode, standard deviation) were required. Estimates of the probable mean values for each input parameter were represented by either a normal or triangular distribution, which was assumed to represent the uncertainty around the mean estimate). Parameter names, symbols, units, selected values (probability distributions or point estimates), comments, and source information are presented in Table D.4-1.

Table D.4-1. Input parameter probability distribution statistics and point estimate values

PARAMETER	SYMBOL	UNIT	VALUES[a]	DISTRIBUTION TYPE	SOURCE/NOTES
Environmental Parameters					
Concentration of total PCBs in water column water	C_{WT}	ng/L	mode = 1.43 mean = 1.59 min = 0.185 max = 3.14	triangular	Mode estimated from EFDC hydrodynamic model output on October 12, 2006. Mode is the mean of 12 monthly averages from bottom three layers in EFDC model. Maximum and minimum values are from King County empirical PCB water data from samples 1 m above bottom (Mickelson and Williston 2006).
Concentration of POC in water column water	χ_{DOC}	kg/L	mean = 2.6×10^{-7} SE = 4.4×10^{-8}	normal	Calculated from unpublished King County 2005 water data (Mickelson 2006) from samples 1 m above bottom. POC is calculated as follows, POC = TOC – DOC. Samples with zero or negative results for POC were replaced with an estimate of POC calculated as follows: POC = 0.0186 × TSS.
DOC in water column water	χ_{POC}	kg/L	mean = 2.2×10^{-6} SE = 2.5×10^{-7}	normal	Unpublished King County 2005 water data (Mickelson 2006) from samples 1 m above bottom.
Proportionality constant describing similarity in phase partitioning of DOC relative to that of octanol	α_{DOC}	unitless	0.08	point estimate	Value from Burkhard (1999), as cited in Arnot and Gobas (Arnot and Gobas 2004a). Used in the bioavailable solute fraction equation for simulating sequestering of chemical by DOC in the water.
Proportionality constant describing similarity in phase partitioning of POC relative to that of octanol	α_{POC}	unitless	0.35	point estimate	Value from Seth et al. (1999) as cited in Arnot and Gobas (Arnot and Gobas 2004a). Used in the bioavailable solute fraction equation for simulation of sequestering of chemical by POC in the water.
Disequilibrium factor for DOC partitioning	D_{DOC}	unitless	1	point estimate	Value from Arnot and Gobas (2004a). Used in the bioavailable solute fraction equation for simulation of sequestering of chemical by DOC in the water. Assumes chemicals in water column water are in equilibrium with DOC.
Disequilibrium factor for POC partitioning	D_{POC}	unitless	1	point estimate	Value from Arnot and Gobas (2004a).Used in the bioavailable solute fraction equation for simulation of sequestering of chemical by POC in the water. Assumes chemicals in water column water are in equilibrium with POC.

PARAMETER	SYMBOL	UNIT	VALUES[a]	DISTRIBUTION TYPE	SOURCE/NOTES
Mean temperature of water column water	T	°C	mean = 11.2 SE = 0.397	normal	Unpublished King County 2005 water data (Mickelson 2006) from samples 1 m above bottom.
Dissolved oxygen concentration in water column water	C_{OX}	mg/L	mean = 7.93 SE = 0.203	normal	Unpublished King County 2005 water data (Mickelson 2006) from samples 1 m above bottom.
TSS concentration in water column water	C_{SS}	kg/L	mean = 5.8×10^{-6} SE =8.8×10^{-7}	normal	Unpublished King County 2005 water data (Mickelson 2006) from samples 1 m above bottom. Used TSS samples filtered with a 45-µm filter to be consistent with POC definition (> 45 µm).
Density of seawater	δ_W	kg/L	1.03	point estimate	Value from Sverdrup et al. (1942). Point estimate assumed because of the narrow range of values in literature.
Concentration of total PCBs in sediment	C_S	µg/kg dw	mean = 380	point estimate	SWAC calculated using IDW on October 20, 2006, based on 1,264 samples from the LDW baseline surface sediment database.
Sediment organic carbon	OC_{sed}	%	mean = 1.91 SE = 0.025	normal	SWAC calculated using Thiessen polygons on October 20, 2006, based on 1,264 samples from the LDW baseline surface sediment database. Sediment OC calculated using Thiessen polygons to allow calculation of SE.
Chemical Parameters					
Log octanol-water partition coefficient for total PCBs	log K_{OW}	L/kg	mean = 6.6 SE = 0.05	normal	Weighted average of log K_{OW} based on PCB congeners analyzed in benthic invertebrate tissue. Log K_{OW}s for each congener from Hawker and Connell (1988).
Proportionality constant expressing the sorption capacity of NLOM for an organic chemical relative to that of octanol	β	unitless	mean = 0.035 SE = 0.005[b]	normal	Mean from Arnot and Gobas (2004a); SE was set equal to the SD reported by Arnot (2005).
Proportionality constant expressing the sorption capacity of NLOC for an organic chemical relative to that of octanol	β_{OC}	L/kg	0.35	point estimate	Value from Seth et al. (1999), as cited in Arnot and Gobas (2004a).
Rate constant for metabolic transformation of total PCBs	k_M	day^{-1}	0	point estimate	Value for k_M assumed to be zero for total PCBs (Arnot 2006b).

PARAMETER	SYMBOL	UNIT	VALUES[a]	DISTRIBUTION TYPE	SOURCE/NOTES
Biological Parameters					
Density of lipids	δ_L	kg/L	mode = 0.9 mean = 0.9 min = 0.8 max = 1	triangular	Data from Arnot (2006a).
Fraction of prey item i in the diet of organism	P_i	fraction	na		See Table D.4-7 for values defining triangular distributions for each dietary item for all species. Prey items consist of organisms (phytoplankton, zooplankton, benthic invertebrates and juvenile fish) and sediment.
Phytoplankton/Algae					
Lipid content	v_{LP}	%	mean = 0.12 SE = 0.05[b]	normal	Data from Mackintosh et al. (2004). SE was set equal to the SD reported by Mackintosh et al. (2004).
Water content[c]	v_{WP}	%	mean = 95.6 SE = 0.55[b]	normal	Data from Mackintosh et al. (2004). SE was set equal to the SD reported by Mackintosh et al. (2004).
Rate constant for growth of phytoplankton/algae	k_G	day^{-1}	0.08	point estimate	Value from Swackhamer and Skoglund (1993) as cited in Arnot and Gobas (2004a). Only phytoplankton/algae has k_G as an input number instead of an equation. This is a mean annual value based on empirical data in which slow-growth conditions (winter) were 0.03 day^{-1} and active-growth conditions (summer) were 0.13 day^{-1}.
Resistance to chemical uptake through aqueous phase for phytoplankton/algae	A	day^{-1}	mean = 6×10^{-5} SE = 1×10^{-5} [b]	normal	Values from Gobas and Arnot (2005). SE was set equal to the SD reported by Gobas and Arnot (2005)
Resistance to chemical uptake through organic phase for phytoplankton/algae	B	unitless	mode = 5.5 mean = 5.5 min = 1.8 max = 9.2	triangular	Values from Gobas and Arnot (2005) and Arnot and Gobas (2004a).
Zooplankton					
Weight	W_B	kg	mean = 1.6×10^{-7} SE = 3.6×10^{-8} [b]	normal	Data from Giles and Cordell (1998). SE was set equal to the SD reported by Giles and Cordell (1998).

PARAMETER	SYMBOL	UNIT	VALUES[a]	DISTRIBUTION TYPE	SOURCE/NOTES
Lipid content	v_{LB}	%	mean = 1.2 SE = 0.3[d]	normal	Data from Kuroshima et al. (1987). SE was set equal to the SD of data reported in Kuroshima et al. (1987), assuming the data represented a distribution of mean values.
Water content[e]	v_{WB}	%	mean = 90 SE = 1.5[d]	normal	Data from Kuroshima et al. (1987). SE was set equal to the SD of data reported in Kuroshima et al. (1987), assuming the data represented a distribution of mean values.
Dietary absorption efficiency of lipids	ε_L	%	mode = 72 mean = 71 min = 55 max = 85	triangular	Data from Conover (1966) as cited in Arnot and Gobas (2004a). Study involved *Calanus hyperboreus* eating diatoms and flagellates from Gulf of Maine.
Dietary absorption efficiency of NLOM	ε_N	%	mode = 72 mean = 71 min = 55 max = 85	triangular	Data from Conover (1966) as cited in Arnot and Gobas (2004a). Study involved *Calanus hyperboreus* eating diatoms and flagellates from Gulf of Maine.
Dietary absorption efficiency of water	ε_W	%	55	point estimate	Value from Gobas and Arnot (2005).
Benthic Invertebrates					
Weight	W_B	kg	mean = 5.1×10^{-5} SE = 2.0×10^{-5}	normal	Values derived from LDWG Phase 2 data. See description of methods for deriving weights in Section D.4.1.3.2.
Lipid content	v_{LB}	%	mean = 0.89 SE = 0.06	normal	LDWG Phase 2 data (n = 20).
Water content[e]	v_{WB}	%	mode = 80 mean = 79 min = 71 max = 87	triangular	Water content range data for bivalves, isopods, amphipods, and cladocerans reported in an Oak Ridge National Laboratory publication were used to derive the mode, maximum, and minimum statistics of a triangular distribution for benthic invertebrate water content (Sample et al. 1997).
Relative fraction of porewater ventilated[f]	m_P	unitless	mode = 0.20 mean = 0.17 min = 0.05 max = 0.25	triangular	Used Winsor et al. (1990), Gobas and Wilcockson (2003), Gobas and Arnot (2005), and knowledge of organism behavior to develop values.
Dietary absorption efficiency of lipids	ε_L	%	mode = 75 mean = 62 min = 15 max = 96	triangular	Data from Roditi and Fisher (1999), Berge and Brevik (1996), Gordon (1966), Parkerton (1993) as cited in Arnot and Gobas (2004a). These studies involved zebra mussels from tidal freshwater section of the Hudson River and polychaetes from Cape Cod intertidal flats.

PARAMETER	SYMBOL	UNIT	VALUES[a]	DISTRIBUTION TYPE	SOURCE/NOTES
Dietary absorption efficiency of NLOM	ε_N	%	mode = 75 mean = 62 min = 15 max = 96	triangular	Data from Roditi and Fisher (1999), Berge and Brevik (1996), Gordon (1966), Parkerton (1993) as cited in Arnot and Gobas (2004a). These studies involved zebra mussels from the tidal freshwater section of the Hudson River and polychaetes from Cape Cod intertidal flats.
Dietary absorption efficiency of water	ε_W	%	55	point estimate	Value from Gobas and Arnot (2005).
Clam					
Weight	W_B	kg	mean = 0.037 SE = 0.0027	normal	Weight calculated using 2004 length data and a weight vs. length regression based on *Mya arenaria* data from the August 8 to 12, 2003, intertidal clam survey in the LDW and the August 13, 2003, catch per unit effort survey.
Lipid content	v_{LB}	**%**	mean = 0.71 SE = 0.026	normal	LDWG Phase 2 data (n = 14).
Water content[e]	v_{WB}	**%**	mean = 85.2 SE = 0.345	normal	LDWG Phase 2 data (n = 14).
Relative fraction of porewater ventilated[f]	m_P	unitless	mode = 0.20 mean = 0.17 min = 0.05 max = 0.25	triangular	Used Winsor et al. (1990), Gobas and Wilcockson (2003), Gobas and Arnot (2005), and knowledge of organism behavior to develop values.
Dietary absorption efficiency of lipids	ε_L	%	mode = 75 mean = 62 min = 15 max = 96	triangular	Data from Roditi and Fisher (1999), Berge and Brevik (1996), Gordon (1966), Parkerton (1993) as cited in Arnot and Gobas (2004a). These studies involved zebra mussels from tidal freshwater section of the Hudson River and polychaetes from Cape Cod intertidal flats.
Dietary absorption efficiency of NLOM	ε_N	%	mode = 75 mean = 62 min = 15 max = 96	triangular	Data from Roditi and Fisher (1999), Berge and Brevik (1996), Gordon (1966), Parkerton (1993) as cited in Arnot and Gobas (2004a). These studies involved zebra mussels from the tidal freshwater section of the Hudson River and polychaetes from Cape Cod intertidal flats.
Dietary absorption efficiency of water	ε_W	%	55	point estimate	Value from Gobas and Arnot (2005).
Filter feeder particle scavenging efficiency	σ	fraction	1	point estimate	Value from Arnot and Gobas (2004a). Used to calculate feeding rate for filter feeders.

PARAMETER	SYMBOL	UNIT	VALUES[a]	DISTRIBUTION TYPE	SOURCE/NOTES
Juvenile Fish					
Weight	W_B	kg	mean = 6×10^{-3} SE = 7×10^{-4}	normal	Based on ≤ 80 mm shiner surfperch from the LDW and background locations from sampled in 2004 and 2005 (n = 16).
Lipid content	v_{LB}	%	mean = 2.5 SE = 0.6	normal	Mean value based on mean lipid content of adult shiner surfperch and English sole collected from the LDW with a correction factor of 0.5 applied based on ratios of juvenile and adult fish lipids described in the literature (Gobas and Arnot 2005; Robards et al. 1999). Standard deviation estimated as 2 × standard error of 19 lipid values (Section D.4.2.1).
Water content[e]	v_{WB}	%	mean = 73.9 SE = 2.0	normal	Based on LDWG Phase 2 data for adult shiner surfperch. Mean of all composite samples (n = 46).
Relative fraction of porewater ventilated[f]	m_P	unitless	mode = 0.01 mean = 0.01 min = 0.005 max = 0.02	triangular	Used Winsor et al. (1990), Gobas and Wilcockson (2003), Gobas and Arnot (2005), and knowledge of organism behavior to develop values.
Dietary absorption efficiency of lipids	ε_L	%	mode = 92 mean = 92 min = 90 max = 95	triangular	Data from Gobas et al. (1999) as cited in Arnot and Gobas (2004a). Based on 73-day laboratory test with adult rainbow trout (*Oncorhynchus mykiss*) and a field study of rock bass (*Ambloplites rupestris*).
Dietary absorption efficiency of NLOM	ε_N	%	mode = 60 mean = 58 min = 50 max = 65	triangular	Data from Nichols et al. (2001) as cited in Arnot and Gobas (2004a). Based on study with tetrachlorobiphenyl and rainbow trout.
Dietary absorption efficiency of water	ε_W	%	55	point estimate	Value from Gobas and Arnot (2005).
Slender Crab					
Weight	W_B	kg	mean = 0.167 SE = 0.0038	normal	LDWG Phase 2 data (n = 13). Values derived using a weight-weighted approach[g] for each crab in a composite sample (see Section D.4.1.3.2 for methods).
Lipid content	v_{LB}	%	mean = 1.1 SE = 0.047	normal	LDWG Phase 2 data (n = 13).

PARAMETER	SYMBOL	UNIT	VALUES[a]	DISTRIBUTION TYPE	SOURCE/NOTES
Water content[e]	v_{WB}	%	mean = 83.8 SE = 0.371	normal	LDWG Phase 2 data (n = 13).
Relative fraction of porewater ventilated[f]	m_P	unitless	mode = 0.02 mean = 0.02 min = 0.01 max = 0.03	triangular	Used Winsor et al. (1990), Gobas and Wilcockson (2003), Gobas and Arnot (2005), and knowledge of organism behavior to develop values.
Dietary absorption efficiency of lipids	ε_L	%	mode = 75 mean = 62 min = 15 max = 96	triangular	Data from Roditi and Fisher (1999), Berge and Brevik (1996), Gordon (1966), Parkerton (1993) as cited in Arnot and Gobas (2004a). These studies involved zebra mussels from tidal freshwater section of the Hudson River and polychaetes from Cape Cod intertidal flats.
Dietary absorption efficiency of NLOM	ε_N	%	mode = 75 mean = 62 min = 15 max = 96	triangular	Data from Roditi and Fisher (1999), Berge and Brevik (1996), Gordon (1966), Parkerton (1993) as cited in Arnot and Gobas (2004a). These studies involved zebra mussels from the tidal freshwater section of the Hudson River and polychaetes from Cape Cod intertidal flats.
Dietary absorption efficiency of water	ε_W	%	55	point estimate	Value from Gobas and Arnot (2005).
Dungeness Crab					
Weight	W_B	kg	mean = 0.528 SE = 0.058	normal	LDWG Phase 2 data (n = 10). Values derived using a weight-weighted[g] approach for each crab in a composite sample (see Section D.4.1.3.2 for methods).
Lipid content	v_{LB}	%	mean = 2.6 SE = 0.40	normal	LDWG Phase 1 and 2 data (n = 12).
Water content[e]	v_{WB}	%	mean = 82 SE = 0.74	normal	LDWG Phase 1 and 2 data (n = 12).
Relative fraction of porewater ventilated[f]	m_P	unitless	mode = 0.02 mean = 0.02 min = 0.01 max = 0.03	triangular	Used Winsor et al. (1990), Gobas and Wilcockson (2003), Gobas and Arnot (2005), and knowledge of organism behavior to develop values.
Dietary absorption efficiency of lipids	ε_L	%	mode = 75 mean = 62 min = 15 max = 96	triangular	Data from Roditi and Fisher (1999), Berge and Brevik (1996), Gordon (1966), Parkerton (1993) as cited in Arnot and Gobas (2004a). These studies involved zebra mussels from tidal freshwater section of the Hudson River and polychaetes from Cape Cod intertidal flats.

PARAMETER	SYMBOL	UNIT	VALUES[a]	DISTRIBUTION TYPE	SOURCE/NOTES
Dietary absorption efficiency of NLOM	ε_N	%	mode = 75 mean = 62 min = 15 max = 96	triangular	Data from Roditi and Fisher (1999), Berge and Brevik (1996), Gordon (1966), Parkerton (1993) as cited in Arnot and Gobas (2004a). These studies involved zebra mussels from the tidal freshwater section of the Hudson River and polychaetes from Cape Cod intertidal flats.
Dietary absorption efficiency of water	ε_W	%	55	point estimate	Value from Gobas and Arnot (2005).
Pacific Staghorn Sculpin					
Weight	W_B	kg	mean = 0.077 SE = 0.0037	normal	LDWG Phase 2 data (n = 28). Values derived using a weight-weighted[g] approach for each fish in a composite sample (see Section D.4.1.3.2 for methods).
Lipid content	v_{LB}	%	mean = 2.1 SE = 0.07	normal	LDWG Phase 2 data (n = 28).
Water content[e]	v_{WB}	%	mean = 79.0 SE = 0.1	normal	LDWG Phase 2 data (n = 28).
Relative fraction of porewater ventilated[f]	m_P	unitless	mode = 0.05 mean = 0.06 min = 0.02 max = 0.1	triangular	Used Winsor et al. (1990), Gobas and Wilcockson (2003), Gobas and Arnot (2005), and knowledge of organism behavior to develop values.
Dietary absorption efficiency of lipids	ε_L	%	mode = 92 mean = 92 min = 90 max = 95	triangular	Data from Gobas et al. (1999) as cited in Arnot and Gobas (2004a). Based on 73-day laboratory test with adult rainbow trout (*Oncorhynchus mykiss*) and a field study of rock bass (*Ambloplites rupestris*).
Dietary absorption efficiency of NLOM	ε_N	%	mode = 60 mean = 58 min = 50 max = 65	triangular	Data from Nichols et al. (2001) as cited in Arnot and Gobas (2004a). Based on study with tetrachlorobiphenyl and rainbow trout.
Dietary absorption efficiency of water	ε_W	%	55	point estimate	Value from Gobas and Arnot (2005).
Shiner Surfperch					
Weight	W_B	kg	mean = 0.019 SE = 0.00043	normal	LDWG Phase 2 data (n = 46). Values derived using a weight-weighted[g] approach for each fish in a composite sample (see Section D.4.1.3.2 for methods).
Lipid content	v_{LB}	%	mean = 4.6 SE = 0.19	normal	LDWG Phase 1 and 2 data (n = 49).

PARAMETER	SYMBOL	UNIT	VALUES[a]	DISTRIBUTION TYPE	SOURCE/NOTES
Water content[e]	v_{WB}	%	mean = 73.9 SE = 0.3	normal	LDWG Phase 2 data (n = 46).
Relative fraction of porewater ventilated[f]	m_P	unitless	mode = 0.01 mean = 0.01 min = 0.005 max = 0.02	triangular	Used Winsor et al. (1990), Gobas and Wilcockson (2003), Gobas and Arnot (2005), and knowledge of organism behavior to develop values.
Dietary absorption efficiency of lipids	ε_L	%	mode = 92 mean = 92 min = 90 max = 95	triangular	Data from Gobas et al. (1999) as cited in Arnot and Gobas (2004a). Based on 73-day laboratory test with adult rainbow trout (*Oncorhynchus mykiss*) and a field study of rock bass (*Ambloplites rupestris*).
Dietary absorption efficiency of NLOM	ε_N	%	mode = 60 mean = 58 min = 50 max = 65	triangular	Data from Nichols et al. (2001) as cited in Arnot and Gobas (2004a). Based on study with tetrachlorobiphenyl and rainbow trout.
Dietary absorption efficiency of water	ε_W	%	55	point estimate	Value from Gobas and Arnot (2005).
English Sole					
Weight	W_B	kg	mean = 0.247 SE = 0.010	normal	LDWG Phase 2 data (n = 42). Values derived using a weight-weighted[g] approach for each fish in a composite sample (see Section D.4.1.3.2 for methods).
Lipid content	v_{LB}	%	mean = 5.5 SE = 0.20	normal	LDWG Phase 2 data (n = 42).
Water content[e]	v_{WB}	%	mean = 75.0 SE = 0.3	normal	LDWG Phase 2 data (n = 42).
Relative fraction of porewater ventilated[f]	m_P	unitless	mode = 0.01 mean = 0.01 min = 0.005 max = 0.02	triangular	Used Winsor et al. (1990), Gobas and Wilcockson (2003), Gobas and Arnot (2005), and knowledge of organism behavior to develop values.

PARAMETER	SYMBOL	UNIT	VALUES[a]	DISTRIBUTION TYPE	SOURCE/NOTES
Dietary absorption efficiency of lipids	ε_L	%	mode = 92 mean = 92 min = 90 max = 95	triangular	Data from Gobas et al. (1999) as cited in Arnot and Gobas (2004a). Based on 73-day laboratory test with adult rainbow trout (*Oncorhynchus mykiss*) and a field study of rock bass (*Ambloplites rupestris*).
Dietary absorption efficiency of NLOM	ε_N	%	mode = 60 mean = 58 min = 50 max = 65	triangular	Data from Nichols et al. (2001) as cited in Arnot and Gobas (2004a). Based on study with tetrachlorobiphenyl and rainbow trout.
Dietary absorption efficiency of water	ε_W	%	55	point estimate	Value from Gobas and Arnot (2005).

[a] The mean value is shown for triangular distributions to facilitate comparison with calibration results only; it was not used in the model. Standard error was used to represent the standard deviation in Crystal Ball™, assuming that values in the distribution were estimates of the mean.

[b] SE was represented by a SD reported in the literature.

[c] NLOC content of phytoplankton (v_{NP}, in units of %) was calculated using the following equation: $v_{NP} = 1 - (v_{LP} + v_{WP})$.

[d] SE was represented by a SD calculated from data assumed to represent a distribution of mean values.

[e] NLOM content of organism (v_{NB}, in units of %) was calculated using the following equation: $v_{NB} = 1 - (v_{LB} + v_{WB})$.

[f] Fraction of overlying water ventilated (m_O, fraction) was calculated using the following equation: $m_O = 1 - m_p$.

[g] The body weight-weighted average for a given composite sample was calculated by multiplying the weight of each individual fish or crab in a composite sample by the fraction of the total composite sample weight each represents and then summing these products. The weight-weighted average for a given composite sample was calculated using the following equation:

$$W_C = \sum_{i=1}^{n} \left(W_i \times \left(\frac{W_i}{\sum W_{i \ldots n}} \right) \right)$$

Where:
W_C = weight-weighted average for a given composite sample (kg)
W_i = individual fish or crab weight from a given composite sample (kg)
n = number of individual fish or crabs included in a given composite sample

DOC – dissolved organic carbon
dw – dry weight
EFDC – Environmental Fluid Dynamics [Computer] Code
IDW – inverse distance weighting
LDWG – Lower Duwamish Waterway Group
max – maximum
min – minimum
NLOC – non-lipid organic carbon
NLOM – non-lipid organic matter
OC – organic carbon
PCB – polychlorinated biphenyl
POC – particulate organic carbon
SD – standard deviation
RBTC – risk-based threshold concentration
SE – standard error
SWAC – spatially weighted average concentration
TSS – total suspended solids

According to the central limit theorem, with sufficient sample size, estimates of the mean approach a normal distribution. Parameters that had adequate site-specific empirical data or literature data with means and standard deviations were assigned a normal distribution. Triangular distributions were assumed for those parameters with more limited data. A triangular distribution requires a mode (a most likely value) and maximum and minimum values for the parameter (Warren-Hicks and Moore 1998). Both mode and mean values are presented for parameters with triangular distributions (Table D.4-1); means were only used for comparison with calibration results, which are presented as mean, maximum, and minimum statistics (Attachment 2). The mean of the triangular distribution was calculated using the following equation:

$$\text{Mean} = \frac{(\text{mode} + \text{minimum} + \text{maximum})}{3} \qquad \textbf{Equation D.4-1}$$

Values and statistical descriptors for each of the FWM parameters were derived from site-specific LDW data, data from the literature (including data from other models), and default values used in previous applications of the Arnot and Gobas model to the Great Lakes (Arnot and Gobas 2004a) or San Francisco Bay (Gobas and Arnot 2005). Default values used in previous applications of the Arnot and Gobas model were also derived from the literature. Table D.4-1 presents the parameters, estimates of relevant statistical descriptors, and the form of the probability distribution selected to represent each parameter. The remainder of this section provides the rationale for selecting individual parameter values or distributions for the biological, environmental, and chemical parameters.

D.4.1 PARAMETER VALUES FROM SITE-SPECIFIC DATA

Site-specific data from the LDW were used to derive values for eight environmental parameters: total PCB concentrations in sediment, percentage of sediment total organic carbon (TOC), total PCB concentrations in water, and five water quality parameters (total suspended solids [TSS], dissolved oxygen [DO], DOC, POC, temperature). These site-specific data were generated from various field sampling events conducted in the LDW.

D.4.1.1 Sediment concentration of total PCBs and organic carbon content

The main reason for developing the FWM was to estimate RBTCs for total PCBs in sediment[12] (as a SWAC) based on RBTCs in tissue. Tissue RBTCs were derived based on the results of the baseline risk assessments (see Section D.8 and Section 8 in the RI).

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[12] RBTCs for sediment are presented in Section 8 of the RI. RBTCs were calculated based on a best-fit estimate and a range based on acceptable output from the model defined by the model performance criterion (see Sections D.5 and D.8 for more details).

The SWAC is considered to be a decision variable[13] in the FWM because the total PCB sediment RBTC (as a SWAC) will be considered in developing PRGs in the feasibility study. Therefore, the total PCB concentration in sediment (as a SWAC) was represented by a single value (point estimate). This is consistent with the approach recommended by Morgan and Henrion (1990) for the treatment of decision variables. Representing the SWAC as a point estimate does not account for the uncertainties in the interpolation methodology or in the true exposure areas for modeled species. Effects of SWAC uncertainties on model estimates are discussed in Section D.6.2.4.

Total PCB concentrations (Aroclor sum) in sediment and OC content were derived using the baseline surface sediment database. The total PCB SWAC was calculated using inverse distance weighting (IDW) interpolations derived from 1,264 surface sediment samples. The IDW parameters (e.g., search radius, weighting factor) were selected to optimize the ability of the IDW interpolation to estimate total PCB concentrations in sediment. The IDW approach used to develop the SWAC for the FWM was described in a technical memorandum on the geographic information system (GIS) interpolation of total PCBs in LDW surface sediment (Windward 2006b). The optimized interpolation resulted in a total PCB SWAC of 380 µg/kg dry weight (dw) for the LDW[14] (Table D.4-1). In order to develop a probability distribution for sediment organic carbon, mean and standard error statistics needed to be calculated. Thiessen polygons were used for calculating sediment organic carbon because calculation of standard error statistics for Thiessen polygons uses only sample concentrations, and therefore, does not incorporate the uncertainty of the estimated concentrations of IDW cells. The sediment OC content was calculated using Thiessen polygons derived from 1,264 surface sediment samples. The spatially weighted average sediment OC content was 1.91% (Table D.4-1).

D.4.1.2 Water data

Water samples for the analysis of conventional parameters were collected in 2005 by King County as part of the Marine Ambient and Outfall Water Column Monitoring Program (Mickelson 2006). Water parameters were estimated for the FWM using these site-specific data, which included DO, temperature, TSS, DOC, and POC. POC was estimated from site-specific values for DOC and TOC in water column water. Water

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[13] Identification of a parameter as a decision variable affects how a parameter is addressed in the calibration of the model; decision variables are best presented as single values to be representative of their likely use in decision-making.

[14] To the extent possible, the same estimation methods (e.g., spatial interpolation, treatment of non-detect data, boundary definitions) used to calculate the SWAC for calibration of the FWM should be used when the model results are applied to support risk management decisions. A new SWAC (340 µg/kg dw) was generated after the calibration of the FWM using a new IDW parameterization (see Section 4.2.1 of the RI). The effects of this new SWAC on model performance are discussed in Section D.6.2.3.

samples for the analysis of PCB congeners were collected in 2005 to assist in the re-calibration of the Environmental Fluid Dynamics [Computer] Code (EFDC) model (King County 2005). Total PCB concentrations in the water column were derived from these site-specific data (Mickelson and Williston 2006) and output from the EFDC model (King County 1999 [Appendix B1]).[15] The distribution of total PCB concentrations in water was assumed to be triangular because few data were available. More data were available for distributions for all other water chemistry parameters, which were assumed to have a normal distribution.

