Appendix D. Food Web Model

List of Tables

List of Figures

List of Maps

This map is included in the Appendix Maps folio.

Acronyms

Acronym	Definition
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
LDWG	Lower Duwamish Waterway Group

Acronym	Definition
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 in the main body of the RI and Section D.9). 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.

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. Comparison of FWM-estimated tissue concentrations to 2007 tissue data is presented in Section D.8. Use of the FWM in the calculation of sediment RBTCs is discussed in Section D.9. A summary is provided in Section D.10.

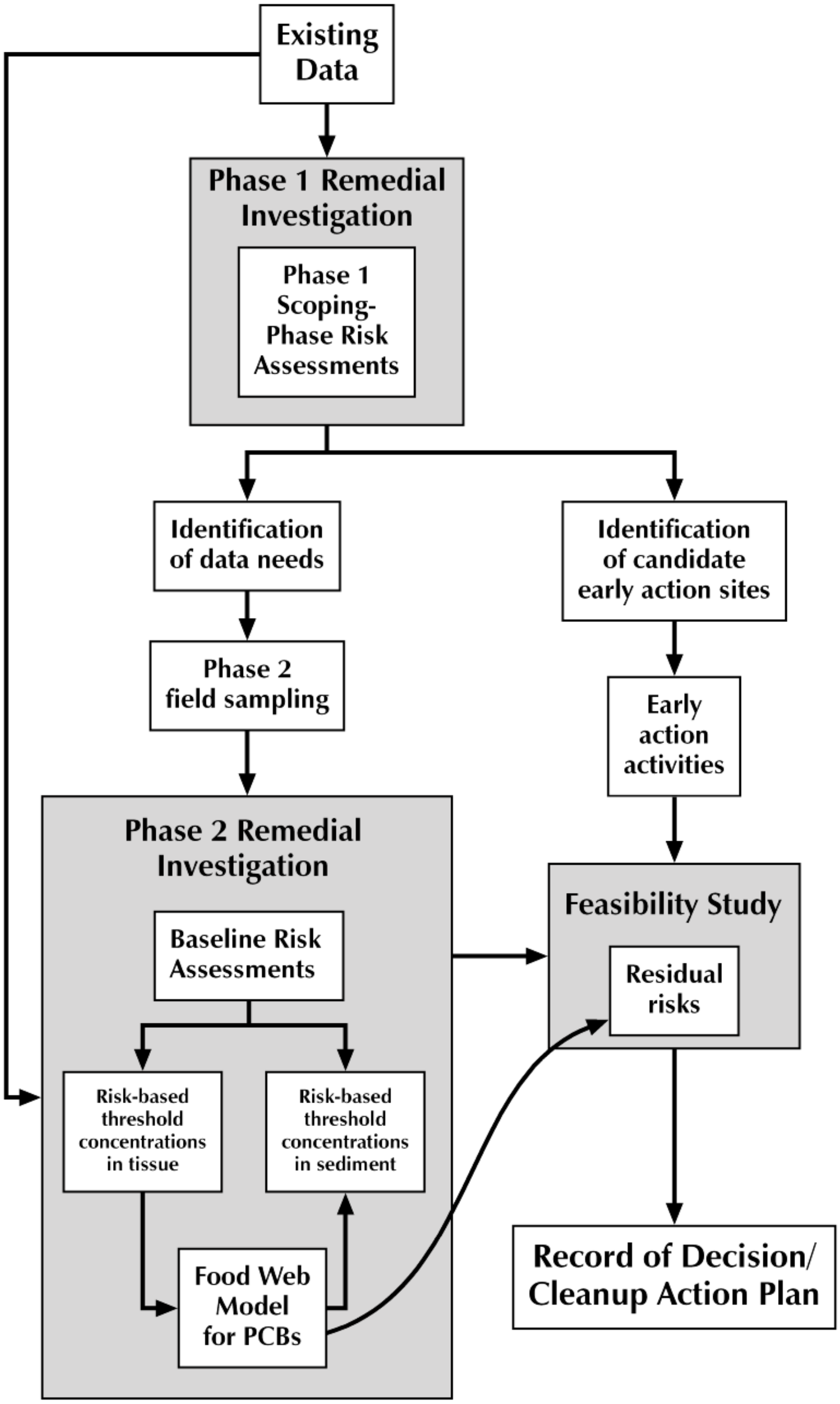


Figure D.1-1. Use of the FWM in the RI/FS process

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 organisms into three components: lipids, non-lipid organic matter (NLOM) or non-lipid organic carbon (NLOC) for phytoplankton and water
- 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 its own unique set of equations. The model has three physical media: sediment, water column water, and porewater.

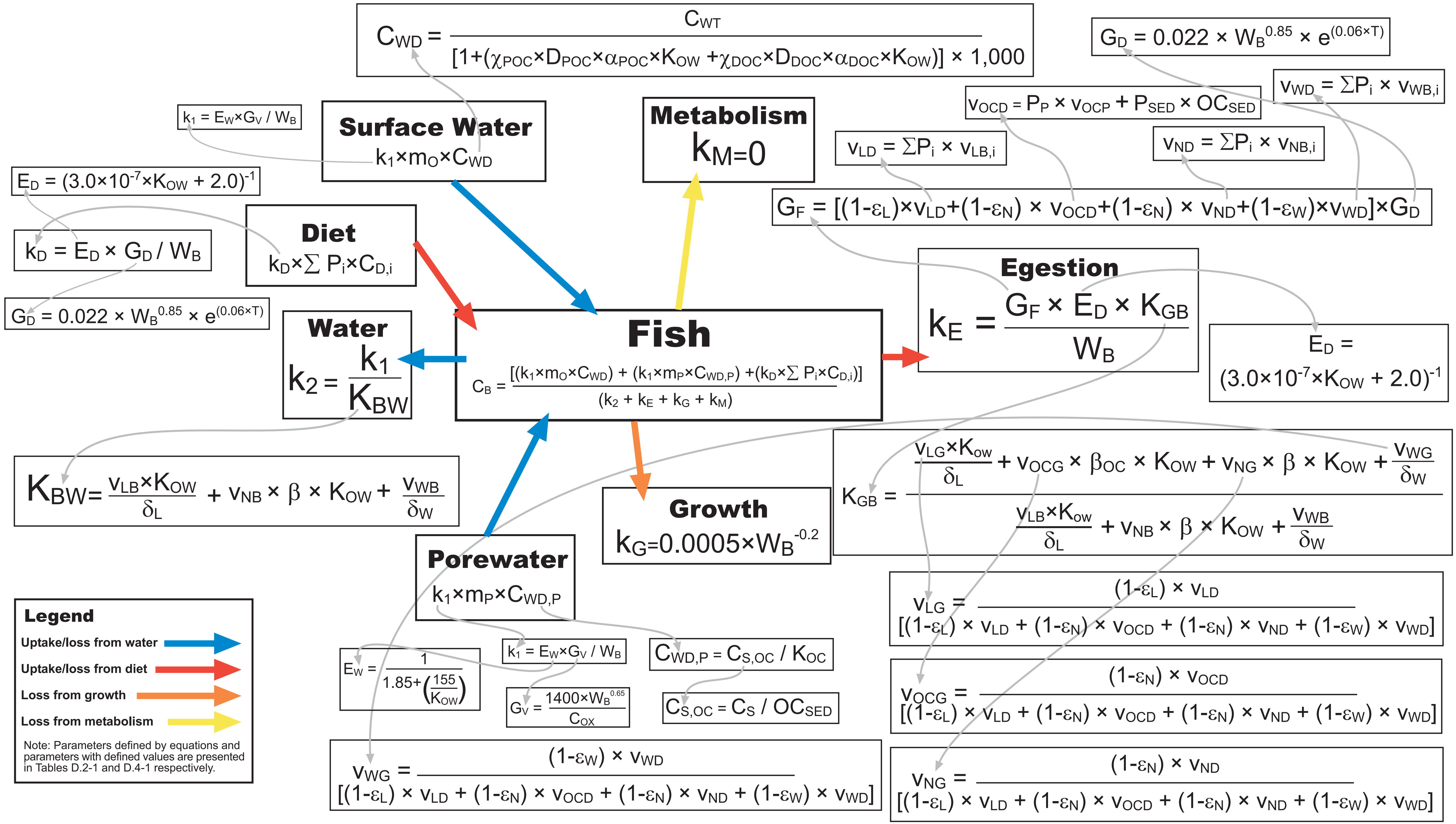
[1] A steady-state assumption means that concentrations of chemicals in tissues are assumed to not change over time or that concentrations of chemicals in tissues maintain a state of relative equilibrium even after undergoing fluctuations or transformations. The steady-state assumption is reasonable 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.

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 NLOM [e.g., proteins and carbohydrates] or 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). Applicability of these assumptions to the LDW is a significant uncertainty that should be considered in interpreting model output. The fact that the Arnot and Gobas model includes species-specific compartments, multiple pathways, and mechanistic equations makes the model more complicated than other available methods, such as the use of a biota-sediment accumulation factors, which represent empirical relationships between few variables. The increased complexity of the Arnot and Gobas model does not necessarily increase the likelihood that the model estimates will be more accurate because the values used for certain parameters are derived from literature (rather than site-specific data). However, the model can be used as a tool to assess the relative importance of various pathways and mechanisms and can potentially be used to enable better estimates under varying conditions. It should also be noted that several different parameter sets can result in the same tissue estimates.

[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.



Windward LLC

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

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). Details on the model equations, including definitions for all model parameters, may be found in Arnot and Gobas (2004a). 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 (C_{WT}) are entered as whole water total PCB concentrations. The dissolved fraction (C_{WD}) 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; thus, sediment exposure is the same regardless of whether exposure occurs while sediments are settled at the bottom of the water column or are suspended in the water column as particulates. 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)
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)

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

Parameter	Symbol	Unit	Equation	Notes	Source
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)
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 NLOC content.	January 2006 (Gobas 2006) update to Arnot and Gobas model (Arnot and Gobas 2004a)

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

PARAMETER	SYMBOL	UNIT	EQUATION	NOTES	SOURCE
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 content of phytoplankton	v_{OCP}	kg NLOC/kg phytoplankton	$v_{OCP} = 1 - (v_{LP} + v_{WP})$		Arnot and Gobas (2004a)
Fraction of non-lipid organic matter in organism *i*	$v_{NB,i}$	kg NLOM/kg organism	$v_{NB,i} = 1 - (v_{LB,i} + v_{WB,i})$	B = biota	Arnot and Gobas (2004a)
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)
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)

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

Information on the foraging ranges, specific habitat utilization, and migratory patterns of the modeled species within the LDW is for the most part unavailable; therefore, the spatial extent of their PCB exposure is uncertain. This section provides an overview of available literature and local expert opinion regarding exposure information for each

---

[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.

of the target species based on inferences from studies conducted in areas outside the LDW. Dietary preferences for each of the target species are discussed in Section D.4.2.2.

According to local fish experts, the FWM target species are likely to have foraging areas that are smaller than the entire LDW, with the possible exception of English sole and Dungeness crabs. However, uncertainty exists regarding the sizes of these areas. Thus, two different spatial scales were modeled (i.e., LDW-wide and modeling area scales). The information available regarding the seasonal movements and home ranges of the target species in the LDW is summarized below.

Adult English sole migrate seasonally out of the LDW system in order to spawn over the course of the winter, with spawning generally occurring in February and March. In Puget Sound, adult populations of English sole congregate in Elliott Bay and Port Gardner for winter spawning and then disperse. Angell et al. (1975) reported the off-season migration of central Puget Sound fish in winter and spring, from Meadow Point to Carkeek Park (northwest of downtown Seattle), at depths of 3 to 30 m. English sole are believed to maintain migration patterns throughout their lives (Day 1976). Home range estimates of approximately 3 km^2 (1.2 square miles) have been developed for English sole using acoustic tracking (O'Neill et al. 2005) and an empirical relationship between sediment PAH concentrations and lesion prevalence (Stern et al. 2003). Estimates of approximately 9 km^2 (3.5 square miles) were reported in the Puget Sound Dredged Disposal Analysis (PSDDA) report based on best professional judgment (PSDDA 1988). During September 2004, 2005, and 2007 trawl sampling throughout the LDW, the abundance of adult English sole (> 200 mm) in the lower waterway (i.e., from RM 0.0 to RM 2.5) was greater than in the upper waterway (RM 2.5 to RM 4.8) (Windward 2005c, 2006a, 2009).

Information available on shiner surfperch suggests that the LDW likely supports resident juveniles and first-year adults in addition to second- and third-year adults that migrate from Puget Sound during summer mating and parturition. February to October monthly beach seine sampling data from locations throughout the LDW and into Elliott Bay indicate that shiner surfperch are rare in the LDW from February through April and abundant from May through October (Shannon 2006). Shiner surfperch abundance in the LDW peaks in the summer, when they bear their young (Miller et al. 1975; Shannon 2006). September 2004, 2005, and 2007 trawl data indicated an increasing abundance of adult shiner surfperch (> 80 mm) from downstream to upstream in the LDW (Windward 2005c, 2006a). In San Francisco Bay, females migrate from nearshore coastal waters in the summer prior to giving birth in the bay. During their first year after birth, most females remain in San Francisco Bay and give birth before migrating to the ocean; males, on the other hand, migrate to the ocean soon after birth. Morrow (1980) also describes the inshore-offshore migration of shiner surfperch in Alaska. No data on shiner surfperch foraging ranges are available for the LDW.

Pacific staghorn sculpin were present in LDW beach seine samples in all months sampled (February through October) (Shannon 2006), although it is not known if they are year-round residents. During September 2004, 2005, and 2007 trawl sampling throughout the LDW, adult Pacific staghorn sculpin (> 150 mm) were collected in all areas sampled with similar abundance throughout. In San Francisco Bay, adult Pacific staghorn sculpin are reported to be present throughout the year in marine areas but seasonally absent from freshwater and slightly saline areas (Jones 1962). Adults are reported to be intolerant of brackish water (Jones 1962). In San Francisco Bay, young-of-the-year move into freshwater areas for rearing and move to more saline waters as they grow (Jones 1962). Tagged subyearlings (< 150 mm) in Tomales Bay, California, were reported to have home ranges less than 800 m (Tasto 1975). No studies reporting the migration of adults were identified; however, PSAMP reports that Pacific staghorn sculpin have restricted home ranges (WDFW 2002b).

Results from a quarterly survey of the LDW indicate that the abundance of Dungeness crabs may not vary substantially throughout the year (Windward 2004a), although it is not known if Dungeness crabs are year-round residents. In California, female Dungeness crabs are reported to have annual home ranges less than 2 km (1.25 miles) (Diamond and Hankin 1985, as cited in Pauley et al. 1986). A separate report states that most migrations in California waters were less than 10 miles, but some individuals moved up to 100 miles, with males moving farther than females (CDFG 2002). PSAMP reports that Dungeness crabs seasonally move between estuaries and offshore waters (WDFW 2002a). Samples collected during late August or early September of 2004, 2005, and 2007 suggest that abundance throughout the LDW is not highly variable, with the exception of RM 1.6 to RM 2.4, where Dungeness crabs were rare during sampling events (Windward 2005c, 2006a, 2009).

Results from a quarterly survey of the LDW suggest that the abundance of slender crabs does not vary greatly throughout the year (Windward 2004a). Slender crabs are able to withstand periods of low salinity but do not actively forage in brackish water areas (Curtis et al. 2007). Trawls conducted at the end of August or early September in 2004, 2005, and 2007 collected higher numbers of slender crab in the lower sections of the LDW (i.e., RM 0.0 to RM 2.5) relative to the upper section of the LDW (i.e., RM 2.5 to RM 6.0) (Windward 2005c, 2006a, 2009). Slender crab movements and home range in the LDW are unknown, and no information was identified on their migrations in other areas.

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).

Statistical analyses were conducted to investigate the relationship between the concentrations of chemicals in tissue from fish and crabs caught in the LDW and concentrations in sediment samples collected in the LDW. These analyses provided modest support for the assumption that during the time that English sole and crab species reside in the LDW, they 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 (880 µg/kg dw); whereas those in Areas T1 (300 µg/kg dw), T2 (270 µg/kg dw), and T4 (190 µg/kg dw) were below the SWAC for the LDW from RM 0.0 to RM 5.25 (380 µg/kg dw) (Figure D.3-1).