In 2005, water samples were collected from two depths (1 m below the water surface [surface samples] and 1 m above the sediment surface [bottom samples]) at each of two stations in the LDW (King County 2005). The two stations were located just south of Harbor Island (LTKE03) and at the 16th Avenue Bridge (LTUM03) (Figure D.3-1). Samples were collected for analysis of conventional parameters (DO, temperature, TSS, DOC, and TOC) monthly from January through December, for a total of 48 samples (i.e., 24 surface samples and 24 bottom samples). Because most of the fish and crab species being modeled spend the majority of their time in more saline, deeper waters in the estuary, means and standard errors for each parameter were calculated from the 24 bottom samples (Table D.4-1).

Water samples collected by King County in August, September, November, and December in 2005 were also analyzed for PCB congeners. These months were selected with the intention of capturing two low-flow events (August and September) and two high-flow events (November and December) in the LDW. The samples were analyzed for all 209 individual PCB congeners, and total PCBs were calculated as the sum of detected congeners. Seven bottom samples were analyzed for PCBs.[16] The maximum and minimum values for the triangular distribution for the total PCB concentrations in water were based on the results of these seven bottom samples, as reported in Table 1 of the *Technical Memorandum: Duwamish River/Elliott Bay/Green River Water Column PCB Congener Survey, Transmittal of Data and Quality Assurance Documentation* (Mickelson and Williston 2006).

The mode of the distribution was defined using the output of the EFDC model, a hydrodynamic model created as part of a water quality assessment for the Duwamish River and Elliott Bay (King County 1999 [Appendix B1]). Since its application to the water quality assessment for the Duwamish River and Elliott Bay in 1999, the EFDC

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[15] The Environmental Fluid Dynamics [Computer] Code model, a hydrodynamic model, was created as part of the water quality assessment for the Duwamish River and Elliott Bay (King County 1999 [Appendix B1]) (Section D.4.1.2).

[16] The laboratory had instrument problems while analyzing the September bottom sample from the Harbor Island (LTKE03) station.

model has been recalibrated (King County in prep).[17] The recalibrated version of EFDC was used to generate output to provide mean total PCB concentrations in the water column for the LDW FWM on an LDW-wide basis and for the four modeling areas. The EFDC model generates estimates of total PCB concentrations every 4 hours in each prediction cell.[18] The EFDC model generated estimates for 1 year. The 4-hour-interval estimated concentrations were averaged within each month to derive 12 monthly average water concentrations for each prediction cell (King County in prep). Average monthly concentrations from all prediction cells in the bottom three water layers of the EFDC model were averaged to represent an average total PCB concentration in the water column for the entire LDW. This average was used to represent the mode of the triangular distribution for total PCB concentrations in the water column.

D.4.1.3 Tissue data

Site-specific tissue data for target species and benthic invertebrates, including percent lipids, percent moisture, body weights, and total PCB concentrations, were generated in a series of sampling events, including the larger datasets derived as part of the RI. Data from different sampling events identified as acceptable for use in the RI (Windward 2005j) were combined and used for the FWM (Table D.4-2). Phase 1 data for Dungeness crabs and shiner surfperch were used; Phase 1 data for other species were not used because Phase 1 composite samples were not whole-body samples (i.e., only fillet [fish] and edible meat [crabs] were available). Body weights, water content, and lipid content data were used as input values for the FWM (Table D.4-2). Total PCB concentrations were used in model calibration, as discussed in Section D.5.

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[17] Updates to the EFDC model included adding LDW slips, changing K_{OW} values for PCB partitioning, and adding and replacing sediment PCB data to reflect conditions after the Duwamish/Diagonal dredging event (King County in prep).

[18] A prediction cell is a three-dimensional space that represents a portion of the LDW in the EFDC model. Prediction cells were defined by dividing the depth, width, and length of the LDW into sections. The depth of the LDW was divided into 10 sections, the width was divided into 3 sections (with the exception of the area around Kellogg Island, which was divided into 7 sections), and the length (i.e., RM 0.0 to RM 5.3) was divided into 30 sections. A typical prediction cell was 820 ft long, 165 ft wide, and one-tenth of the depth of the LDW (which varies by tidal cycle and location).

Table D.4-2. Tissue datasets used in the FWM

Year	Species	Tissue Type	No. of Individuals per Composite Tissue Sample	No. of Composite Tissue Samples Analyzed	Parameter	Source
LDW RI						
2005	Dungeness crab	edible meat	5	3	weight, lipid content, water content (from % solids), PCB Aroclors	Windward (2006a)
		hepatopancreas	5	3		
	slender crab	edible meat	5	1		
		hepatopancreas	10	1		
	English sole	whole body	5	11		
		paired skin-on fillet and remainder[a]	5	10		
	shiner surfperch	whole body	10	22		
	Pacific staghorn sculpin	whole body	10	4		
2004	benthic invertebrates	whole body	> 100	20	weight, lipid content, PCB Aroclors, PCB congeners[b]	Windward (2005a; 2005b)
	clams	whole body	19 – 52	14	weight (from length data), lipid content, water content (from % solids), PCB Aroclors	
	Dungeness crab	edible meat	5	7	weight, lipid content, water content (from % solids), PCB Aroclors	Windward (2005c; 2005e)
		hepatopancreas	6 – 15	3		
	slender crab	edible meat	5	12		
		hepatopancreas	15 – 18	4		
	English sole	whole body	5	21		
	Pacific staghorn sculpin	whole body	7 – 10	24		
	shiner surfperch	whole body	9 – 10	24		

YEAR	SPECIES	TISSUE TYPE	NO. OF INDIVIDUALS PER COMPOSITE TISSUE SAMPLE	NO. OF COMPOSITE TISSUE SAMPLES ANALYZED	PARAMETER	SOURCE
King County CSO water quality assessment for the Duwamish River and Elliott Bay						
1997	Dungeness crab	edible meat	3	2	lipid content, water content (from % solids), PCB Aroclors	King County (1999)
		hepatopancreas	3	1		
	shiner surfperch	whole body	10	3	lipid content, PCB Aroclors	

[a] The remainder is the portion of fish that remains after the removal of the skin-on fillet. These remainder and fillet data were used to estimate whole-body English sole concentrations as specified in the quality assurance project plan (Windward 2005i) and the data report (Windward 2006a).

[b] PCB congener data were used in the derivation of log K_{OW} values.

CSO – combined sewer overflow

FWM – food web model

PCB – polychlorinated biphenyl

D.4.1.3.1 Lipid and water content

Tissue composite samples collected from the LDW were used to determine mean and standard error estimates for lipid content (v_{LB}) and water content (v_{WB}) for fish, crabs, clams, and benthic invertebrates. Water content for benthic invertebrates and lipid content for juvenile fish were derived from the literature (Table D.4-1). Water content (v_{WB}) was calculated from total solids using the following equation:

$$v_{WB} = (100 - \text{total solids}) \qquad \textbf{Equation D.4-2}$$

Ten of the twenty-one English sole samples in the 2005 tissue dataset were paired English sole fillet and remainder samples. Whole-body lipid content for each of these English sole whole-body composite samples was calculated using the following equation:

$$v_{LWB} = \left(\left(\frac{W_F}{W_F + W_R}\right) \times v_{LF}\right) + \left(\left(\frac{W_R}{W_F + W_R}\right) \times v_{LR}\right) \qquad \textbf{Equation D.4-3}$$

Where:

v_{LWB} = lipid content of calculated whole-body composite sample (%)
v_{LF} = lipid content of fillet composite sample (%)
v_{LR} = lipid content of remainder composite sample (%)
W_F = weight of fillet composite sample (kg)
W_R = weight of remainder composite sample (kg)

Percent total solids content used to calculate water content for each of these English sole whole-body composite samples was calculated using the following equation:

$$v_{TSWB} = \left(\left(\frac{W_F}{W_F + W_R}\right) \times v_{TSF}\right) + \left(\left(\frac{W_R}{W_F + W_R}\right) \times v_{TSR}\right) \qquad \textbf{Equation D.4-4}$$

Where:

v_{TSWB} = total solids content of calculated whole-body composite sample (%)
v_{TSF} = total solids content of fillet composite sample (%)
v_{TSR} = total solids content of remainder composite sample (%)
W_F = weight of fillet composite sample (kg)
W_R = weight of remainder composite sample (kg)

Mean and standard error estimates of whole-body lipid and water contents were calculated for Dungeness and slender crabs based on a combination of edible meat composite samples with corresponding hepatopancreas composite samples from the same crabs. Whole-body percentages of lipid or moisture content for Dungeness and slender crabs were estimated using the following equation:

$$v_{wb} = (v_h \times F_h) + (v_{em} \times F_{em})$$ **Equation D.4-5**

Where:

v_{wb} = lipid or moisture content in whole-body crabs (%)
v_h = lipid or moisture content in hepatopancreas of crabs (%)
v_{em} = lipid or moisture content in edible meat of crabs (%)
F_h = fraction of whole-body weight consisting of hepatopancreas weight
F_{em} = fraction of whole-body weight consisting of edible meat weight

The hepatopancreas and edible meat fractions were estimated to be 0.31 and 0.69, respectively, based on the ratio of wet masses of these tissues in a 16.6-cm Dungeness crab[19] dissected at Windward Environmental LLC.[20] Similar relative masses for edible meat and hepatopancreas were presented in Atar and Secer (2003).

Juvenile fish in the FWM represent small fish that would serve as prey for fish and crab species, such as Pacific staghorn sculpin and crabs. Juvenile shiner surfperch and juvenile starry flounder were the most abundant small fish (< 100 mm) captured in trawls during Phase 2 sampling events conducted in late summer (Windward 2005c, 2006a). Juvenile shiner surfperch and juvenile starry flounder represented 54 and 30%, respectively, of the non-target fish catch in 2004,[21] and 40 and 42%, respectively, in the 2005 sampling event. Thus, these species are likely prey for Pacific staghorn sculpin and crabs in the LDW.

Because they were not target fish during 2004 and 2005 sampling events, tissue data for juvenile starry flounder and juvenile shiner surfperch were not available (with the exception of limited weight data). Therefore, estimates for juvenile fish mean lipid content were calculated using Phase 2 adult shiner surfperch and adult English sole data (Table D.4-1). Because juvenile fish lipids are approximately 50% of adult lipid values (Gobas and Arnot 2005; Robards et al. 1999), mean lipid content for juvenile fish (2.5%) was estimated as 50% of the combined mean lipid content of adult shiner surfperch and adult English sole. The selection of this value was supported by the fact that 2.5% was both the median and the mode of mean lipid content values reported for 19 juvenile and small fish species eaten by salmon in the Bering Sea (Nomura and Davis 2005). Juvenile fish water content was based on Phase 2 adult shiner surfperch data.[22]

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[19] Maximum width of the shell from tip of spine to tip of spine.

[20] A live Dungeness crab was purchased and dissected at Windward to determine the relative weights of edible meat and hepatopancreas. The weights of the crab's edible meat and hepatopancreas were 158 g and 49 g, respectively.

[21] Non-target fish were individual fish not retained for tissue analysis either because they were too small or the wrong species. Each non-target fish captured was identified to species, measured (length), counted, and then returned to the LDW.

[22] Lipid content values for juvenile fish were based on the literature (Table D.4-1).

D.4.1.3.2 Body weights

Mean and standard error estimates for fish and crab weights (W_B) were calculated based on the average whole-body weight of fish and crabs included in composite samples (W_C) collected in 2004 and 2005. The average whole-body weight for each fish or crab composite sample was calculated as a body weight-weighted average to account for the fact that composite samples included fish (or crabs) with different weights (kg), and thus some fish (or crabs) contributed more tissue mass (kg) to the composite sample than others. The body weight-weighted average for a given composite sample was calculated using Equation D.4-5.

$$W_C = \sum_{i=1}^{n} \left(W_i \times \left(\frac{W_i}{\sum W_{i\ldots n}} \right) \right)$$

Equation D.4-6

Where:

W_C = body weight-weighted average for a given composite sample (kg)
W_i = individual fish or crab weight from a given composite sample (kg)
n = number of individual fish or crabs included in a given composite sample

Mean weights of all composite samples were then calculated using the following equation:

$$W_B = \frac{\sum W_{C\,(i\ldots n)}}{n}$$

Equation D.4-7

Where:

W_B = mean weight for a given species of fish or crab (weight of biota) (kg)
W_C = body weight-weighted average for a given composite sample (kg)
n = number of fish or crab composite samples

Because the benthic invertebrate compartment was defined as a species assemblage, an estimate of the mean body weight across species (or other taxonomic groups) was needed to define mean and standard error values for benthic invertebrates. Estimates of benthic invertebrate body weights in samples analyzed for PCBs were based on abundances of major taxonomic groups (i.e., annelids, crustaceans, mollusks, and miscellaneous taxa) of benthic invertebrates in taxonomy samples collected in 2004 (Windward 2005d) combined with weight data of major taxonomic groups from samples analyzed for PCBs (Windward 2005b).

To estimate individual clam weights in the LDW, a regression relationship was developed between length and weight data for 609 individual *Mya arenaria* clams from the 2003 LDW intertidal and catch-per-unit effort surveys[23] (Windward 2004). This regression was needed because lengths, but not weights, were determined in the 2004 sampling event for clams; clams collected in 2004 were analyzed for PCBs. Average clam weight estimates for the 14 clam composite samples collected in 2004 were calculated using 2004 mean length data from those samples (Windward 2005b) and the following regression equation developed from the 2003 data:

$$W_{Clam} = 0.106 \times (L_{Clam})^{2.9974}$$ **Equation D.4-8**

Where:

W_{Clam} = weight of clam (g)
L_{Clam} = length of clam (cm)

Average and standard error estimates of clam weights were calculated from the 14 mean composite sample weights calculated using Equation D.4-7.

D.4.1.4 Estimation of log K_{OW} for PCBs

The log K_{OW} for PCBs was estimated using site-specific concentrations of PCB congeners in benthic invertebrate tissue and log K_{OW} values for individual PCB congeners from the literature. A concentration-weighted average log K_{OW} was calculated using Equation D.4-8 for the eight benthic invertebrate tissue samples for which all 209 individual PCB congeners were analyzed (Windward 2005a) (Table D.4--3). PCB congener-specific log K_{OW}s were taken from Hawker and Connell (1988).

$$\text{Average } \log K_{OW} = \frac{\sum_{i=1}^{n} C_i \times \log K_{OWi}}{\sum C_i}$$ **Equation D.4-9**

Where:

C_i = Detected concentration of PCB congener i (µg/kg ww)
Log K_{OWi} = log K_{OW} of PCB congener i (L/kg)
n = number of detected PCB congeners

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[23] The regression was developed using *Mya arenaria* data. Clam tissue samples collected from the LDW consisted mostly of *Mya arenaria*. A few composite samples had 2 to 3 *Macoma nasuta* individuals compared to 17 to 19 *Mya arenaria*. All other composite samples were composed only of *Mya arenaria*.

Table D.4-3. Weighted log K_{OW}s for benthic invertebrate composite samples

SAMPLE ID	WEIGHTED LOG K_{OW} (L/kg)
LDW-B1b-T	6.64
LDW-B2a-T	6.55
LDW-B3b-T	6.47
LDW-B4b-T	6.54
LDW-B5a-T	6.42
LDW-B8a-T	6.87
LDW-B9b-T	6.51
LDW-B10a-T	6.53
Mean	**6.57**
Standard error	**0.05**

ID – identification

K_{ow} – octanol water partition coefficient

The mean and standard error of the eight weighted log K_{OW} values were used to define the normal distribution for log K_{OW}.

D.4.2 PARAMETER VALUES FROM LITERATURE DATA

Literature sources were used to derive water and lipid content for phytoplankton, weight and water and lipid content for zooplankton, water content for benthic invertebrates, and lipid content for juvenile fish. In addition, literature sources were used to derive values for fraction of porewater ventilated for all species, diets for all species, and densities for lipids and water (see Table D.4-1 for a description of methods and sources). Methods for determining values for these parameters are discussed below.

D.4.2.1 Values for organism lipid, water, and NLOC content and weight

Phytoplankton water and lipid content were derived from one study that reported lipid and NLOC content data for phytoplankton and macroalgae in False Creek, Burrard Inlet, Vancouver, British Columbia (Mackintosh et al. 2004). Data for green algae, brown algae, and phytoplankton were used because the phytoplankton/algae compartment in the model represents both phytoplankton and macroalgae. In Mackintosh et al. (2004), green and brown macroalgae samples were collected by hand, and plankton samples were collected using a 236-µm plankton tow net. The plankton tow net collected both phytoplankton and microzooplankton. Because microzooplankton are the same size as phytoplankton (20 to 200 µm), they are normally included in bulk analyses of phytoplankton as part of a constituent analysis (Olson 2006). Therefore, most marine FWMs include microzooplankton as part of their phytoplankton compartment (Olson 2006).

Mackintosh et al. (2004) reported lipid and NLOC content data for these species assemblages. Because phytoplankton and algae have low lipid concentrations, NLOC is an important organic chemical storage phase in these organisms. NLOC, which makes up a fraction of NLOM, is used rather than NLOM for phytoplankton/algae because it is a better predictor of organic chemical content in phytoplankton (Skoglund and Swackhamer 1999). Water content for phytoplankton was calculated from NLOC using the following equation:

$$\text{water content} = (100 - \text{NLOC}) \qquad \textbf{Equation D.4-10}$$

Where:

NLOC = non-lipid organic carbon content (%)

Mean and standard error values of water and lipid content percentages were calculated across green algae, brown algae, and plankton (Table D.4-1).

Zooplankton lipid and water content were derived from a study in Maizura Bay, Japan (Kuroshima et al. 1987). In this study, five 1-month average values for lipid and water content were reported. Water content for each monthly average was used to convert lipid content from dry to wet weight.

Zooplankton body weights were derived from a study in Budd Inlet, Puget Sound, Washington (Giles and Cordell 1998). Twenty-one zooplankton samples were collected from six stations over 12 months. Zooplankton samples contained crustaceans, cnidarians, larvaceans, and polychaetes. Dry weights were converted to wet weights assuming 90% water content.

Benthic invertebrate water content was derived from the literature. Mean and range data for the water content of bivalves, isopods, amphipods, and cladocerans reported in an Oak Ridge National Laboratory publication (Sample et al. 1997) were used to derive mode, maximum, and minimum statistics of a triangular distribution for benthic invertebrate water content.

The standard deviation value for juvenile fish lipids[24] (0.6%) was derived from a study of salmon prey fish in the Bering Sea (Nomura and Davis 2005) as the standard error of lipid content for 19 juvenile and small fish species (0.3%) multiplied by a factor of 2. The standard error was multiplied by 2 to account for variation in lipid values within species. In the Bering Sea study, samples were collected during the summer and fall of a single year and thus did not capture potential variation throughout the entire year or from year to year.

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[24] Mean lipid content for juvenile fish (2.5%) was estimated as 50% of the combined mean lipid content of adult shiner surfperch and English sole (Section D.4.1.3.1).

D.4.2.2 Diets

Simplifying assumptions must be made when estimating diets of aquatic species because ecosystems are complex, dynamic environments that cannot be fully characterized in a quantitative manner without a high level of uncertainty. Ecology, behavior, feeding observation studies, and stomach content analyses were considered in the creation of the simplified uptake routes and plausible dietary scenarios were developed to reflect average diets. Stomach content analyses were the dominant sources used in the creation of dietary scenarios.

Different dietary scenarios were created to represent the variability and uncertainty in the diets of the species being modeled (Windward 2005h). To support the probabilistic approach used to calibrate the FWM, it was necessary to develop probability distributions for each dietary item for each species. Triangular distributions were assumed for each dietary item with mode, maximum, and minimum values derived from the dietary scenarios.[25]

Dietary scenarios were established for all species except phytoplankton and zooplankton. Although some phytoplankton species consume other plankton or detritus (e.g., mixotrophic dinoflagellates), the phytoplankton/algae compartment was assumed to represent only photosynthesizing organisms. The diets of zooplankton were assumed to consist entirely of phytoplankton.

D.4.2.2.1 Fish and crab dietary scenarios

Three dietary scenarios were created for each target fish and crab species, with the exception of Dungeness crab, for which four dietary scenarios were created. Diets of fish and crabs are difficult to characterize because they likely vary by location, season, age, and size class. Fish and crab diets are also difficult to quantify in terms of mass or volume fractions because stomach content analyses favor items that are digested more slowly. In addition, certain feeding habits, such as scavenging or extensive mastication of food items, make food-item species identification difficult.

In the FWM, there are four compartments available to serve as dietary items for fish or crabs: phytoplankton/algae, zooplankton, benthic invertebrates, and juvenile fish. These compartments have been populated in the FWM as species assemblages that represent different trophic levels and exposure environments (e.g., pelagic vs. benthic). Zooplankton represent herbivorous invertebrates exposed to chemicals in the water column.[26] Benthic invertebrates represent suspension- and deposit-feeding invertebrates that are exposed to chemicals in both the water column and the

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[25] Dietary triangular distributions for clams were derived from the literature and best professional judgment (see Table D.4-7).

[26] Weight, lipid and water content, and dietary absorption efficiencies for the zooplankton compartment were derived solely from literature data for macrozooplankton (copepods, crustaceans, cnidarians, larvaceans, and polychaetes).

sediment. Sediment is also a dietary item for fish or crabs. In order to create dietary scenarios for each fish and crab species, it was necessary to assign each species or organism type identified in stomach content studies to one of the four compartments above or to sediment. Fish and crabs consume a diversity of prey items, some of which were not represented in the above compartments (e.g., juvenile crabs and shrimp). As discussed below, shrimp and juvenile crabs were represented by benthic invertebrates or zooplankton in the dietary scenarios.

Three dietary scenarios were created for fish species and slender crab, which are all opportunistic feeders. Four dietary scenarios were created for Dungeness crabs (Table D.4-4). In general, Dietary Scenarios 1 and 2 were statistical estimates of the organisms' diets based on stomach content analyses presented in the literature. Dietary Scenario 2 was similar to Dietary Scenario 1, except that juvenile crab or shrimp prey items in the dietary studies were represented by zooplankton instead of benthic invertebrates. Zooplankton are a reasonable surrogate for juvenile crabs and shrimp because zooplankton, juvenile crabs, and shrimp are primarily exposed to PCBs in water, unlike benthic invertebrates, which are in closer association with the sediment. Dietary Scenario 3 was created from studies that considered organism ecology and behavior in addition to stomach content analyses. Dietary Scenario 3 was the only scenario that included sediment as a fraction of the diet; sediment was assumed to be 10% of the diet of all fish and crab species for this scenario. Dungeness crab was the only species with a fourth dietary scenario. This scenario was based on an additional literature source that quantified stomach contents using a different metric (Gotshall 1978). These dietary scenarios were used to develop probability distributions applied in the FWM, as discussed in Section D.4.2.2.3.

Table D.4-4. Fraction of prey items consumed by fish and crab species in the four dietary scenarios

Species	Dietary Surrogate	Fraction of Diet[a]				Sources
		Scenario 1[b]	Scenario 2[c]	Scenario 3[d]	Scenario 4[b]	
Juvenile fish	zooplankton	0.07	0.17	0.05	na	Fresh et al. (1979); Miller et al. (1977); Wingert et al. (1979)
	benthic invertebrates	0.93	0.83	0.85	na	
	sediment	0	0	0.10	na	
Slender crab	zooplankton	0	0.12	0	na	Bernard (1979)
	benthic invertebrates	0.99	0.87	0.90	na	
	juvenile fish	0.01	0.01	0	na	
	sediment	0	0	0.10	na	

SPECIES	DIETARY SURROGATE	FRACTION OF DIET[a]				SOURCES
		SCENARIO 1[b]	SCENARIO 2[c]	SCENARIO 3[d]	SCENARIO 4[b]	
Dungeness crab	zooplankton	0	0.48	0	0	Stevens et al. (1982) for Scenarios 1 and 2; Gotshall (1978) for Scenario 4
	benthic invertebrates	0.63	0.16	0.75	0.75	
	juvenile fish	0.37	0.36	0.15	0.25	
	sediment	0	0	0.10	0	
Pacific staghorn sculpin	zooplankton	0	0.37	0.25	na	Fresh et al. (1979); Miller et al. (1977); Wingert et al. (1979)
	benthic invertebrates	0.56	0.19	0.50	na	
	fish	0.44	0.44	0.15	na	
	sediment	0	0	0.10	na	
Shiner surfperch	zooplankton	0.14	0.21	0.10	na	Fresh et al. (1979); Miller et al. (1977); Wingert et al. (1979)
	benthic invertebrates	0.86	0.79	0.80	na	
	sediment	0	0	0.10	na	
English sole	phytoplankton/ algae	0.08	0.07	0	na	Fresh et al. (1979); Wingert et al. (1979)
	zooplankton	0	0.05	0	na	
	benthic invertebrates	0.92	0.88	0.90	na	
	sediment	0	0	0.10	na	

[a] Unidentifiable prey items were not included in calculations (fractions were normalized without unidentified items).

[b] Crab and shrimp prey were assigned to the benthic invertebrate compartment.

[c] Crab and shrimp prey were assigned to the zooplankton compartment.

[d] Ten percent incidental sediment consumption was assumed for all fish and crab species. For Pacific staghorn sculpin, crab and shrimp prey were assigned to the zooplankton compartment.

na – not available; no scenario investigated

D.4.2.2.2 Benthic invertebrate dietary scenarios

Benthic invertebrate communities in the LDW are composed of many species from numerous phyla within multiple feeding guilds. The 20 benthic invertebrate composite tissue samples collected from the LDW in 2004 consisted primarily of annelids (polychaetes), crustaceans (e.g., amphipods, isopods, cumaceans, copepods, decapods), and small mollusks (e.g., bivalves [*Macoma* sp.] and gastropods). Miscellaneous invertebrates included flatworms (Platyhelminthes), cnidarians, nematodes, and nemertines. Two dietary scenarios were created for benthic invertebrates to encompass the diversity of feeding modes in this multi-species compartment (Table D.4-5).

Table D.4-5. Fraction of prey items consumed by benthic invertebrates under the two dietary scenarios

Benthic Invertebrate Dietary Item	Dietary Fraction			
	Dietary Scenario 1		Dietary Scenario 2	
	Mean	Range	Mean	Range
Phytoplankton/algae	0.11	0.01 – 0.16	0.11	0.01 – 0.16
Zooplankton	0.05	0.01 – 0.07	0.12	0.02 – 0.17
Sediment	0.84	0.77 – 0.99	0.77	0.67 – 0.97

Benthic invertebrate dietary scenarios were established by estimating percent feeding guilds in benthic invertebrate samples and then assigning percentages of each dietary item to feeding guilds. Average percent feeding guilds (deposit feeders or detritivores, suspension feeders, and carnivores) were estimated for all LDW subtidal[27] benthic samples based on the literature[28] and information on major taxonomic groups in each sample (Windward 2005b). Each feeding guild was assigned percentages of benthic invertebrate dietary items, including phytoplankton, zooplankton, and sediment. Two dietary scenarios were developed by having two different sets of assumptions about what dietary items were consumed by carnivores.

Dietary Scenario 1 was constructed assuming that carnivores consumed 100% sediment. Dietary Scenario 2 was constructed assuming that carnivores consumed 50% zooplankton and 50% sediment. Because the FWM does not allow for a fraction of a modeled species diet coming from their own model compartment and because some of the species in the benthic invertebrate samples are carnivores that eat other species also in the benthic invertebrate samples, it was necessary to assign a surrogate prey item to represent “cannibalism” within benthic invertebrates. Because total PCB concentrations in sediment were more similar to those in benthic invertebrates than in plankton or juvenile fish and because benthic invertebrates are in close association with sediment, sediment was used as a surrogate for benthic invertebrate prey consumed by benthic invertebrate carnivores. Zooplankton were used as a dietary item for carnivores to represent prey items exposed primarily to the water column. Both dietary scenarios assumed that suspension feeders consumed 30% zooplankton and 70% phytoplankton/algae and that deposit feeders consumed 100% sediment. Even though suspension feeders and deposit feeders consume a significant amount of

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[27] Subtidal samples were used because is was necessary to compare species composition in samples collected for chemical analysis of tissue and samples collected for taxonomy, and sampling procedures were consistent for tissue and taxonomy samples in the subtidal.

[28] Various sources were used to determine feeding types of invertebrates identified in the LDW benthic invertebrate samples (Barnes and Mann 1980; California Academy of Sciences 2002; Cruz-Rivera and Hay 2001; Fauchald and Jumars 1979; Harbo 2001; Jensen 1995; Kozloff 1983; MarLIN 2002, 2004, 2005; Museum Victoria 1996; Palaeos 2004; Ricketts et al. 1985; Shimek 2003, 2004; Word 1990).

detritus, a "detritus" compartment was not modeled because there were insufficient data to generate values for such a compartment. Surrogate prey items for detritus included both sediment (benthic detritus) and phytoplankton (water column detritus).

D.4.2.2.3 Probability distributions for diets

To calibrate the FWM using a probabilistic approach, probability distributions were developed for diets. Triangular distributions were assumed for diets because there were limited data (Table D.4-6). Because the dietary scenarios for each species were created using different assumptions, they represented a range of variability and uncertainty in the diets. Therefore, dietary scenarios served as the source of information from which dietary probability distributions were developed. Input on the relative fractions of phytoplankton and/or zooplankton consumed by benthic invertebrates, clams, juvenile fish, and shiner surfperch from NOAA and EPA also contributed to the development of dietary distributions (Field 2006). Mean values are presented in addition to modes (Table D.4-6) to facilitate comparison with post calibrated values, which are presented as mean, maximum, and minimum values (Attachment 2).

Table D.4-6. Summary of triangular dietary distributions for LDW food web model

	Dietary Item																			
	Sediment				Phytoplankton				Zooplankton				Benthic Invertebrates				Juvenile Fish			
Species	Min	Max	Mode	Mean	Min	Max	Mode	Mean	Min	Max	Mode	Mean	Min	Max	Mode	Mean	Min	Max	Mode	Mean
Benthic invertebrates	0.62	0.93	0.79	0.78	0.06	0.21	0.16	0.14	0.01	0.17	0.05	0.08	0	0	0	0	0	0	0	0
Clam	0.30	0.60	0.45	0.45	0.40	0.60	0.50	0.50	0	0.10	0.05	0.05	0	0	0	0	0	0	0	0
Juvenile fish	0	0.01	0	0	0	0	0	0	0.3	0.87	0.50	0.56	0.13	0.70	0.50	0.44	0	0	0	0
Slender crab	0	0.05	0	0.02	0	0	0	0	0	0.12	0.12	0.08	0.86	0.99	0.87	0.91	0.01	0.01	0.01	0.01
Dungeness crab	0	0.05	0	0.02	0	0	0	0	0	0.68	0.48	0.39	0.16	0.84	0.16	0.39	0.16	0.58	0.36	0.37
Pacific staghorn sculpin	0	0.05	0	0.02	0	0	0	0	0	0.50	0.25	0.25	0.04	0.83	0.50	0.46	0.17	0.68	0.25	0.37
Shiner surfperch	0	0.01	0.01	0.01	0	0	0	0	0.15	0.72	0.35	0.41	0.28	0.85	0.64	0.59	0	0	0	0
English sole	0	0.10	0.01	0.04	0.05	0.10	0.06	0.07	0	0.09	0.05	0.05	0.86	0.90	0.88	0.88	0	0	0	0

LDW – Lower Duwamish Waterway

D.4.2.3 Default values

For several parameters, literature-derived values from previous applications of the Arnot and Gobas model (Arnot and Gobas 2004a; Gobas and Arnot 2005) were used to estimate values and statistical descriptors. There were insufficient site-specific data and limited new literature data to derive new values or probability distributions for these parameters.

Point estimate values for eight parameters were taken directly from applications of the model for the Great Lakes (Arnot and Gobas 2004a) and San Francisco Bay (Gobas and Arnot 2005). The eight parameters were the filter feeder particle scavenging efficiency (σ), the disequilibrium factors for DOC and POC partitioning (D_{DOC}, D_{POC}), the proportionality constants that quantify the similarity in phase partitioning of DOC and POC relative to that of octanol (α_{DOC}, α_{POC}), the proportionality constant that expresses the sorption capacity of NLOC relative to octanol (β_{OC}), the dietary absorption efficiency of water (ε_W), and the rate constant for the growth of phytoplankton/algae (k_G).

Values for statistical descriptors (e.g., mean and standard deviation) of probability distributions for the proportionality constant expressing the sorption capacity of NLOM relative to that of octanol, fractions of porewater and overlying water ventilated by all species (except plankton), dietary absorption efficiencies of lipids and NLOM, as well as values for resistance to chemical uptake through aqueous and organic phases for phytoplankton/algae were also derived from these previous applications of the Arnot and Gobas FWM.

D.5 Calibration

Calibration is a process of deriving a set of FWM parameter values that optimizes the ability of the FWM to estimate total PCB concentrations in tissues that match empirical data as closely as possible. This process is important because proper calibration should improve the FWM's performance in estimating RBTCs in sediment (Section D.8). However, improving the ability of the model to match empirical data does not necessarily mean that the "true" values for each parameter have been identified. Numerous combinations of parameters can result in similar estimates.

The FWM is a steady-state model (i.e., it assumes that concentrations do not change as a function of time). Thus, it does not model perturbations in the system (e.g., seasonal variations, inter-annual variations, or short-term disturbances). The FWM estimates average conditions that are assumed to be stable as a function of time. Therefore, the empirical dataset selected for the calibration process is important; it should represent "average" conditions expected in the LDW.

Empirical data (i.e., total PCB concentrations in tissues collected from the LDW) are available from the late 1990s, 2004, 2005, and 2006. The largest datasets were collected in 2004 and 2005; 2006 data were not used for FWM calibration because they were not available at that time and were only collected from Area T1 for two fish species. A number of large dredging events occurred in and around the LDW in 2004 (see Map-4-7a-d in RI). Though it is not known if these dredging events resulted in a significant increase in exposure of aquatic organisms in the LDW to PCBs, total PCB concentrations in tissue samples collected in 2004 (months after the dredging events) were significantly higher than those in samples collected in the late 1990s, 2005, and 2006 (see Section 4.2.1.4.1 in the RI). This result is consistent with tissue data collected in other areas where dredging has occurred (Stern et al. 2005; Stern and Patmont 2006; Patmont and Palermo 2007).

Thus, because it is not known if the 2004 tissue data were higher as a result of the short-term dredging event (and thus should be excluded from the empirical dataset) or whether they are a reflection of natural variability (and thus should in included in the average), the FWM was calibrated using two different empirical datasets. Calibration 1 included tissue data collected in the late 1990s, 2004, and 2005. Calibration 2 included tissue data collected only in the late 1990s and 2005 (i.e., the 2004 tissue data were excluded).

D.5.1 METHODS

The FWM was calibrated probabilistically in order to systematically explore the plausible combinations of parameter values and their ability to estimate empirical data. The calibration process involved three steps:

- Step 1. Monte Carlo simulation
- Step 2. Model performance filtering
- Step 3. Identification of the best-fit parameter set

Each step is discussed in detail in the following subsections.

D.5.1.1 Monte Carlo simulation

The FWM was run probabilistically in Excel® with Crystal Ball® software. For each of the thousands of Monte Carlo simulations, parameter values were randomly selected from the parameter probability distributions described in Section D.4. The resulting set of parameter values selected in each model run is termed a "parameter set."[29] Each parameter set yielded an estimate of total PCB concentrations in tissues of the modeled species.

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[29] Point estimates were assigned for some parameters so that the same value was selected for that parameter for each Monte Carlo simulation.

During the Monte Carlo simulation, the probability distributions of dietary items for each species were treated as independent random variables, which meant that the sum of the dietary fractions had to be normalized (because dietary fractions must sum to 1). Dietary fractions for each species in the FWM were normalized by dividing each dietary fraction by the sum of all dietary fractions for a given species. Treating the dietary fractions as independent random variables greatly simplified the Monte Carlo simulation. However, as a consequence, the normalized dietary fractions for some parameter sets fell outside of their specified probability distributions. The easiest way to address this issue was to apply a diet filter. Therefore, the last action taken in the Monte Carlo simulation step was to discard parameter sets if any of the normalized dietary fractions fell outside of their assigned ranges as defined in Table D.4-6. This step was a bookkeeping step, the only effect of which was to correct for an artifact of the way dietary fractions were defined.

D.5.1.2 Model performance filtering

The model performance filter step consisted of comparing estimated total PCB concentrations in tissues with available empirical data (i.e., total PCB concentrations detected in species collected in the LDW). The parameter sets that resulted in estimated concentrations that were outside specified bounds for empirical data (i.e., a difference greater than a factor of 2) were rejected. The remaining parameter sets were retained for use in the next step (i.e., identification of the best-fit parameter set) and also in the sensitivity and uncertainty analyses.

As discussed in Section D.5, two calibrations were performed because of the uncertainty in selecting the most appropriate empirical dataset. Therefore, model performance was evaluated by comparing FWM-estimated concentrations in tissues to two different empirical datasets. Mean and range information from these datasets is presented in Tables D.5-1 and D.5-2 for Calibrations 1 and 2, respectively.

Table D.5-1. Empirical dataset for Calibration 1: Total PCB concentrations detected in fish, crab, and benthic invertebrate tissues collected in Phase 1 (late 1990s) and Phase 2 (2004 and 2005)

SPECIES	TOTAL PCB CONCENTRATION IN TISSUES (µg/kg ww)		NO. OF COMPOSITE SAMPLES	NOTES	DATASET SUMMARY
	MEAN	RANGE			
Benthic invertebrates	200	na	20	Mean was estimated using surface sediment total PCB SWAC of 380 µg/kg dw for the entire LDW and the following tissue-sediment regression equation (described further in Attachment 1): $\log10[C_{BI}] = 1.40 + 0.35 \times \log10[C_S]$.	Phase 2 (2004) benthic invertebrate tissue data and co-located sediment data used for the tissue-sediment regression (n = 20), and Phase 1 and Phase 2 sediment data used for the total PCB SWAC
Slender crab	670	250 – 838	13	combined edible meat and hepatopancreas tissue samples[a]	Phase 2 (2004, n = 12) and Phase 2 (2005, n = 1)
Dungeness crab	1,100	420 – 1,900	12	combined edible meat and hepatopancreas tissue samples[a]	Phase 1 (n = 2), Phase 2 (2004, n = 7), and Phase 2 (2005, n = 3)
Pacific staghorn sculpin	900	430 – 2,800	28	whole-body tissue samples	Phase 2 (2004, n = 24) and Phase 2 (2005, n = 4)
Shiner surfperch	1,800[b]	350 – 18,400	49	whole-body tissue samples	Phase 1 (n = 3), Phase 2 (2004, n = 24), and Phase 2 (2005, n = 22)
English sole	2,300	610 – 4,700	42	whole-body tissue samples[c]	Phase 2 (2004, n = 21) and Phase 2 (2005, n = 21)

[a] Concentrations in whole-body crab tissue (i.e., edible meat plus hepatopancreas) were calculated for each edible meat sample assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004.

[b] Mean would be 1,400 µg/kg ww if the 18,400-µg/kg ww sample in Area M2 were excluded.

[c] Ten English sole samples (three each from Areas M1, M2, and M3 and one from Area M4) from 2005 were “calculated whole-body” from paired fillet and remainder samples.

dw – dry weight
LDW – Lower Duwamish Waterway
na – not applicable
PCB – polychlorinated biphenyl
ww – wet weight

Table D.5-2. Empirical dataset for Calibration 2: Total PCB concentrations detected in fish and crab tissues collected in Phase 1 (late 1990s) and Phase 2 (2005 only)

SPECIES	TOTAL PCB CONCENTRATIONS IN TISSUES (µg/kg ww)		NO. OF COMPOSITE SAMPLES	NOTES	DATASET SUMMARY
	MEAN	RANGE			
Slender crab	250	na	1	combined edible meat and hepatopancreas tissue samples[a]	Phase 2 (2005)
Dungeness crab	510	420 – 650	5	combined edible meat and hepatopancreas tissue samples[a]	Phase 1 (n = 2) and Phase 2 (2005, n = 3)
Pacific staghorn sculpin	590	430 – 720	4	whole-body tissue samples	Phase 2 (2005)
Shiner surfperch	1,000	350 – 2,400	25	whole-body tissue samples	Phase 1 (n = 3) and Phase 2 (2005, n = 22)
English sole	1,600	610 – 2,400	21	whole-body tissue samples[b]	Phase 2 (2005)

[a] Concentrations in whole-body crab tissue (i.e., edible meat plus hepatopancreas) were calculated for each edible meat sample assuming 69% (by weight) edible meat and 31% hepatopancreas based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004.

[b] Ten English sole samples (three each from Areas M1, M2, and M3 and one from Area M4) from 2005 were "calculated whole-body" from paired fillet and remainder samples.

LDW – Lower Duwamish Waterway

PCB – polychlorinated biphenyl

na – not applicable

ww – wet weight

Model estimates were compared to mean concentrations of total PCBs in composite samples of fish and crabs collected from the LDW. Mean total PCB tissue concentrations were used rather than single composite sample values because the biological compartments in the FWM were assumed to represent populations, not individual organisms.

Benthic invertebrate tissue data were not used directly in the calibration because these data were not collected to provide a representative sampling of total PCB concentrations in benthic invertebrate tissue throughout the LDW. Instead, benthic invertebrate sampling was designed to sample a range of total PCB concentrations in sediment from various locations throughout the LDW. The data were collected in this manner to explore the relationship between total PCB concentrations in tissue and sediment through the use of a regression, so that total PCB concentrations in benthic invertebrate tissues could be estimated from an average total PCB concentration in sediment. A tissue-sediment regression (Equation D.5-1) was used to estimate a single total PCB concentration in benthic invertebrate tissues based on a SWAC of 380 µg/kg dw (the LDW-wide spatially weighted average total PCBs concentration in sediment).

$$\log10[C_{BI}] = 1.40 + 0.35 \times \log10[C_S] \qquad \textbf{Equation D.5-1}$$

Where:

C_{BI} = total PCB concentration in benthic invertebrate tissue (µg/kg ww)
C_S = total PCB concentration in sediment (µg/kg dw)

Estimated total PCB concentrations in benthic invertebrates were compared to the single concentration of total PCBs in benthic invertebrates generated by the tissue-sediment regression. The benthic invertebrate tissue-sediment regression was based on organisms that were collected in 2004, so the calibration dataset for Calibration 2 (which excluded 2004 tissue data) did not include calibration to the benthic invertebrate empirical dataset.

Clams were included as target species in the FWM to support calculations of sediment RBTCs for human health consumption scenarios (see Section D.8). The FWM was not calibrated for clams because clams are present only in intertidal areas in the LDW with suitable habitat. The best-fit parameter set for Calibration 2 was applied at specific intertidal locations to assess the ability of the model to estimate total PCB concentrations in clam tissue (Section D.7). No empirical data existed for phytoplankton, zooplankton, or juvenile fish, so the model was not calibrated for those species.

A species predictive accuracy factor (SPAF) was selected as the metric for model performance evaluation (i.e., to quantitatively compare model estimates and empirical data). The SPAF is the ratio of estimated to empirical total PCB concentrations in tissue for a given species, or the inverse of that ratio, whichever is greater (i.e., the SPAF will always be a number greater than 1). Accordingly, if the estimated concentration was greater than the empirical concentration, Equation D.5-2 was used to calculate the SPAF:

$$\text{SPAF} = \frac{C_M}{C_E}$$ **Equation D.5-2**

Where:

C_M = model-estimated total PCB concentration in tissue (µg/kg ww)
C_E = mean empirical total PCB concentration in tissue (µg/kg ww)

If the estimated concentration was less than empirical concentration, the reciprocal ratio (Equation D.5-3) was used:

$$\text{SPAF} = \frac{C_E}{C_M}$$ **Equation D.5-3**

A perfect correlation between model-estimated and mean empirical concentrations would result in a SPAF of 1. Any difference between the model-estimated and mean empirical tissue concentrations would result in a SPAF > 1.

To meet the selected model performance criterion, SPAFs for all species had to be $\leq$ 2. If the SPAF of any species was > 2, the corresponding parameter set was rejected. This model performance criterion was selected at a meeting on October 6, 2006, by participating parties, including LDWG, EPA, and NOAA.

In order to understand a model performance assessment, it is important to understand the metric used. If a model run has a SPAF of X, the model's estimate differs from the empirical data to which it is being compared by a factor of X. Thus, model estimates with equal distance but opposite direction from an empirical data point (e.g., ±100 µg/kg ww from a mean concentration) will have different SPAFs, with the over-estimate always having a higher SPAF. So, if the mean empirical total PCB concentration in Pacific staghorn sculpin tissue is 900 µg/kg ww, and for one parameter set the model estimate is 1,000 µg/kg ww (i.e., 100 µg/kg ww greater than the mean empirical concentration) and for another parameter set the model estimate is 800 µg/kg ww (i.e., 100 µg/kg ww less than the mean empirical concentration), the percent difference of both model estimates from the mean empirical tissue chemical concentration is 11.1%, but the SPAFs are 1.11 and 1.13, respectively. SPAF and percent difference metrics are both useful tools for assessing model performance. The SPAF metric was used to assess model performance.

Parameter sets that met the model performance criterion (SPAF ≤ 2 for all species) were checked to ensure that unrealistic relationships among parameters did not occur (e.g., if temperature and DO, which should be negatively correlated, were found to be positively correlated with an r-value greater than 0.3). These combinations could occur by chance during Monte Carlo sampling. None of the parameter sets that met the model performance criterion were excluded based on this review.

D.5.1.3 Identification of the best-fit parameter set

The final step in the FWM calibration was to identify the parameter set that produced estimates most similar to the empirical data (i.e., mean total PCB concentrations in tissues). This parameter set was defined as the parameter set with the lowest mean SPAF across all species with empirical data. To identify this parameter set, the average SPAF across species was calculated for each parameter set that passed the model performance filter. Parameter sets were then sorted by average SPAF across species, and the set with the lowest average SPAF was identified. A best-fit parameter set was identified for both Calibration 1 and Calibration 2.

D.5.2 RESULTS

The calibration process identified FWM parameter sets that estimated total PCB concentrations for all species within a factor of 2 of empirical data (i.e., SPAF ≤ 2).

The mean SPAFs across species for parameter sets passing the model performance criterion for Calibrations 1 and 2 were 1.4 and 1.5, respectively (Table D.5-3). The SPAF for the best-fit parameter sets for both Calibrations 1 and 2 was 1.2.

Table D.5-3. Summary of model performance for Calibrations 1 and 2

Species	Summary of SPAFs from Parameter Sets Passing the Model Performance Filter							
	Calibration 1[a]				Calibration 2[a]			
	Closest to Empirical (by species)	Greatest Under-prediction (by species)	Greatest Over-prediction (by species)	Best Fit (for all species)	Closest to Empirical (by species)	Greatest Under-prediction (by species)	Greatest Over-prediction (by species)	Best Fit (for all species)
Benthic invertebrate	1.2	na[b]	2.0	1.5	nd	nd	nd	nd
Slender crab	1.0	**2.0**	2.0	1.0	1.2	na[b]	2.0	1.5
Dungeness crab	1.0	**2.0**	2.0	1.1	1.0	**1.3**	2.0	1.1
Pacific staghorn sculpin	1.0	**1.2**	2.0	1.2	1.2	na[b]	2.0	1.4
Shiner surfperch	1.0	**2.0**	1.2	**1.2**	1.0	**1.6**	1.8	**1.2**
English sole	1.0	**1.3**	1.7	1.1	1.0	**1.1**	1.9	1.1
Average SPAF	1.0	**1.6**	1.8	1.2	1.1	**1.3**	1.9	1.2

[a] Calibration 1 dataset included all Phase 1 (1997) and Phase 2 (2004 and 2005) empirical tissue data. The Calibration 2 dataset included Phase 1 tissue data and Phase 2 tissue data from 2005 (Phase 2 tissue data from 2004 were excluded).

[b] There were no underpredictions for benthic invertebrates in Calibration 1; there were no underpredictions for slender crab or Pacific staghorn sculpin in Calibration 2.

na – not applicable

nd – no data (no empirical data were available for benthic invertebrates for Calibration 2)

SPAF – species predictive accuracy factor

BOLD indicates an underprediction.

The benthic invertebrate tissue concentration, estimated using the tissue-sediment regression and an LDW-wide SWAC of 380 µg/kg dw was similar to the benthic invertebrate tissue concentration estimated by the FWM using the best-fit parameter set from Calibration 2 and was lower than the FWM estimate using the best-fit parameter set from Calibration 1 (Figure D.5-1). Benthic invertebrates were not included in Calibration 2 because the only benthic invertebrate tissue samples were collected in 2004, which are not in the Calibration 2 dataset.[30] Empirical data were not available for total PCB concentrations in phytoplankton, zooplankton, and juvenile fish tissues and hence were not included in the tabulated summary of model performance (Table D.5-3). However, estimated total PCB concentrations in those tissues were relatively similar between the two calibrations (Figure D.5-1).

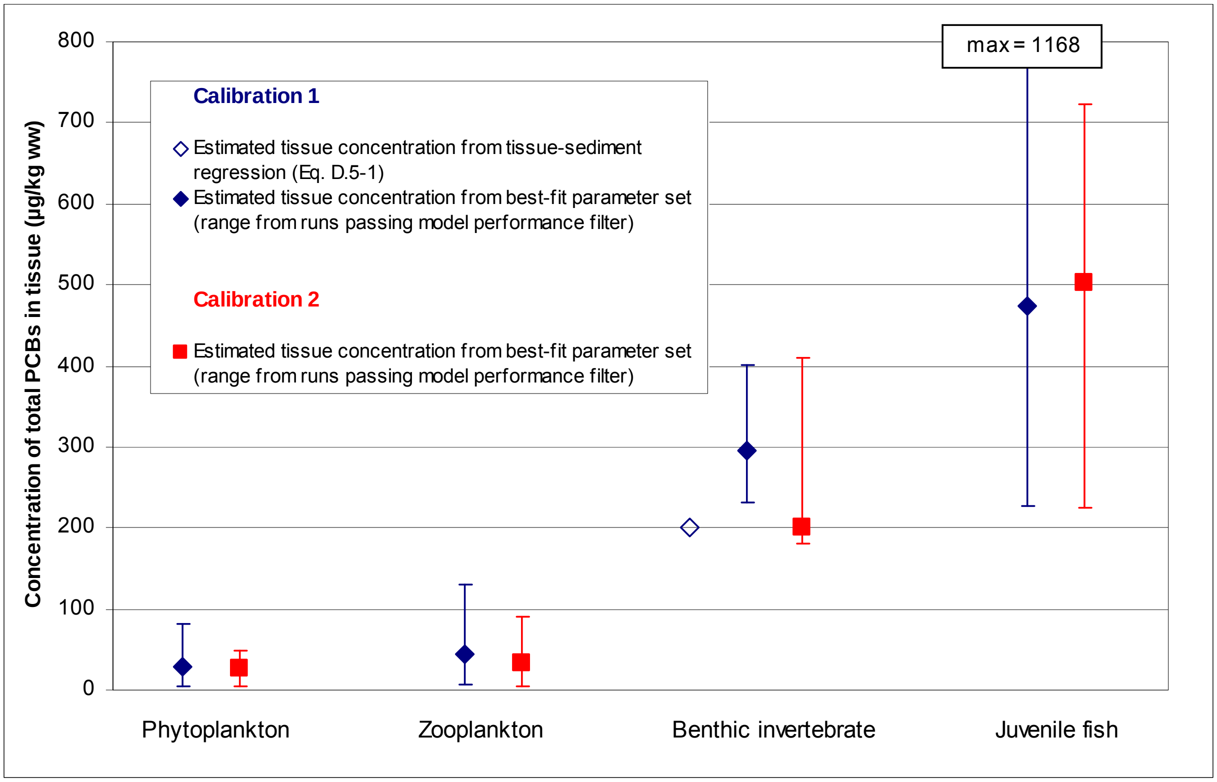

Figure D.5-1. **Estimated total PCB concentrations in tissues of prey species for parameter sets that passed the model performance filter in Calibrations 1 and 2 relative to empirical data**

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[30] Inclusion of benthic invertebrates in Calibration 2 would likely have had negligible impact on the model performance and selection of the best-fit parameter set for that calibration because only one parameter set was rejected when parameter sets were filtered based on the Calibration 2 empirical dataset plus the value for benthic invertebrates from the tissue-sediment regression (Equation D.5-1).

Estimated total PCB concentrations in fish and crab tissues associated with the best-fit parameter sets for both calibrations were generally similar to mean empirical data for each species (Figure D.5-2). The estimates associated with the best-fit parameter sets were generally higher than the mean empirical data, with the exception of shiner surfperch in Calibration 2. Possible reasons for overestimation are discussed in Section D.6.2.4.

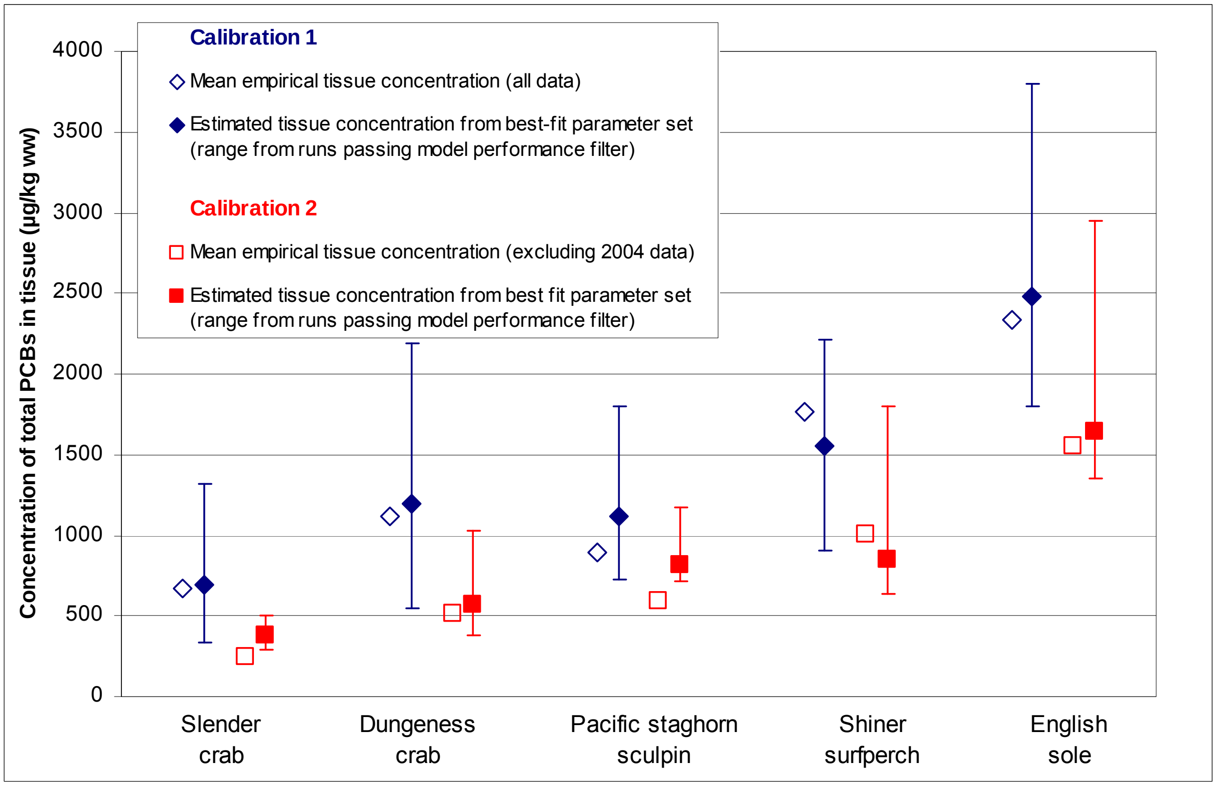

Figure D.5-2. Estimated total PCB concentrations in tissues of adult fish and crab species for parameter sets that passed the model performance filter in Calibrations 1 and 2 relative to empirical data

The calibration process rejected parameter sets that resulted in estimated tissue concentrations greater than a factor of 2 from empirical values for any species. Therefore, as part of the calibration process, parameter values were adjusted to optimize the fit of the model estimates to empirical total PCB data. The same FWM parameters tended to be affected in both calibrations; in most cases, the changes in parameter values from the original values (as presented in Table D.4-1) to those that passed the model performance criterion (Attachment 2) were similar for both calibrations. Relative to the original values, (Table D.4-1) the two best-fit parameter sets from Calibrations 1 and 2 (Table D.5-4) generally had:

- Lower total PCB concentrations in the water column compared to the average predicted by the EFDC model (1.43 ng/L)
- Lower uptake of total PCBs by benthic invertebrates (e.g., lower lipid content, lower dietary absorption efficiencies, greater fraction of zooplankton instead of sediment in diet)
- Higher dietary fraction of plankton (which was also intended to partially represent detritus) and a lower dietary fraction benthic invertebrates and sediment for some species

Table D.5-4. Best-fit parameter sets for Calibration 1 and 2

PARAMETER DESCRIPTION	UNIT	CALIBRATION 1	CALIBRATION 2
Environmental Parameters			
Concentration of total PCBs in the water column	ng/L	1.22	1.11
Concentration of POC in the water column	kg/L	2.3×10^{-7}	3.2×10^{-7}
Concentration of DOC in the water column	kg/L	2.2×10^{-6}	1.6×10^{-6}
Mean water temperature	°C	11.0	10.8
Concentration of dissolved oxygen in the water column	mg/L	8.15	8.02
Concentration of total suspended solids in the water column	kg/L	5.4×10^{-6}	4.7×10^{-6}
Concentration of total PCBs in sediment	µg/kg dw	380	380
Sediment total organic carbon	%	1.91	1.92
Chemical Parameters			
Octanol-water partition coefficient for total PCBs (log K_{ow})	unitless	6.5	6.5
Biological Parameters			
Proportionality constant expressing the sorption capacity of NLOM relative to that of octanol (β)	unitless	0.031	0.037
Resistance to chemical uptake through aqueous phase for phytoplankton/ algae (A)	day-1	6×10^{-5}	6×10^{-5}
Resistance to chemical uptake through organic phase for phytoplankton/ algae (B)	unitless	6.2	4.6
Density of lipids	kg/L	0.9	0.9
Phytoplankton			
Lipid content of organism	%	0.14	0.12
Water content of organism	%	95.7	95.6
Zooplankton			
Organism weight	kg	2.2×10^{-7}	1.6×10^{-7}
Lipid content	%	1.4	0.96
Water content of organism	%	92	89

PARAMETER DESCRIPTION	UNIT	CALIBRATION 1	CALIBRATION 2
Dietary absorption efficiency of lipids (ε_L)	%	66	71
Dietary absorption efficiency of NLOM (ε_N)	%	72	72
Benthic Invertebrates			
Organism weight	kg	4.1×10^{-5}	1.5×10^{-6}
Lipid content	%	0.83	0.81
Water content of organism	%	82	84
Relative fraction of porewater ventilated	unitless	0.13	0.074
Dietary absorption efficiency of lipids (ε_L)	%	30	79
Dietary absorption efficiency of NLOM (ε_N)	%	56	61
Juvenile Fish			
Organism weight	kg	6×10^{-3}	5×10^{-3}
Lipid content	%	1.5	3.1
Water content of organism	%	74.3	71.4
Relative fraction of porewater ventilated	unitless	0.01	0.02
Dietary absorption efficiency of lipids (ε_L)	%	92	91
Dietary absorption efficiency of NLOM (ε_N)	%	54	64
Slender Crab			
Organism weight	kg	0.165	0.167
Lipid content	%	1.1	1.1
Water content of organism	%	83.7	83.2
Relative fraction of porewater ventilated	unitless	0.03	0.02
Dietary absorption efficiency of lipids (ε_L)	%	75	66
Dietary absorption efficiency of NLOM (ε_N)	%	76	54
Dungeness Crab			
Organism weight	kg	0.653	0.529
Lipid content	%	3.4	2.6
Water content of organism	%	81	82
Relative fraction of porewater ventilated	unitless	0.02	0.02
Dietary absorption efficiency of lipids (ε_L)	%	71	36
Dietary absorption efficiency of NLOM (ε_N)	%	59	68
Pacific Staghorn Sculpin			
Organism weight	kg	0.075	0.082
Lipid content	%	2.1	2.2
Water content of organism	%	79	79
Relative fraction of porewater ventilated	unitless	0.03	0.06
Dietary absorption efficiency of lipids (ε_L)	%	93	91
Dietary absorption efficiency of NLOM (ε_N)	%	50	52
Shiner Surfperch			
Organism weight	kg	0.019	0.018
Lipid content	%	4.6	4.6

PARAMETER DESCRIPTION	UNIT	CALIBRATION 1	CALIBRATION 2
Water content of organism	%	74.0	73.5
Relative fraction of porewater ventilated	unitless	0.02	0.01
Dietary absorption efficiency of lipids (ε_L)	%	94	90
Dietary absorption efficiency of NLOM (ε_N)	%	56	58
English Sole			
Organism weight	kg	0.230	0.246
Lipid content	%	5.5	5.7
Water content of organism	%	75.0	74.8
Relative fraction of porewater ventilated	unitless	0.1	0.06
Dietary absorption efficiency of lipids (ε_L)	%	92	94
Dietary absorption efficiency of NLOM (ε_N)	%	59	53
Dietary Fraction			
Benthic Invertebrates			
Sediment	fraction	0.70	0.73
Phytoplankton	fraction	0.18	0.15
Zooplankton	fraction	0.12	0.12
Juvenile Fish			
Sediment	fraction	0.00	0.00
Zooplankton	fraction	0.53	0.60
Benthic invertebrates	fraction	0.47	0.40
Slender Crab			
Sediment	fraction	0.02	0.02
Zooplankton	fraction	0.09	0.09
Benthic invertebrates	fraction	0.88	0.88
Juvenile fish	fraction	0.01	0.01
Dungeness Crab			
Sediment	fraction	0.00	0.00
Zooplankton	fraction	0.37	0.19
Benthic invertebrates	fraction	0.24	0.41
Juvenile fish	fraction	0.39	0.39
Pacific Staghorn Sculpin			
Sediment	fraction	0.00	0.00
Zooplankton	fraction	0.22	0.34
Benthic invertebrates	fraction	0.54	0.44
Juvenile fish	fraction	0.24	0.22
Shiner Surfperch			
Sediment	fraction	0.00	0.00
Zooplankton	fraction	0.23	0.42
Benthic invertebrates	fraction	0.76	0.57

PARAMETER DESCRIPTION	UNIT	CALIBRATION 1	CALIBRATION 2
English Sole			
Sediment	fraction	0.04	0.03
Phytoplankton	fraction	0.05	0.07
Zooplankton	fraction	0.05	0.03
Benthic invertebrates	fraction	0.86	0.87

DOC – dissolved organic carbon
dw – dry weight
NLOM – non-lipid organic matter
PCB – polychlorinated biphenyl
POC – particulate organic carbon

D.5.3 IDENTIFICATION OF PARAMETER SET FOR FUTURE MODEL APPLICATIONS

As described above, the FWM was calibrated to two empirical datasets (Calibration 1, which included tissue data from the late 1990s, 2004, and 2005; and Calibration 2, which included all tissue data included in the Calibration 1 dataset, except for data from 2004, when major dredging events occurred). In both calibrations, the best-fit parameter sets estimated total PCB concentrations in tissue within a factor of 1.2 of empirical data as an average across all species. Also, the same FWM parameters tended to be affected in both calibrations (Table D.4-1 and Attachment 2).