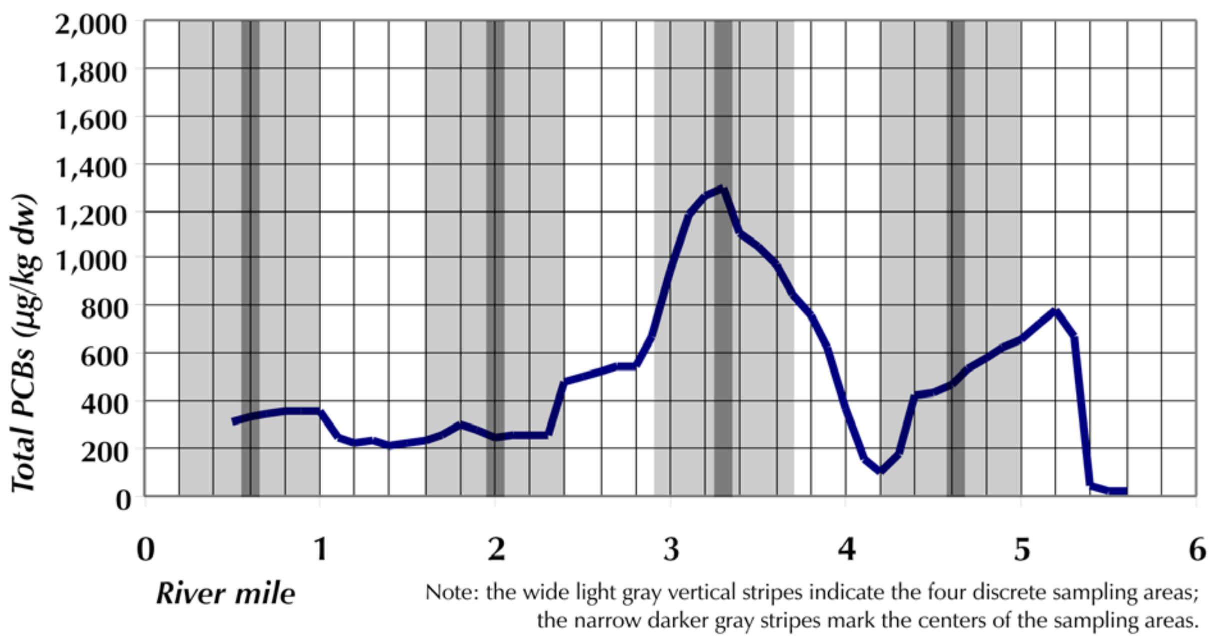


Figure D.3-1. One-mile rolling average total PCB concentration in LDW surface sediment

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 tissue concentrations 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]

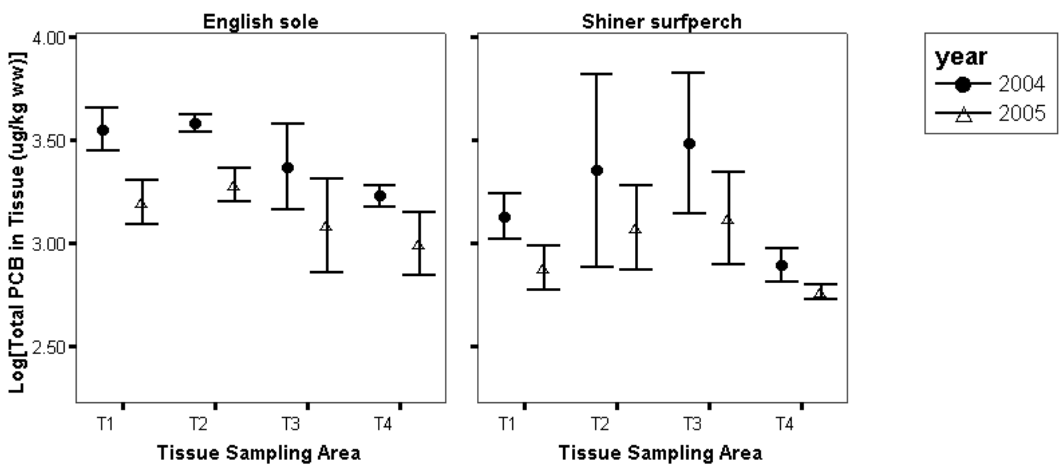


Figure D.3-2. Mean and 95% confidence intervals for total PCBs in English sole and shiner surfperch tissues by tissue sampling area

---

[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 ($p = 0.075$).

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]

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 (n = 24 in 2004, n = 22 in 2005) and from 23 of 24 subareas[12] for Pacific staghorn sculpin (n = 24 in 2004, n = 4 in 2005). Other species were sampled on an area-wide basis (see Map 4-9 in the main body of the RI). Regression relationships were analyzed using raw data, square root-transformed data, and log-transformed data. Relationships between dry and OC-normalized sediment concentrations were also examined. Regression relationships with the highest R^2 values (each had $p < 0.05$) are presented in Figures D.3-3 through D.3-5. 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 strong (Figure D.3-5, $R^2 = 0.29$). A regression analysis was not conducted using 2005 data 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.

[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.

[12] Two composite samples were collected from one subarea.

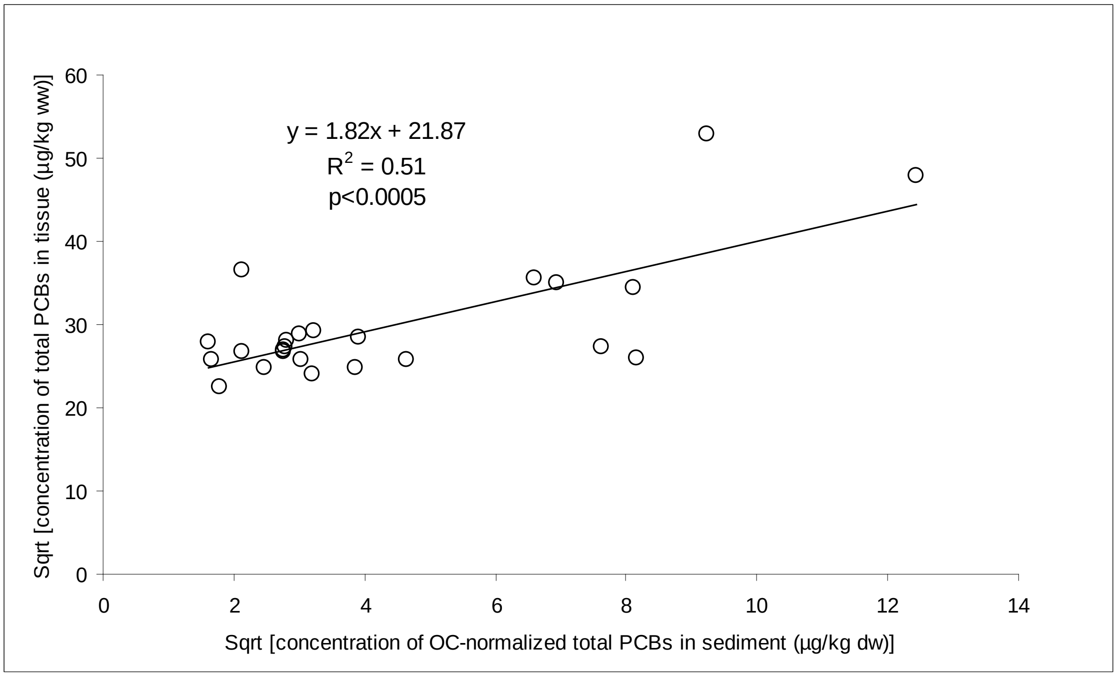


Figure D.3-3. Regression between total PCB concentrations in sediment and 2004 Pacific staghorn sculpin tissue on a subarea basis

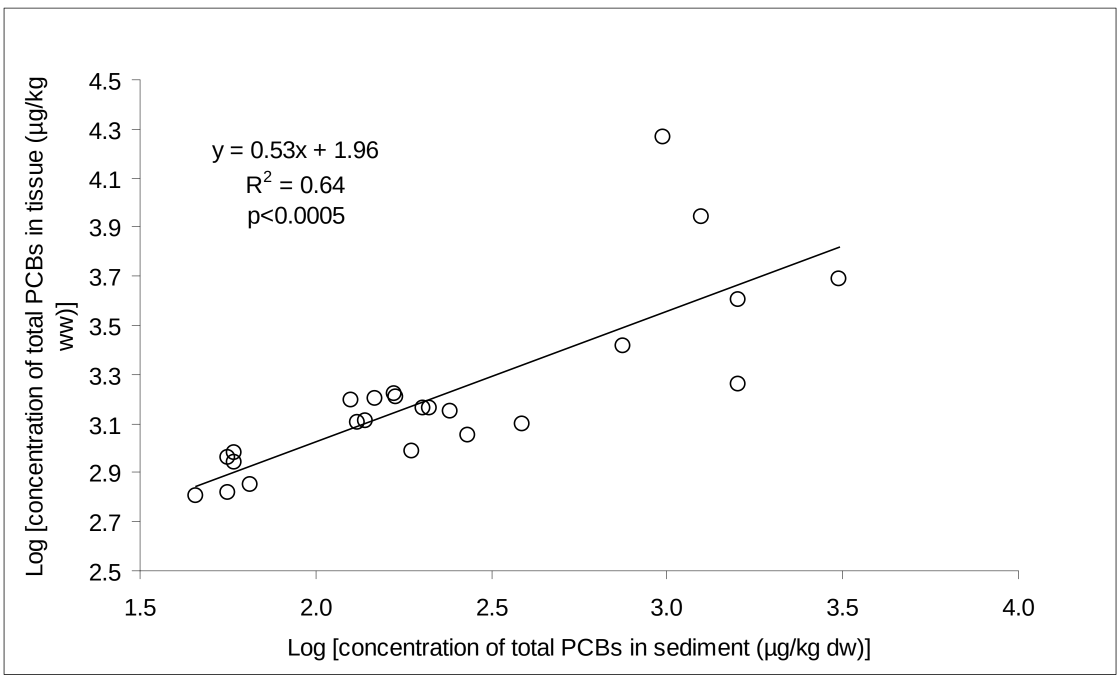


Figure D.3-4. Regression between total PCB concentrations in sediment and 2004 shiner surfperch tissue on a subarea basis

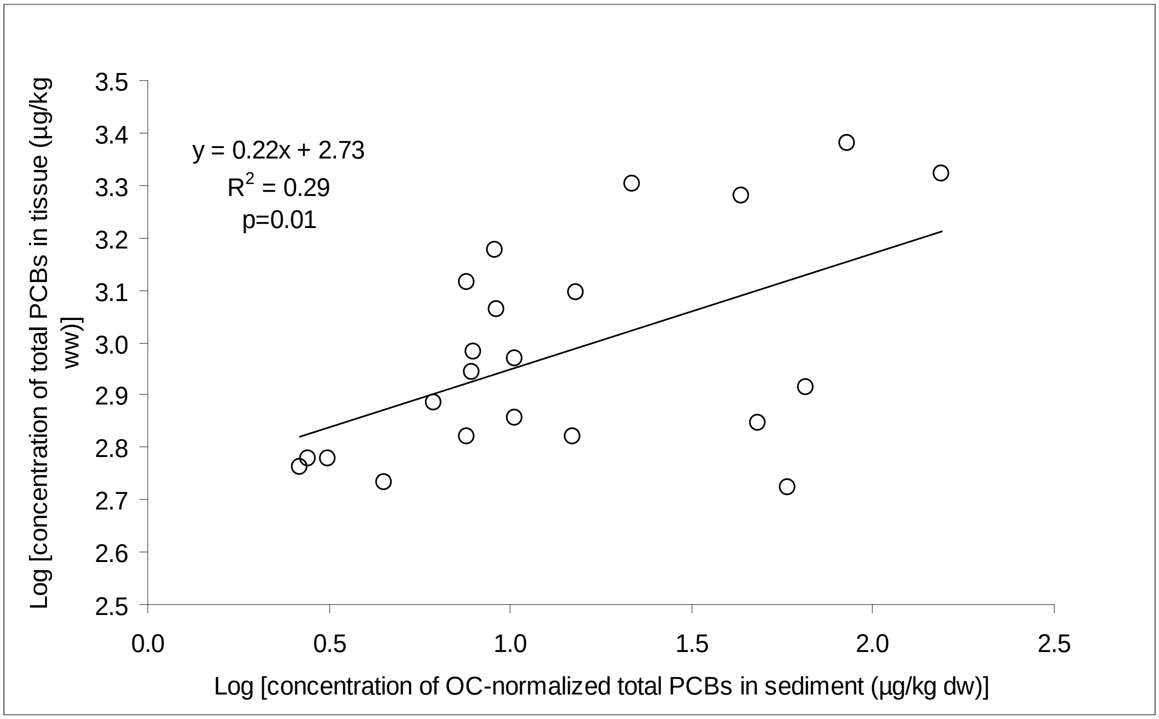


Figure D.3-5. Regression between total PCB concentrations in sediment and 2005 shiner surfperch tissue on a subarea basis

Results from the ANOVAs and regressions indicate that the application of the FWM at areas smaller than the LDW could be appropriate for shiner surfperch and Pacific staghorn sculpin because tissue concentrations varied among tissue sampling areas, patterns of tissue concentrations roughly corresponded to patterns of total PCB concentrations in sediment, and regressions of tissue sediment data were significant at the subarea scale. At the area scale for Pacific staghorn sculpin, a regression of log-transformed area mean tissue and sediment concentrations (weighted by area) was also significant with an R^2 of 99%. For shiner surfperch, regressions of area mean log-transformed tissue and sediment concentrations were not significant.

Tissue data were not available for English sole and crab at the subarea scale. Although the ANOVAs indicated differences in area mean tissue concentrations at the area scale for these two species, regressions of the log of area mean tissue concentrations vs. the log of area mean sediment concentrations were not significant for either species in either 2004 or 2005. 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.

One shiner surfperch composite sample collected from Subarea T2E in 2004 had a total PCB concentration (18,400 µg/kg ww) that was significantly higher than the rest of the data.[13] The total PCB concentration based on the sum of PCB congeners for that sample was also very high (12,230 µg/kg ww) and provided laboratory confirmation of the initial Aroclor results. To better understand the variability of total PCB concentrations in shiner surfperch collected from Subarea T2E in 2004, 10 archived fish from this subarea were analyzed individually.[14] Total PCB concentrations in these individual fish ranged from 172 to 1,140 µg/kg ww, with a mean concentration of 640 µg/kg ww. Based on these data, it is likely that one or more of the 10 fish included in the composite sample with a total PCB concentration of 18,400 µg/kg ww had a very high concentration of total PCBs.

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. Therefore, 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. In addition, a second type of analysis was conducted to evaluate the performance of the model at spatial scales smaller than that of the modeling area. This analysis focused on the shallow bench areas of the river, on either side of the navigation channel, and was designed to investigate the impact of exposure on species that may have smaller home ranges, specifically Pacific staghorn sculpin and shiner surfperch. The results of this analysis are also 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.

[13] Using Rosner's test for outliers from a log-normal distribution, this value was considered a statistical outlier ($p < 0.005$).

[14] A total of 20 shiner surfperch (> 80 mm) were collected in subarea T2E in 2004. Ten of these fish were included in the initial composite tissue sample for this area, and the ten remaining fish were archived frozen as individual fish.

To characterize a parameter distribution, several statistical descriptors (e.g., mean, mode, standard deviation [SD]) 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.

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 SDs 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 as discussed in Section D.4.2.2.3. 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.

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 used for the distribution is equivalent to the mean of 12 monthly averages from bottom three layers in EFDC model (Nairn 2009). Mean presented here is based on Equation D.4-1. 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.

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

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 between RM 0.0 and RM 5.25 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 between RM 0.0 and RM 5.25 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).

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

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 Section D.4.2.2.3 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.
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.
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).

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

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.

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

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.

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

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 × SE 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).

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

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.

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

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).

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

Parameter	Symbol	Unit	Values[a]	Distribution Type	Source/Notes
Lipid content	v_{LB}	%	mean = 4.6 SE = 0.19	normal	LDWG Phase 1 and 2 data (n = 49).
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.

Table D.4-1, cont. Input parameter probability distribution statistics and point estimate 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 SD in Crystal Ball™, assuming that values in the distribution were estimates of the mean.

b SE was represented by an 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 an 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:

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

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

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[15] (as a SWAC) based on RBTCs in tissue. Tissue RBTCs were derived based on the results of the baseline risk assessments (see Section D.9 and Section 8 in the main body of the RI).

The SWAC is considered to be a decision variable[16] 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 (Figure D.4-1). The total PCB SWAC was calculated using inverse distance weighting (IDW) interpolations derived from 1,264 surface sediment samples (Windward 2006b). The SWAC provides a less-biased estimate of average concentrations in areas where spatially biased sampling has occurred. The use of a SWAC does not address differential habitat use within the LDW by various species; instead, application of the SWAC assumes that all areas of the LDW are used equally by all species. This assumption is likely an over-simplification of habitat use, as discussed in Section D.7.

---

[15] 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).

[16] 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.

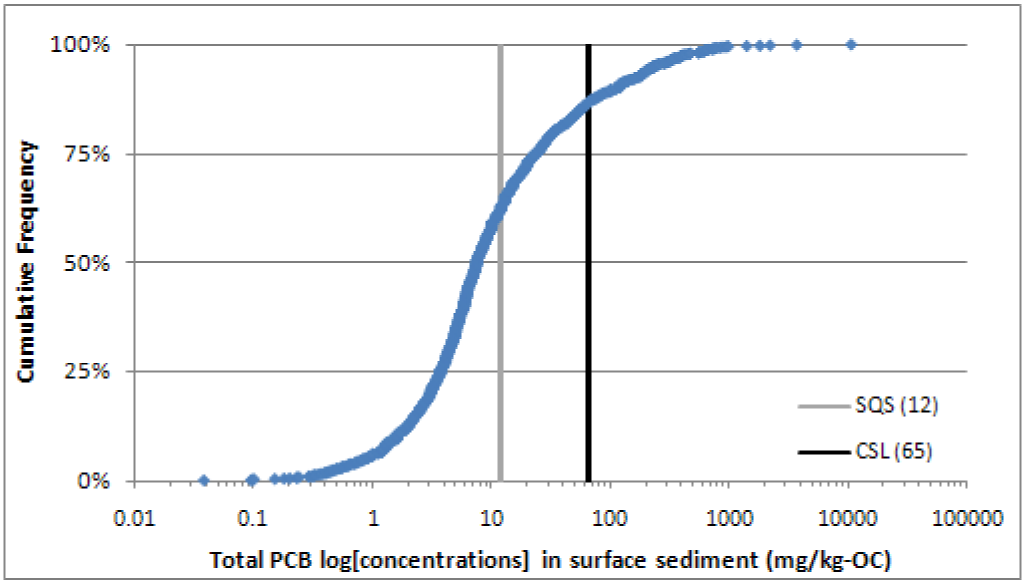


Figure D.4-1. Cumulative frequency of OC-normalized total PCB concentrations in surface sediment (log-scale)

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). Interpolation was required because most of the sampling in the LDW has been focused on areas that were known to have elevated concentrations of chemicals of concern. The comparatively large number of samples collected from contaminated areas biases the overall average concentration high and also imparts a spatial bias (i.e., regions that were more densely sampled are emphasized).

Interpolation methods were used to estimate the total PCB SWAC in the LDW surface sediment dataset to reduce the potential for bias. The process for creating SWAC estimates was developed in consultation with EPA and Ecology. 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 method for interpolation was selected for both technical and practical reasons, including the accuracy of the estimates. IDW is a deterministic method in which interpolated estimates are made based on concentrations at nearby locations. The IDW method creates a continuous surface of grid cells (10 x 10 ft), in which each cell is represented by a single estimated concentration. These individual grid cell concentrations are estimated as a function of the empirically determined concentrations at nearby locations; empirical data points are weighted by the inverse of their distance to the estimated cell, with the effect that nearby data points are given more weight than those farther away. The optimized interpolation resulted in a total PCB SWAC of 380 µg/kg dry weight (dw) for the LDW from RM 0.0 to RM 5.25[17] (Table D.4-1).

In order to develop a probability distribution for sediment organic carbon, mean and standard error (SE) statistics needed to be calculated. Thiessen polygons were used for calculating sediment organic carbon because calculation of SE 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 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 as monthly averages from the EFDC model (Nairn 2006, 2009).[18] The distribution of total PCB concentrations in water was assumed to be triangular because few site-specific 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 near RM 0.0 west 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

---

[17] 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 (350 µg/kg dw) was generated after the calibration of the FWM using a new IDW parameterization (see Section 4.2.2 of the RI) and the inclusion of additional surface sediment data collected as part of the RI. The effects of this new SWAC on model performance are discussed in Section D.6.2.3.

[18] 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]).

the fish and crab species being modeled spend the majority of their time in more saline, deeper waters in the estuary, means and SEs 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 PCB congeners. Seven bottom samples were analyzed for PCBs.[19] 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). These empirical data are summarized in Table D-4.2.

Table D.4-2. Total PCB concentrations in water based on empirical data and estimates from the EFDC model

Source of Water Data	Concentration (ng/L)		
	Mean	Minimum	Maximum
Empirical data for bottom water samples (Mickelson and Williston 2006)	1.31[a]	0.185	3.14
EFDC model data for bottom three cells (Nairn 2009)	1.43[b]		
EFDC model data for cells that correspond to water samples (Nairn 2009)	1.43[b]	0.1[c]	5.4[d]

a Mean of empirical data collected at two locations (two depths each) in August, September, November, and December 2005.

b Yearly mean based on monthly mean estimates from the bottom three cells throughout the LDW estimated from the EFDC LDW-wide model.

c Minimum EFDC model estimate (based on 3-hour-interval model prediction) for the location where the sample with the minimum total PCB concentration was detected (LTUM03, bottom).

d Maximum EFDC model estimate (based on 3-hour-interval model prediction) for the location where the sample with the maximum total PCB concentration was detected (LTUM03, bottom).

EFDC – Environmental Fluid Dynamics [Computer] Code

LDW – Lower Duwamish Waterway

PCB – polychlorinated biphenyl

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

The mode of the water distribution was estimated 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 model has been recalibrated (Nairn 2009).[20] The recalibrated version of EFDC was used to generate output to provide total PCB concentrations for the LDW FWM on an LDW-wide basis and for the four modeling areas. The EFDC model generated estimates of total PCB concentrations every 3 hours in each prediction cell for 1 year.[21] The 3-hour-interval estimated concentrations were then averaged within each month to derive 12 monthly average water concentrations for each prediction cell (Nairn 2009). Average monthly concentrations from all prediction cells in the bottom three water layers of the EFDC model were then averaged to represent an annual average total PCB concentration in the water column for the entire LDW; this concentration was used as the mode for the FWM (Table D.4-1). The EFDC model was also used to generate maximum and minimum concentrations for the locations where the maximum and minimum empirical water concentration data were collected (Table D.4-2). The model estimates for those locations bounded the range of empirical data; however, as described in Table D.4-1, the maximum and minimum of the empirical data were used to bound the triangular distribution used for the FWM. The EFDC modeling effort for predicting PCB water concentrations is further described in Attachment 3.

Average exposure concentrations were used to represent long-term exposure conditions averaged over the LDW because the modeled species spend long periods of time in the LDW (months to years) and integrate their exposure over their foraging range. In reality, total PCB concentration in water can vary on smaller scales and can

[20] 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.

[21] 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). Because depth varies from 3 to 36 ft in the LDW, the depth of water represented by the bottom three cells of the EFDC model varied from approximately 0.5 to 12 ft. Note that the EFDC model referred to in Section 3 of the main body of the RI, where the sediment transport model is discussed, had 7 model cells across the LDW versus the 3 model cells discussed above. The longitudinal grid resolution was also greater in the along-channel direction, using 80 cells of variable length to replace 30 cells in the configuration used to support the FWM. The coarser grid configuration was used in the EFDC model supporting the FWM in order to take advantage of the previous configuration and calibration for PCB and other chemical simulations and to avoid the lengthy simulation times that would be required with the more refined grid configuration.

also fluctuate daily and seasonally, as demonstrated by the EFDC model. For example, on a prediction cell basis, the maximum annual average concentration was predicted to be 16 ng/L, compared with an LDW-wide average of 1.3 ng/L. The total PCB concentration in individual prediction cells can vary by a factor of 6 or more during a day, as tidal conditions change (Nairn 2009). Therefore, while the FWM evaluates exposures over the long-term, some species such as English sole could have short-term exposures to water PCB concentrations much higher than the long-term spatial average concentrations. Seasonal variability has also been demonstrated in the LDW and elsewhere. Studies of the Kalamazoo, Hudson, and Grasse Rivers have demonstrated that during late spring and summer (i.e., during periods of low flow), PCBs can be transferred from the sediment to the water column (dissolved phase) at rates that are higher than those estimated by standard chemical fate and transport models (Thibodeaux and Bierman 2003). The model predicted just such seasonal changes during low-flow periods. Higher concentrations in water just above the sediment surface were estimated on the benches in a few specific areas with higher sediment concentrations. These higher concentrations were included in the EFDC model output used to generate the average concentrations used in the FWM. Dilution and mixing tend to diminish the importance of this contribution over time.

The EFDC model does not simulate the effect of temperature on the physical properties of PCB compounds, nor does it include any simulation of biological activity. The temperature in the LDW surface water varied between 4°C and 17°C, and the near-bottom water ranged between 7°C and 14°C based on historical King County data. No information is available on the relative amount of biological activity or the influence of that activity on stabilizing or destabilizing sediments. The EFDC model was calibrated to water column PCB concentrations obtained between August and December. This period of calibration should encompass variations resulting from changes in temperature and biological activity. Even without simulating temperature or biological effects, predicted concentrations tend to increase in late summer and decline afterwards, following the same trend as the empirical observations (Nairn 2009). While consideration of these processes could lead to potential improvements in model calibration, the existing level of calibration suggests that the impact is likely to be small.

Because the LDW water samples (Table D.4-2) were collected in the middle of the navigation channel rather than directly above the benches, where PCB concentrations are known to be the highest, it is possible that the extreme high end of the water PCB range was not captured in the empirical data. However, because the PCB water concentration used in the FWM represents a yearly average of the exposure throughout the entire LDW, the uncertainty associated with the variability in LDW water concentrations is minimized.

The bottom three cells from the EFDC model, which represent 1 to 12 ft of water depth, were selected to represent the exposure of all species to PCBs in the water column (i.e., was set equal to the mode of water concentration triangular distribution). Although different species occupy different water depths in the LDW, and thus could be exposed to somewhat different concentrations of waterborne PCBs, most of the species modeled (e.g., English sole, crabs, clams, benthic invertebrates) spend the majority of their lives at or near the sediment/water interface. The FWM is not amenable to assigning different PCB concentrations in water to different species. Instead, the potentially higher exposure of organisms dwelling on the river bottom to chemicals present in sediment is accounted for by the species-specific porewater ventilation rate. The degree of porewater exposure was set proportional to the porewater ventilation rate assumed for each modeled species. Species with high porewater exposure (e.g., benthic invertebrates) had high porewater ventilation rates; species with low porewater exposure (e.g., shiner surfperch) had low porewater ventilation rates. The total PCB concentrations in porewater were estimated in the FWM based on equilibrium partitioning with sediment.

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.4-3). 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.4-3). Total PCB concentrations were used in model calibration, as discussed in Section D.5.

Table D.4-3. 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, b)
	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, e)
		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 SE 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)

Lipid content is a particularly important parameter in the bioaccumulation of hydrophobic chemicals. The lipid content of fish and crab collected from the LDW varied within each sampling event and from year to year. Seasonal differences may have also contributed to differences in lipid content in the samples collected in the 1990s relative to those collected for the RI. The variability in lipid contents could be the result of variability in food abundance, food type, changing dietary preferences, or a myriad of other factors that could affect the condition of fish and crabs. Figure D.4-2 shows the available lipid content data for whole-body English sole and shiner surfperch and in Dungeness and slender crab edible tissue samples. Graphs presenting variability in lipid and PCB concentrations over time are presented in Appendix E.5.

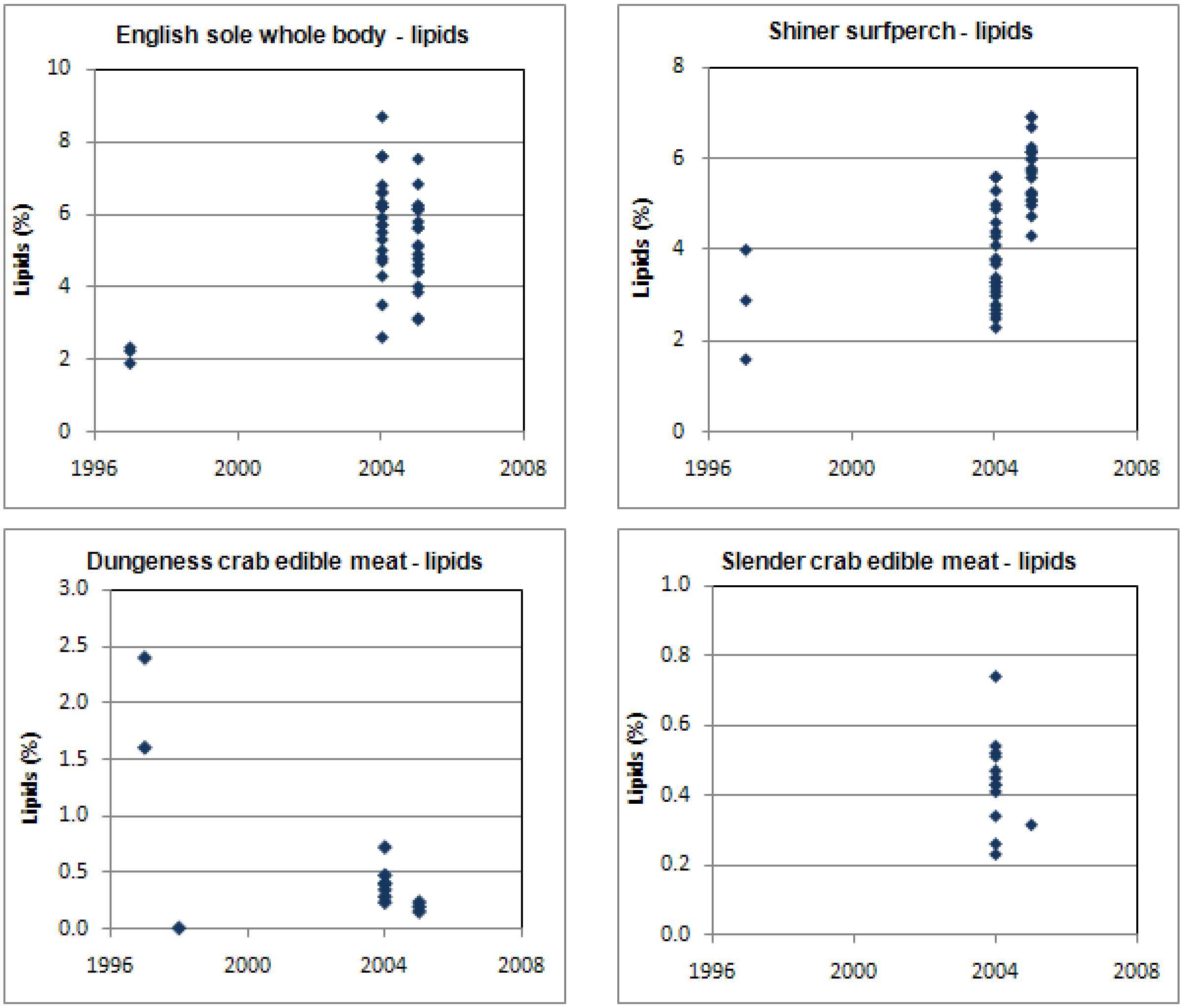


Figure D.4-2. Variability in fish and crab lipid content

The 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 SE 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:

$$V_{wb} = (V_h \times F_h) + (V_{em} \times F_{em}) \qquad \textbf{Equation D.4-5}$$

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[22] dissected at Windward Environmental LLC.[23] 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,[24] 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

---

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

[23] 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.

[24] 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.

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.[25]

D.4.1.3.2 Body weights

Mean and SE 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.4-6.

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

$$W_B = \frac{\sum W_{C\,(i...n)}}{n}$$ **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 SE 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)

---

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

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[26] (Windward 2004b). 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:

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

Where:

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

Average and SE 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 concentration-weighted average log K_{OW} value for total PCBs was estimated using site-specific concentrations of individual PCB congeners in benthic invertebrate tissue and the log K_{OW} values for individual PCB congeners from the literature (Equation D.4-9). A concentration-weighted average log K_{OW} was calculated using Equation D.4-9 for the eight benthic invertebrate tissue samples for which all 209 individual PCB congeners were analyzed (Windward 2005a) (Table D.4-4). PCB congener-specific log K_{OW}s were taken from Hawker and Connell (1988).