In order to estimate sediment RBTCs for total PCBs in the RI (see Section D.8), it was necessary to select a calibrated model. The best-fit parameter set for Calibration 2 was selected for this purpose because tissue data from 2004 may have been influenced by the large dredging events that took place months before the tissue samples were collected. These events may have resulted in a short-term increase in exposure to chemicals in the water column, as has been documented at several sites nationwide (Alcoa Inc 1995; EPA 2006). Using these data to calibrate the model may have violated the steady-state intent of the modeling effort. While this hypothesis can not be definitively proven with available data, dredging events stir up bottom sediments, which can mobilize sequestered PCBs from bottom sediments into the water column and increase aquatic organisms' exposure to PCBs in the short term. There were no other obvious changes in environmental conditions between 2004 and 2006 that could explain the differences in total PCB tissue concentrations. Tissue data were collected in 2007, which will provide additional temporal information for the LDW. Also as discussed in Section D.5.2, the FWM tended to overestimate total PCB tissue concentrations when compared to either calibration 1 or 2, suggesting that using calibration 2 should still be conservative in its estimates. Thus, the best-fit parameter set from Calibration 2 was selected to derive sediment RBTCs (Section D.8). The performance of the best-fit parameter set from Calibration 2 was also tested at a smaller spatial scale within the LDW and at clam intertidal locations (Section D.7).

D.6 Sensitivity and Uncertainty Analyses

Sensitivity and uncertainty analyses were performed to assess the sensitivity of the FWM to individual input parameters in combination with the uncertainty in the estimates of those parameters. These analyses provide insight into uncertainties in the application of FWM results.

An uncertainty analysis is an evaluation of how uncertainties in model parameters affect the reliability of the model's output. Uncertainties can be reducible (i.e., they can be eliminated by gathering more information and/or considering available information differently) or irreducible (i.e., they cannot be eliminated because there is an element of either chance or variability in the parameter's distribution, such as variability across individuals in a population or within an individual over time).

A sensitivity analysis is an evaluation of how model estimates respond to changes in input values. The greater the response to a particular change (or set of changes), the higher the sensitivity to that parameter or parameters. A sensitivity analysis can thus provide information regarding the relative importance of uncertainties by examining their potential influence on model output.

A number of uncertainties were not evaluated in the sensitivity and uncertainty analyses:

- Mean of the empirical data as an estimate of true mean tissue total PCB concentrations in the LDW
- True uptake and depuration processes described by the FWM equations
- Distributions assigned to FWM parameters

In addition, the ability of the SWAC to serve as an estimate of the true mean sediment concentration to which the modeled species were exposed is uncertain. Because the SWAC was not allowed to vary in the calibration (i.e., it was treated deterministically as described in Section D.4.1.1), the influence of sediment concentration on model predictions was not examined as part of the three sensitivity and uncertainty analyses described in this section. As discussed in Section D.5.2, the FWM tended to overestimate total PCB concentrations in tissues when compared to either calibration dataset (Figures D.5-1 and D.5-2). The following assumptions made in defining the SWAC for the FWM could have contributed to the model's tendency to overestimate tissue concentrations for most species in the LDW.

- The interpolation method used to generate the SWAC (i.e., IDW) has uncertainties.
- The SWAC used in the FWM assumed that fish and crab species in the LDW use all areas of the LDW equally. In reality, some or all of the fish and crab species may preferentially use some areas of the LDW with more suitable

habitat (e.g., better food sources or refuge from predators) more than other areas.

- The SWAC used in the FWM assumed that all modeled species use the LDW 100% of the time. No site use factors were applied for species that may move out of the LDW for part of the year.

Uncertainty regarding the value of the SWAC resulted in uncertainty about the accuracy of calibrated parameters. Because the SWAC is an influential input parameter and was treated deterministically, any error in the point estimate of the SWAC used in calibration was countered by offsetting adjustments in other FWM parameters. Thus, the parameter sets identified through the calibration process were highly influenced by the SWAC. For these reasons, which underlie the importance of this parameter to FWM calibration and predictions, the sensitivity of the FWM to total PCB concentrations in sediment was also investigated further (Section D.6.1.3).

All models are simplifications of the processes and parameters that they describe. Therefore, it is important to assess the potential uncertainties in a FWM so these uncertainties can be acknowledged in its application. The following sensitivity and uncertainty analyses were conducted and are described in more detail in the following subsections:

- Correlation coefficient analysis
- Nominal range sensitivity (NRS) analysis
- SWAC sensitivity and uncertainty analysis

D.6.1 METHODS

The parameter set used in the sensitivity and uncertainty analyses was the best-fit parameter set from Calibration 1. This parameter set was used because these analyses were conducted prior to the decision to use Calibration 2 to estimate RBTCs. It is likely that the results of these three analyses would not change significantly if the best-fit parameter set for Calibration 2 had been used instead because the values for parameters to which the model is most sensitive (e.g., concentration of total PCBs in the water column, log K_{OW}, density of lipids, lipid contents of organisms and dietary absorption efficiencies) were similar for the best-fit parameter sets for Calibrations 1 and 2. The methods used for each of the three analyses are described in the following subsections.

In contrast, the SWAC sensitivity analysis was conducted after Calibration 2 was selected to estimate RBTCs, and therefore, the SWAC sensitivity analysis used the best-fit parameter set from Calibration 2. The results from this analysis are more directly relevant to sensitivity associated with the selected sediment RBTCs.

D.6.1.1 Correlation coefficient analysis

Pearson product-moment correlation coefficients (r-values) were calculated to characterize the strength of correlations between each FWM parameter and estimated total PCB concentrations in tissues. For each parameter, the absolute values of the correlation coefficients were averaged across all species in the FWM to get a general sense of the degree of covariance between a given parameter and predicted total PCB concentrations in tissues of all species combined. The 20 parameters that correlated most strongly with tissue concentration estimates (i.e., had the highest average absolute r-values) were carried forward into the NRS analysis. Parameters for which correlations were lower were not evaluated further because they had relatively low influence on model estimates.

Because the correlation coefficient analysis used output from the Monte Carlo runs, it accounted for parameter interactions as opposed to univariate analyses, which hold all other parameter values constant while changing the value for one parameter at a time. The NRS (Section D.6.1.2) is a univariate analysis. Because the correlation analysis incorporated parameter interactions, it was the most suitable analysis for identifying the 20 most important parameters.

D.6.1.2 Nominal range sensitivity analysis

In the NRS analysis, the input values for each of the top 20 parameters were varied, one at a time, from their minimum to their maximum values while all other FWM parameters were held at their best-fit parameter set values.[31] Minimum and maximum parameter values were identified in the sets passing the model performance filter for each of the top 20 parameters (Table D.6-1).

Table D.6-1 Minimum and maximum values for each parameter evaluated in the NRS

PARAMETER DESCRIPTION	UNIT	VALUES FROM PARAMETER SETS THAT PASSED MODEL PERFORMANCE FILTER FROM CALIBRATION 1	
		MINIMUM	MAXIMUM
Concentration of total PCBs in the water column	ng/L	0.218	2.940
Log octanol-water partition coefficient for PCBs (log K_{ow})	unitless	6.4	6.8
Density of lipids	kg/L	0.8	1.0
Zooplankton lipid content	%	0.2%	2.3%
Weight of benthic invertebrates	kg	7.1×10^{-8}	1.2×10^{-4}

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[31] Nominal range sensitivity analysis is conventional terminology, but this analysis can also be referred to as an uncertainty analysis because it provides information about how uncertainties in model parameters affect the reliability of the model's output. The term "sensitivity" was adopted for this section to emphasize the comparative nature of the analysis.

PARAMETER DESCRIPTION	UNIT	VALUES FROM PARAMETER SETS THAT PASSED MODEL PERFORMANCE FILTER FROM CALIBRATION 1	
		MINIMUM	MAXIMUM
Lipid content of benthic invertebrates	%	0.69%	1.05%
Water content of benthic invertebrates	%	71%	87%
Relative fraction of porewater ventilated by benthic invertebrates	unitless	0.050	0.247
Dietary absorption efficiency of NLOM (ε_N) for benthic invertebrates	%	17%	93%
Lipid content juvenile fish	%	0.6%	4.6%
Dietary absorption efficiency of lipids (ε_L) for slender crab	%	16%	95%
Dietary absorption efficiency of NLOM (ε_N) for slender crab	%	16%	95%
Lipid content of Dungeness crab	%	1.1%	4.2%
Dietary absorption efficiency of lipids (ε_L) for Dungeness crab	%	16%	96%
Relative fraction of zooplankton in juvenile fish diet	fraction	0.35	0.81
Relative fraction of benthic invertebrates in juvenile fish diet	fraction	0.18	0.65
Relative fraction of zooplankton in Pacific staghorn sculpin diet	fraction	0.01	0.50
Relative fraction of juvenile fish in Pacific staghorn sculpin diet	fraction	0.172	0.661
Relative fraction of zooplankton in shiner surfperch diet	fraction	0.188	0.689
Relative fraction of benthic invertebrate in shiner surfperch diet	fraction	0.304	0.803

NLOM – non-lipid organic matter
PCB – polychlorinated biphenyl

Each of the minimum and maximum values was substituted, in turn, into the best-fit parameter set, yielding 40 new estimates of total PCB concentrations in each species' tissue. For each of the 20 parameters, NRS was calculated for each species as:

$$\mathrm{NRS} = \left|\left(C_{\mathrm{Max}} - C_{\mathrm{Min}}\right)\right| \qquad \textbf{Equation D.6-1}$$

Where:

C_{Max} = estimated total PCB concentration in tissue when the maximum value for the parameter being tested was substituted into the best-fit parameter set
C_{Min} = estimated total PCB concentration in tissue when the minimum value for the parameter being tested was substituted into the best-fit parameter set

A parameter's NRS value is a measure of the relative influence that parameter has on the uncertainty of FWM tissue estimates for each species.

D.6.1.3 SWAC sensitivity and uncertainty analysis

To explore the effects of SWAC uncertainty on FWM estimates and on the tendency of the FWM to overestimate concentrations of total PCBs in tissue (Section D.6), the best-fit parameter set was run six times, each time with a lower SWAC, starting at the initial estimate of 380 µg/kg dw. Model estimates were compared to empirical data to determine which SWAC resulted in the best fit for the FWM.

D.6.2 RESULTS

D.6.2.1 Correlation coefficient analysis

The 20 parameters with the highest average absolute value correlation coefficients across species are presented in Table D.6-2. A positive correlation indicates that an increase in a parameter value led to an increase in estimated total PCB concentrations in tissue for a given species. A negative correlation indicates that an increase in a parameter value led to a decrease in the estimated concentrations for a given species. In general, parameter values that most strongly correlated with estimates for at least one tissue type included those that:

- Affected PCB exposure in the water column, particularly the concentration of total PCBs (for phytoplankton and zooplankton)
- Contributed to the uptake of total PCBs, including dietary adsorption efficiencies (for crabs) and lipid content (for various species)
- Characterized dietary preferences (e.g., pelagic vs. benthic components of the food web) (for shiner surfperch, juvenile fish, Pacific staghorn sculpin)
- Affected the uptake of total PCBs by benthic invertebrates (e.g., porewater ventilation) (for English sole)

Table D.6-2. Parameters most strongly correlated with estimated total PCB concentrations in tissues

	Correlation Coefficient										
Parameter	Maximum Correlation	Average Correlation (Absolute Value)	Phyto-plankton	Zooplankton	Benthic Inverte-brates	Juvenile Fish	Slender Crab	Dungeness Crab	Pacific Staghorn Sculpin	Shiner Surfperch	English Sole
Concentration of total PCBs in the water column	**0.96**	0.37	**0.96**	**0.86**	0.09	**0.31**	0.06	0.17	**0.33**	**0.32**	0.19
Dietary absorption efficiency of lipids for slender crab	**0.75**	0.10	-0.02	-0.01	-0.03	-0.03	**0.75**	-0.01	-0.01	-0.01	-0.02
Fraction of benthic invertebrates in diet of shiner surfperch	**0.68**	0.10	-0.04	-0.03	-0.04	-0.01	-0.03	-0.05	-0.001	**0.68**	-0.05
Fraction of zooplankton in diet of shiner surfperch	**-0.68**	0.10	0.04	0.03	0.04	0.01	0.03	0.05	0.002	**-0.68**	0.05
Dietary absorption efficiency of lipids for Dungeness crab	**0.67**	0.11	-0.04	-0.04	-0.04	-0.03	-0.03	**0.67**	-0.03	-0.06	-0.02
Lipid content of juvenile fish	**0.61**	0.16	-0.07	-0.08	-0.07	**0.61**	-0.05	0.06	0.27	-0.09	-0.12
Fraction of benthic invertebrates in diet of juvenile fish	**0.46**	0.15	-0.04	-0.06	-0.11	**0.46**	-0.05	0.17	0.25	-0.10	-0.14
Fraction of zooplankton in diet of juvenile fish	**-0.46**	0.15	0.04	0.06	0.11	**-0.46**	0.05	-0.17	-0.26	0.10	0.14
Dietary absorption efficiency of NLOM for slender crab	**0.46**	0.07	-0.01	0.01	0.02	0.01	**0.46**	-0.03	0.01	-0.02	0.01
Lipid content of zooplankton	**0.41**	0.07	-0.01	**0.41**	-0.03	0.04	-0.02	-0.07	0.02	0.01	-0.04
Relative fraction of porewater ventilated by benthic invertebrates	**0.36**	0.14	-0.07	-0.10	0.28	0.03	0.13	0.05	0.11	0.13	**0.36**
Lipid content of Dungeness crab	**0.30**	0.06	-0.03	-0.02	-0.03	-0.001	-0.04	**0.30**	-0.03	-0.02	-0.06
Fraction of zooplankton in diet of Pacific staghorn sculpin	**-0.30**	0.09	0.07	0.07	0.04	0.13	0.05	0.04	**-0.30**	0.03	0.09
Density of lipids	**-0.29**	0.09	0.10	0.04	-0.02	0.02	-0.03	-0.04	-0.12	-0.16	**-0.29**
Water content of benthic invertebrates	**-0.28**	0.09	0.03	0.03	**-0.28**	-0.02	0.11	0.03	0.06	0.05	0.19
Fraction of juvenile fish in diet of Pacific staghorn sculpin	**-0.25**	0.12	-0.11	-0.10	-0.11	**-0.25**	-0.05	-0.11	0.14	-0.10	-0.12
Dietary absorption efficiency of NLOM for benthic invertebrates	**0.25**	0.10	-0.10	-0.11	0.20	0.01	0.11	0.04	0.02	0.07	**0.25**
Lipid content of benthic invertebrates	**0.24**	0.06	-0.05	-0.05	**0.24**	0.01	-0.08	-0.05	-0.02	-0.004	-0.06
Log octanol-water partition coefficient (log K_{OW}) for total PCBs	**0.2**	0.10	-0.003	0.05	0.05	0.08	0.15	0.06	**0.20**	0.16	0.12
Weight of benthic invertebrates	**0.2**	0.08	-0.03	-0.04	0.17	0.02	0.04	0.02	0.06	0.11	**0.20**

NLOM – non-lipid organic matter

PCB – polychlorinated biphenyl

Bold indicates the maximum correlation for that parameter.

D.6.2.2 Nominal range sensitivity analysis

NRS values for each parameter for each species are presented in Table D.6-3. NRS values ranked by maximum NRS value across species indicate the relative potential effect of a given parameter on the uncertainty of FWM estimates. In order to understand the importance of a parameter, it is necessary to compare the NRS value to the estimated total PCB concentration for each modeled species (Table D.6-3). This comparison provides a sense of the magnitude of the uncertainty associated with a specific parameter relative to the estimate.

Parameters that influenced estimates for all species are concentration of total PCBs in the water column, log K_{OW}, and density of lipids (Table D.6-3). All five benthic invertebrate parameters had an effect on all species except phytoplankton and zooplankton. Parameters specific to an adult fish or crab species (e.g., dietary absorption efficiency of lipids (ε_L) for Dungeness crab) influenced tissue estimates for that species only.

The results of the correlation coefficient analysis (Table D.6-2) and the NRS analysis (Table D.6-3) are different. These differences can be partly explained by the fact that correlation coefficients take parameter interaction into account, whereas NRS values are based on the effect of changing one parameter value at a time while all other values are held constant.

NRS values for benthic invertebrates, juvenile fish, and fish and crab species are presented graphically in Figures D.6-1 to D.6-7. Estimated correlation coefficients from the correlation analysis discussed in Section D.6.2.1 are also included for reference. Parameters with NRS values of zero are not shown on figures for individual species.

The total PCB concentrations in tissue shown in bold on the figures are the estimated concentrations resulting from the best-fit parameter set for Calibration 1. The bars range from C_{Max} (the estimated concentration in tissue that results when the maximum value for a given parameter is used) to C_{Min} (the estimated concentration in tissue that results when the minimum value for a given parameter is used) (see Table D.6-1). NRS is the absolute value of the difference between C_{Max} and C_{Min} (Equation D.6-1).

Table D.6-3 Nominal range sensitivity values for the top 20 parameters ranked by maximum NRS

Parameter	Nominal Range Sensitivity (µg/kg ww)								
	Phyto-plankton	Zoo-plankton	Benthic Inverte-brates	Juvenile Fish	Slender Crab	Dungeness Crab	Pacific Staghorn Sculpin	Shiner Surfperch	English Sole
Dietary absorption efficiency of lipids (ε_L) for Dungeness crab	0	0	0	0	0	**1,200**	0	0	0
Weight of benthic invertebrates	0	0	130	160	280	400	410	610	**920**
Lipid content of Dungeness crab	0	0	0	0	0	**840**	0	0	0
Relative fraction of benthic invertebrates in the shiner surfperch diet	0	0	0	0	0	0	0	**830**	0
Relative fraction of porewater ventilated by benthic invertebrates	0	0	110	140	240	350	360	530	**800**
Relative fraction of zooplankton in the shiner surfperch diet	0	0	0	0	0	0	0	**790**	0
Concentration of total PCBs in the water column	63	100	61	280	190	**740**	520	560	600
Lipid content of juvenile fish	0	0	0	**680**	5.3	450	510	0	0
Dietary absorption efficiency of NLOM (ε_N) for benthic invertebrates	0	0	86	110	180	270	270	410	**620**
Dietary absorption efficiency of lipids (ε_L) for slender crab	0	0	0	0	**570**	0	0	0	0
Log octanol-water partition coefficient (Log K_{OW}) for total PCBs	6.6	20	69	240	270	**560**	550	560	540
Relative fraction of zooplankton in the juvenile fish diet	0	0	0	340	7.5	**560**	300	0	0
Relative fraction of benthic invertebrates in the juvenile fish diet	0	0	0	340	7.5	**560**	300	0	0
Density of lipids	0.5	8.4	36	110	130	310	320	380	**550**
Lipid content of benthic invertebrates	0	0	75	86	77	190	210	330	**460**
Relative fraction of zooplankton in Pacific staghorn sculpin diet	0	0	0	0	0	0	**460**	0	0
Relative fraction of juvenile fish in Pacific staghorn sculpin diet	0	0	0	0	0	0	**380**	0	0
Dietary absorption efficiency of NLOM (ε_N) for slender crab	0	0	0	0	**310**	0	0	0	0
Lipid content of zooplankton	0	57	24	6.4	91	**230**	40	75	170
Water content of benthic invertebrates	0	0	92	59	93	130	56	**160**	140
FWM-estimated total PCB concentrations in tissue (for reference)	28	45	300	470	690	1,200	1,100	1,600	2,500

NLOM – non-lipid organic matter

NRS – nominal range sensitivity

PCB – polychlorinated biphenyl

ww – wet weight

Bold indicates maximum NRS for that parameter.

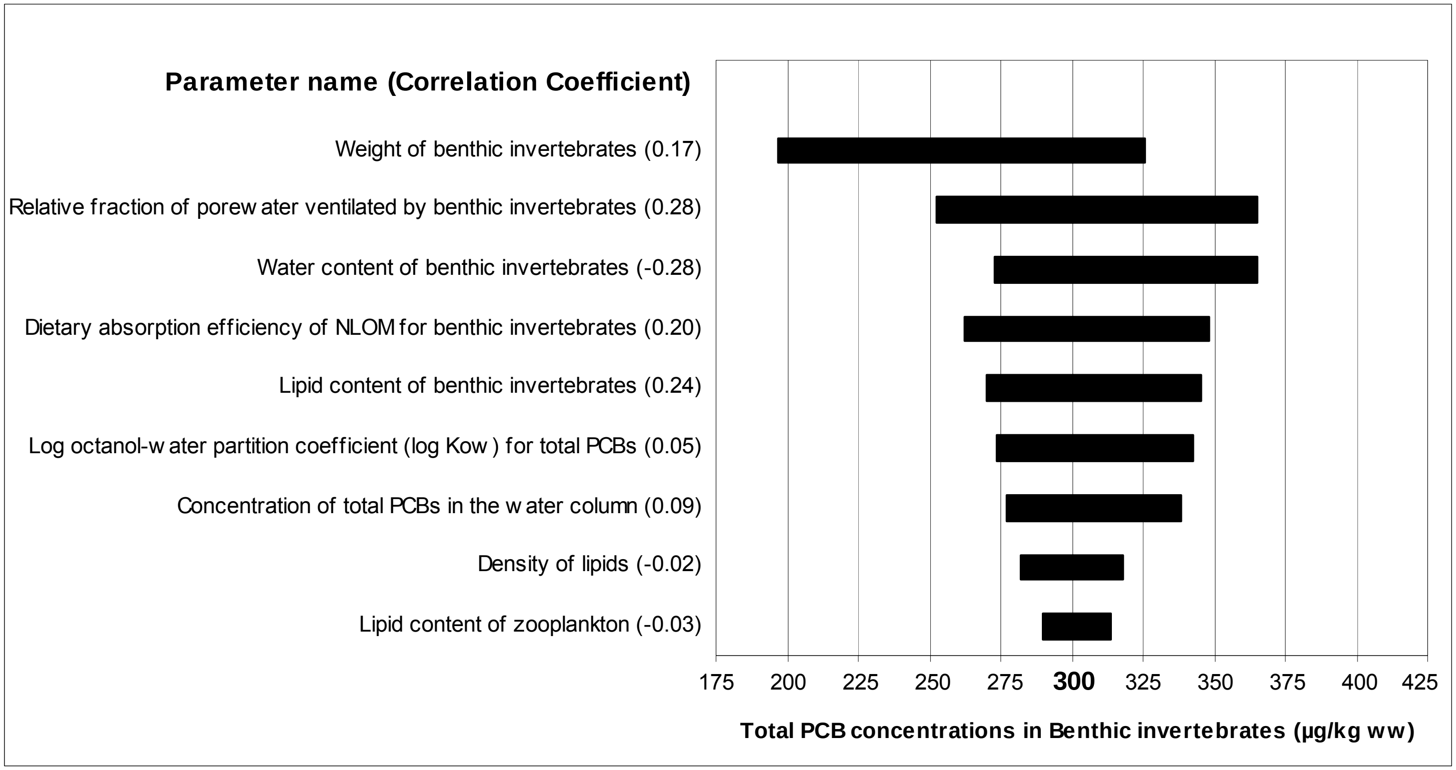

Figure D.6-1. Results of the nominal range sensitivity analysis for benthic invertebrates

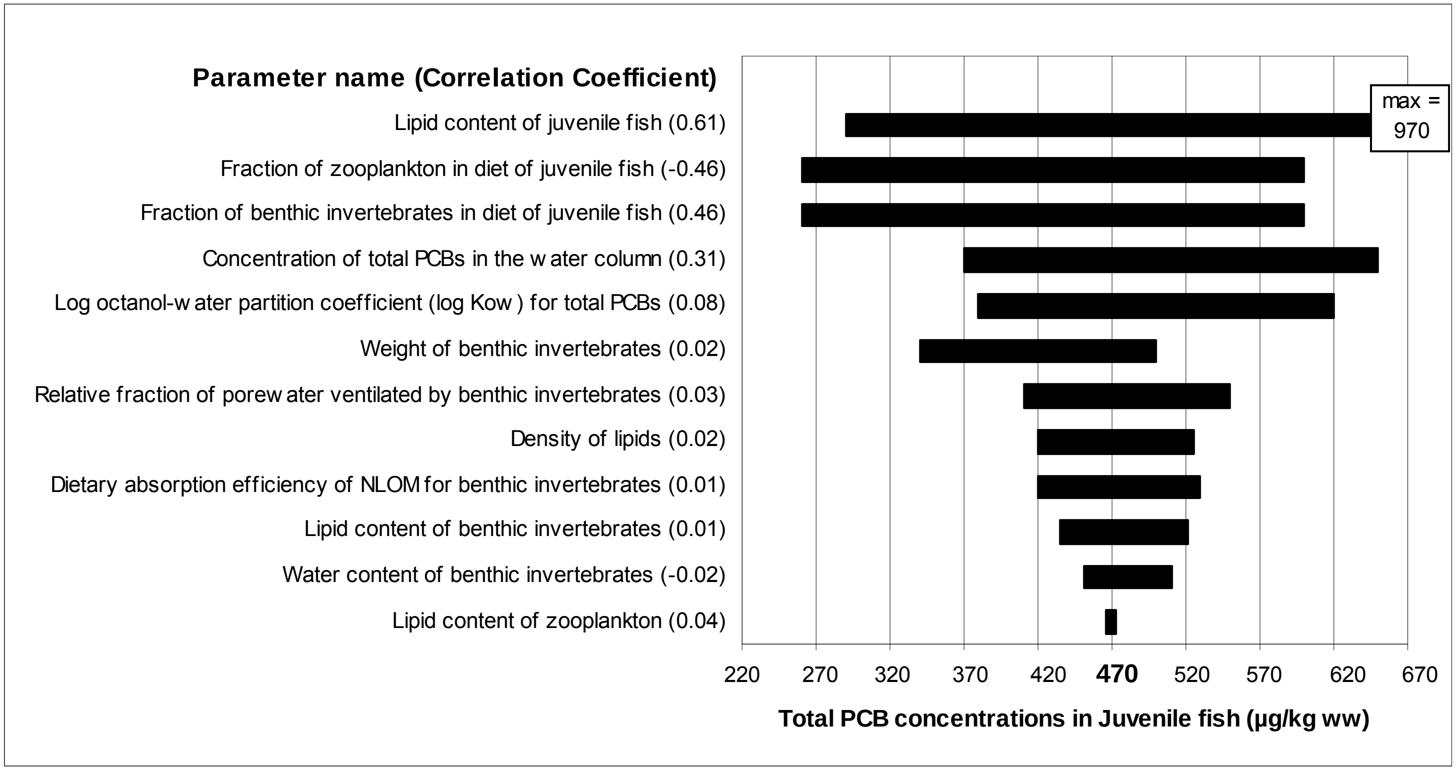

Figure D.6-2. Results of the nominal range sensitivity analysis for juvenile fish