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

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

[26] 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.4-4. Weighted log K_{OW}s for benthic invertebrate composite samples

Sample ID	Weighted Log K_{OW} (L/kg)
LDW-B1b-T	6.6
LDW-B2a-T	6.5
LDW-B3b-T	6.5
LDW-B4b-T	6.5
LDW-B5a-T	6.4
LDW-B8a-T	6.9
LDW-B9b-T	6.5
LDW-B10a-T	6.5
Mean	**6.6**
Standard error	**0.05**

ID – identification
K_{ow} – octanol water partition coefficient

The decision to use a log K_{OW} based on benthic invertebrate tissue data was made in collaboration with EPA and NOAA. The mean and SE of the eight weighted log K_{OW} values for benthic invertebrates were used to define the normal distribution for log K_{OW}. The average log K_{OW} derived based on the pattern of PCB congeners in all available tissue types (i.e., fish, crabs, shellfish, and invertebrates) was the same as the average derived using only benthic invertebrate samples (Table D.4-5).

Table D.4-5. Average weighted log K_{OW}s for LDW species

Species	Number of Samples	Weighted Log K_{OW} (L/kg)	
		Range	Average
English sole	7	6.5 – 6.6	6.6
Pacific staghorn sculpin	8	6.6 – 6.8	6.7
Starry flounder	1	6.6	6.6
Shiner surfperch	9	6.4 – 7.0	6.7
Dungeness crab	5	6.5 – 6.7	6.6
Slender crab	7	6.6	6.6
Benthic invertebrates	8	6.4 – 6.9	6.6
Clams	8	6.2 – 6.8	6.4
Overall	**53**	**6.2 – 7.0**	**6.6**[a]

[a] The overall average was calculated using data from the 53 individual samples; it was not determined from the species averages.
K_{ow} – octanol water partition coefficient
LDW – Lower Duwamish Waterway

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 SE 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 SD value for juvenile fish lipids[27] (0.6%) was derived from a study of salmon prey fish in the Bering Sea (Nomura and Davis 2005) as the SE of lipid content for 19 juvenile and small fish species (0.3%) multiplied by a factor of 2. The SE 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.

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 average feeding preference of the LDW resident 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.[28]

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

---

[27] 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).

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

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.

Although the FWM has five compartments, four compartments serve to categorize dietary prey species for fish or crabs: phytoplankton/algae, zooplankton, benthic invertebrates, and juvenile fish. The two benthic invertebrate compartments were combined into a single prey compartment to represent scavenger/predator/detritivore benthic invertebrates as well as small filter-feeding benthic invertebrates.[29] These species were combined into a single compartment because the empirical benthic invertebrate data from the LDW were available as composite samples with multiple species and because prey preference information is limited at the level of specific benthic invertebrate species. Zooplankton represent herbivorous invertebrates exposed to chemicals in the water column.[30]

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.4-6). 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

---

[29] Large clams, *Mya arenaria*, were modeled separately (see Section D.7.3).

[30] 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).

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 1977). These dietary scenarios were used to develop probability distributions applied in the FWM, as discussed in Section D.4.2.2.3.

Table D.4-6. 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	
Dungeness crab	zooplankton	0	0.48	0	0	Stevens et al. (1982) for Scenarios 1 and 2; Gotshall (1977) 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.4-7).

Table D.4-7. 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[31] benthic samples based on the literature[32] 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

[31] 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.

[32] 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).

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 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.4-8). Because the dietary scenarios for each species were created using different assumptions, they represented a range of variability and uncertainty in the average diets (e.g., variability and uncertainty in mean population-level feeding preferences). 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.4-8) to facilitate comparison with post calibrated values, which are presented as mean, maximum, and minimum values (Attachment 2).

Table D.4-8. Summary of triangular dietary distributions for LDW food web model

Species	Dietary Item																			
	Sediment				Phytoplankton				Zooplankton				Benthic Invertebrates				Juvenile Fish			
	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 SD) 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.9). 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 fact that a model has the ability to accurately estimate concentrations using the calibration dataset does not necessarily indicate that the model will accurately predict actual concentrations under all conditions.

The FWM is a steady-state model (i.e., it assumes that concentrations do not change as a function of time). Thus, it was not designed to estimate changes in tissue concentrations resulting from short-term physical perturbations in the system (e.g., seasonal influences, such as changing prey availability, or short-term physical disturbances, such as dredging). 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; to be effective, the calibration data set should represent "average" conditions expected in the LDW.

The empirical data (i.e., total PCB concentrations in tissues collected from the LDW) used for calibrating the FWM were collected in the late 1990s, 2004, and 2005. The largest datasets were collected in 2004 and 2005. Based on empirical data, total PCB concentrations in tissue (as a sum of Aroclors) were higher in several species in 2004 than in other years (see Section 4.2.2.4 and Appendix E.5). Although additional tissue data were collected in 2006 and 2007, these data were not used for FWM calibration because they were not available at that time.

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."[33] Each parameter set yielded an estimate of total PCB concentrations in tissues of the modeled species.

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

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

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.4-8. 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. Mean and range information for the empirical dataset used in model calibration is presented in Table D.5-1.

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).

$$C_{BI} = (0.34 \times C_S) + 75 \qquad \textbf{Equation D.5-1}$$

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.

Table D.5-1. Empirical dataset for calibration: 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): $C_{BI} = (0.34 \times C_S) + 75$.	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
n – number of composite samples
na – not applicable
PCB – polychlorinated biphenyl
ww – wet weight

Clams were included as target species in the FWM to support calculations of sediment RBTCs for human health consumption scenarios (see Section D.9). The FWM was not calibrated for clams because clams are present only in intertidal areas in the LDW with suitable habitat.[34] 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:

$$SPAF = \frac{C_M}{C_E}$$ **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:

$$SPAF = \frac{C_E}{C_M}$$ **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 Lower Duwamish Waterway Group (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

[34] The FWM was not calibrated using empirical clam tissue data because the empirical data were not considered representative of the entire area being modeled because large clams for which empirical data are available (i.e., *Mya arenaria*) were only collected in intertidal areas where clamming might occur. Model estimates of PCB concentrations in clam tissues were compared to empirical data from the various LDW areas to assess model performance. This assessment was needed to ensure that the estimates were sufficiently similar to empirical data to be used in the derivation of RBTCs (see Section D.7.3).

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. For example, 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.

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 SPAF across species for parameter sets passing the model performance criterion was 1.4 (Table D.5-2). The SPAF for the best-fit parameter set was 1.2. 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.5-2).

Table D.5-2. Summary of model performance

Species	SPAFs from Parameter Sets that Passed the Model Performance Filter			
	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[a]	2.0	1.5
Slender crab	1.0	**2.0**	2.0	1.0
Dungeness crab	1.0	**2.0**	2.0	1.1
Pacific staghorn sculpin	1.0	**1.2**	2.0	1.2
Shiner surfperch	1.0	**2.0**	1.2	**1.2**
English sole	1.0	**1.3**	1.7	1.1
Average SPAF	1.0	**1.6**	1.8	1.2

a There were no under-predictions for benthic invertebrates.

na – not applicable

SPAF – species predictive accuracy factor

Bold indicates an under-prediction.

Estimated total PCB concentrations in fish and crab tissues associated with the best-fit parameter sets were generally similar to mean empirical data for each species (Figure D.5-1). The estimates associated with the best-fit parameter sets were generally higher than the mean empirical data, with the exception of shiner surfperch. Possible reasons for overestimation are discussed in Section D.6.2.4. The benthic invertebrate tissue concentration, estimated using the tissue-sediment regression and an LDW-wide SWAC of 380 µg/kg dw, was lower than the benthic invertebrate tissue concentration estimated by the FWM using the best-fit parameter set (Figure D.5-2).

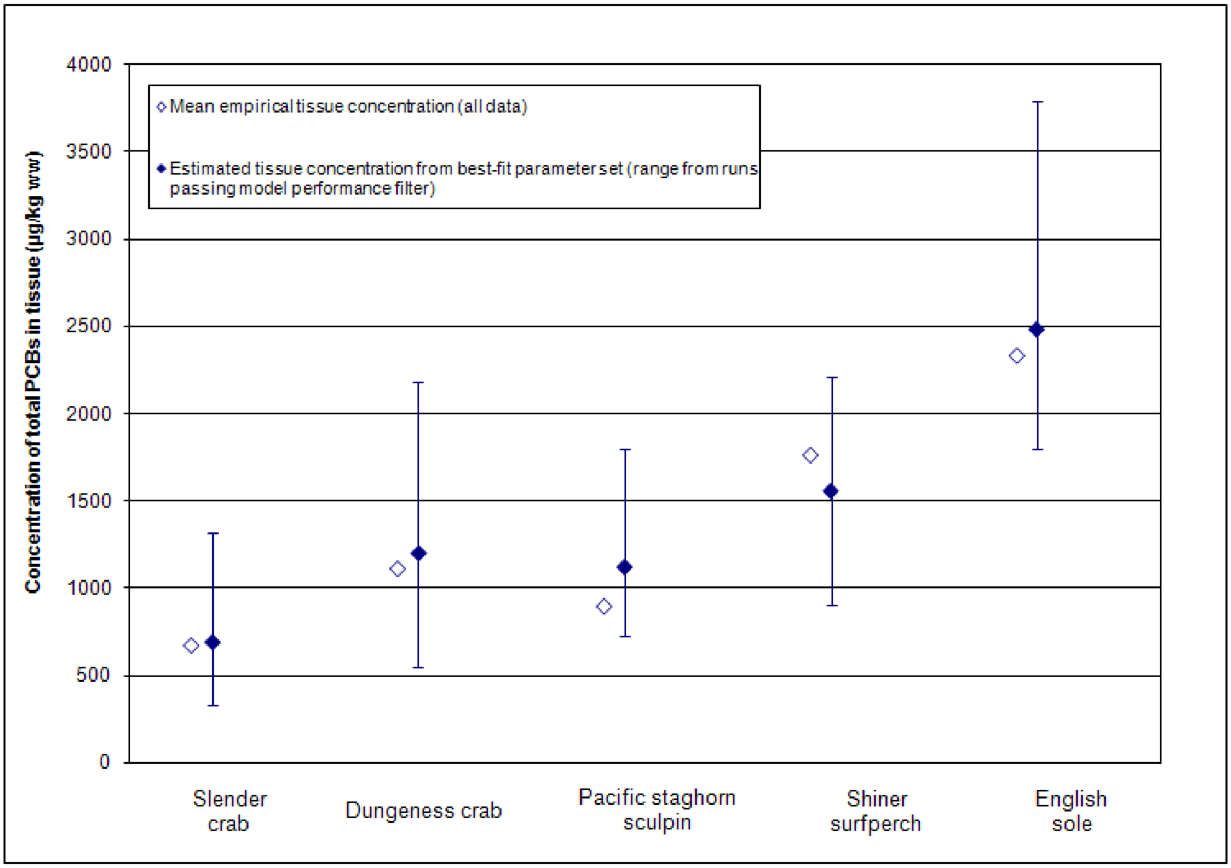


Figure D.5-1. Estimated total PCB concentrations in tissues of adult fish and crab species for parameter sets that passed the model performance filter in the best-fit model parameter set relative to empirical data

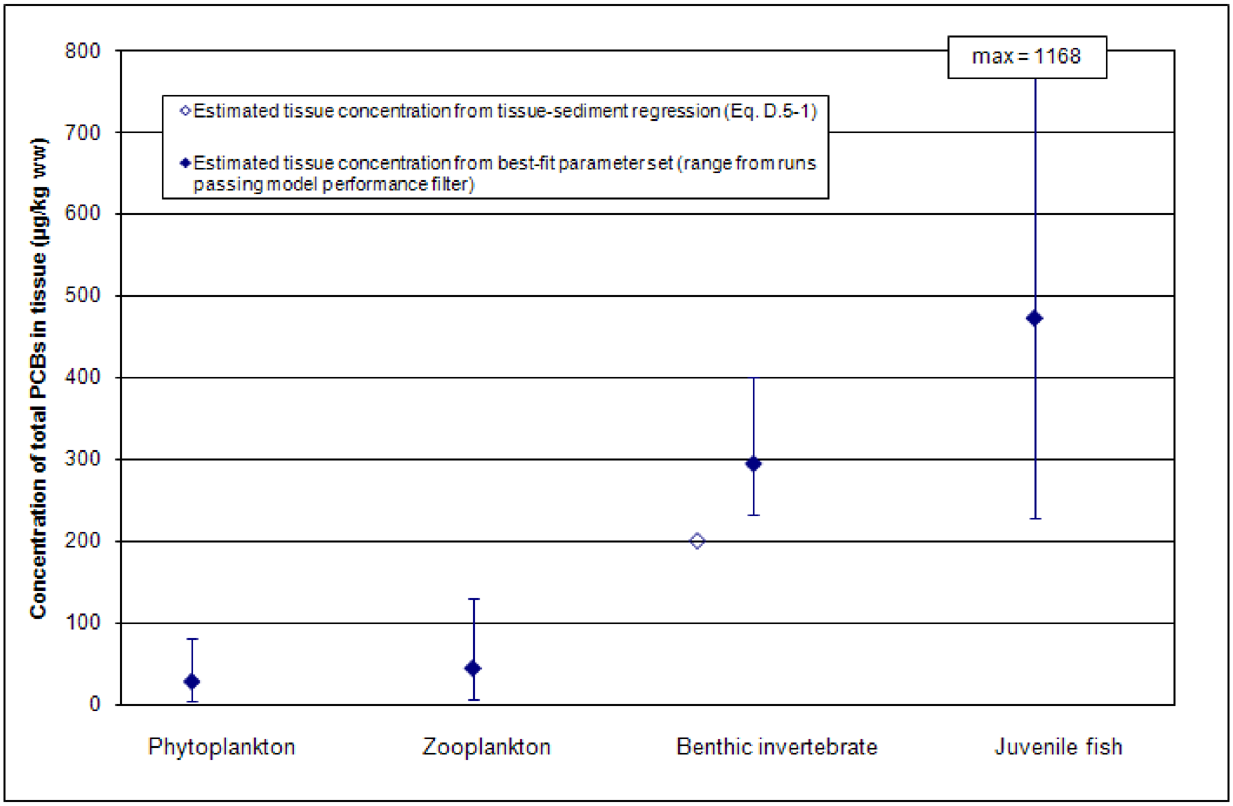


Figure D.5-2. Estimated total PCB concentrations in tissues of prey species for parameter sets that passed the model performance filter in the best-fit model parameter set relative to empirical data

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. Relative to the original model input values (Table D.4-1), the best-fit parameter set (Table D.5-3) 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 [surrogate for detritus] 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.5-3. Best-fit parameter set for the calibrated model

Parameter Description	Unit	Best-Fit Parameter Set
Environmental Parameters		
Concentration of total PCBs in the water column	ng/L	1.22
Concentration of POC in the water column	kg/L	2.3×10^{-7}
Concentration of DOC in the water column	kg/L	2.2×10^{-6}
Mean water temperature	°C	11.0
Concentration of dissolved oxygen in the water column	mg/L	8.15
Concentration of total suspended solids in the water column	kg/L	5.4×10^{-6}
Concentration of total PCBs in sediment	µg/kg dw	380
Sediment total organic carbon	%	1.91
Chemical Parameters		
Octanol-water partition coefficient for total PCBs (log K_{ow})	unitless	6.5
Biological Parameters		
Proportionality constant expressing the sorption capacity of NLOM relative to that of octanol (β)	unitless	0.031
Resistance to chemical uptake through aqueous phase for phytoplankton/ algae (A)	day-1	6×10^{-5}
Resistance to chemical uptake through organic phase for phytoplankton/ algae (B)	unitless	6.2
Density of lipids	kg/L	0.9
Phytoplankton		
Lipid content of organism	%	0.14
Water content of organism	%	95.7
Zooplankton		
Organism weight	kg	2.2×10^{-7}
Lipid content	%	1.4
Water content of organism	%	92
Dietary absorption efficiency of lipids (ε_L)	%	66
Dietary absorption efficiency of NLOM (ε_N)	%	72
Benthic Invertebrates		
Organism weight	kg	4.1×10^{-5}
Lipid content	%	0.83
Water content of organism	%	82
Relative fraction of porewater ventilated	unitless	0.13
Dietary absorption efficiency of lipids (ε_L)	%	30

Table D.5-3. cont. Best-fit parameter set for the calibrated model

Parameter Description	Unit	Best-Fit Parameter Set
Dietary absorption efficiency of NLOM (ε_N)	%	56
Juvenile Fish		
Organism weight	kg	6×10^{-3}
Lipid content	%	1.5
Water content of organism	%	74.3
Relative fraction of porewater ventilated	unitless	0.01
Dietary absorption efficiency of lipids (ε_L)	%	92
Dietary absorption efficiency of NLOM (ε_N)	%	54
Slender Crab		
Organism weight	kg	0.165
Lipid content	%	1.1
Water content of organism	%	83.7
Relative fraction of porewater ventilated	unitless	0.03
Dietary absorption efficiency of lipids (ε_L)	%	75
Dietary absorption efficiency of NLOM (ε_N)	%	76
Dungeness Crab		
Organism weight	kg	0.653
Lipid content	%	3.4
Water content of organism	%	81
Relative fraction of porewater ventilated	unitless	0.02
Dietary absorption efficiency of lipids (ε_L)	%	71
Dietary absorption efficiency of NLOM (ε_N)	%	59
Pacific Staghorn Sculpin		
Organism weight	kg	0.075
Lipid content	%	2.1
Water content of organism	%	79
Relative fraction of porewater ventilated	unitless	0.03
Dietary absorption efficiency of lipids (ε_L)	%	93
Dietary absorption efficiency of NLOM (ε_N)	%	50
Shiner Surfperch		
Organism weight	kg	0.019
Lipid content	%	4.6
Water content of organism	%	74.0
Relative fraction of porewater ventilated	unitless	0.02
Dietary absorption efficiency of lipids (ε_L)	%	94
Dietary absorption efficiency of NLOM (ε_N)	%	56

Table D.5-3. cont. Best-fit parameter set for the calibrated model

Parameter Description	Unit	Best-Fit Parameter Set
English Sole		
Organism weight	kg	0.246
Lipid content	%	5.5
Water content of organism	%	75.0
Relative fraction of porewater ventilated	unitless	0.1
Dietary absorption efficiency of lipids (ε_L)	%	92
Dietary absorption efficiency of NLOM (ε_N)	%	59
Dietary Fraction		
Benthic Invertebrates		
Sediment	fraction	0.70
Phytoplankton	fraction	0.18
Zooplankton	fraction	0.12
Juvenile Fish		
Sediment	fraction	0.00
Zooplankton	fraction	0.53
Benthic invertebrates	fraction	0.47
Slender Crab		
Sediment	fraction	0.02
Zooplankton	fraction	0.09
Benthic invertebrates	fraction	0.88
Juvenile fish	fraction	0.01
Dungeness Crab		
Sediment	fraction	0.00
Zooplankton	fraction	0.37
Benthic invertebrates	fraction	0.24
Juvenile fish	fraction	0.39
Pacific Staghorn Sculpin		
Sediment	fraction	0.00
Zooplankton	fraction	0.22
Benthic invertebrates	fraction	0.54
Juvenile fish	fraction	0.24
Shiner Surfperch		
Sediment	fraction	0.00
Zooplankton	fraction	0.23
Benthic invertebrates	fraction	0.76

Table D.5-3. cont. Best-fit parameter set for the calibrated model

Parameter Description	Unit	Best-Fit Parameter Set
English Sole		
Sediment	fraction	0.04
Phytoplankton	fraction	0.05
Zooplankton	fraction	0.05
Benthic invertebrates	fraction	0.86

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

The minimum, maximum, mean, and range for parameter values for the 10 best performing model runs (based on lowest average SPAF across species) are presented in Attachment 2.

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 both qualitatively and quantitatively. 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.

All models are simplifications of the processes and parameters that they describe. The calibrated FWM is designed to represent, to the extent possible, the complicated relationship between sediment and tissue, including aquatic organism life histories and foraging strategies across the food web. It is important to assess the potential uncertainties in the FWM so that these uncertainties can be acknowledged in the model's application. The following two sensitivity and uncertainty analyses were conducted using the best-fit parameter set and are described in this section:

- Correlation coefficient analysis
- Nominal range sensitivity (NRS) analysis

Because the SWAC was not varied 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 correlation coefficient and NRS analyses described in this section.

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 investigated (see Section D.6.3).

D.6.1 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.6.2) is a univariate analysis. Because the correlation analysis incorporated parameter interactions, it was the most suitable analysis for identifying the 20 most important parameters.

The 20 parameters with the highest average absolute value correlation coefficients across species are presented in Table D.6-1. 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 absorption 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.6-1. Parameters most strongly correlated with estimated total PCB concentrations in tissues

Parameter	Correlation Coefficient										
	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	**0.20**	0.10	-0.003	0.05	0.05	0.08	0.15	0.06	**0.20**	0.16	0.12
Weight of benthic invertebrates	**0.20**	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 identifies the maximum correlation for each parameter.

D.6.2 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.[35] Minimum and maximum parameter values were identified in the sets passing the model performance filter for each of the top 20 parameters (Table D.6-2).

Table D.6-2. 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}
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

[35] 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.

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:

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

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.

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.

Table D.6-3. NRS values for the top 20 parameters

Parameter	NRS (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.

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.6-1) 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.6.1 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.6-2). NRS is the absolute value of the difference between C_{Max} and C_{Min} (Equation D.6-1).