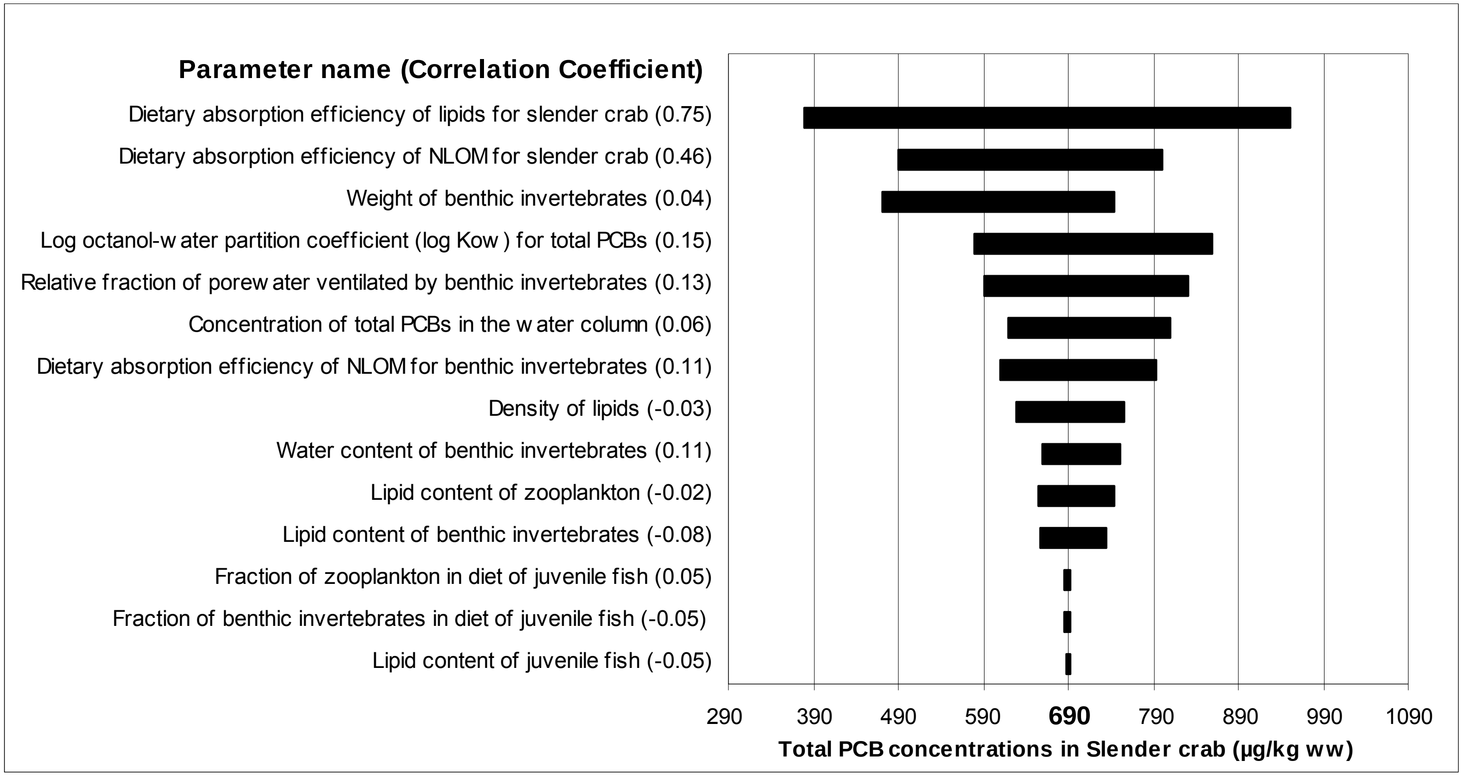

Figure D.6-3. Results of the nominal range sensitivity analysis for slender crab

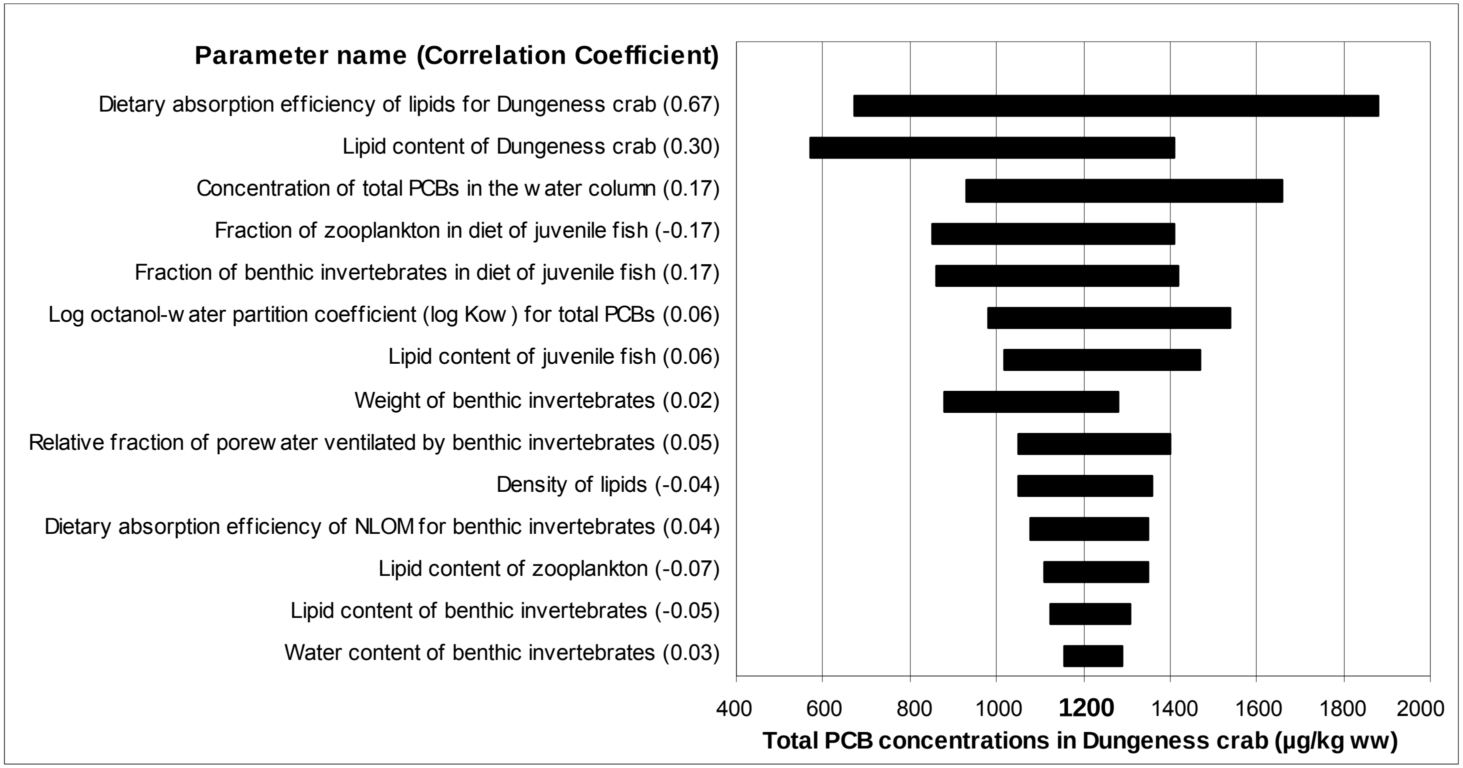

Figure D.6-4. Results of the nominal range sensitivity analysis for Dungeness crab

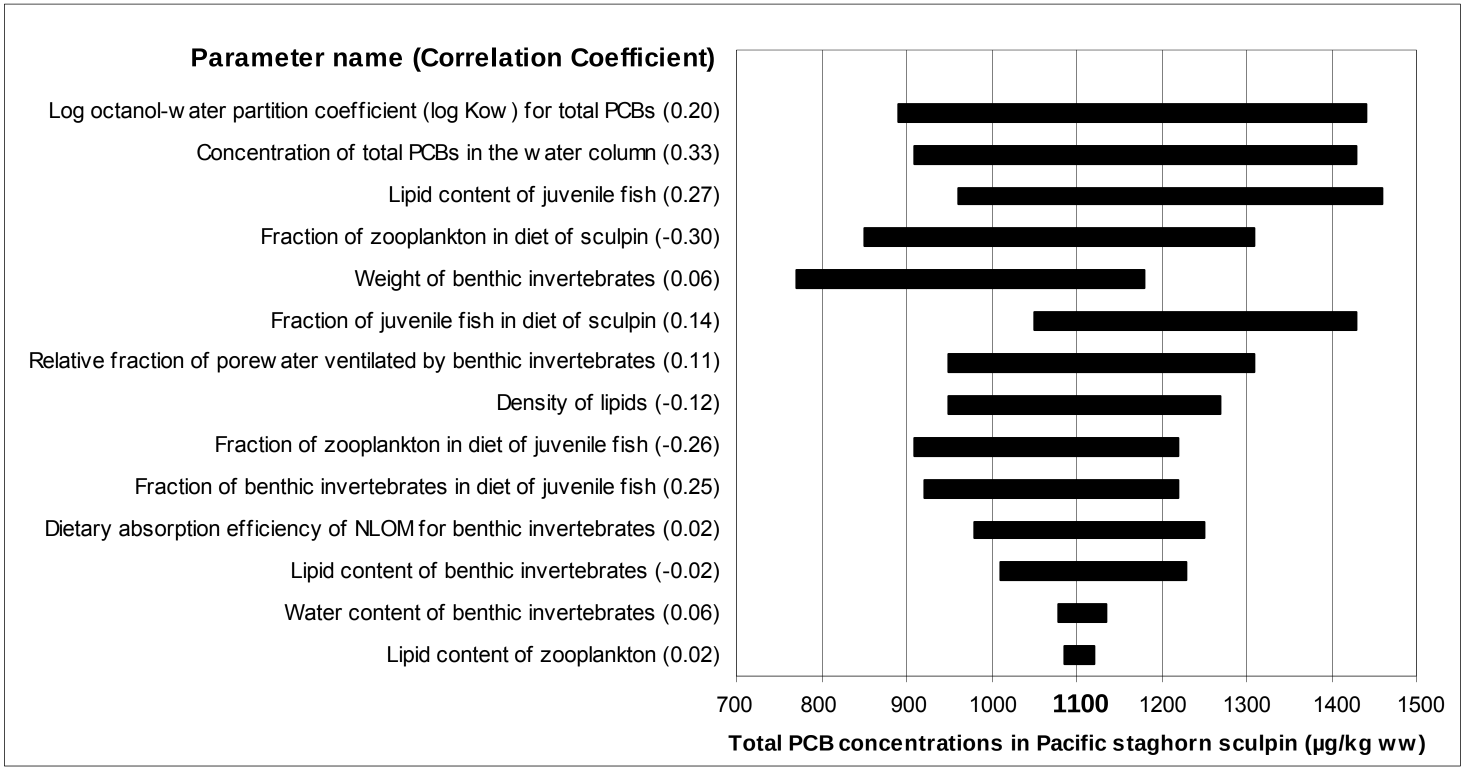

Figure D.6-5. Results of the nominal range sensitivity analysis for Pacific staghorn sculpin

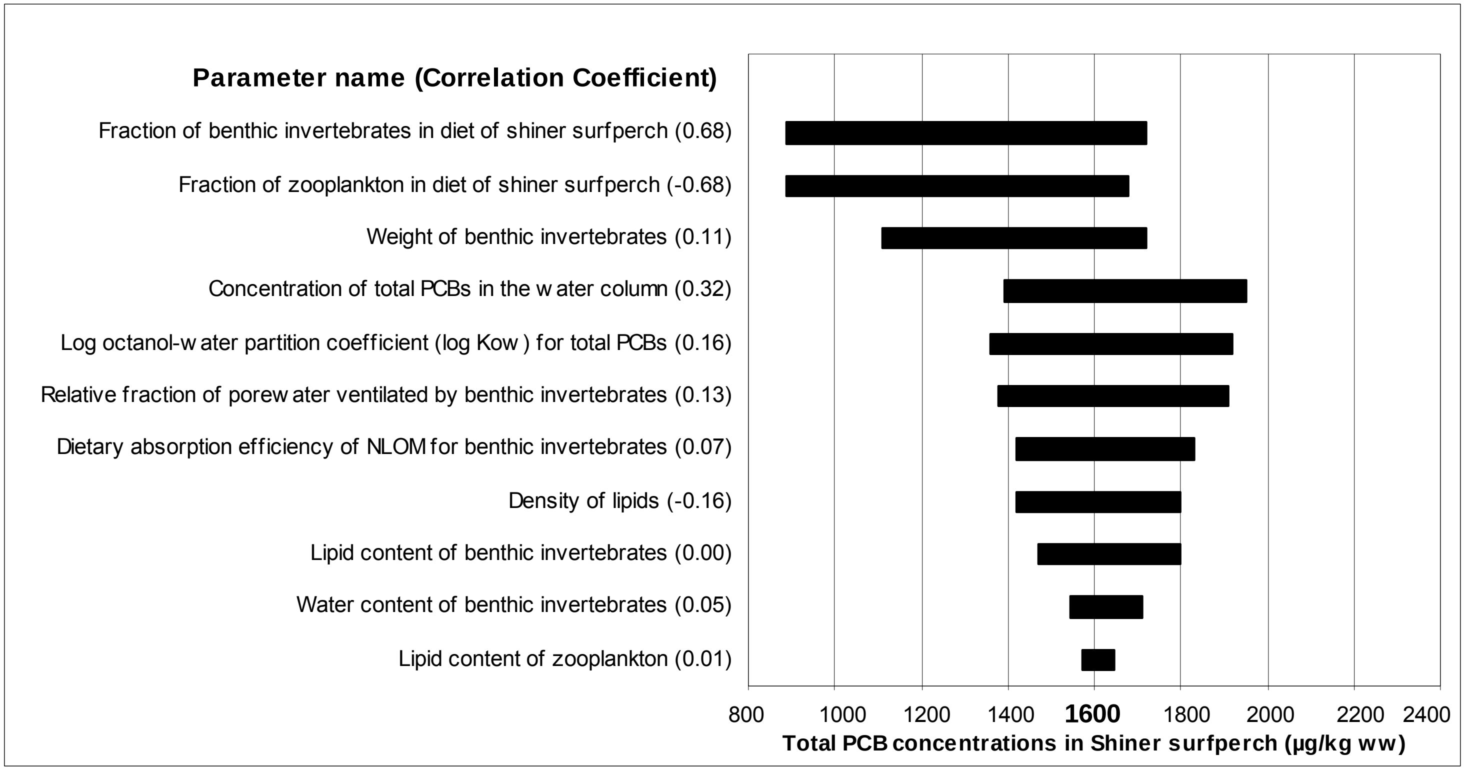

Figure D.6-6. Results of the nominal range sensitivity analysis for shiner surfperch

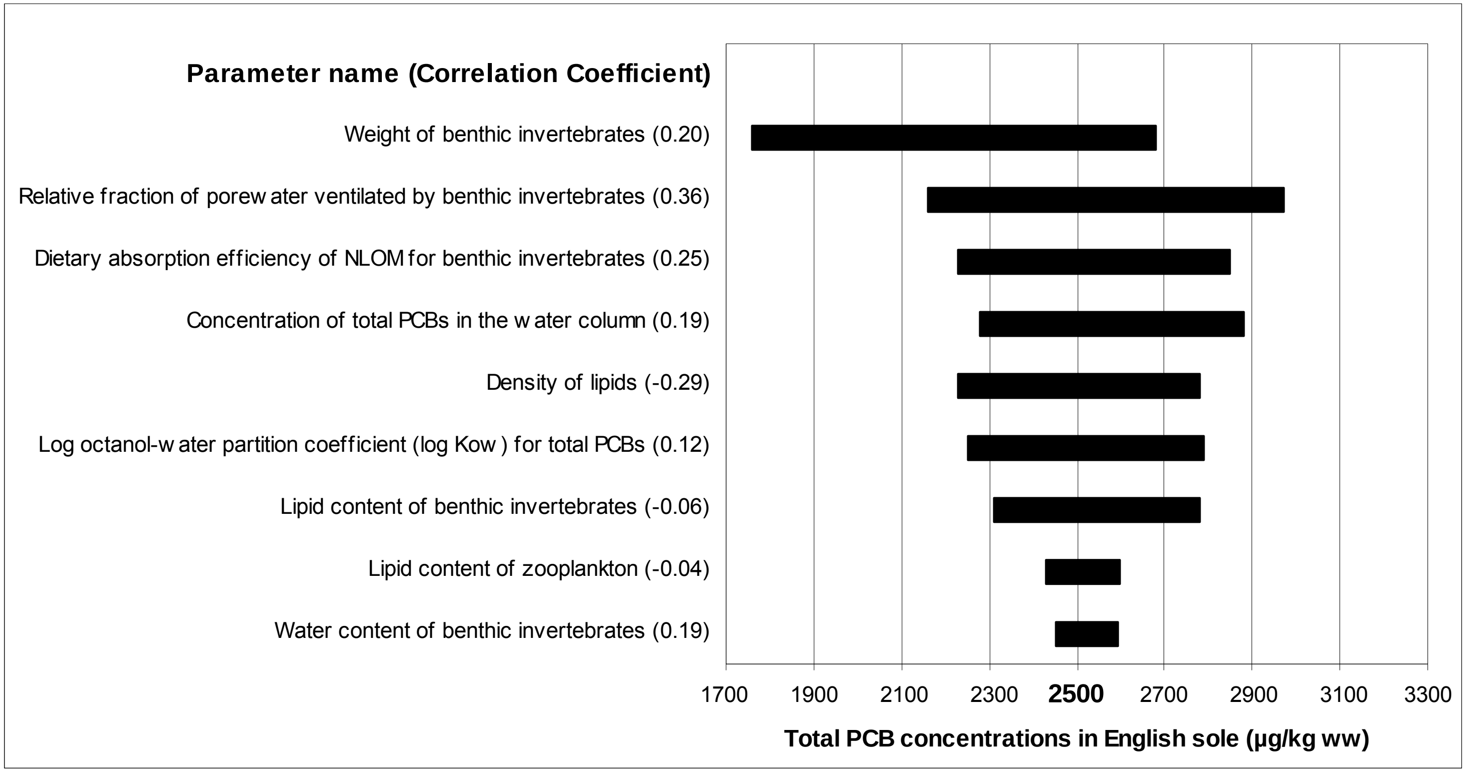

Figure D.6-7. Results of the nominal range sensitivity analysis for English sole

Log K_{OW} had a significant influence on estimates of total PCBs in tissue for all species (i.e., its NRS ranked in the top six parameters for all species). Log K_{OW} is a key parameter for total PCB uptake and loss in the FWM. The range of possible input values for this parameter is high (i.e., the maximum log K_{OW} value is 60% greater than the mean, and the minimum log K_{OW} value is 40% less than the mean), which may contribute to the high NRS values.

The benthic invertebrate-specific parameters of body weight, relative fraction of porewater ventilated, and dietary absorption efficiency of NLOM had a relatively significant influence on model estimates for many species. All target fish and crab species modeled were assumed to consume benthic invertebrates as a significant component of their diet. The broad range of input values assumed for benthic invertebrate weight (i.e., 7.1×10^{-8} kg to 1.2×10^{-4} kg), contributed to the high NRS. Compared to other species consumed by fish and crab species, benthic invertebrates had the greatest range of fraction of porewater ventilated, and consequently NRS values for this parameter also ranked high. Benthic invertebrates and the species that consume them were sensitive to the benthic invertebrate dietary absorption efficiency of NLOM because the diet of benthic invertebrates is composed of items with very low lipid content (i.e., sediment, phytoplankton, and zooplankton). Benthic invertebrate lipid and water content had less of an influence on the FWM estimates because of the relatively narrow range of values around the mean defined for these parameters.

Total PCB concentrations in the water column had a significant influence on estimated total PCB concentrations in phytoplankton and zooplankton. Other species affected by the total PCB concentration in water were organisms that consume at least 25% zooplankton in their diets (i.e., juvenile fish, Dungeness crab, Pacific staghorn sculpin, and shiner surfperch). In addition, because juvenile fish were assumed to consume 57% zooplankton, estimated tissue total PCB concentrations in species that consume juvenile fish (e.g., Dungeness crab and Pacific staghorn sculpin) had additional sensitivity to this parameter.

Estimated total PCB concentrations in crabs were highly influenced by dietary absorption efficiencies (Figures D.6-3 and D.6-4). Model estimates for slender crabs were sensitive to lipid and NLOM dietary absorption efficiencies; model estimates for Dungeness crabs were sensitive to dietary absorption efficiency of lipids. Dietary absorption efficiencies for crabs had a broad range of defined mean values (i.e., both NLOM and lipid dietary absorption efficiencies ranged from 16 to 96 percent),[32] which may explain the significant influence of these parameters.

Estimated total PCB concentrations in Pacific staghorn sculpin were influenced by dietary assumptions and juvenile fish lipid content (Figure D.6-5). Pacific staghorn

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[32] For comparison, the dietary absorption efficiency ranges for fish were 50 to 65% for NLOM and 90 to 95% for lipids.

sculpin were assumed to consume an average of 24% zooplankton and 33% juvenile fish, but the ranges for these dietary fractions were allowed to increase up to 50% zooplankton or 66% juvenile fish. Because juvenile fish were assumed to have higher lipid contents and are higher in the food chain than zooplankton, the relative consumption of juvenile fish and zooplankton had a significant effect on estimated total PCB concentrations in Pacific staghorn sculpin.

Estimated total PCB concentrations in shiner surfperch were influenced by the relative dietary fractions of zooplankton vs. benthic invertebrates (Figure D.6-6), which is highly uncertain. Greater amounts of zooplankton in the diet of shiner surfperch would decrease estimated total PCB concentrations in their tissue (because zooplankton have lower estimated tissue total PCB concentrations than do benthic invertebrates).

Benthic invertebrates make up 86 to 90% of the diet of English sole. Consequently, estimated total PCB concentrations in English sole were heavily influenced by benthic invertebrate-specific parameters (Table D.6-3 and Figure D.6-7).

The nominal range sensitivity analysis provided a sense of which parameters had the greatest potential to influence FWM estimates. It is not surprising that the parameters identified as the "most sensitive" through the NRS analysis were generally the same parameters that were adjusted through calibration (Section D.5.2). In general, the parameters that had the largest influence on model uncertainty were those with values that were derived from the literature and had broad ranges.

D.6.2.3 SWAC sensitivity and uncertainty analysis

The SWAC was not evaluated in the correlation coefficient or NRS analyses (Sections D.6.2.1 and D.6.2.2) because the SWAC is a decision variable and thus had only one value for calibration. The results of an analysis of the sensitivity of the FWM to the SWAC and the potential influence of the SWAC on the uncertainty of FWM estimates are presented in this section. As mentioned earlier, because this evaluation was conducted after the identification of the best-fit parameter set to be used in applications of the FWM, the best-fit parameter set from Calibration 2 was used.

The FWM was run six times using a range of sediment chemical concentrations to explore the effects of SWAC uncertainty on FWM estimates and the tendency of the FWM to over-estimate concentrations of total PCBs in tissue. The initial run used a SWAC of 380 µg/kg dw, which was the SWAC for the calibrated model; each additional run used a lower SWAC (see Table D.6-4) starting at the initial estimate of 380 µg/kg dw (Table D.6-5). Lower SWACs were investigated because the FWM over-estimated tissue concentrations for most species at 380 µg/kg dw and because SWACs generated from the baseline sediment database using Thiessen polygons and a new IDW parameterization (see Section 4.2.1.1 of the RI) resulted in lower values.

Table D.6-4 Sensitivity of FWM estimates to the SWAC

SPECIES	MEAN EMPIRICAL TOTAL PCB CONCENTRATION IN TISSUE (CALIBRATION 2) (µg/kg ww)	ESTIMATED TOTAL PCB CONCENTRATIONS IN TISSUE (µg/kg ww) BASED ON DIFFERENT SWACs (µg/kg dw)[a]						
		380	350	340[b]	300	250	200	150
Slender crab	250	381	356	348	315	274	**232**	191
Dungeness crab	510	565	534	**524**	484	434	383	333
Pacific staghorn sculpin	590	814	770	755	696	**622**	548	473
Shiner surfperch	1,000	**845**	797	781	716	635	555	474
English sole	1,600	**1,644**	1,534	1,498	1,352	1,169	987	804

[a] Best-fit parameter set from Calibration 2 was used for model runs.

[b] The SWAC of 340 µg/kg ww is from the most recent IDW parameterization using the baseline surface sediment database.

FWM – food web model

PCB – polychlorinated biphenyl

SWAC – spatially weighted average concentration

ww – wet weight

Bold values are estimates closest to mean empirical tissue data for that species.

Underlined values are underestimates.

Table D.6-5 Effects of SWAC on FWM performance

Species	SPAFs Based on FWM Runs Using Different SWACs (µg/kg dw)[a]						
	380	350	340[b]	300	250	200	150
Slender crab	1.5	1.4	1.4	1.3	1.1	**1.1**	1.3
Dungeness crab	1.1	1.0	**1.0**	1.1	1.2	1.3	1.5
Pacific staghorn sculpin	1.4	1.3	1.3	1.2	**1.1**	1.1	1.2
Shiner surfperch	**1.2**	1.3	1.3	1.4	1.6	1.8	2.1
English sole	**1.1**	1.0	1.0	1.1	1.3	1.6	1.9
Average SPAF	**1.3**	**1.2**	**1.2**	**1.2**	**1.2**	**1.4**	**1.6**

[a] Best-fit parameter set from Calibration 2 was used for model runs.

[b] The SWAC of 340 µg/kg ww is from the most recent IDW parameterization using the baseline surface sediment database.

dw – dry weight

FWM – food web model

IDW – inverse distance weighted

SPAF – species-predictive accuracy factor

SWAC – spatially weighted average concentration

ww – wet weight

Bold values are the best-fit estimate for that species compared to empirical tissue data.

Underlined values are SPAFs calculated from underestimated tissue concentrations.

The SWAC that produced the lowest average SPAF across species for the best-fit parameter set from Calibration 2 was 340 µg/kg dw (Table D.6-5), although SPAFs for each individual species were less than 2 for all SWACs greater than or equal to 200 µg/kg dw. Interestingly, the SWAC presented in the RI (Section 4.2.1.1 of the RI), based on an updated IDW interpolation, is 340 µg/kg dw.

When total PCB concentrations in sediment were reduced from 380 to 150 µg/kg dw, a change of 61%, the average change in tissue concentrations, across all species, was 54%. This indicates that the FWM responds in a proportional manner to changes in total PCB concentrations in sediment when the concentration of total PCBs in water is held constant. However, because the FWM was overestimating for all species (except shiner surfperch) when the SWAC was 380 µg/kg dw and underestimating for all species when the SWAC was 200 µg/kg dw, the average SPAF across species was not highly influenced. Overall, when the FWM was run using the best fit parameter set from Calibration 2, the average SPAF across species was similar when a SWAC of 340 µg/kg dw was used (SWAC that provided the optimal model performance and is currently used in the RI) compared to when a SWAC of 380 µg/kg dw was used (current estimate in the FWM).

D.7 Testing the FWM at Different Spatial Scales

To test the performance of the calibrated model for areas smaller than the LDW, the best-fit parameter set for Calibration 2 was applied to the four modeling areas (M1, M2, M3, and M4), and model estimates for each area were compared to area-specific empirical tissue data. These tests were conducted because EPA expressed an interest in potentially running the FWM at a scale smaller than the entire LDW and there were sufficient empirical data to test model performance at the scale of modeling areas. The best-fit parameter set for Calibration 2 was also used to test the performance of the FWM at specific intertidal locations to assess the ability of the model to estimate total PCB concentrations in clam tissue. Clams were modeled to support calculations of RBTCs in sediment for human health consumption scenarios. The model was not calibrated for clams because clams that are harvested for human use are present only in select intertidal areas, where the habitat is suitable, and the model was calibrated for the entire LDW.

D.7.1 MODELING AREAS

The FWM was applied to the four modeling areas (M1, M2, M3, and M4) (Figure D.3-1) to assess model performance for fish and crab species at a spatial scale smaller than the LDW.[33] Site-specific input parameters that were changed for modeling area runs were the total PCB concentration in the water column, the total PCB concentration in sediment, and the sediment OC content (Table D.7-1).

Table D.7-1. Modeling area-specific input parameter values

MODELING AREA	TOTAL PCB CONCENTRATION IN THE WATER COLUMN [a] (ng/L)	TOTAL PCB CONCENTRATION IN SEDIMENT (SWAC) [b] (µg/kg dw)	SEDIMENT ORGANIC CARBON CONTENT (SWAC) [c] (%)
M1	1.06	300	2.00
M2	1.29	270	2.05
M3	2.72	880	1.76
M4	2.16	190	1.72
LDW-wide	1.11	380	1.92

[a] Total PCB concentrations in the water column were derived for each modeling area from EFDC model output (as the average of 12 monthly averages in cells from the bottom three layers of the model for each modeling area) (Nairn 2006).

[b] SWACs of total PCBs in sediment (calculated using the 2006 IDW interpolation method) were calculated for modeling areas using the same interpolation grids generated for the entire LDW but clipped to modeling areas.

[c] Spatially weighted average percentages of sediment OC were calculated for modeling areas using Thiessen polygons generated for the entire LDW but clipped to modeling areas.

dw – dry weight LDW – Lower Duwamish Waterway PCB – polychlorinated biphenyl

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[33] The performance of the FWM was not tested at a subarea scale because fewer composite tissue samples were available at that scale.

At a modeling area scale, all estimates were within a factor of 2 of empirical data[34] for Areas M1, M2, and M4 (Table D.7-2). In Area M3, estimates for Dungeness crab, Pacific staghorn sculpin, and English sole ranged from 2.8 to 3.3 times higher than empirical data (Table D.7-2). The model performed reasonably well for shiner surfperch in Area M3 (estimates were 1.4 times higher than empirical data).

Table D.7-2. Application of the FWM to individual modeling areas

Modeling Area	Species	Mean Empirical Total PCB Concentration (µg/kg ww)	n	Best-Fit Parameter Set (Calibration 2) Estimated Total PCB Concentration (µg/kg ww)	SPAF	Over (+) or Under (-) Estimate
M1	slender crab	na	0	308	na	na
	Dungeness crab	570	3	470	1.2	-
	Pacific staghorn sculpin	720	1	675	1.1	-
	shiner surfperch	690	9	697	1.0	+
	English sole	1,600	6	1,321	1.2	-
M2	slender crab	250	1	295	1.2	+
	Dungeness crab	na	0	476	na	na
	Pacific staghorn sculpin	620	1	680	1.1	+
	shiner surfperch	1,300	6	693	1.9	-
	English sole	2,000	6	1,258	1.6	-
M3	slender crab	na	0	916	na	na
	Dungeness crab	420	1	1,364	3.2	+
	Pacific staghorn sculpin	590	1	1,970	3.3	+
	shiner surfperch	1,500	6	2,038	1.4	+
	English sole	1,400	6	3,954	2.8	+
M4	slender crab	na	0	294	na	na
	Dungeness crab	420	1	555	1.3	+
	Pacific staghorn sculpin	430	1	785	1.8	+
	shiner surfperch	580	4	772	1.3	+
	English sole	1,000	3	1,227	1.2	+

FWM – food web model
LDW – Lower Duwamish Waterway
n – number of composite samples
na – no empirical data available for modeling area
PCB – polychlorinated biphenyl
SPAF – species predictive accuracy factor
ww – wet weight

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[34] The modeling areas test was also run using the best fit parameter set from Calibration 1 and the Calibration 1 empirical dataset to compare results between Calibration 1 and 2. If one composite tissue sample for shiner surfperch in Area M2 with a high concentration of 18,400µg/kg was removed, the FWM performed equally well at the modeling area scale whether Calibration 1 or 2 was used, except that the model performance criterion was met for Pacific staghorn sculpin in M3 in Calibration 1 but not in Calibration 2.

Literature and statistical analyses of empirical total PCB tissue data (Section D.3.2) suggested that the FWM may perform better at the modeling area scale for Pacific staghorn sculpin and possibly better for shiner surfperch than for English sole and Dungeness and slender crab. English sole and crabs appear to be wider-ranging species relative to the spatial scale of the modeling areas (Section 4.2.1.4.2 of the RI).

Differences in home range size could possibly explain the poorer performance of the FWM for Dungeness crab and English sole in Area M3 relative to the better performance for shiner surfperch in Area M3. However, the poor performance of Pacific staghorn sculpin in Area M3 is not consistent with the hypothesis that Pacific staghorn sculpin may have smaller home ranges that roughly correspond with the modeling area scale (Section D.3.2). SWACs for Areas M1, M2, and M4 varied from 190 to 300 µg/kg dw, a difference of 80 to 190 µg/kg dw from the LDW-wide SWAC of 380 µg/kg dw. The SWAC for Area M3 was 880 µg/kg dw, a difference of 500 µg/kg dw from the LDW-wide SWAC of 380 µg/kg dw. If the exposure areas for Dungeness crab and English sole include the entire LDW, then the SWACs for these species would have been reasonably approximated by the LDW-wide SWAC of 380 µg/kg dw. Therefore, the good performance of the FWM for these species in Areas M1, M2, and M4 does not necessarily indicate that the modeling area SWACs represented the full exposure area (i.e., home ranges), but instead could be explained by the similarity of the SWACs in these modeling areas to the LDW-wide SWAC. If the home range of shiner surfperch is smaller than the LDW and corresponds roughly with the modeling areas, then sediment exposure should have been better approximated by modeling area SWACs.

In summary, the FWM performed well for slender crab in Area M2; however, it is difficult to make any conclusions about modeling areas and slender crabs because Area M2 was the only modeling area with available empirical tissue data for this species. For Dungeness crab, Pacific staghorn sculpin, and English sole, the FWM performed within the SPAF model performance criterion (SPAF ≤ 2) for all modeling areas (with available empirical data) except Area M3. Some loss of performance is to be expected if the model is applied on a smaller spatial scale because of the following:

- Greater standard error because of smaller tissue sample sizes when the data are split by tissue sampling area
- Potential differences in diet at the modeling area scale versus the LDW-wide scale because of potential differences in the relative abundance of different types of prey
- Potential differences in the spatial distributions of habitat and sediment contamination (both for the modeled species and their prey)
- Potential differences in factors that affect the bioavailability of PCBs (e.g., differences in PCB congener patterns or in sediment organic carbon content)

- Potential differences in water exposure at the modeling area spatial scale relative to LDW-wide
- Movement of individuals (of the sampled population or their prey) across modeling area boundaries

Applying the FWM at the modeling area scale may not be appropriate for Dungeness crab, Pacific staghorn sculpin, and English sole.

For shiner surfperch, the FWM performed within the SPAF model performance criterion (SPAF ≤ 2) for all modeling areas. These results indicate that applying the FWM at the smaller modeling area scale for this species may be appropriate, although uncertainty is higher at a modeling area scale, as discussed in Section D.3.2.

D.7.2 CLAM INTERTIDAL AREAS

To test how well the model estimated total PCB concentrations in clam tissue, the model was run for the 10 clam intertidal areas, and estimated total PCB tissue concentrations in clams were compared to empirical clam tissue data. Four of the 10 intertidal areas (i.e., C2, C3, C7, and C10) had two sampling locations each, for a total of 14 locations (Figure D.3-1). Co-located tissue and sediment samples were collected at each of the 14 clam sampling locations.

The best-fit parameter set for Calibration 2 was used for all 14 clam sampling locations, except for three parameters that were location-specific. Location-specific input parameters that were changed for clam runs were the total PCB concentration in the water column, the total PCB concentration in sediment, and the sediment OC content (Table D.7-3).

Table D.7-3. Location-specific input parameters values for 14 clam intertidal locations in the LDW

LOCATION ID	MODELING AREA	TOTAL PCB CONCENTRATION IN THE WATER COLUMN[a] (ng/L)	TOTAL PCB CONCENTRATION IN SEDIMENT[b] (µg/kg dw)	ORGANIC CARBON IN SEDIMENT[b] (%)
C1	M1	1.1	3.1	0.47
C2-1	M1	1.1	56	1.82
C2-2	M1	1.1	99	1.06
C3-1	M1	1.1	52	0.93
C3-2	M1	1.1	20 U	1.31
C4	M2	1.3	69	1.4
C5	M2	1.3	53	0.32
C6	M2	1.3	61	1.24
C7-1	M3	2.7	1,000	1.55
C7-2	M3	2.7	380	0.78
C8	M3	2.7	3,300	2.11
C9	M3	2.7	35	0.56

LOCATION ID	MODELING AREA	TOTAL PCB CONCENTRATION IN THE WATER COLUMN[a] (ng/L)	TOTAL PCB CONCENTRATION IN SEDIMENT[b] (µg/kg dw)	ORGANIC CARBON IN SEDIMENT[b] (%)
C10-1	M3	2.7	6,600	1.63
C10-2	M3	2.7	15,000	2.27

[a] The total PCB concentration in the water column for each clam intertidal area was assumed to be the same as the corresponding modeling area based on output from the bottom three layers of the EFDC model.

[b] Total PCB concentrations in sediment and organic carbon content at specific intertidal locations were based on composite sediment samples collected at the same locations as the clam tissue samples. These sediment samples represented total PCB concentrations and organic carbon content over the area from which clams were collected at a given intertidal location.

dw – dry weight
ID – identification
LDW – Lower Duwamish Waterway
PCB – polychlorinated biphenyl
U – not detected at the reporting limit shown

Compared to the empirical data for clams, estimated total PCB concentrations in clams for 12 of the 14 clam intertidal locations had SPAFs < 2 (Table D.7-4). These results indicate that the model generally performed well for locations where total PCB concentrations in the sediment are 3,300 µg/kg dw or lower (Table D.7-4).

The FWM overestimated total PCB concentrations in clam tissues at two locations (C10-1 and C10-2) with high concentrations in sediment (6,600 and 15,000 µg/kg dw, respectively). SPAFs at these two locations were 4.1 and 7.0, respectively. Empirical total PCB concentrations in clam tissue at locations C10-1 and C10-2 (320 and 330 µg/kg ww, respectively) were in the same range as tissue concentrations (220 to 580 µg/kg ww) from other locations with total PCB concentrations in sediment ranging from 380 to 3,300 µg/kg dw (Table D.7-4). These results may indicate that less-contaminated areas adjacent to C10-1 and C10-2 may influence total PCB exposure of clams at those locations or that the FWM does not perform well for clams at locations with total PCB concentrations higher than 3,300 µg/kg ww.

Table D.7-4. Application of the FWM for clams

LOCATION ID	EMPIRICAL TOTAL PCB CONCENTRATIONS IN CLAM TISSUES (µg/kg ww)	FWM-ESTIMATED TOTAL PCB CONCENTRATION IN CLAM TISSUES (µg/kg ww)	SPAF	OVER (+) OR UNDER (-) ESTIMATE
C1	24	23	1.0	-
C2-1	24	31	1.3	+
C2-2	29	48	1.7	+
C3-1	33	37	1.1	+
C3-2	32	26	1.2	-
C4	31	41	1.3	+
C5	43	68	1.6	+

Location ID	Empirical Total PCB Concentrations in Clam Tissues (µg/kg ww)	FWM-Estimated Total PCB Concentration in Clam Tissues (µg/kg ww)	SPAF	Over (+) or Under (-) Estimate
C6	34	41	1.2	+
C7-1	220	253	1.2	+
C7-2	250	188	1.3	-
C8	580	575	1.0	-
C9	50	72	1.4	+
C10-1	320	1,312	4.1	+
C10-2	330	2,301	7.0	+

FWM – food web model

ID – identification

PCB – polychlorinated biphenyl

SPAF – species predictive accuracy factor

ww – wet weight

An NRS analysis for clams was conducted using the same methods described in Section D.6.1.2. In the NRS analysis, input values for a given set of parameters were varied, one at a time, from their minimum to their maximum values in the parameter sets that passed the model performance filter. All other FWM parameters were held at their best-fit parameter set values. In order to understand the importance of a parameter, it is helpful to compare the NRS value to the estimated total PCB concentration (Table D.7-5). This comparison provides a sense of the magnitude of the uncertainty associated with a specific parameter relative to the estimate).

Table D.7-5 Results of NRS analysis at three clam intertidal locations

Parameter	NRS Value for clams (µg/kg ww)		
	Clam Location C3-1	Clam Location C7-2	Clam Location C10-2
Concentration of total PCBs in the water column	48	48	47
Relative fraction of porewater ventilated by clams[a]	29	280	3800
Density of lipids	5.4	33	430
Log octanol-water partition coefficient (Log K_{OW}) for total PCBs	4.4	6.5	69
Dietary absorption efficiency of NLOM (ε_N) for clams[a]	2.8	16	270
Lipid content of zooplankton	0.045	1	25
For reference:			
Estimated total PCB concentration in clam tissue based on best-fit parameter set (µg/kg ww)	37	188	2,301
Total PCB concentration in sediment at clam intertidal location (µg/kg dw)	52	380	15,000

[a] For the NRS analysis, the maximum and minimum fraction of porewater ventilation for clams was assumed to be the same as the values used for benthic invertebrates.

dw – dry weight
PCB – polychlorinated biphenyl
NLOM – non-lipid organic matter
NRS – nominal range sensitivity
ww – wet weight

The NRS analysis was conducted for three intertidal locations, representing a range of total PCB concentrations in sediment (52, 380, and 15,000 µg/kg dw). Testing the sensitivity and uncertainty of the FWM at three locations with differing total PCB concentrations in sediment provides insight into how the sensitivity of the FWM changes with environmental conditions. Six of the 20 parameters tested in the NRS analysis had an effect on estimated total PCB concentrations in clams (Table D.7-5).

The two parameters with the greatest potential influence on estimated total PCB concentrations in clam tissues were the total PCB concentrations in the water column and the relative fraction of porewater ventilated by clams. Use of the EFDC model at the scale of the clam intertidal areas (e.g., C10) may be explored in the future to potentially reduce the uncertainty associated with estimates of total PCB concentrations in the water column. The fraction of porewater ventilated by clams is a highly uncertain value.

Because NRS values for total PCB concentrations in the water column were similar at each of the three locations, and estimated tissue concentrations decreased with decreasing total PCB concentration in sediment, the relative influence of total PCB concentrations in the water column increased with decreasing sediment concentrations (Table D.7-5). These results indicate that as total PCB concentrations in sediment decrease, FWM estimates of total PCB concentrations for clams become more sensitive to total PCB concentrations in water.

In summary, for the six parameters that had an effect on FWM clam tissue estimates, the relative rankings of parameters by NRS value, and thus the relative influence of those parameters on the uncertainty of FWM estimates, were similar to those for other modeled species in the LDW (Section D.6.2.2).

D.8 Application of the FWM to Calculate Sediment RBTCs

RBTCs represent the concentrations that correspond to specific thresholds of risk.[35] In Section 8 of the RI, RBTCs were estimated for various human exposure pathways for risk driver chemicals identified in the baseline risk assessments (Appendices A and B). The FWM was used to generate sediment RBTCs for total PCBs for exposure through the ingestion of aquatic species (seafood) by humans and river otter.

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[35] For example, a 1×10^{-6} RBTC is the tissue concentration (or the associated sediment concentration) at which the excess cancer risk equals 1×10^{-6} for a specific human exposure scenario.

As discussed in Section D.5.3, parameter sets from Calibration 2 of the FWM were selected for use in estimating RBTCs. This section describes the four main steps of the process used to generate estimates of sediment RBTCs for total PCBs. Briefly, sediment and water input parameters were selected, and then the model was run iteratively to estimate the tissue concentrations that correspond to each set of input parameters. The estimated tissue concentrations were then used in the human health risk equations, and the sediment concentrations associated with particular risk thresholds were identified. Details for each of these steps are discussed below.

Step 1. Estimate total PCB concentrations in surface sediment and in overlying water in the water column

To estimate sediment RBTCs, the FWM required paired inputs of total PCB concentrations in surface sediment and overlying water; both of these input parameters are important for the model. The surface sediment concentrations was represented by the SWAC for the LDW, which has been estimated to be 380 µg/kg dw. [36] The EFDC model[37] estimated that the total PCB concentration in the water column ranged from 1.06 to 2.72 ng/L (mean of 1.4 ng/L) in the three cells of the model that represent the three bottom layers of the LDW.[38] The total PCB concentration in water grab samples collected in the LDW just above the bottom in 2005 ranged from 0.13 to 3.2 ng/L (Mickelson and Williston 2006).

In the future, total PCB concentrations in sediment and water are likely to be lower following sediment remediation and source control actions within the LDW. Because these concentrations are not yet known, and instead of artificially confining the range based on estimates of future concentrations, the FWM was run with total PCB concentrations in sediment ranging from 0 to 380 µg/kg dw. Total PCB concentrations in sediment will never be 0 µg/kg dw because of local, regional, and global sources of PCBs. The low end of the range (approaching zero PCBs in sediment) was modeled to estimate total PCB concentrations in tissues at very low concentrations in sediment.

The EFDC model was not used to estimate future total PCB concentrations in the water column for each concentration in sediment; these estimates would have been highly uncertain because of the numerous modeling assumptions that would have been required (e.g., assumed spatial distributions of PCBs in sediment, including

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[36] The 2006 IDW parameterization used to estimate the SWAC for the FWM was discussed in the *Technical Memorandum: GIS Interpolation of Total PCBs in LDW Surface Sediment* (Windward 2006b). The baseline surface sediment dataset used in this application was the same dataset used in the risk assessments and thus did not include surface sediment data collected during Round 3 in 2006.

[37] Estimates from the EFDC model were received in October 2006. Additional information on the EFDC model is provided in a memo produced by King County (Nairn 2007).

[38] Most of the fish and crab species being modeled spend the majority of their time in deeper waters in the LDW; thus the EFDC model predictions for the bottom three cells were deemed most appropriate for use in the FWM.

values for East and West Waterway). In addition to these uncertainties, the simulation run time required to process each sediment scenario would have required significant computational time (Nairn 2007). Instead, total PCB concentrations in the water column were simulated for current conditions and for a uniform bed concentration of 40 ug/kg dw. These endpoints were used to interpolate total PCBs concentrations in water corresponding to sediment concentrations; this interpolation was divided into three ranges to reflect the large uncertainty in the model input parameters and to simplify the execution and analysis of the FWM.

Because the EFDC model was not used, future total PCB concentrations in the water column were divided into three general ranges. To define these ranges, total PCB concentrations in the water column and in surface sediment were assumed to be related. For total PCB concentrations in surface sediment between 250 and 380 µg/kg dw, a water concentration of 1.2 ng/L was assumed (Table D.8-1). This concentration is slightly below the EFDC model-estimated LDW-wide mean concentration of 1.4 ng/L. The LDW-wide mean estimated by the EFDC model was not used to represent the upper end of the sediment range because that value (1.4 ng/L) corresponds to the current sediment values. The water concentration selected for the upper end of the sediment range was required to correspond to a range of sediment concentrations that trended downwards from the current sediment concentration. Thus, a lower water concentration (1.2 ng/L) was selected to represent the water concentration at the upper end of the sediment range (250 to 380 µg/kg dw). For the lower sediment ranges, total PCB concentrations in water were assumed to be proportionately lower (Table D.8-1). As a point of reference, total PCB concentrations in water from the Green River, which is the upstream source of surface water to the LDW, ranged from 0.1 to 0.8 ng/L in 2005 and from 0.04 to 1.5 ng/L in 2007 (Mickelson and Williston 2006; Williston 2007). The total PCB concentration in water in Elliott Bay, the source of saline water to the LDW, ranged from 0.056 to 0.089 ng/L in 2005 (Mickelson and Williston 2006).

Table D.8-1. Assumed relationship between total PCB concentrations in sediment and overlying water

RANGE OF TOTAL PCB CONCENTRATIONS IN SEDIMENT (µg/kg dw)	ASSUMED TOTAL PCB CONCENTRATIONS IN THE WATER COLUMN(ng/L)
0 – 100	0.6
100 – 250	0.9
250 – 380	1.2

dw – dry weight

PCB – polychlorinated biphenyl

Step 2. Run the model probabilistically using Monte Carlo simulation

The FWM was run probabilistically as a Monte Carlo simulation using Crystal Ball® software, allowing numerous model runs for small incremental changes in total PCB concentrations in sediment, with concentrations ranging from 0 to 380 µg/kg dw. The total PCB concentration in water for each of these runs also varied, per the relationship described in Table D.8-1.

Results of these model runs (i.e., estimates of total PCB concentrations in tissues) using the best-fit parameter set are displayed graphically in Figure D.8-1. The "steps" in estimated total PCB concentrations in tissue occurred at total PCB concentrations in sediment corresponding to the three sediment/water intervals defined in Step 1.

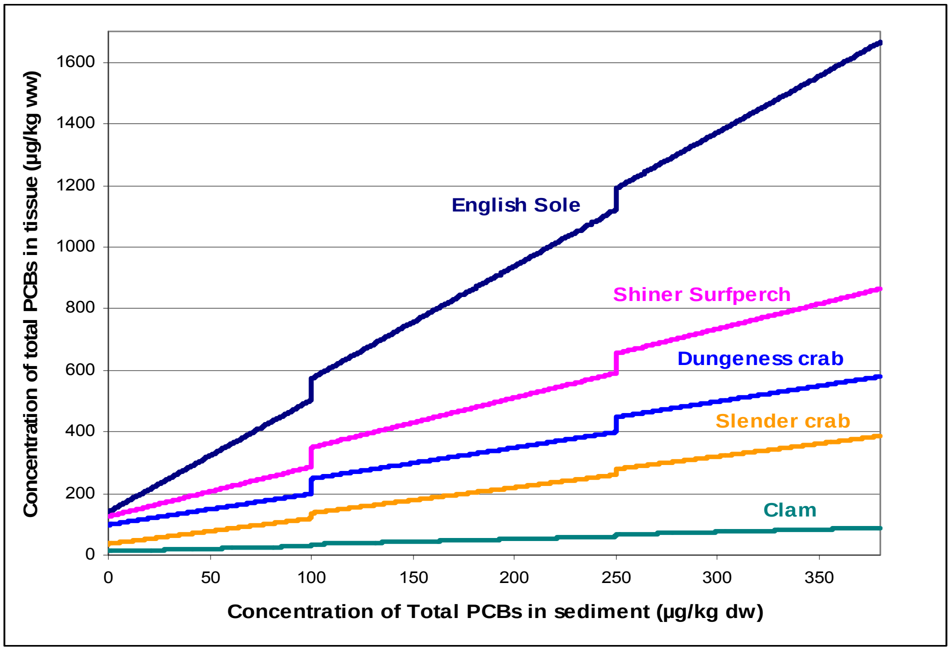

Figure D.8-1. Total PCB concentrations in whole-body tissues of seafood species as a function of total PCB concentrations in sediment

The FWM was also used to estimate a range of total PCB concentrations in each tissue type. Parameter sets that passed the model performance criterion (SPAF ≤ 2 for all species) were reviewed to determine which set produced the highest and lowest estimated total PCB concentrations for each species, regardless of the performance of other species.

Figures D.8-2 and D.8-3 present the results for Dungeness crab and English sole, respectively, as example tissues. The red lines represent the FWM estimates using the best-fit parameter set. The yellow and orange lines are the lower- and upper-bound estimates, respectively.

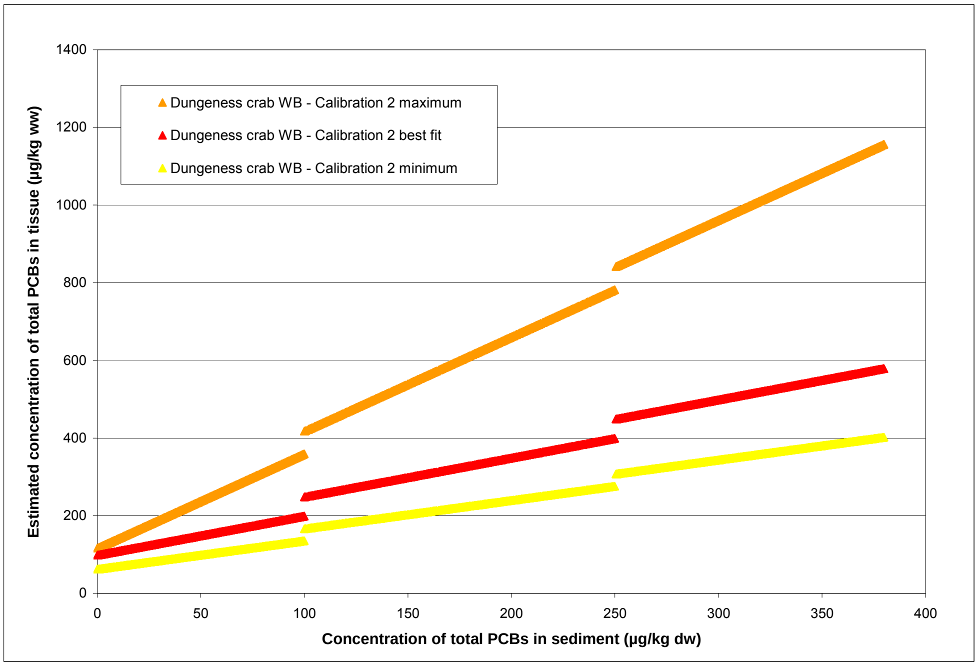

Figure D.8-2. **Estimated total PCB concentrations in whole-body Dungeness crab using best-fit, maximum, or minimum parameter sets as a function of total PCB concentration in sediment**

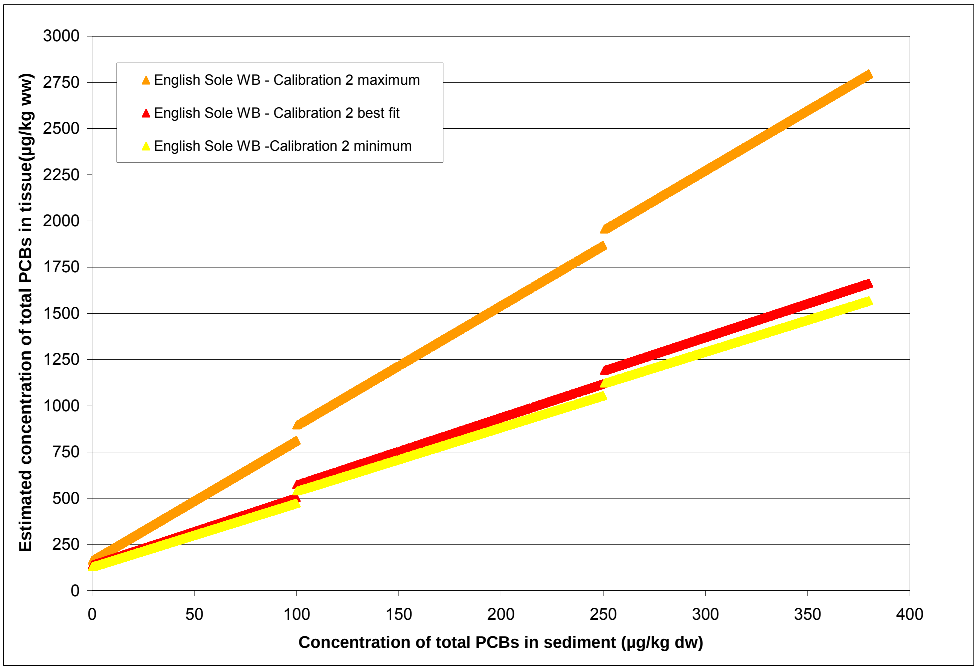

Figure D.8-3. **Estimated total PCB concentrations in whole-body English sole using best-fit, maximum, or minimum parameter sets as a function of total PCB concentration in sediment**

Because of the way the range of estimates was defined, the upper-bound estimate (orange line) could exceed the best-fit estimate (red line) by up to a factor of 2 at any given sediment total PCB concentration, as was frequently the case. Similarly, the lower-bound estimate (yellow line) could be as low as 50% of the best-fit estimate (red line). However, the lower-bound estimates were more similar to the best-fit estimates because the greatest underestimate of the FWM was 36% (vs. the 50% allowed). Therefore, the specific model criterion selected (SPAF ≤ 2 for all species) did not result in the elimination of any of the parameter sets that underestimated the mean and thus did not influence the lower-bound estimate. The upper-bound estimate would have been higher if the criterion had been less stringent (i.e., a SPAF threshold > 2).

Step 3. Calculate risk estimates using the output generated by each FWM run

The estimated total PCB concentrations in tissue for the modeled species,[39] corresponding to each of the thousands of FWM runs associated with incremental steps in total PCB concentration in sediment, were entered into the human health and ecological receptor risk equations. These estimated tissue concentrations were used in the risk equations in the same way that exposure point concentrations (EPCs) were used in the risk assessments.

Excess cancer risks and non-cancer hazards were estimated using these estimates for each of the seafood ingestion scenarios evaluated in the HHRA (Appendix B) and in the ERA (Appendix A) for river otters. Risks were calculated using the best-fit, maximum, and minimum estimates over the full range of paired total PCB concentrations in sediment and water.

To determine the upper-bound EPCs for each risk scenario, the highest estimates for each species were combined to estimate the total PCB concentration in a given market basket selection. To determine the lower-bound EPCs for each risk scenario, the lowest estimates for each species were combined to estimate the lowest total PCB concentration in a particular market basket selection. For receptors that consume multiple species, this approach may lead to an over- or underestimate of possible exposures and associated risks; parameter sets were selected on a species-by-species basis rather than as a single set of parameters that resulted in the highest (or lowest) tissue concentrations across all species consumed by a particular receptor.

This range in EPCs reflects some of the uncertainty in the FWM. Other uncertainties associated with the FWM or with risk assumptions were not quantified and thus were not captured by the RBTC range. Uncertainties associated with the risk assumptions

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[39] The FWM estimated total PCB concentrations in whole-body organisms. In the HHRA, some of the seafood ingestion scenarios included the consumption of edible meat (crab) or fillet (English sole). Therefore, conversion factors were developed. The conversion factors used to convert total PCB concentrations in whole-body organisms to lower concentrations in edible meat or fillet concentrations were 0.295 for slender crab, 0.139 for Dungeness crab, and 0.526 for English sole.

are discussed in Appendices A and B, and FWM uncertainties are discussed in Section D.6.