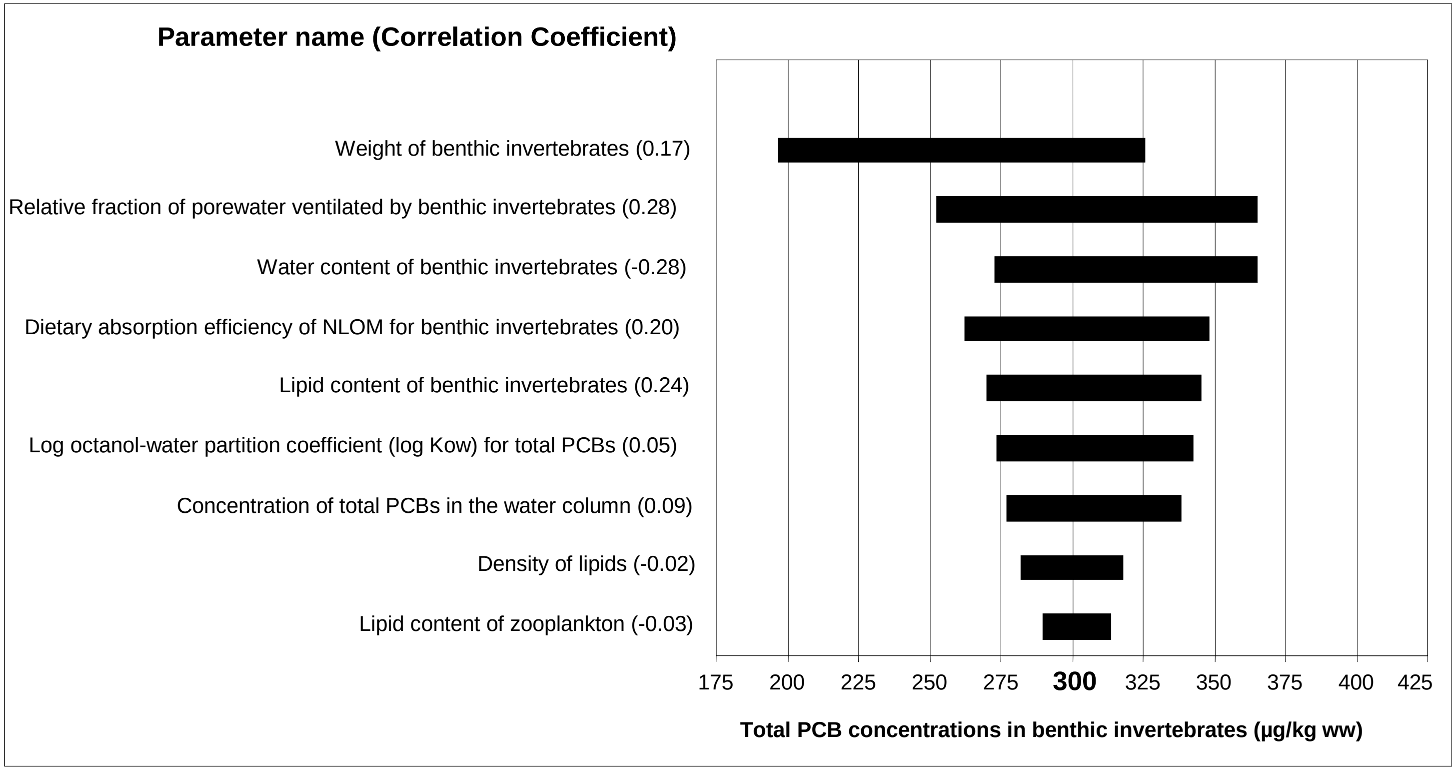


Figure D.6-1. Results of the NRS analysis for benthic invertebrates

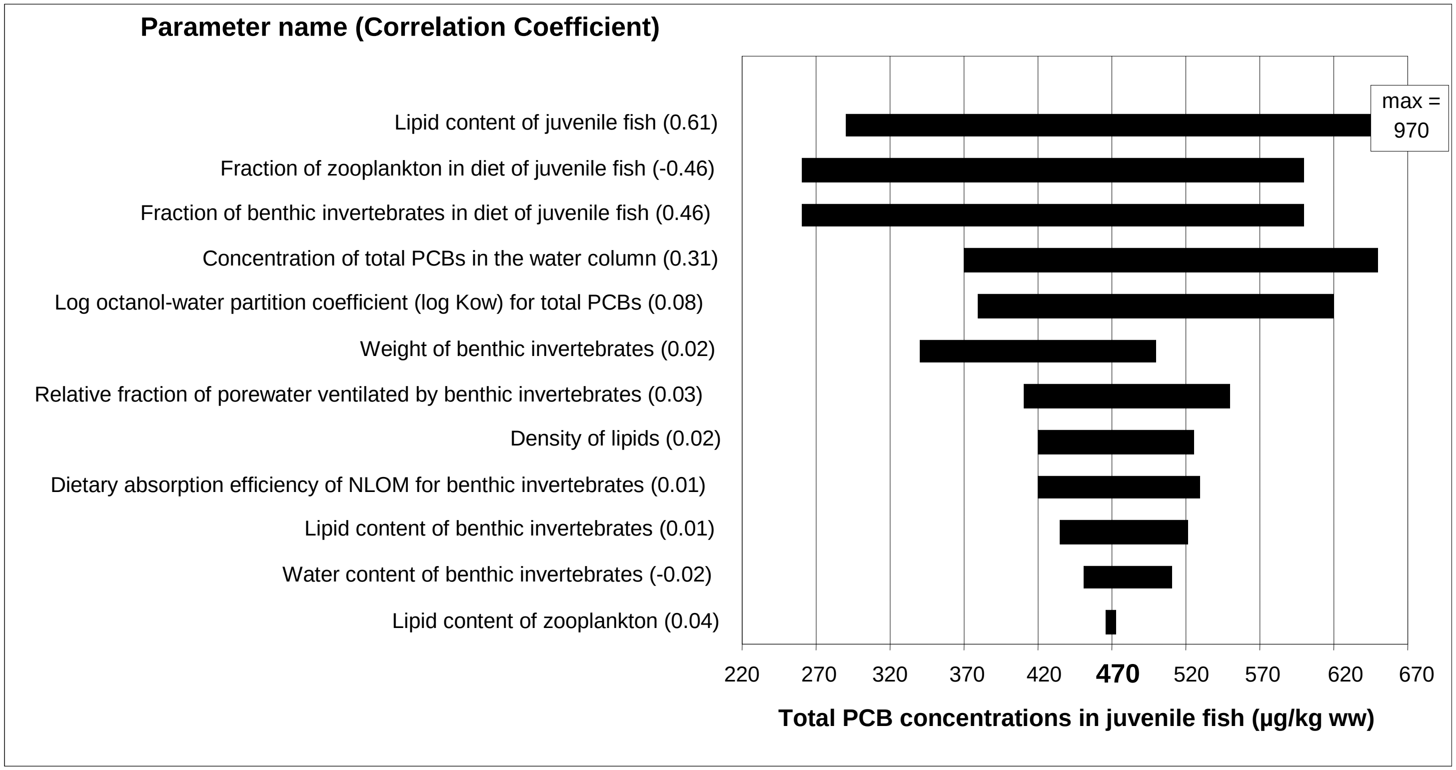


Figure D.6-2. Results of the NRS analysis for juvenile fish

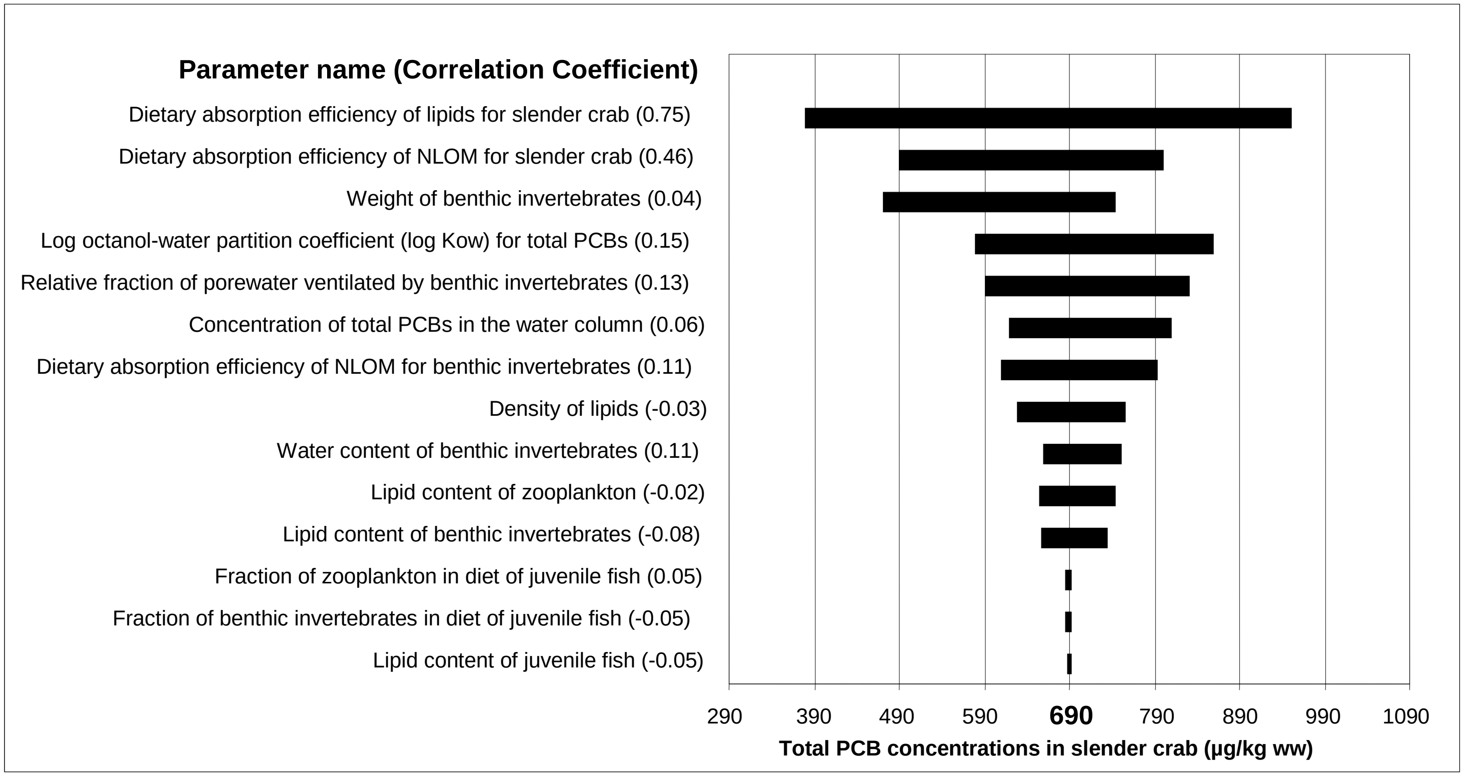


Figure D.6-3. Results of the NRS analysis for slender crabs

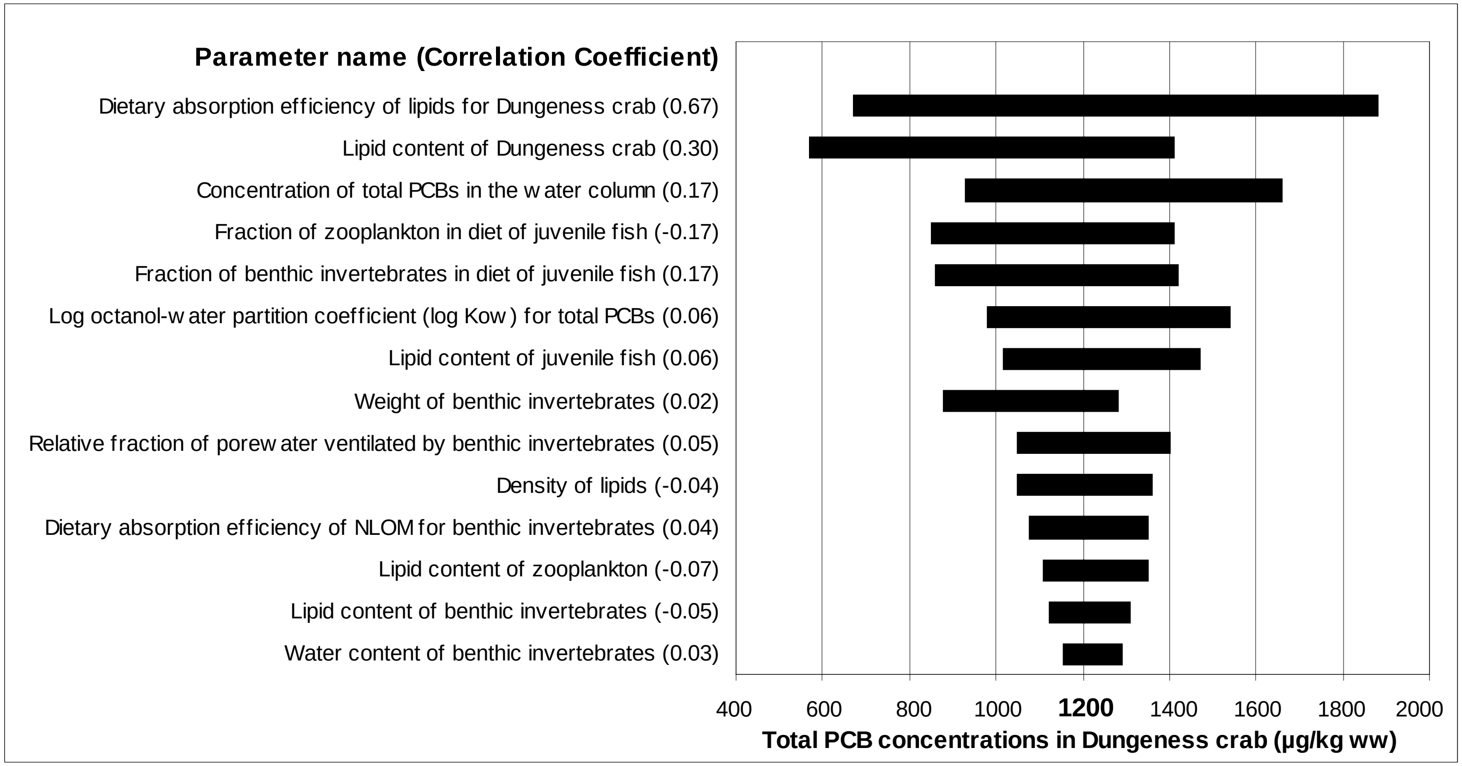


Figure D.6-4. Results of the NRS analysis for Dungeness crabs

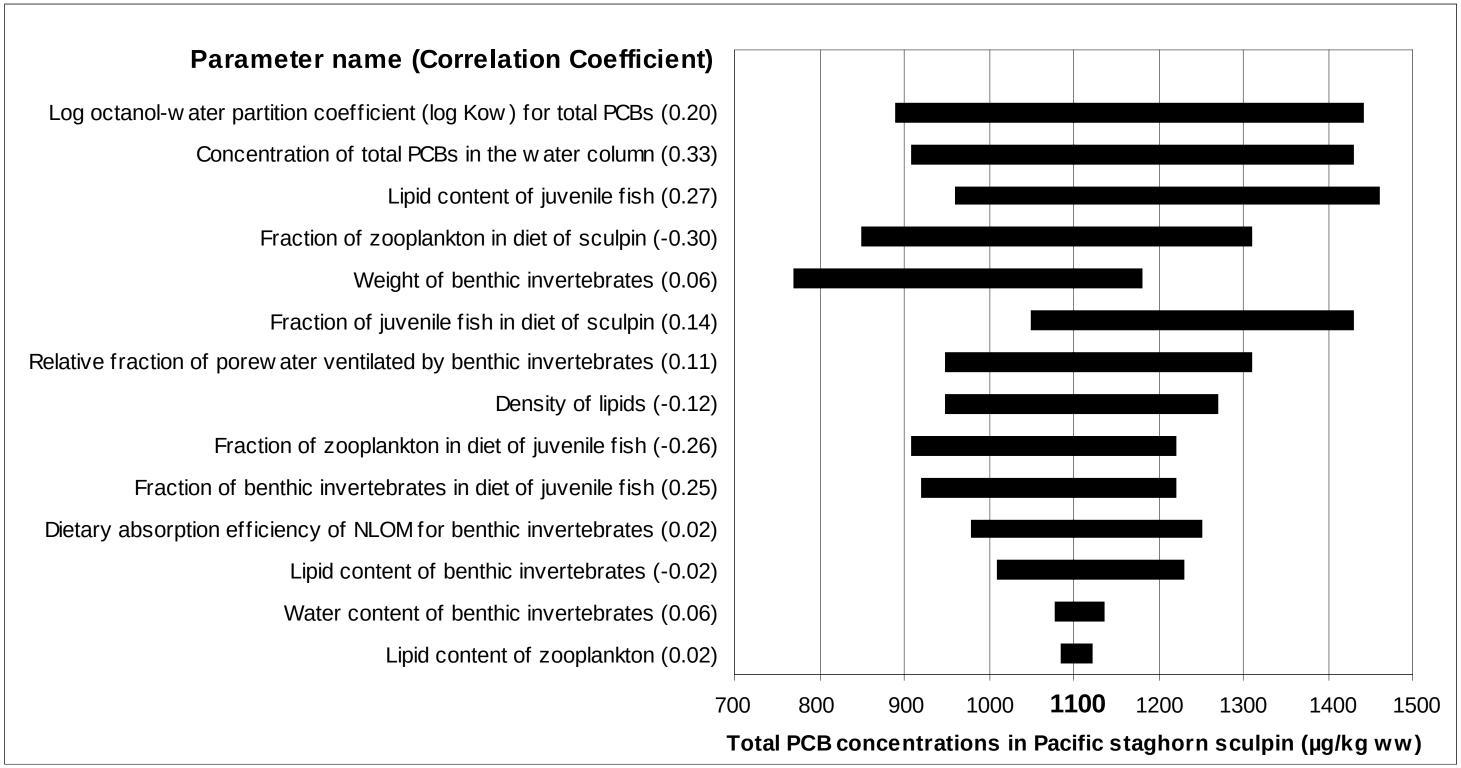


Figure D.6-5. Results of the NRS analysis for Pacific staghorn sculpin

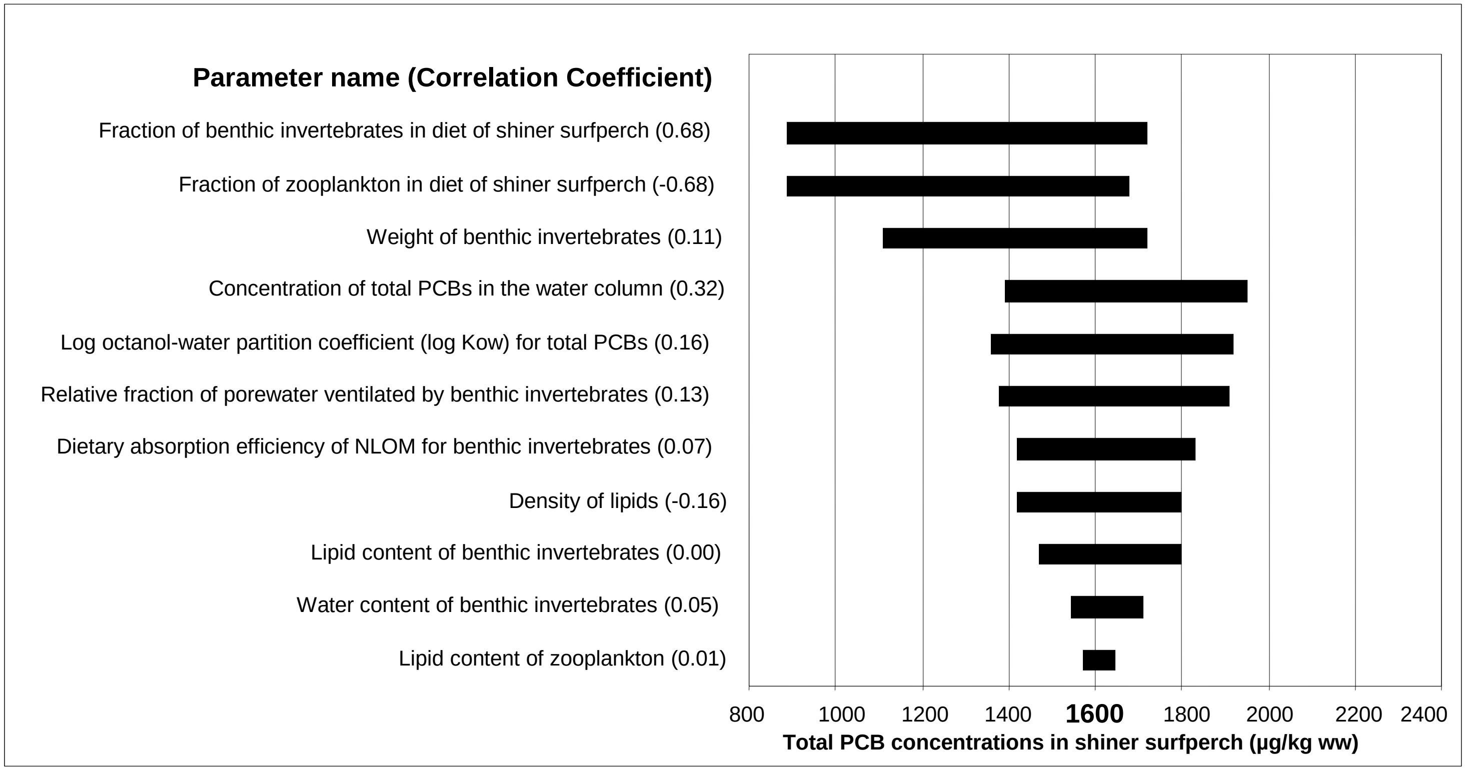


Figure D.6-6. Results of the NRS analysis for shiner surfperch

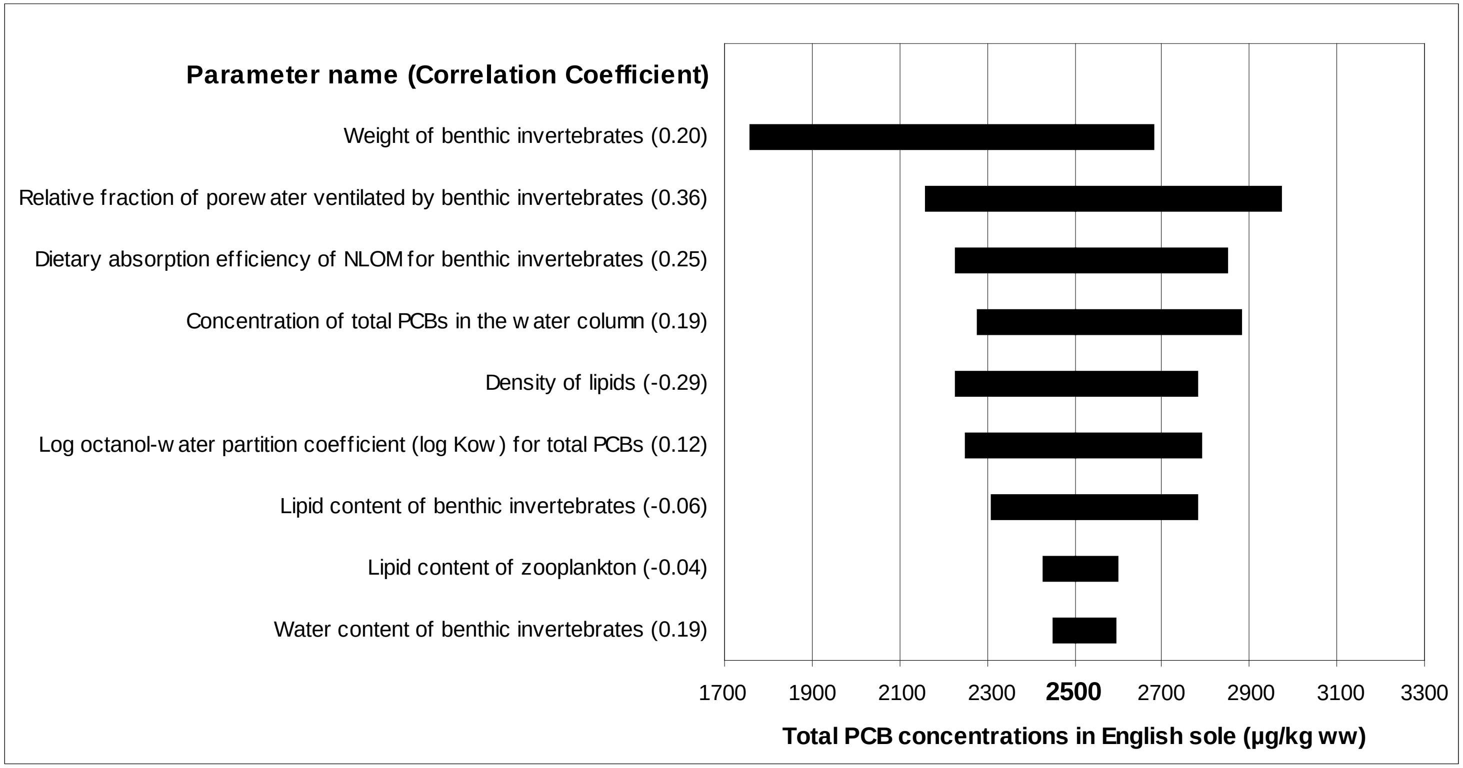


Figure D.6-7. Results of the NRS analysis for English sole

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 crabs 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),[36] 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 sculpin were assumed to consume an average of 24% zooplankton and 33% juvenile

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

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 NRS 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.6.3 SWAC Sensitivity and Uncertainty Analysis

The SWAC was not evaluated in the correlation coefficient or NRS analyses (Sections D.6.1 and D.6.2) 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 discussed in Section D.5.2, the FWM tended to overestimate total PCB concentrations in tissues (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 target 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).
- 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.

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 an additional eight 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. The water PCB concentration was held constant in order to illustrate the impact of the sediment PCB concentration on model estimates.

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 (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 4.2.3.1 in the main body 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 (µg/kg ww)	Estimated Total PCB Concentrations in Tissue for Selected SWACs (µg/kg ww)[a]								
		380	350	340	300	250	200	150	100	50
Slender crab	670	**690**	642	626	563	483	404	324	245	165
Dungeness crab	1,100	1,201	1,132	**1,109**	1,018	903	789	674	560	445
Pacific staghorn sculpin	900	1,122	1,052	1,028	**935**	818	701	585	468	351
Shiner surfperch	1,800	**1,558**	1,455	1,420	1,283	1,111	939	767	595	423
English sole	2,300	2,485	**2,310**	2,252	2,019	1,727	1,435	1,144	852	561

[a] Best-fit parameter set was used for model runs. SWACs are in µg/kg dw.

dw – dry weight

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.