Step 4. Identify the sediment RBTC associated with a given risk threshold

Because of the large number of tissue predictions and risks generated for each scenario, it was necessary to devise a method to organize the data so that RBTCs could be efficiently identified for any of the risk thresholds of interest (i.e., 1 x 10^{-4}, 1 x 10^{-5}, and 1 x 10^{-6}). Thus, the risk estimates described in Step 3 were compiled in a matrix table to facilitate the identification of the total PCB concentration in sediment corresponding to a selected excess cancer risk threshold (1 x 10^{-4}, 1 x 10^{-5}, or 1 x 10^{-6}) or a non-cancer hazard (hazard quotient = 1) for each of the exposure scenarios (Table D.8-2).

Table D.8-2 demonstrates the manner in which sediment RBTCs were identified for two of the seafood consumption scenarios. The full matrix table, which included all of the seafood consumption scenarios evaluated in the HHRA (Appendix B) and the river otter scenario evaluated in the ERA (Appendix A), was too large to reproduce in this format.

Table D.8-2 presents 16 of the many model runs that were conducted. The right-hand columns show excess cancer risk for adult Tulalip seafood consumption scenarios, and the bold cells identify specific excess cancer risk levels (1 x 10^{-4} for the adult Tulalip reasonable maximum exposure (RME) and 1 x 10^{-5} for adult Tulalip central tendency [CT]). The sediment value corresponding to those excess cancer risk values are shown in bold type. For the adult tribal RME scenario based on Tulalip data, a sediment RBTC of 19 µg/kg dw total PCBs was associated with the 1 x 10^{-4} excess risk level; for the adult tribal CT scenario based on Tulalip data, a sediment RBTC of 53 µg/kg dw total PCBs was associated with the 1 x 10^{-5} excess risk level. Sediment RBTCs for other risk scenarios and risk thresholds are presented in Section 8.0 of the main document.

In total, three sediment RBTCs were identified for each risk scenario/risk threshold: a best-fit sediment RBTC (based on the best-fit parameter set) and upper and lower bound RBTCs. These sediment RBTCs are presented in Figure 8.6 in the RI.

Table D.8-2. Matrix table relating excess risk levels for two seafood consumption scenarios to sediment concentrations to generate sediment RBTCs for total PCBs

Total PCB Concentrations Used as Input Values		Estimated Total PCB Tissue Concentration (µg/kg ww)										Excess Cancer Risk Estimates Based on FWM Output	
Sediment (µg/kg dw)	Water (ng/L)	Clam	Juvenile Fish	Slender Crab WB	Slender Crab EM	Dungeness Crab WB	Dungeness Crab EM	Pacific Staghorn Sculpin	Shiner Surf-perch	English Sole WB	English Sole Fillet	Adult tribal RME (Tulalip data)	Adult tribal CT (Tulalip data)
1	0.6	12	100	36	11	99	14	137	126	141	74	7.8×10^{-5}	5.5×10^{-6}
10	0.6	14	108	44	13	108	15	151	142	175	92	8.9×10^{-5}	6.3×10^{-6}
19	0.6	15	115	52	15	117	16	164	156	208	110	**1.0×10^{-4}**	7.1×10^{-6}
30	0.6	17	124	61	18	128	18	180	174	248	131	1.1×10^{-4}	8.0×10^{-6}
40	0.6	19	133	69	20	139	19	195	190	285	150	1.3×10^{-4}	8.9×10^{-6}
50	0.6	21	141	77	23	149	21	210	206	321	169	1.4×10^{-4}	9.7×10^{-6}
53	0.6	21	144	80	24	152	21	214	211	333	175	1.4×10^{-4}	**1.0×10^{-5}**
60	0.6	22	149	86	25	159	22	225	222	358	188	1.5×10^{-4}	1.1×10^{-5}
70	0.6	24	158	94	28	169	23	239	238	394	207	1.6×10^{-4}	1.1×10^{-5}
90	0.6	27	174	110	33	189	26	269	270	467	246	1.9×10^{-4}	1.3×10^{-5}
100	0.6	29	183	119	35	199	28	284	287	504	265	2.0×10^{-4}	1.4×10^{-5}
140	0.9	42	266	170	50	288	40	411	414	719	378	2.9×10^{-4}	2.0×10^{-5}
160	0.9	45	283	186	55	309	43	441	446	792	417	3.1×10^{-4}	2.2×10^{-5}
180	0.9	49	299	203	60	328	46	470	478	864	454	3.4×10^{-4}	2.4×10^{-5}
200	0.9	52	316	219	65	348	48	499	510	936	492	3.6×10^{-4}	2.5×10^{-5}
250	0.9	60	358	261	77	399	55	574	591	1,120	589	4.2×10^{-4}	3.0×10^{-5}

Note: Values shown are excerpt of the full table used to estimate RBTCs. The excess cancer risk estimate on the right side of the table corresponds with the sediment concentration on the left side of the table for each row.

CT – central tendency
EM – edible meat
FWM – food web model
PCB – polychlorinated biphenyl
RBTC – risk-based threshold concentration
RME – reasonable maximum exposure
WB – whole-body
ww – wet weight

Bold values are those called out in the example discussed in the text.

D.9 Summary

The FWM was developed to estimate the relationship between total PCB concentrations in tissue and sediment in order to estimate RBTCs in sediment for the RI. The FWM may also be used in the FS to assess residual risks that may remain following various sediment cleanup alternatives.

The FWM structure was based on the Arnot and Gobas model (Arnot and Gobas 2004a), a steady-state bioaccumulation model. The FWM provides estimates of total PCB concentrations in the tissues of nine species or species groups, based on bioaccumulation of total PCBs from the sediment and water column. Many of the species included in the FWM were ecological receptors, prey for ecological receptors, or consumed by humans, as described in the risk assessments (Appendices A and B).

Input parameter values and distributions for the model were based on literature-derived and site-specific environmental data. The model was then calibrated to identify sets of parameter values that best estimated empirical tissue total PCB concentration data. For many model input parameters, distributions of estimates of mean values were developed to reflect uncertainty in their values. Calibration was performed using a probabilistic approach in order to systematically explore all combinations of plausible parameter sets and their corresponding estimated total PCB concentrations in tissue.

Through the calibration process, a best-fit parameter set was identified that estimated total PCB concentrations for all modeled fish and crab species within a factor or 2 (1.2 on average) of empirical data.

To better understand the strengths and limitations of the model, model sensitivities and uncertainties were evaluated. The parameters that most influenced model uncertainty were dietary absorption for crabs, relative fractions of benthic versus pelagic food items in the diet of various modeled species, and parameters that characterized prey species (such as lipid content and porewater ventilation rate). In general, the parameters that most influenced model uncertainty had broad ranges of values derived from the literature.

The FWM was calibrated at a LDW-wide spatial scale. It was tested at smaller scales within the LDW to assess its performance, in part because home ranges of many of the modeled species were uncertain. Based on these analyses, application of the FWM appeared to be inappropriate at the modeling area scale for most species. The FWM was performed well for clams at locations with sediment total PCB concentrations of 3,300 µg/kg dw or lower.

The FWM was used to develop sediment RBTCs for total PCBs. Following a four-step process, sediment RBTCs associated with various risk thresholds for various seafood ingestion scenarios were identified. Best-fit sediment RBTCs were identified as well as upper- and lower-bound RBTCs. Upper and lower bounds were developed based on the model performance criterion and do not reflect the total range of uncertainty in the sediment RBTCs. Sediment RBTCs are presented in Section 8 of the RI.

D.10 References

Alcoa Inc. 1995. The Grasse River: Non-time critical removal action documentation report, December 1995 [online]. Alcoa Inc. [Cited October 11, 2006]. Available from: http://www.thegrasseriver.com/ppt/Nov20CAP/NTCRA.htm.

Angell CL, Miller BS, Wellings SR. 1975. Epizootiology of tumors in a population of juvenile English sole (*Parophrys vetulus*) from Puget Sound, Washington. J Fish Res Bd Can 32:1723-1732.

Arnot J. 2005. Personal communication (e-mail to Fiona McNair, Windward Environmental, LLC, Seattle, WA, regarding sensitivity and uncertainty analyses). Canadian Environmental Modeling Center, Peterborough, Ontario, Canada. July 26.

Arnot J. 2006a. Personal communication (e-mail to Fiona McNair, Windward Environmental, LLC, Seattle, WA, regarding food web model lipid density range). Consultant, Canadian Environmental Modeling Center, Peterborough, Ontario, Canada. February 10, 2006.

Arnot JA. 2006b. Personal communication (e-mail to Fiona McNair, Windward Environmental, regarding PCB metabolism). Canadian Environmental Modelling Centre, Trent University, Peterborough, ON. January 4, 2006.

Arnot JA, Gobas FAPC. 2004a. A food web bioaccumulation model for organic chemicals in aquatic ecosystems. Environ Toxicol Chem 23:2343-2355.

Arnot JA, Gobas FAPC. 2004b. A food web bioaccumulation model for organic chemicals in aquatic ecosystems: Environ Toxicol Chem, SETAC Supplemental Data Archive. (http://etc.allenpress.com).

Atar HH, Secer S. 2003. Width/length-weight relationships of the blue crab (*Callinectes sapidus* Rathbun 1896) population living in Beymelek Lagoon Lake. Turk J Vet Anim Sci 27:443-447.

Barnes RSK, Mann KH. 1980. Fundamentals of aquatic ecosystems.

Berge JA, Brevik EM. 1996. Uptake of metals and persistent organochlorines in crabs (*Cancer pagurus*) and flounder *Platichthys flesus*) from contaminated sediments: mesocosm and field experiments. Mar Pollut Bull 33(1-6):46-55.

Bernard FR. 1979. The food of Hecate Strait crabs, August 1977. Fisheries and Marine Service Manuscript Report No. 1464. Pacific Biological Station, Department of Fisheries and the Environment, Nanaimo, BC.

California Academy of Sciences. 2002. *Platynereis bicanaliculata* (Baird, 1963). Research, IZG, San Francisco 2000, Animal images, Annelida, Polychaeta, Family Nereididae, "errant" polychaetes [online]. California Academy of Sciences, San Francisco, CA. [Cited 12/8/06]. Available from: http://www.calacademy.org/research/izg/SFBay2K/Platynereis%20bicaniculata.htm.

Conover RJ. 1966. Assimilation of organic matter by zooplankton. Limnol Oceanogr 11:338-345.

Cruz-Rivera E, Hay ME. 2001. Macroalgal traits and the feeding and fitness of an herbivorous amphipod: the roles of selectivity, mixing, and compensation. Mar Ecol Prog Ser 218:249-266.

Day DE. 1976. Homing behavior and population stratification in central Puget Sound English sole (*Parophrys vetulus*). J Fish Res Board Can 33:287-282.

EPA. 2006. Case study data and information. United Heckathorn Superfund Site, Richmond, CA, May 16, 2006. U.S. Environmental Protection Agency.

. US Environmental Protection Agency, Washington, DC.

Fauchald K, Jumars P. 1979. The diet of worms: a study of polychaete feeding guilds. Oceanogr Mar Biol Ann Rev 17:193-284.

Field J. 2006. Personal communication (comments on dietary fractions of phytoplankton and zooplankton for benthic invertebrates, juvenile fish and shiner surfperch; at 8/12/06 meeting of LDWG, EPA, and NOAA representatives; subsequent telephone conversation with F. McNair, Windward Environmental on 5/17/06). Office of Response and Restoration, National Oceanic and Atmospheric Administration, Seattle, WA. August 12 and 17, 2006.

Fresh KL, Rabin D, Simenstad CA, Salo EO, Garrison K, Matheson L. 1979. Fish ecology studies in the Nisqually Reach area of southern Puget Sound, Washington. FRI-UW-7904. Prepared for Weyerhauser Company. Fisheries Research Institute, University of Washington, Seattle, WA.

Giles SL, Cordell JR. 1998. Zooplankton composition and abundance in Budd Inlet, Washington. Puget Sound Research 1998, March 12-13, 1998, Seattle, WA.

Gobas FAPC. 1988. Bioaccumulation of hydrophobic organic chemicals in fish. PhD thesis. University of Toronto, Toronto, ON, Canada.

Gobas FAPC. 1993. A model for predicting the bioaccumulation of hydrophobic organic chemicals in aquatic food-webs: application to Lake Ontario. Ecol Model 69:1-17.

Gobas FAPC, Arnot J. 2005. San Francisco Bay PCB food web bioaccumulation model: final technical report. Prepared for the Clean Estuary Partnership, San Francisco, CA.

Gobas FAPC. 2006. Mathematical models available for download [online]. Environmental Toxicology Research Group, School of Resource and Environmental Management, Simon Fraser University, Burnaby, BC, Canada. Updated 11/23/05. [Cited 2/12/06]. Available from: http://www.rem.sfu.ca/toxicology/models/models.htm.

Gobas FAPC, Mackay D. 1987. Dynamics of hydrophobic organic chemical bioconcentration in fish. Environ Toxicol Chem 6:495-504.

Gobas FAPC, MacLean LG. 2003. Sediment-water distribution of organic contaminants in aquatic ecosystems: the role of organic carbon mineralization. Environ Sci Tech 37(4):735-741.

Gobas FAPC, Wilcockson J. 2003. San Francisco Bay PCB food-web model: RMP technical report. Regional Monitoring Program, San Francisco, CA.

Gobas FAPC, McCorquodale JR, Haffner GD. 1993. Intestinal absorption and biomagnification of organochlorines. Environ Toxicol Chem 12:567-576.

Gobas FAPC, Wilcockson JB, Russell RW, Haffner GD. 1999. Mechanism of biomagnification in fish under laboratory and field conditions. Environ Sci Technol 33:133-141.

Gordon DC, Jr. 1966. The effects of the deposit feeding polychaete *Pectinaria gouldii* on the intertidal sediments of Barnstable Harbor. Limnol Oceanogr 11:327-332.

Gotshall DW. 1978. Catch-per-unit-of-effort studies of northern California Dungeness crabs, *Cancer magister*. Cal Fish Game 64(3):189-199.

Harbo RM. 2001. Shells & shellfish of the Pacific Northwest. Harbour Publishing, Madeira Park, BC.

Hawker DW, Connell DW. 1988. Octanol-water partition coefficients of polychlorinated biphenyl congeners. Environ Sci Tech 22:382-387.

Jensen GC. 1995. Pacific coast crabs and shrimps. Sea Challengers, Monterey, CA.

King County. 1999. King County combined sewer overflow water quality assessment for the Duwamish River and Elliott Bay. Vol 1: Overview and interpretation, plus appendices. King County Department of Natural Resources, Seattle, WA.

King County. 2005. Duwamish River, Elliott Bay, and Green River water column PCB congener survey: sampling and analysis plan. King County Department of Natural Resources and Parks, Seattle, WA.

Kozloff EN. 1983. Seashore life of the northern Pacific coast: an illustrated guide to northern California, Oregon, Washington, and British Columbia. University of Washington Press, Seattle, WA.

Kraaij R, Seinen W, Tolls J, Cornelissen G, Belfroid AC. 2002. Direct evidence of sequestration in sediments affecting the bioavailability of hydrophobic organic chemicals to benthic deposit feeders. Environ Sci Technol 36:3525-3529.

Kuroshima R, Sato M, Yoshinaka R, Ikeda S. 1987. Nutritional quality of the wild zooplankton as a living feed for fish larvae. Suisanzoshoku 35(2):113-117.

Lassuy DR. 1989. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (Pacific Northwest). English sole. USFW biological report 82(11.101). Coastal Ecology Group, US Army Corps of Engineers, Vicksburg, MS and National Wetlands Research Center, US Fish and Wildlife Service, Slidell, LA.

Mackintosh CE, Maldonado J, Hongwu J, Hoover N, Chong A, Ikonomou MG, Gobas FAPC. 2004. Distribution of phthalate esters in a marine aquatic food web: comparison to polychlorinated biphenyls. Environ Sci Tech 38:2011-2020.

MarLIN. 2002. *Macoma balthica* and *Abra alba* in infralittoral muddy sand or mud. Marine Life Information Network (MarLIN): Biology and sensitivity key information sub-programme [online]. Marine Biological Association of the United Kingdom, Plymouth, England. Updated 9/5/2002. [Cited 12/8/2006]. Available from: http://www.marlin.ac.uk/biotopes/Bio_BasicInfo_IMS.MacAbr.htm.

MarLIN. 2004. Ecology of Cirratulids and *Cerastoderma edule* in littoral mixed sediment. Marine Life Information Network (MarLIN): Biology and sensitivity key information Sub-programme [online]. Marine Biological Association of the United Kingdom, Plymouth, England. Updated 11/29/2004. [Cited 12/8/2006]. Available from: http://www.marlin.ac.uk/biotopes/bio_eco_LS.LMX.MX.cirCer.htm.

MarLIN. 2005. *Capitella capitata*. Gallery worm. Marine Life Information Network (MarLIN): Biology and sensitivity key information Sub-programme [online]. Marine Biological Association of the United Kingdom, Plymouth, England. [Cited 12/8/2006]. Available from: http://www.marlin.ac.uk/species/Capitellacapitata.htm.

Mickelson S. 2006. Personal communication (3 emails to Fiona McNair, Windward Environmental, with unpublished King County water quality data from Jan-Dec 2005). Water Quality Project Manager, Marine and Sediment Assessment Group, King County Department of Natural Resources and Parks, Seattle, WA. 1/18/06, 1/19/06, and 3/30/06.

Mickelson S, Williston D. 2006. Technical memorandum: Duwamish River/Elliott Bay/Green River water column PCB congener survey: transmittal of data and quality assurance documentation. King County Department of Natural Resources, Seattle, WA.

Miller BS, Simenstad CA, Moulton LL, Fresh KL, Funk FC, Karp WA, Borton SF. 1977. Puget Sound baseline program nearshore fish survey. Final report, July 1974-June 1977. Prepared for Washington Department of Ecology. Fisheries Research Institute, University of Washington, Seattle, WA.

Morgan MG, Henrion M. 1990. Uncertainty: a guide to dealing with uncertainty in quantitative risk and policy analysis. Cambridge University Press, Cambridge, UK.

Morrison HA, Gobas FAPC, Lazar R, Haffner GD. 1996. Development and verification of a bioaccumulation model for organic contaminants in benthic invertebrates. Environ Sci Technol 30:3377-3384.

Museum Victoria. 1996. Marine crustaceans of southern Australia: Amphipods: *Ampithoe* sp. [online]. Museum Victoria, Melbourne, Australia. [Cited 12/8/06]. Available from: http://www.mov.vic.gov.au/crust/movampi.html.

Nairn B. 2006. Personal communication (e-mail to F. McNair, Windward Environmental, regarding updated EFDC numbers, with attached Excel spreadsheet). King County, Seattle, WA. October 12, 2006.

Nichols JW, Fitzsimmons PN, Whiteman FW, Kuehl DW, Butterworth BC, Jenson CT. 2001. Dietary uptake kinetics of 2,2',5,5'-tetrachlorobiphenyl in rainbow trout. Drug Metab Dispos 29:1013–1022.

Nomura T, Davis ND. 2005. Lipid and moisture content of salmon prey organisms and stomach contents of chum, pink, and sockeye salmon in the Bering Sea. In: BASIS 2004: Salmon and marine ecosystems in the Bering Sea and adjacent waters. NPAFC Technical Report no. 6. North Pacific Anadromous Fish Commission, Vancouver, BC.

Olson B. 2006. Personal communication (e-mail to Fiona McNair, Windward Environmental, regarding lipid content and organic carbon content of phytoplankton and zooplankton in Puget Sound). PhD student, School of Oceanography, University of Washington, Seattle, WA. September 27, 2006.

Palaeos. 2004. Echinodermata: Class Ophiouroidea [online]. Palaeos: the trace of life on earth. Updated 04/08/01. [Cited 12/8/06]. Available from: http://www.palaeos.com/Invertebrates/Echinoderms/Ophiuroidea/Ophiuroidea.htm.

Parkerton TF, Connolly JP, Thomann RV, Uchrin CG. 1993. Do aquatic effects or human health end points govern the development of sediment-quality criteria for nonionic organic chemicals? Environ Toxicol Chem 12:507-523.

Patmont C, Palermo M. 2007. Case studies, environmental dredging residuals and management implications. Paper D-066 in Remediation of Contaminated Sediments-2007: Proceedings of the 4th International Conference on

Remediation of Contaminated Sediments, January 2007, Savannah, GA. Battelle Press, Columbia OH.

Ricketts EG, Calvin J, Hedgepeth J, Phillips DW. 1985. Between Pacific tides. Revised by J Hedgepeth. 5th ed. Stanford University Press, Palo Alto, CA.

Robards MD, Anthony JA, Rose GA, Piatt JF. 1999. Changes in proximate composition and somatic energy content for Pacific sand lance (*Ammodytes hexapterus*) from Kachemak Bay, Alaska relative to maturity and season. J Exp Mar Bio Ecol 242:245-258.

Roditi H, Fisher N. 1999. Rates and routes of trace element uptake in zebra mussels. Limnol Oceanogr 44:1730-1749.

Sample BE, Aplin MS, Efroymson RA, Suter GW, II, Welsh CJE. 1997. Methods and tools for estimation of the exposure of terrestrial wildlife to contaminants. Publ. no. 4650. Oak Ridge National Laboratory, Oak Ridge, TN.

Seth R, Mackay D, Muncke J. 1999. Estimating the organic carbon partition coefficient and its variability for hydrophobic chemicals. Environ Sci Tech 33:2390-2394.

Shimek RL. 2003. Invertebrate strippers [online]. Reefkeeping Magazine, ReefCentral, LLC. Updated September 2003. [Cited 12/8/2006]. Available from: http://www.reefkeeping.com/issues/2003-09/rs/index.php.

Shimek RL. 2004. Peanuts [online]. Reefkeeping Magazine, ReefCentral, LLC. Updated April 2004. [Cited 12/8/2006]. Available from: http://www.reefkeeping.com/issues/2004-04/rs/index.php.

Skoglund R, Swackhamer DL. 1999. Evidence for the use of organic carbon as the sorbing matrix in the modeling of PCB accumulation in phytoplankton. Environ Sci Technol 3:1516-1519.

Stern J, Patmont C, Hennessy D. 2005. Evaluation and management of dredging residuals to meet sediment remediation and risk reduction goals. Presentation at the SETAC North America 26th Annual Meeting, November 2005, Baltimore, MD. SETAC Abstracts 26, p. 154. Society of Environmental Toxicology and Chemistry, Pensacola, FL.

Stern J, Patmont C. 2006. Evaluation of post-dredge monitoring results to assess net risk reduction of different sediment cleanup options. Presentation at the SETAC North America 27th Annual Meeting, Nov. 2007, Montreal, Canada. SETAC Abstracts 27, pp. 385-386. Society of Environmental Toxicology and Chemistry, Pensacola, FL.

Stern JH, Hennessy DP, Patmont CR. 2003. Improving estimates of contaminant exposure for mobile organisms: an assessment of area-weighted home range exposure estimates applied to the relationship between sediment chemistry and liver lesions in English sole. Puget Sound Research Conference 2003, Vancouver, BC.

Sverdrup HU, Johnson MW, Fleming RH. 1942. The oceans: their physics, chemistry and general biology, p. 1087. Prentice -Hall, Englewood Cliffs, NJ.

Swackhamer DL, Skoglund RS. 1993. Bioaccumulation of PCB's by algae: Kinetics versus equilibrium. Environ Toxicol Chem 12:831-838.

Thomann RV, Connolly JP, Parkerton TF. 1992. An equilibrium model of organic chemical accumulation in aquatic food webs with sediment interaction. Environ Toxicol Chem 11:615-629.

Warren-Hicks WJ, Moore DRJ, eds. 1998. Uncertainty analysis in ecological risk assessment. Society of Environmental Toxicology and Chemistry, Pensacola, FL.

Weininger D. 1978. Accumulation of PCBs by lake trout in Lake Michigan. PhD thesis. University of Wisconsin, Madison, WI.

Windward. 2004. Lower Duwamish Waterway remedial investigation. Intertidal clam survey data report. Prepared for Lower Duwamish Waterway Group. Windward Environmental LLC, Seattle, WA.

Windward. 2005a. Lower Duwamish Waterway remedial investigation. Benthic data report addendum: PCB congener data and SIM analyses. Prepared for Lower Duwamish Waterway Group. Windward Environmental LLC, Seattle, WA.

Windward. 2005b. Lower Duwamish Waterway remedial investigation. Data report: Chemical analyses of benthic invertebrate and clam tissue samples and co-located sediment samples. Prepared for Lower Duwamish Waterway Group. Windward Environmental LLC, Seattle, WA.

Windward. 2005c. Lower Duwamish Waterway remedial investigation. Data report: Fish and crab tissue collection and chemical analyses. Prepared for Lower Duwamish Waterway Group. Windward Environmental LLC, Seattle, WA.

Windward. 2005d. Lower Duwamish Waterway remedial investigation. Data report: Taxonomic identifications of benthic invertebrate communities. Prepared for Lower Duwamish Waterway Group. Windward Environmental LLC, Seattle, WA.

Windward. 2005e. Lower Duwamish Waterway remedial investigation. Fish and crab data report addendum: PCB congener data, MS/MSD analyses, and DDT confirmation. Prepared for Lower Duwamish Waterway Group. Windward Environmental LLC, Seattle, WA.

Windward. 2005f. Lower Duwamish Waterway remedial investigation. Food web model memorandum 1: objectives, conceptual model, and selection of food web model. Draft. Prepared for Lower Duwamish Waterway Group. Windward Environmental LLC, Seattle, WA.

Windward. 2005g. Lower Duwamish Waterway remedial investigation. Food web model memorandum 2: modeling approach. Draft. Prepared for Lower Duwamish Waterway Group. Windward Environmental LLC, Seattle, WA.

Windward. 2005h. Lower Duwamish Waterway remedial investigation. Food web model memorandum 3: preliminary model results. Prepared for Lower Duwamish Waterway Group. Windward Environmental LLC, Seattle, WA.

Windward. 2005i. Lower Duwamish Waterway remedial investigation. Quality assurance project plan: Fish and crab tissue collection and chemical analyses. Addendum for additional fish and crab sampling in the Lower Duwamish Waterway. Prepared for Lower Duwamish Waterway Group. Windward Environmental LLC, Seattle, WA.

Windward. 2005j. Technical Memorandum: Summary of sediment and tissue chemistry datasets to be used in the Phase 2 RI/FS. Prepared for Lower Duwamish Waterway Group. Windward Environmental LLC, Seattle, WA.

Windward. 2006a. Lower Duwamish Waterway remedial investigation. Data report: chemical analyses of fish and crab tissue samples collected in 2005. Prepared for Lower Duwamish Waterway Group. Windward Environmental LLC, Seattle, WA.

Windward. 2006b. Lower Duwamish Waterway remedial investigation. Technical memorandum: GIS interpolation of total PCBs in LDW surface sediment. Windward Environmental LLC, Seattle, WA.

Wingert RC, Terry CB, Miller BS. 1979. Food and feeding habits of ecologically important nearshore and demersal fishes in central Puget Sound. FRI-UW-7903. Prepared for Washington Department of Ecology. Fisheries Research Institute, University of Washington, Seattle, WA.

Winsor MH, Boese BL, Lee H, Randall RC, Specht DT. 1990. Determination of the ventilation rates of interstitial and overlying water by the clam *Macoma nasuta*. Environ Toxicol Chem 9:209-213.

Word JQ. 1990. The infaunal trophic index, a functional approach to benthic community analyses. Doctoral dissertation. School of Fisheries, University of Washington, Seattle, WA.

Attachment 1 Benthic Invertebrate Tissue-Sediment Regression

Total PCB tissue concentrations for benthic invertebrates were derived from tissue-sediment regressions. Benthic invertebrate tissue and co-located surface sediment samples were collected from 20 locations in the LDW (10 intertidal locations and 10 subtidal locations). Linear least-squares regression was used to model the relationship between total PCB concentrations[1] in benthic invertebrate tissue and co-located sediment. The log-log relationship provided a reasonable linear fit with homogeneous residuals (Figure 1),[2] except for two extreme points (locations B5a-1 and B8a). Location B5a-1 had a low-moderate sediment total PCB concentration and a high tissue concentration. The sediment had very low organic carbon content, so this point was not extreme when the data were organic carbon-normalized. Location B8a had a high total PCB sediment concentration. This point was exerting undue influence on the regression estimates and was far higher than the total PCB concentrations in sediment for which total PCB concentrations were to be estimated in tissue. The R^2 value with the two outliers included was 0.51. Without these two outliers, the regression provided a good fit to the data in the range for which total PCB concentrations will be estimated in tissue. The R^2 value with the outliers removed was 0.74. The regression parameters were estimated with full reporting-limit concentrations for the two non-detect samples.[3] The equation for the line with outliers removed is presented as Equation 1.

$$\log10[C_{BI}] = 1.40 + 0.35 \times \log10[C_S]$$ **Equation 1**

Where:

C_{BI} = total PCB concentration in benthic invertebrate tissue (µg/kg ww)
C_S = total PCB concentration in sediment (µg/kg dw)

Total PCB concentrations in benthic invertebrate tissues for the entire LDW and for each modeling area were estimated from total PCBs in sediment using the equation above. The sediment concentrations (C_S) used were the spatially weighted average concentrations (SWACs) from corresponding areas of the LDW (Table D.4-5 in Appendix D).