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

Species	SPAFs Based on FWM Runs that Used Selected SWACs[a]								
	380	350	340	300	250	200	150	100	50
Slender crab	**1.0**	1.0	1.1	1.2	1.4	1.7	2.1	2.7	4.1
Dungeness crab	1.1	1.0	**1.0**	1.1	1.2	1.4	1.6	2.0	2.5
Pacific staghorn sculpin	1.2	1.2	1.1	**1.0**	1.1	1.3	1.5	1.9	2.6
Shiner surfperch	**1.2**	1.2	1.3	1.4	1.6	1.9	2.3	3.0	4.3
English sole	1.1	**1.0**	1.0	1.1	1.3	1.6	2.0	2.7	4.1

Species	SPAFs Based on FWM Runs that Used Selected SWACs[a]								
	380	350	340	300	250	200	150	100	50
Slender crab	**1.0**	<u>1.0</u>	<u>1.1</u>	<u>1.2</u>	<u>1.4</u>	<u>1.7</u>	<u>2.1</u>	<u>2.7</u>	<u>4.1</u>
Average SPAF	**1.1**	**1.1**	**1.1**	**1.2**	**1.3**	**1.6**	**1.9**	**2.5**	**3.5**

[a] Best-fit parameter set was used for model runs. SWACs are in µg/kg dw.

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 a species compared to empirical tissue data.
<u>Underlined</u> values are SPAFs calculated from underestimated tissue concentrations.

The SWAC that produced the lowest average SPAF across species for the best-fit parameter set was 350 µg/kg dw (Table D.6-5), although average SPAFs were similar for 380 and 340 µg/kg dw, and SPAFs for each individual species were less than 2 for all SWACs ≥ 200 µg/kg dw. Interestingly, the SWAC presented in Section 4.2.2.1 in the main body of the RI, based on an updated IDW interpolation, is 350 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 63%. This indicates that the average of FWM estimates across species 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.

D.6.4 Uncertainty in Other Input Parameters

A number of uncertainties were not evaluated in the sensitivity and uncertainty analyses presented above. These uncertainties include:

- True uptake and depuration processes described by the FWM equations
- Applicability of basic assumptions of the Arnot and Gobas FWM to LDW organisms and conditions (i.e., primary routes of chemical uptake, homogeneous distribution of chemicals within organisms, assumptions about equilibrium between organisms and the environment) (Arnot and Gobas 2004b)
- Mean of the empirical data as an estimate of true mean tissue total PCB concentrations in the LDW
- Impact of temporal differences among datasets for different media
- Distributions assigned to FWM parameters

The model's quantitative description of uptake and depuration processes is an important uncertainty of the model. Biological processes are highly complex and were necessarily simplified for the creation of the model. The degree to which this simplification appropriately captures the critical elements of these processes for predicting current and particularly future conditions is unknown. With regard to current conditions, the model reasonably estimates current PCB tissue concentrations, providing some confidence in its design.

The degree to which the model is appropriate for the LDW organisms and conditions is another source of uncertainty. The Gobas model was originally developed for a freshwater lake very different from the LDW (Gobas 1993). The model has since been applied in a various freshwater and marine environments (deBruyn et al. 2004; Gobas and Arnot 2005). However, each system is unique and the model assumptions related to primary routes of chemical uptake, homogeneous distribution of chemicals within organisms, and assumptions about equilibrium between organisms and the environment (Arnot and Gobas 2004b) are violated to some degree in any system.

Empirical data for each species tended to be highly variable; minimum and maximum total PCB concentrations in the tissues of different species ranged from 2 to 10 times the species' mean tissue concentrations. Factors that contribute to the variance in tissue concentrations include laboratory protocols, time, and spatial heterogeneity. The variability in the empirical dataset reflects uncertainties that carry over into the calibration process. Although empirical data represent the best approximation of tissue concentrations in the LDW, the variability in the data suggest the potential for uncertainty in estimates of the mean.

Another source of uncertainty is the temporal relationship among the datasets. Ideally, all empirical data would have been collected concurrently. Most of the tissue total PCB concentrations used in the calibration were collected in 2004 and 2005, and most of the sediment total PCB concentrations were collected in the late 1990s, 2004, and 2005. These sediment data along with water data collected in 2005 were used in the EFDC model, which generated the water concentration estimates used in the FWM. Inclusion of data from multiple years increases the level of uncertainty. However, the larger dataset, particularly for sediment, was believed to provide a better estimate of average conditions throughout the LDW than a smaller, concurrent dataset.

One of the temporal factors that complicated the selection of empirical calibration and input data for the FWM involved the dredging that occurred in 2003/2004 in the Duwamish/Diagonal area within Area T1. The surface sediment layer used to derive the SWAC used in the FWM was based on baseline (pre-dredging) conditions (Section D.4.1.1), consistent with the dataset used in the risk assessments. The surface sediment layer used in the EFDC model for estimation of water column concentrations was somewhat different; 2005 post-dredging surface sediment data were used in the area around the Duwamish/Diagonal dredging project instead of the pre-dredging data in the baseline dataset. These post-dredging data were used to better coincide with the PCB surface water data collected by King County in 2005 (Nairn 2009).

The fact that the surface sediment data used in the Duwamish/Diagonal area were different in the FWM and the EFDC modeling efforts generates some uncertainty. However, the water column total PCB concentrations predicted by the EFDC model, and thus used in the FWM, would be similar with either set of sediment data because the EFDC model was calibrated to empirical water column total PCB concentrations (Section D.4.1.2). Calibration to the different sediment data would still estimate water concentrations very similar to those estimated from the current EFDC model. Similarly, the FWM was calibrated to empirical tissue total PCB concentrations, and thus the difference in sediment total PCB concentrations would likely not have had a large effect on the estimated total PCB concentrations in tissues.

As an additional assessment, SWACs were generated using IDW interpolation to assess the effect of using pre-dredging vs. post-dredging data in the Duwamish/Diagonal area. The FWM used an LDW-wide SWAC for total PCBs. LDW-wide (i.e., RM 0.0 to RM 5.25) total PCB SWACs were 380 and 340 µg/kg dw using the pre-dredging and post-dredging Duwamish/Diagonal data, respectively. Based on the analysis presented in Table D.6-5, a change in sediment SWAC from 380 to 340 µg/kg dw did not result in a change in the average SPAF (1.1), and none of the SPAFs changed by more than 0.1 for individual species.

The effect of using the post-dredging data was also evaluated on a sampling-area basis to assess the potential influence of using these data on smaller scales (Section D.7). The Area T1 total PCB SWAC would have decreased from 300 to 230 µg/kg dw if the post-dredging data had been used in the Duwamish/Diagonal area rather than the pre-dredging data in the Duwamish/Diagonal area. The sensitivity of the FWM to the SWAC was not evaluated at this scale. However, if the influence of the SWAC on SPAFs was similar to that calculated for the LDW-wide scale, the decrease in SWAC would have increased the average SPAF from 1.2 to approximately 1.4, with changes on the order of 0.2 to 0.3 for individual species. All estimated tissue concentrations could still be within a factor of 2 of the empirical tissue dataset.

Distributions were assigned to many of the input parameters to describe uncertainty in their values. The type of distribution selected (e.g. normal, triangular, uniform, etc.) indicates something about the how well the parameter was characterized and/or what type of information was available. The distribution assigned for concentration of PCBs in water is illustrative of this issue. As discussed in Section D.4.1.2, the distribution assigned to water was based on both empirical and EFDC model estimates. Empirical data from the two mid-channel locations were used to define the upper and lower bounds of PCB concentrations in water for the FWM. The mode used in the data distribution was generated by the EFDC model, which was calibrated with several large empirical datasets for many parameters. Because the LDW water samples were collected mid-channel rather than directly above the benches (Map 4-11a), where PCB concentrations would be expected to be the highest, it is possible that the extreme high end of the water PCB range was not captured in the empirical data. The model estimated higher concentrations in prediction cells just above the sediment surface on the benches in a few specific areas of higher sediment concentrations. These higher concentrations were included in the EFDC model output used to generate the average concentrations. The mode of the total PCB concentration in water used in the FWM represents a yearly average of the exposure throughout the entire LDW estimated by the EFDC model. As discussed in Section D.3.2.2, because the modeled species integrate their exposure over space and time, identifying upper and lower limits is less important than identifying a reasonable average exposure concentration. Therefore, modeled monthly averages should be a better estimate of actual exposure than empirical data because of the inherent variability in water concentrations over small spatial and temporal scales. Although the baseline water PCB concentration represents a source of uncertainty in the FWM, it is not expected to be the most significant input parameter for any of the target species.

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 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. Modeling area tests were also used to investigate the potential impact of the uncertainty associated with home ranges of species used in the FWM.

The best-fit parameter set 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.[37] 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).

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

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

Modeling Area	Total PCB Concentration in the Water Column (ng/L)[a]	Total PCB Concentration in Sediment (SWAC) (µg/kg dw)[b]	TOC (%)[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

[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 2009).

[b] SWACs of total PCBs in sediment were calculated using the 2006 IDW interpolation method and pre-Round 3 sediment data 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 TOC were calculated using the 2006 IDW interpolation method and pre-Round 3 sediment data for modeling areas using Thiessen polygons generated for the entire LDW but clipped to modeling areas.

dw – dry weight
IDW – inverse distance weighting
LDW – Lower Duwamish Waterway
PCB – polychlorinated biphenyl
SWAC – spatially weighted average concentration
TOC – total organic carbon

At a modeling area scale, estimates were within a factor of 2 of empirical data for Areas M1, M2, and M4 (Table D.7-2) with a few exceptions. In Area M2, the estimate for shiner surfperch was 2.4 times lower than the empirical average. However, as discussed in Section D.3.2.2 and Section 4.2.2.4 (in the main body of the RI), there was one shiner surfperch composite sample with very a high concentration of PCBs (18,400 µg/kg ww). If that sample had been removed from the dataset for Area M2, the model would have predicted within a factor of 2. In Area M3, estimates for benthic invertebrates, Dungeness crab, slender crab, and English sole ranged from 2.2 to 3.0 times higher than empirical data (Table D.7-2). The model performed reasonably well for shiner surfperch and Pacific staghorn sculpin in Area M3 (estimates were 1.4 and 1.9 times higher than empirical data, respectively).

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 4.2.2.4 in the main body of the RI).

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: Estimated Total PCB Concentration (µg/kg ww)	SPAF	Over (+) or Under (-) Estimate
M1	benthic invertebrates	180[a]	6	231	1.3	**+**
	slender crab	650	3	542	1.2	**-**
	Dungeness crab	990	6	960	1.0	**-**
	Pacific staghorn sculpin	720	7	889	1.2	**+**
	shiner surfperch	970	15	1,229	1.3	**+**
	English sole	2,600	12	1,946	1.3	**-**
M2	benthic invertebrates	170[b]	6	214	1.3	**+**
	slender crab	700	7	507	1.4	**-**
	Dungeness crab	na	0	948	na	na
	Pacific staghorn sculpin	750	7	858	1.1	**+**
	shiner surfperch	2,800	12	1,165	2.4	**-**
	English sole	2,900	12	1,808	1.6	**-**
M3	benthic invertebrates	370[c]	4	702	1.9	**+**
	slender crab	631	3	1,636	2.6	**+**
	Dungeness crab	1,300	4	2,820	2.2	**+**
	Pacific staghorn sculpin	1,400	7	2,648	1.9	**+**
	shiner surfperch	2,600	12	3,689	1.4	**+**
	English sole	2,000	12	5,918	3.0	**+**
M4	benthic invertebrates	140[d]	4	188	1.3	**+**
	slender crab	na	0	467	na	na
	Dungeness crab	1,200	2	1,040	1.2	**-**
	Pacific staghorn sculpin	730	7	879	1.2	**+**
	shiner surfperch	710	10	1,127	1.6	**+**
	English sole	1,400	6	1,639	1.2	**+**

[a] The mean “empirical” total PCB concentration for benthic invertebrates in Area M1 was estimated using a total PCB SWAC of 300 µg/kg dw and the benthic invertebrate tissue-sediment regression: concentration of total PCBs in benthic invertebrate tissue = 0.34 x (sediment PCB concentration) + 75 (see Attachment 1).

[b] The mean “empirical” total PCB concentration for benthic invertebrates in Area M2 was estimated using a total PCB SWAC of 270 µg/kg dw and the benthic invertebrate tissue-sediment regression described in Footnote a.

[c] The mean “empirical” total PCB concentration for benthic invertebrates in Area M3 was estimated using a total PCB SWAC of 880 µg/kg dw and the benthic invertebrate tissue-sediment regression described in Footnote a.

[d] The mean “empirical” total PCB concentration for benthic invertebrates in Area M4 was estimated using a total PCB SWAC of 190 µg/kg dw and the benthic invertebrate tissue-sediment regression described in Footnote a.

FWM – food web model
LDW – Lower Duwamish Waterway
n – number of composite samples
na – not available
PCB – polychlorinated biphenyl
SPAF – species predictive accuracy factor
SWAC – spatially weighted average concentration
ww – wet weight

Differences in home range size could possibly explain the poorer performance of the FWM for Dungeness and slender crabs and English sole relative to the performance for shiner surfperch and Pacific staghorn sculpin in Area M3. 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 and slender crabs 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 addition, the home-range hypothesis for the good performance of Pacific staghorn sculpin and shiner surfperch in Area M3 is supported by the ANOVAs and regressions performed with the empirical tissue data among the four modeling areas for these species (Section D.3.2.2). These analyses indicated that Pacific staghorn sculpin, and to a lesser extent shiner surfperch, may integrate their exposure over areas smaller than the LDW.

In summary, for Pacific staghorn sculpin and shiner surfperch, the FWM performed within the SPAF criterion (SPAF ≤ 2) for all modeling areas when the 18,400-µg/kg ww sample result was removed from the shiner surfperch Area M2 dataset. These results indicate that applying the FWM at the modeling area scale for these species may be appropriate, although uncertainty is higher at the modeling area scale, as discussed in Section D.3.2. For Dungeness and slender crabs and English sole, the FWM performed within the SPAF criterion (SPAF ≤ 2) for all modeling areas except Area M3. The fact that the SPAF criterion was met in Areas M1, M2, and M4 but not in M3 (the modeling area with the highest sediment SWAC) indicates that these species may have home ranges that are larger than the modeling areas.

Regardless of the species, some loss of performance is to be expected if the model is applied on a smaller spatial scale because of the following:

- Greater SE 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

In summary, because of their larger home range and the likelihood that exposure occurs on a scale larger than that of the modeling areas, the application of the FWM at the modeling area scale may not be appropriate for Dungeness and slender crabs and English sole, particularly if the SWACs within the smaller areas are dramatically different than the LDW-wide SWAC. This issue would need to be considered if the LDW FWM is applied on a smaller scale in the future.

D.7.2 BENCH AREA SCALE

In addition to evaluating FWM performance at the modeling area scale, the model was evaluated on a bench area scale, per EPA request, to assess the potential for preferential use of nearshore areas. Modeling Area 3 was selected for this analysis because of the variation in sediment concentrations for the nearshore bench areas. The benches were defined as the area outside of the navigation channel. Model inputs for sediment PCB concentration and TOC were estimated as SWAC values, and water concentrations were estimated from EFDC prediction cells in those areas. Table D.7-3 presents the input values and results of the bench analysis. It should be noted that the fish tissue data available for comparison to model predictions were not collected exclusively from the benches but were instead collected as available within the entire sampling subareas, which generally bisected the LDW.