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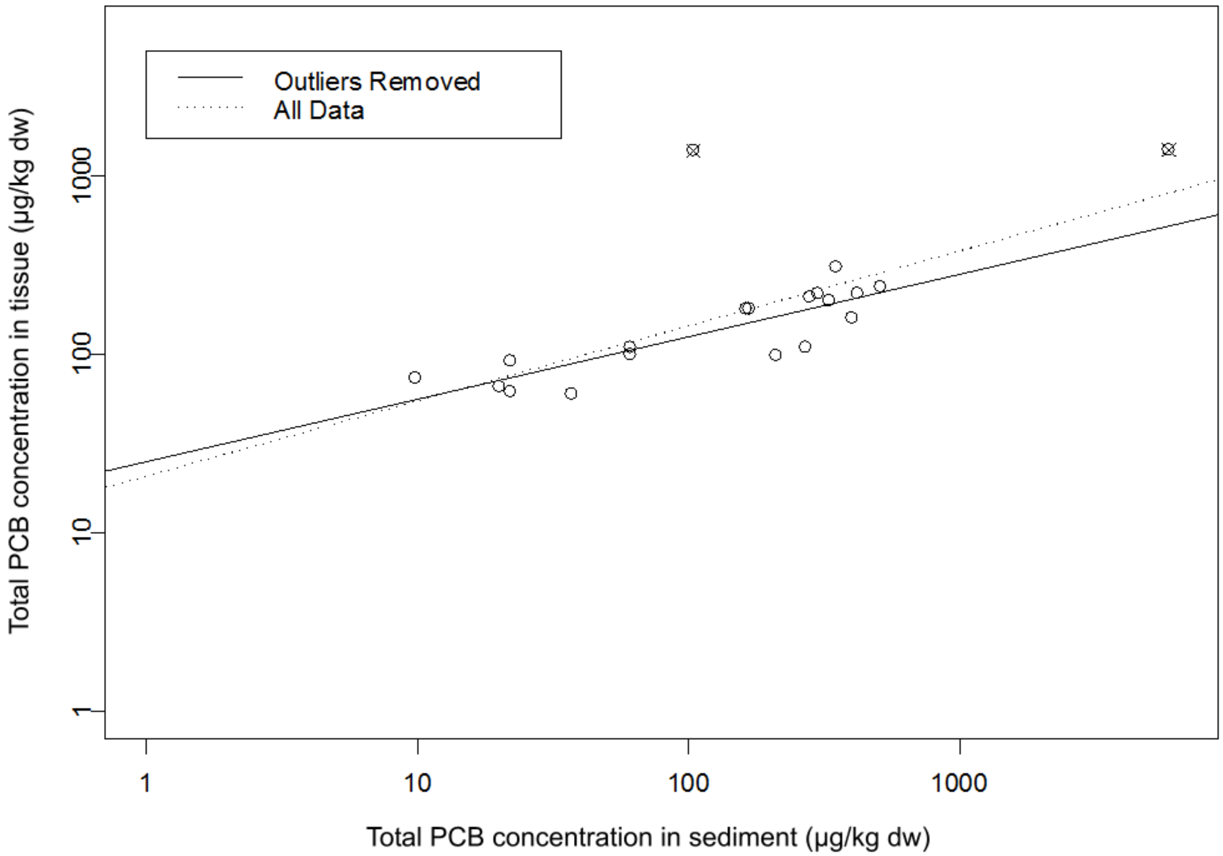
[1] The relationship between organic carbon-normalized total PCB concentrations in sediment and lipid-normalized total PCB concentrations in tissue was also tested, but the relationship without normalization provided a better fit to the data.

[2] The regression analysis was conducted by Alice Shelly of Terrastat Consulting Group.

[3] There was one non-detect sediment concentration (B1a; reporting limit = 20 µg/kg dw) and one non-detect tissue concentration (B4a; reporting limit = 200 µg/kg ww).



Points that are crossed are outliers.

Figure 1. Linear least-squares fit to log-transformed total PCB concentration in benthic invertebrate tissue as a function of log-transformed total PCB concentration in sediment

Attachment 2 Statistics for Calibration 1 and Calibration 2

		CALIBRATION 1									CALIBRATION 2								
		PASSED MODEL PERFORMANCE FILTER					TOP 10 RUNS (Ranked by average SPAF across species)				PASSED MODEL PERFORMANCE FILTER					TOP 10 RUNS (Ranked by average SPAF across species)			
PARAMETER DESCRIPTION	UNIT	MIN	MAX	MEAN	RANGE	BEST FIT	MIN	MAX	MEAN	RANGE	MIN	MAX	MEAN	RANGE	BEST FIT	MIN	MAX	MEAN	RANGE
Environmental Parameters																			
Concentration of total PCBs in the water column	ng/L	0.218	2.940	1.322	2.721	1.22	0.470	2.13	1.34	1.66	0.194	2.672	0.957	2.478	1.11	0.208	1.41	0.722	1.20
Concentration of particulate organic carbon (POC) in the water column	kg/L	$1.0E^{-7}$	4.0×10^{-7}	2.6×10^{-7}	3.0×10^{-7}	2.3×10^{-7}	1.8×10^{-7}	3.5×10^{-7}	2.5×10^{-7}	1.7×10^{-7}	1.2×10^{-7}	3.5×10^{-7}	2.7×10^{-7}	2.3×10^{-7}	3.2×10^{-7}	2.1×10^{-7}	3.2×10^{-7}	2.7×10^{-7}	1.1×10^{-7}
Dissolved organic carbon (DOC) in the water column	kg/L	1.3×10^{-6}	3.0×10^{-6}	2.2×10^{-6}	1.7×10^{-6}	2.2×10^{-6}	1.7×10^{-6}	2.4×10^{-6}	2.2×10^{-6}	6.6×10^{-7}	1.6×10^{-6}	2.8×10^{-6}	2.2×10^{-6}	1.2×10^{-6}	1.6×10^{-6}	1.6×10^{-6}	2.6×10^{-6}	2.2×10^{-6}	1.0×10^{-6}
Mean water column temperature	°C	9.9	12.5	11.2	2.5	11.0	10.8	11.9	11.2	1.1	10.3	12.0	11.1	1.8	10.8	10.5	11.7	11.1	1.3
Dissolved oxygen concentration in the water column	mg/L	7.12	8.56	7.91	1.44	8.15	7.75	8.19	7.96	0.44	7.46	8.46	7.92	1.00	8.02	7.78	8.27	8.02	0.49
Total suspended solids in the water column	kg/L	3.1×10^{-6}	8.6×10^{-6}	5.8×10^{-6}	5.5×10^{-6}	5.4×10^{-6}	4.2×10^{-6}	8.4×10^{-6}	6.1×10^{-6}	4.2×10^{-6}	3.4×10^{-6}	8.0×10^{-6}	5.7×10^{-6}	4.6×10^{-6}	4.7×10^{-6}	4.7×10^{-6}	6.8×10^{-6}	6.0×10^{-6}	2.1×10^{-6}
Concentration of PCBs in sediment	µg/kg dw	380	380	380	0	380	380	380	380	0	380	380	380	0	380	380	380	380	0
Sediment total organic carbon	%	1.82%	1.98%	1.91%	0.17%	1.91%	1.89%	1.95%	1.92%	0.06%	1.83%	1.97%	1.91%	0.14%	1.92%	1.83%	1.94%	1.90%	0.11%
Chemical Parameters																			
Octanol-water partition coefficient for PCBs (log K_{ow})	unitless	6.4	6.8	6.6	0.4	6.5	6.5	6.7	6.6	0.2	6.4	6.7	6.6	0.3	6.5	6.4	6.6	6.5	0.2
Biological Parameters																			
Proportionality constant expressing the sorption capacity of NLOM relative to that of octanol (β or MAF)	unitless	0.016	0.050	0.033	0.034	0.031	0.022	0.040	0.032	0.018	0.022	0.045	0.033	0.022	0.037	0.026	0.040	0.032	0.013
Resistance to chemical uptake through aqueous phase for phytoplankton/ algae (A)	day-1	2×10^{-5}	1×10^{-4}	6×10^{-5}	8×10^{-5}	6×10^{-5}	4×10^{-5}	8×10^{-5}	6×10^{-5}	4×10^{-5}	3×10^{-5}	8×10^{-5}	6×10^{-5}	5×10^{-5}	6×10^{-5}	3×10^{-5}	8×10^{-5}	6×10^{-5}	4×10^{-5}
Resistance to chemical uptake through organic phase for phytoplankton/ algae (B)	unitless	2.0	9.2	5.5	7.2	6.2	3.1	8.0	5.8	4.8	2.6	8.6	5.5	6.0	4.6	4.1	7.8	5.9	3.7
Density of lipids	kg/L	0.8	1.0	0.9	0.2	0.9	0.8	1.0	0.9	0.1	0.8	1.0	0.9	0.2	0.9	0.9	1.0	0.9	0.1
Phytoplankton																			
Lipid content of organism	%	0.00%	0.28%	0.12%	0.28%	0.14%	0.09%	0.21%	0.15%	0.13%	0.00%	0.23%	0.12%	0.23%	0.12%	0.05%	0.17%	0.12%	0.12%
Water content of organism	%	93.7%	97.2%	95.6%	3.5%	95.7%	94.8%	96.7%	95.5%	2.0%	93.9%	97.2%	95.6%	3.2%	95.6%	94.1%	95.9%	95.1%	1.8%
Zooplankton																			
Organism weight	kg	2.2×10^{-8}	2.7×10^{-7}	1.6×10^{-7}	2.5×10^{-7}	2.2×10^{-7}	5.3×10^{-8}	2.4×10^{-7}	1.6×10^{-7}	1.9×10^{-7}	5.6×10^{-8}	2.6×10^{-7}	1.5×10^{-7}	2.0×10^{-7}	1.6×10^{-7}	8.5×10^{-8}	2.0×10^{-7}	1.4×10^{-7}	1.1×10^{-7}
Lipid content	%	0.2%	2.3%	1.2%	2.1%	1.4%	0.7%	1.7%	1.3%	1.0%	0.4%	2.1%	1.2%	1.8%	1.0%	0.8%	2.1%	1.2%	1.4%
Water content of organism	%	85%	96%	90%	10%	92%	88%	92%	91%	4%	86%	94%	90%	8%	89%	89%	94%	91%	5%

		Calibration 1									Calibration 2								
		Passed Model Performance Filter					Top 10 Runs (Ranked by average SPAF across species)				Passed Model Performance Filter					Top 10 Runs (Ranked by average SPAF across species)			
Parameter Description	Unit	Min	Max	Mean	Range	Best Fit	Min	Max	Mean	Range	Min	Max	Mean	Range	Best Fit	Min	Max	Mean	Range
Dietary absorption efficiency of lipids (ε_L)	%	55%	85%	71%	30%	66%	61%	81%	71%	20%	56%	83%	71%	27%	71%	64%	82%	70%	17%
Dietary absorption efficiency of NLOM (ε_N)	%	55%	85%	71%	29%	72%	58%	83%	70%	25%	58%	84%	70%	27%	72%	60%	73%	68%	13%
Benthic Invertebrates																			
Organism weight	kg	7.1×10^{-8}	1.2×10^{-4}	4.5×10^{-5}	1.1×10^{-4}	4.1×10^{-5}	4.1×10^{-5}	8.5×10^{-5}	5.9×10^{-5}	4.4×10^{-5}	3.5×10^{-8}	9.1×10^{-5}	3.5×10^{-5}	9.1×10^{-5}	1.5×10^{-6}	1.5×10^{-6}	3.9×10^{-5}	1.4×10^{-5}	3.7×10^{-5}
Lipid content	%	0.69%	1.05%	0.86%	0.35%	0.83%	0.69%	0.90%	0.80%	0.21%	0.74%	1.05%	0.86%	0.31%	0.81%	0.78%	0.94%	0.85%	0.16%
Water content of organism	%	71%	87%	81%	15%	82%	76%	85%	83%	10%	73%	87%	81%	14%	84%	75%	87%	82%	11%
Relative fraction of porewater ventilated	unitless	0.050	0.247	0.142	0.197	0.134	0.059	0.22	0.13	0.161	0.055	0.217	0.121	0.162	0.074	0.057	0.109	0.081	0.052
Dietary absorption efficiency of lipids (ε_L)	%	16%	95%	61%	80%	30%	30%	89%	59%	58%	17%	94%	60%	78%	79%	53%	83%	68%	30%
Dietary absorption efficiency of NLOM (ε_N)	%	17%	93%	52%	77%	56%	18%	76%	43%	58%	17%	87%	47%	71%	61%	24%	87%	53%	63%
Juvenile Fish																			
Organism weight	kg	3×10^{-3}	8×10^{-3}	6×10^{-3}	5×10^{-3}	6×10^{-3}	5×10^{-3}	7×10^{-3}	6×10^{-3}	2×10^{-3}	4×10^{-3}	8×10^{-3}	6×10^{-3}	3×10^{-3}	5×10^{-3}	5×10^{-3}	7×10^{-3}	6×10^{-3}	2×10^{-3}
Lipid content	%	0.6%	4.6%	2.4%	4.0%	1.5%	1.1%	3.1%	1.9%	2.0%	0.4%	3.5%	2.2%	3.1%	3.1%	1.7%	3.1%	2.4%	1.4%
Water content of organism	%	65.9%	82.0%	74.0%	16.1%	74.3%	69.8%	76.3%	73.2%	6.5%	69.6%	79.5%	74.0%	9.9%	71.4%	71.4%	78.3%	74.2%	6.9%
Relative fraction of porewater ventilated	unitless	0.01	0.02	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.02	0.01	0.01	0.02	0.01	0.02	0.01	0.01
Dietary absorption efficiency of lipids (ε_L)	%	90%	95%	92%	5%	92%	91%	93%	92%	2%	90%	94%	92%	4%	91%	91%	94%	92%	3%
Dietary absorption efficiency of NLOM (ε_N)	%	50%	65%	58%	15%	54%	54%	61%	58%	7%	51%	65%	58%	14%	64%	54%	64%	58%	10%
Slender Crab																			
Organism weight	kg	0.152	0.180	0.167	0.028	0.165	0.163	0.175	0.167	0.012	0.159	0.176	0.167	0.018	0.167	0.164	0.171	0.167	0.007
Lipid content	%	0.9%	1.2%	1.1%	0.3%	1.1%	1.0%	1.2%	1.1%	0.1%	1.0%	1.3%	1.1%	0.3%	1.1%	1.1%	1.1%	1.1%	0.1%
Water content of organism	%	82.5%	85.1%	83.8%	2.7%	83.7%	83.4%	84.5%	83.9%	1.1%	82.7%	84.8%	83.8%	2.2%	83.2%	83.2%	84.3%	83.7%	1.1%
Relative fraction of porewater ventilated	unitless	0.01	0.03	0.02	0.02	0.03	0.01	0.03	0.02	0.02	0.01	0.03	0.02	0.02	0.02	0.01	0.03	0.02	0.01
Dietary absorption efficiency of lipids (ε_L)	%	16%	95%	62%	79%	75%	39%	90%	68%	51%	17%	85%	45%	69%	66%	27%	85%	56%	58%
Dietary absorption efficiency of NLOM (ε_N)	%	16%	95%	62%	79%	76%	39%	89%	68%	51%	20%	89%	50%	68%	54%	20%	63%	45%	43%
Dungeness Crab																			
Organism weight	kg	0.328	0.719	0.527	0.391	0.653	0.431	0.653	0.570	0.222	0.410	0.677	0.521	0.267	0.529	0.443	0.641	0.517	0.198
Lipid content	%	1.1%	4.2%	2.6%	3.1%	3.4%	2.3%	3.4%	2.8%	1.1%	1.6%	3.7%	2.5%	2.1%	2.6%	2.1%	3.2%	2.7%	1.1%
Water content of organism	%	79%	84%	82%	5%	81%	81%	83%	82%	2%	80%	84%	82%	4%	82%	81%	82%	82%	2%
Relative fraction of porewater ventilated	unitless	0.01	0.03	0.02	0.02	0.02	0.02	0.03	0.02	0.01	0.01	0.03	0.02	0.02	0.02	0.01	0.03	0.02	0.02

Parameter Description	Unit	Calibration 1									Calibration 2								
		Passed Model Performance Filter					Top 10 Runs (Ranked by average SPAF across species)				Passed Model Performance Filter					Top 10 Runs (Ranked by average SPAF across species)			
		Min	Max	Mean	Range	Best Fit	Min	Max	Mean	Range	Min	Max	Mean	Range	Best Fit	Min	Max	Mean	Range
Dietary absorption efficiency of lipids (ε_L)	%	16%	96%	61%	79%	71%	47%	82%	66%	35%	22%	92%	55%	71%	36%	31%	63%	49%	32%
Dietary absorption efficiency of NLOM (ε_N)	%	18%	95%	62%	77%	59%	48%	82%	65%	34%	20%	94%	60%	74%	68%	37%	81%	58%	44%
Pacific Staghorn Sculpin																			
Organism weight	kg	0.062	0.089	0.077	0.026	0.075	0.065	0.078	0.074	0.013	0.068	0.085	0.077	0.017	0.082	0.071	0.085	0.078	0.014
Lipid content	%	1.9%	2.3%	2.1%	0.4%	2.1%	2.0%	2.2%	2.1%	0.2%	1.9%	2.3%	2.1%	0.3%	2.2%	2.0%	2.2%	2.1%	0.2%
Water content of organism	%	79%	79%	79%	1%	79%	79%	79%	79%	0%	79%	79%	79%	0%	79%	79%	79%	79%	0%
Relative fraction of porewater ventilated	unitless	0.02	0.10	0.06	0.08	0.03	0.03	0.09	0.06	0.06	0.02	0.09	0.05	0.07	0.06	0.04	0.09	0.06	0.05
Dietary absorption efficiency of lipids (ε_L)	%	90%	95%	92%	5%	93%	90%	94%	92%	3%	90%	95%	92%	4%	91%	91%	92%	92%	2%
Dietary absorption efficiency of NLOM (ε_N)	%	50%	65%	58%	14%	50%	50%	62%	58%	12%	51%	63%	58%	12%	52%	52%	63%	57%	10%
Shiner Surfperch																			
Organism weight	kg	0.017	0.021	0.019	0.003	0.019	0.018	0.020	0.019	0.001	0.018	0.020	0.019	0.002	0.018	0.018	0.020	0.019	0.002
Lipid content	%	3.9%	5.3%	4.6%	1.3%	4.6%	4.2%	4.9%	4.6%	0.7%	4.0%	5.1%	4.6%	1.0%	4.6%	4.3%	4.7%	4.5%	0.3%
Water content of organism	%	72.8%	75.2%	73.9%	2.4%	74.0%	73.3%	74.4%	73.9%	1.0%	73.2%	74.7%	73.9%	1.4%	73.5%	73.5%	74.4%	74.0%	0.9%
Relative fraction of porewater ventilated	unitless	0.01	0.02	0.01	0.01	0.02	0.01	0.02	0.01	0.01	0.01	0.02	0.01	0.01	0.01	0.01	0.01	0.01	0.01
Dietary absorption efficiency of lipids (ε_L)	%	90%	95%	92%	5%	94%	91%	94%	92%	3%	90%	95%	92%	5%	90%	90%	94%	92%	3%
Dietary absorption efficiency of NLOM (ε_N)	%	50%	65%	58%	15%	56%	52%	63%	58%	11%	51%	64%	58%	13%	58%	55%	63%	59%	7%
English Sole																			
Organism weight	kg	0.212	0.282	0.247	0.070	0.246	0.231	0.258	0.246	0.027	0.212	0.271	0.248	0.059	0.246	0.237	0.263	0.252	0.026
Lipid content	%	4.7%	6.2%	5.5%	1.4%	5.5%	5.2%	6.0%	5.5%	0.7%	5.0%	6.1%	5.5%	1.1%	5.7%	5.0%	5.8%	5.4%	0.7%
Water content of organism	%	74.0%	76.0%	75.0%	2.0%	75.0%	74.4%	75.3%	75.0%	1.0%	73.9%	75.8%	75.0%	1.9%	74.8%	74.5%	75.6%	75.1%	1.1%
Relative fraction of porewater ventilated	unitless	0.01	0.20	0.10	0.19	0.15	0.04	0.1	0.09	0.11	0.01	0.18	0.11	0.17	0.06	0.05	0.18	0.10	0.12
Dietary absorption efficiency of lipids (ε_L)	%	90%	95%	92%	5%	92%	91%	94%	93%	3%	90%	95%	92%	5%	94%	91%	95%	93%	4%
Dietary absorption efficiency of NLOM (ε_N)	%	50%	65%	58%	15%	59%	54%	63%	59%	9%	51%	64%	59%	13%	53%	52%	62%	56%	10%
Dietary Fraction Statistics																			
Benthic Invertebrates																			
Sediment	fraction	0.66	0.91	0.77	0.24	0.70	0.70	0.82	0.76	0.12	0.67	0.85	0.77	0.19	0.73	0.70	0.83	0.77	0.13
Phytoplankton	fraction	0.07	0.21	0.15	0.14	0.18	0.10	0.18	0.16	0.08	0.07	0.20	0.15	0.13	0.15	0.09	0.16	0.14	0.07
Zooplankton	fraction	0.01	0.17	0.08	0.16	0.12	0.04	0.12	0.08	0.08	0.03	0.15	0.08	0.12	0.12	0.07	0.14	0.09	0.07

		CALIBRATION 1									CALIBRATION 2								
		PASSED MODEL PERFORMANCE FILTER					TOP 10 RUNS (Ranked by average SPAF across species)				PASSED MODEL PERFORMANCE FILTER					TOP 10 RUNS (Ranked by average SPAF across species)			
PARAMETER DESCRIPTION	UNIT	MIN	MAX	MEAN	RANGE	BEST FIT	MIN	MAX	MEAN	RANGE	MIN	MAX	MEAN	RANGE	BEST FIT	MIN	MAX	MEAN	RANGE
Juvenile Fish																			
Sediment	fraction	0.00	0.01	0.00	0.01	0.00	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00
Zooplankton	fraction	0.35	0.81	0.57	0.47	0.53	0.51	0.78	0.60	0.27	0.43	0.83	0.63	0.40	0.60	0.43	0.75	0.61	0.31
Benthic invertebrate	fraction	0.18	0.65	0.42	0.46	0.47	0.22	0.49	0.39	0.27	0.17	0.56	0.36	0.39	0.40	0.25	0.56	0.38	0.31
Slender Crab																			
Sediment	fraction	0.000	0.049	0.015	0.048	0.021	0.00	0.02	0.01	0.018	0.000	0.049	0.017	0.049	0.017	0.004	0.042	0.019	0.038
Zooplankton	fraction	0.004	0.118	0.076	0.114	0.094	0.016	0.115	0.075	0.099	0.008	0.116	0.076	0.108	0.092	0.061	0.110	0.087	0.049
Benthic invertebrate	fraction	0.860	0.976	0.899	0.115	0.876	0.863	0.968	0.903	0.105	0.860	0.970	0.897	0.110	0.882	0.864	0.913	0.884	0.050
Juvenile fish	fraction	0.009	0.011	0.010	0.002	0.009	0.009	0.010	0.010	0.001	0.009	0.011	0.010	0.002	0.009	0.009	0.010	0.010	0.001
Dungeness Crab																			
Sediment	fraction	0.00	0.05	0.01	0.05	0.00	0.000	0.040	0.016	0.04	0.00	0.05	0.01	0.05	0.00	0.00	0.03	0.01	0.03
Zooplankton	fraction	0.01	0.59	0.33	0.57	0.37	0.07	0.39	0.31	0.32	0.06	0.56	0.34	0.50	0.19	0.13	0.48	0.33	0.36
Benthic invertebrate	fraction	0.16	0.73	0.33	0.57	0.24	0.18	0.53	0.31	0.34	0.17	0.57	0.32	0.41	0.41	0.17	0.53	0.32	0.36
Juvenile fish	fraction	0.16	0.58	0.32	0.41	0.39	0.28	0.42	0.37	0.14	0.16	0.49	0.32	0.33	0.39	0.19	0.43	0.33	0.23
Pacific Staghorn Sculpin																			
Sediment	fraction	0.00	0.05	0.02	0.05	0.00	0.00	0.02	0.01	0.02	0.00	0.05	0.02	0.05	0.00	0.00	0.03	0.01	0.03
Zooplankton	fraction	0.01	0.50	0.24	0.49	0.22	0.22	0.40	0.32	0.18	0.08	0.48	0.28	0.40	0.34	0.14	0.40	0.29	0.26
Benthic invertebrate	fraction	0.073	0.744	0.415	0.671	0.543	0.27	0.54	0.39	0.277	0.107	0.619	0.394	0.512	0.436	0.185	0.590	0.379	0.405
Juvenile fish	fraction	0.172	0.661	0.325	0.489	0.236	0.176	0.335	0.280	0.159	0.173	0.604	0.310	0.431	0.219	0.219	0.469	0.316	0.249
Shiner Surfperch																			
Sediment	fraction	0.00	0.01	0.01	0.01	0.00	0.004	0.009	0.007	0.01	0.00	0.01	0.01	0.01	0.00	0.00	0.01	0.01	0.01
Zooplankton	fraction	0.188	0.689	0.403	0.501	0.230	0.23	0.37	0.32	0.137	0.198	0.626	0.410	0.429	0.422	0.258	0.531	0.409	0.273
Benthic invertebrate	fraction	0.304	0.803	0.591	0.499	0.765	0.629	0.765	0.677	0.135	0.367	0.798	0.584	0.430	0.574	0.464	0.735	0.585	0.270
English Sole																			
Sediment	fraction	0.00	0.08	0.02	0.08	0.04	0.003	0.037	0.022	0.03	0.00	0.06	0.02	0.06	0.03	0.00	0.04	0.02	0.04
Phytoplankton	fraction	0.05	0.10	0.07	0.05	0.05	0.05	0.08	0.07	0.03	0.05	0.10	0.07	0.04	0.07	0.05	0.09	0.07	0.03
Zooplankton	fraction	0.00	0.08	0.04	0.08	0.05	0.01	0.06	0.04	0.05	0.01	0.06	0.03	0.06	0.03	0.01	0.05	0.03	0.05
Benthic invertebrate	fraction	0.86	0.90	0.88	0.04	0.86	0.86	0.89	0.87	0.03	0.86	0.90	0.88	0.04	0.87	0.86	0.89	0.88	0.03
Estimated Total PCB Concentrations in Biota																			
Estimated total PCB concentration in phytoplankton tissue	µg/kg ww	5	82	31	77	28	11	61	33	50	4	49	22	45	27	6	32	17	26
Estimated total PCB concentration in zooplankton tissue	µg/kg ww	7	130	45	120	45	15	76	49	61	4	91	32	87	34	4	37	22	33
Estimated total PCB concentration in benthic invertebrate tissue	µg/kg ww	230	400	360	170	300	270	350	310	77	180	410	300	230	200	180	290	240	110
Estimated total PCB concentration in juvenile fish tissue	µg/kg ww	230	1200	700	940	470	410	680	530	270	220	720	470	500	502	260	500	400	240

		CALIBRATION 1									CALIBRATION 2								
		PASSED MODEL PERFORMANCE FILTER					TOP 10 RUNS (Ranked by average SPAF across species)				PASSED MODEL PERFORMANCE FILTER					TOP 10 RUNS (Ranked by average SPAF across species)			
PARAMETER DESCRIPTION	UNIT	MIN	MAX	MEAN	RANGE	BEST FIT	MIN	MAX	MEAN	RANGE	MIN	MAX	MEAN	RANGE	BEST FIT	MIN	MAX	MEAN	RANGE
Estimated total PCB concentration in slender crab tissue	µg/kg ww	340	1,300	670	980	690	570	740	660	170	290	500	430	200	380	300	430	360	140
Estimated total PCB concentration in Dungeness crab tissue	µg/kg ww	550	2,200	1,100	1,600	1,200	910	1,200	1,100	320	380	1,000	730	640	570	410	770	570	360
Estimated total PCB concentration in Pacific staghorn sculpin tissue	µg/kg ww	720	1,800	1,500	1,100	1,100	1,100	1,300	1,100	180	710	1,200	1,100	470	810	710	1,100	860	350
Estimated total PCB concentration in shiner surfperch tissue	µg/kg ww	900	2,200	1,500	1,300	1,600	1,200	1,700	1,500	490	640	1,800	1,200	1,200	850	640	1,200	1,000	560
Estimated total PCB concentration in English sole tissue	µg/kg ww	1,800	3,800	2,726	2,000	2,500	2,100	2,800	2,500	700	1,400	3,000	2,300	1,600	1,600	1,400	2,200	1,800	860
Species Predictive Accuracy Factor (SPAF)																			
Benthic invertebrate SPAF	unitless	1	2	2	1	1.5	1.4	1.7	1.5	0.4	na	na	na	na	na	na	na	na	na
Slender crab SPAF	unitless	1.0	2.0	1.2	1.0	1.0	1.0	1.2	1.1	0.2	1.2	2.0	1.7	0.8	1.5	1.2	1.7	1.4	0.5
Dungeness crab SPAF	unitless	1.0	2.0	1.3	1.0	1.1	1.0	1.2	1.1	0.2	1.0	2.0	1.4	1.0	1.1	1.0	1.5	1.2	0.5
Pacific staghorn sculpin SPAF	unitless	1.0	2.0	1.7	1.0	1.2	1.2	1.4	1.3	0.2	1.2	2.0	1.8	0.8	1.4	1.2	1.8	1.5	0.6
Shiner surfperch SPAF	unitless	1.0	2.0	1.2	1.0	1.2	1.1	1.5	1.2	0.4	1.0	1.8	1.2	0.8	1.2	1.0	1.6	1.2	0.6
English sole SPAF	unitless	1.0	1.7	1.2	0.7	1.1	1.0	1.2	1.1	0.2	1.0	1.9	1.5	0.9	1.1	1.0	1.4	1.2	0.4
Average SPAF	unitless	1.2	1.7	1.4	0.5	1.18	1.2	1.2	1.2	0.05	1.2	1.8	1.5	0.6	1.2	1.2	1.3	1.3	0.1

DOC – dissolved organic carbon
NLOM – non-lipid organic matter
PCB – polychlorinated biphenyl
POC – particulate organic carbon
SPAF – species predictive accuracy factor
ww – wet weight