Table D.7-3. Input parameters and results of bench area FWM analysis

Species Name	Exposure Area	FWM Input Parameters			Empirical Total PCB Concentration in Tissue (µg/kg ww)[d]	FWM Output	
		Total PCB Concentration in Water (ng/L)[a]	Total PCB SWAC in Surface Sediment (µg/kg dw)[b]	TOC (%)[c]		Estimated Total PCB Concentration in Tissue (µg/kg ww)	SPAF
Pacific staghorn sculpin	LDW-wide	1.2	380	1.9	900	1,122	1.2
	M3	2.7	880	1.76	1,400	2,648	1.9
	M3, navigation channel excluded	3.5	1,065	1.79	1,400	3,228	2.3
	M3, west bench	3.5	428	1.62	940	1,748	1.9
	M3, east bench	3.5	1,586	1.92	1,700	4,365	2.6

SPECIES NAME	EXPOSURE AREA	FWM INPUT PARAMETERS			EMPIRICAL TOTAL PCB CONCENTRATION IN TISSUE (µg/kg ww)[d]	FWM OUTPUT	
		TOTAL PCB CONCENTRATION IN WATER (ng/L)[a]	TOTAL PCB SWAC IN SURFACE SEDIMENT (µg/kg dw)[b]	TOC (%)[c]		ESTIMATED TOTAL PCB CONCENTRATION IN TISSUE (µg/kg ww)	SPAF
Shiner surfperch	LDW-wide	1.2	380	1.9	1,800	1,558	1.2
	M3	2.7	880	1.76	2,600	3,689	1.4
	M3, navigation channel excluded	3.5	1,065	1.79	2,600	4,482	1.7
	M3, west bench	3.5	428	1.62	2,500	2,302	1.1
	M3, east bench	3.5	1,586	1.92	2,800	6,159	2.2

a LDW-wide water concentration based on water concentration in the best-fit parameter set (Table D.5-3). Other water concentrations were estimated from EFDC model output for the specified exposure area (Nairn 2009).

b SWACs of total PCBs in sediment were calculated using the 2006 IDW interpolation method and pre-Round 3 sediment data using the same interpolation grids generated for the entire LDW but clipped to specific areas.

c Spatially weighted average percentages of sediment TOC were calculated using the 2006 IDW interpolation method and pre-Round 3 sediment data using Thiessen polygons generated for the entire LDW but clipped to specific areas.

d Mean total PCB concentrations in fish collected from specified exposure areas. The west bench included fish from Subareas 3B, 3D, and 3F. The east bench included fish from subareas 3A, 3C, and 3E.

dw – dry weight

EFDC – Environmental Fluid Dynamics [Computer] Code

FWM – food web model

LDW – Lower Duwamish Waterway

PCB – polychlorinated biphenyl

SPAF – species predictive accuracy factor

TOC – total organic carbon

SPAFs ranged from 1.2 (LDW-wide) to 2.6 (east bench) for Pacific staghorn sculpin and from 1.1 (west bench) to 2.2 (east bench) for shiner surfperch. The FWM tended to overestimate PCB concentrations in tissue when the sediment SWAC was assumed to be higher (i.e., the concentration for the benches). However, because the fish tissue data were not collected exclusively from the benches, results of this assessment cannot be considered conclusive for characterizing the exposure of Pacific staghorn sculpin or shiner surfperch to contamination on the benches relative to that in the entire subarea.

D.7.3 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 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.7-4).

Table D.7-4. Location-specific input parameter values for 14 clam intertidal locations in the LDW

Location ID	Modeling Area	Total PCB Concentration in the Water Column (ng/L)[a]	Total PCB Concentration in Sediment (□g/kgdw)[b]	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
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.7-5). 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.7-5).

Table D.7-5. Application of the calibrated FWM for clams

Location ID	Empirical Total PCB Concentration in Clam Tissue (µg/kg ww)	FWM-Estimated Total PCB Concentration in Clam Tissue (µg/kg ww)	SPAF	Over (+) or Under (-) Estimate
C1	24	22	1.1	-
C2-1	24	34	1.4	+
C2-2	29	61	2.1	+
C3-1	33	43	1.3	+
C3-2	32	26	1.2	-
C4	31	46	1.5	+
C5	43	91	2.1	+
C6	34	45	1.3	+
C7-1	220	352	1.6	+
C7-2	250	259	1.0	+
C8	580	828	1.4	+
C9	50	75	1.5	+
C10-1	320	1,973	6.2	+
C10-2	330	3,395	10.3	+

FWM – food web model
ID – identification
PCB – polychlorinated biphenyl
SPAF – species predictive accuracy factor
ww – wet weight

Total PCB concentrations were overestimated at locations C10-1 and C10-2 (SPAFs of 6.2 and 10.3, respectively); these locations had the highest total PCB concentrations in sediment (6,600 and 15,000 µg/kg dw, respectively). Empirical clam tissue total PCB concentrations at locations C10-1 and C10-2 (320 and 330 µg/kg ww, respectively) were in the same range as clam tissue concentrations (220 to 580 µg/kg ww) from locations with sediment total PCB concentrations that ranged from 380 to 3,300 µg/kg dw (Table D.7-4). These results may indicate that clam tissue concentrations are not greatly influenced by local sediment total PCB concentrations and may be more a function of some other parameter. The other two locations with SPAFs > 2 (C2-2 and C5, each with a SPAF of 2.1) had total PCBs concentrations in sediment similar to those for the other areas.

An NRS analysis for clams was conducted using the same methods described in Section D.6.2. 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. The higher the NRS value, the greater the sensitivity of the model to that parameter.

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 twenty parameters tested in the NRS analysis had an effect on estimated total PCB concentrations in clams (Table D.7-6).

Table D.7-6. Results of NRS analysis at three clam intertidal locations

Parameter	NRS Value for Clams (µg/kg ww)		
	Intertidal Location C3-1[a]	Intertidal Location C7-2[a]	Intertidal Location C10-2[a]
Estimated concentration of total PCBs in the water column	48	48	47
Relative fraction of porewater ventilated by clams[a]	29	280	3,800
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

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

b The total PCB concentrations in co-located sediment at locations C3-1, C7-2, and C10-2 were 52, 380, and 15,000 µg/kg dw, respectively.

dw – dry weight

PCB – polychlorinated biphenyl

NLOM – non-lipid organic matter

NRS – nominal range sensitivity

ww – wet weight

The six parameters that had an effect on FWM clam tissue estimates were also identified as important parameters for the other modeled species (Figures D.6-1 through D.6-7). The influence of total PCB concentrations in the water column, relative to the influence of other parameters, increased with decreasing sediment concentrations (Table D.7-6). 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.

A regression model was also evaluated to assess its ability to estimate total PCB concentrations in clam tissue. When both sediment and tissue concentrations were log transformed to help meet the assumptions of a regression analysis (linearity of the relationship and homogeneous variance of the dependent variable around the regression line), the sediment variable explained 80% of the variance in tissue concentrations ($R^2 = 0.80$) and the regression was significant ($p < 0.0005$) (Figure D.7-1).

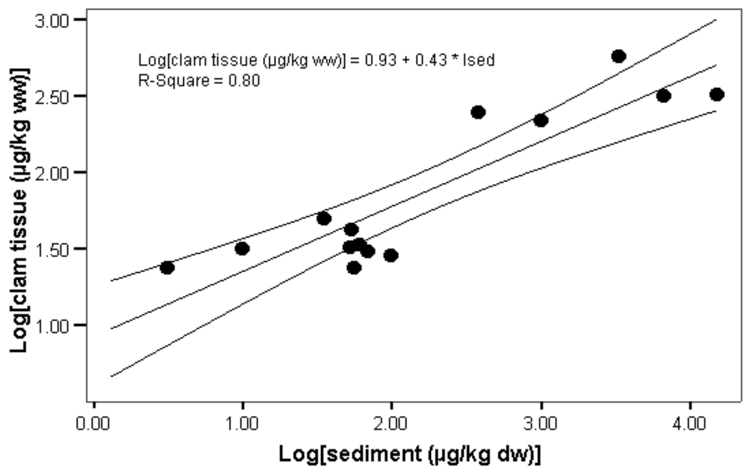


Figure D.7-1. PCB regression using log(10)-transformed concentrations of co-located clam tissue and sediment samples.

The regression model provided better estimates of total PCB concentrations in clam tissue at high sediment concentrations (i.e., PCB concentrations in sediment greater than 6,600 µg/kg dw) but provided estimates similar to the FWM (in terms of SPAFs) at the low end of the sediment scale, particularly in the range of sediment RBTCs (Table D.7-7). Because the regression model performed similarly to the FWM in the lower range of total PCB concentrations in sediment, the FWM was selected to estimate the clam tissue sediment relationship in the derivation of sediment RBTCs, which is discussed in Section D.9.

Table D.7-7. Comparison of empirical total PCB concentrations in clam tissue relative to estimates made using the FWM and a regression equation

Location ID	Sediment PCB Concentration (µg/kg dw)	Empirical Tissue PCB Concentration (µg/kg ww)	FWM Estimate		Regression Estimate	
			Tissue PCB Concentration (µg/kg ww)	SPAF	Tissue PCB Concentration (µg/kg ww)	SPAF
C1	3.1	24	22	1.1	14	1.7
C2-1	56	24	34	1.4	48	2.0
C2-2	99	29	61	2.1	61	2.1
C3-1	52	33	43	1.3	47	1.4
C3-2	10	32	26	1.2	23	1.4
C4	69	31	46	1.5	53	1.7
C5	53	43	91	2.1	47	1.1
C6	61	34	45	1.3	50	1.5
C7-1	1,000	220	352	1.6	166	1.3

Location ID	Sediment PCB Concentration (µg/kg dw)	Empirical Tissue PCB Concentration (µg/kg ww)	FWM Estimate		Regression Estimate	
			Tissue PCB Concentration (µg/kg ww)	SPAF	Tissue PCB Concentration (µg/kg ww)	SPAF
C7-2	380	250	259	1.0	109	2.3
C8	3,300	580	828	1.4	277	2.1
C9	35	50	75	1.5	39	1.3
C10-1	6,600	320	1,973	6.2	374	1.2
C10-2	15,000	330	3,395	10.3	532	1.6

dw – dry weight
FWM – food web model
ID – identification
PCB – polychlorinated biphenyl
SPAF – species predictive accuracy factor
ww – wet weight

D.8 Comparison of FWM Estimates to 2007 Tissue Data

Most of the tissue total PCB data used in the FWM calibration were collected in 2004 and 2005, with a smaller amount of data from the 1990s. Additional tissue data were collected in the LDW in 2006 and 2007. These additional data were collected after the FWM was calibrated so they were not included in the calibration dataset. In this section, FWM-estimated concentrations in tissue are compared to the 2007 data as an informational exercise.

As discussed in Section 4.2.2.4 in the main body of the RI, total PCB concentrations (Aroclor sum) in fish, crabs, and clams were generally lower in 2006 and 2007 than in 2004 and 2005. The reason for this decrease is not known. Possible hypotheses include higher total PCB concentrations in tissue samples collected in 2004 because of the dredging of PCB-contaminated sediments in 2004 that could have mobilized PCBs and made them more available to organisms, a gradual decline in total PCB concentrations in sediment and water over time because of natural recovery (which will be discussed in greater detail in the FS), reductions in surface sediment concentrations from the Duwamish/Diagonal dredging lowering site-wide exposures after 2004, analytical uncertainties associated with the use of different laboratories in 2004 versus 2005, 2006, and 2007 (see Section 4.2.2.4 in the main body of the RI), or a combination of these factors.

Interest has been expressed by agency reviewers in seeing how well the FWM would predict the 2007 tissue concentrations. Because there are no alternative input parameters (e.g., surface sediment total PCB concentrations, surface water total PCB concentrations) that could be used in the FWM to be reflective of the more recent conditions, this exercise does not constitute a validation of the FWM. For example, the baseline surface sediment dataset that served as the basis for the SWAC used in the

FWM represents samples collected from 1990 through 2005; there are too few samples from the latter years to estimate an LDW-wide SWAC that might more accurately reflect the exposure regime of the organisms collected in 2006 or 2007. Nevertheless, for the sake of demonstration, mean tissue PCB concentrations (sum of detected Aroclors) for species collected in 2007 were compared with the total PCB concentrations estimated for those species using the FWM and the best-fit parameter set identified in calibration of the FWM (Table D.8-1). Similar comparisons cannot be made for tissue samples collected in 2006 because tissue samples in that year were available for only two of the target species and only from Area T1. Clam data were also not included in this comparison because those data were not considered spatially representative of the entire LDW. SPAFs were then calculated by comparing the empirical 2007 tissue PCB data to the FWM-estimated total PCB concentrations.

Table D.8-1. FWM-estimated total PCB concentrations in tissue compared to empirical data from 2007, tissue data in the calibration dataset, and the combined tissue dataset

Species	Mean Total PCB Concentration in Tissue (µg/kg ww)				SPAF		
	FWM Estimate Using the Best-Fit Parameter Set[a]	2007 Tissue Data[b]	Tissue Data in the Calibration Dataset (1990s – 2005)	All Tissue Data (1990s – 2007)[b, c]	2007 Tissue Data	Tissue Data in the Calibration Dataset (1990s – 2005)	All Tissue Data (1990s – 2007)[c]
Benthic invertebrates	300	nd	200	nc	nd	1.5	nc
Slender crab	690	155	670	510	4.5	1.0	1.4
Dungeness crab	1,200	200	1,100	890	6.0	1.1	1.3
Pacific staghorn sculpin	1,100	nd	900	nc	nd	1.2	nc
Shiner surfperch	1,600	452	1,800	1,400	3.5	**1.2**	1.1
English sole	2,500	683	2,300	1,800	3.7	1.1	1.4
Average	nc	nc	nc	nc	**4.4**	**1.2**	**1.3**

[a] FWM estimates were determined without changing any of the input parameters to attempt to reflect any changes in the exposure regime of organisms sampled in 2007.

[b] Mean total PCB concentrations are based on the sum of detected Aroclors.

[c] Does not include 2006 data because these data were not considered spatially representative.

FWM – food web model
LDW – Lower Duwamish Waterway
nd – no data (not sampled in 2007)
nc – not calculated
PCB – polychlorinated biphenyl
SPAF – species predictive accuracy factor
ww – wet weight

Although the FWM estimates were within a factor of 2 of the tissue concentrations in the FWM calibration dataset (i.e., SPAFs ranged from 1.1 to 1.5, with an average SPAF of 1.2), the FWM slightly overpredicted tissue concentrations (i.e., estimated higher concentrations) for five of the six modeled species. Given that tissue concentrations in 2007 were substantially lower than those in the calibration dataset, it is not surprising that SPAFs were higher still; SPAFs calculated using the 2007 empirical tissue data ranged from 3.5 to 6.0, with an average of 4.4 across species.

In addition to comparing FWM estimates to the 2007 data, the FWM estimates were also compared to a combined LDW dataset (including all tissue data from the 1990s through 2007, except data from 2006). This comparison resulted in SPAFs that were very similar to those generated from the FWM calibration dataset. The average SPAF was 1.3 compared to an average SPAF of 1.2 for the FWM calibration dataset. Thus, if it is assumed that conditions in the system did not change significantly between the period when the FWM calibration data set was collected and 2007, then the FWM estimates this larger dataset quite well.

The FWM's underestimation of 2007 tissue concentrations should not be taken as an indication that the FWM performed poorly; true validation of the FWM would only be possible if there was a sufficiently robust exposure dataset with significantly different PCB concentrations that could be used in the FWM to reflect the actual exposure of the organisms sampled in 2007. In the absence of a robust synoptic sediment, water, and tissue dataset, the 2007 tissue data cannot be used to "validate" the FWM. Comparison of model predictions to the larger dataset, including FWM calibration data and 2007 data, indicated that the model predicts this larger dataset quite well. Although the FWM may be used to estimate future tissue concentrations under alternative exposure scenarios, the best indication of possible linkage between sediment and tissue PCB concentrations will result from a thoughtfully designed monitoring program to be implemented after remediation has begun.

D.9 Application of the FWM to Calculate Sediment RBTCs

RBTCs represent the concentrations that correspond to specific thresholds of risk.[38] In Section 8 in the main body 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. This use of the FWM carries an implicit assumption that risks associated with tissue concentrations of PCBs are a predictable function of sediment PCB concentrations and that risks from PCBs can thus be predictably reduced by lowering sediment concentrations.

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

[38] 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.

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 concentration was represented by the SWAC for the LDW from RM 0.0 to RM 5.25, which has been estimated to be 380 µg/kg dw.[39] The EFDC model[40] estimated an annual LDW-wide mean total PCB concentration in water of 1.43 ng/L using the three bottom cells of the EFDC model. These concentrations represent water in the lower portion of the water column, closer to the sediment surface, where most of the fish and crab species being modeled spend the majority of their time.

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, 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 background 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 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 2009).

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 based on the best-fit parameter set (Table D.5-3). This concentration is slightly below the LDW-wide mean concentration

---

[39] 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.

[40] 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 2009).

of 1.43 ng/L (Table D.4-1) estimated by the EFDC model. For the lower sediment ranges, total PCB concentrations in water were assumed to be proportionately lower (Table D.9-1). 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 0.04 to 0.8 ng/L in 2005 and from 0.04 to 2.4 ng/L in 2007 (Mickelson and Williston 2006; Williston 2008). 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). The selection of a single water value was necessary for each sediment range because the FWM can only accommodate a single value for overlying water. The porewater concentration parameter (estimated by the model) provides a mechanism for the FWM to account for the potentially higher concentrations of chemicals within the sediment-water interface.

Table D.9-1. Assumed relationship between total PCB concentrations in sediment and overlying water for the calculation of RBTCs in sediment

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
RBTC – risk-based threshold concentration

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.9-1.

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.9-1. 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.

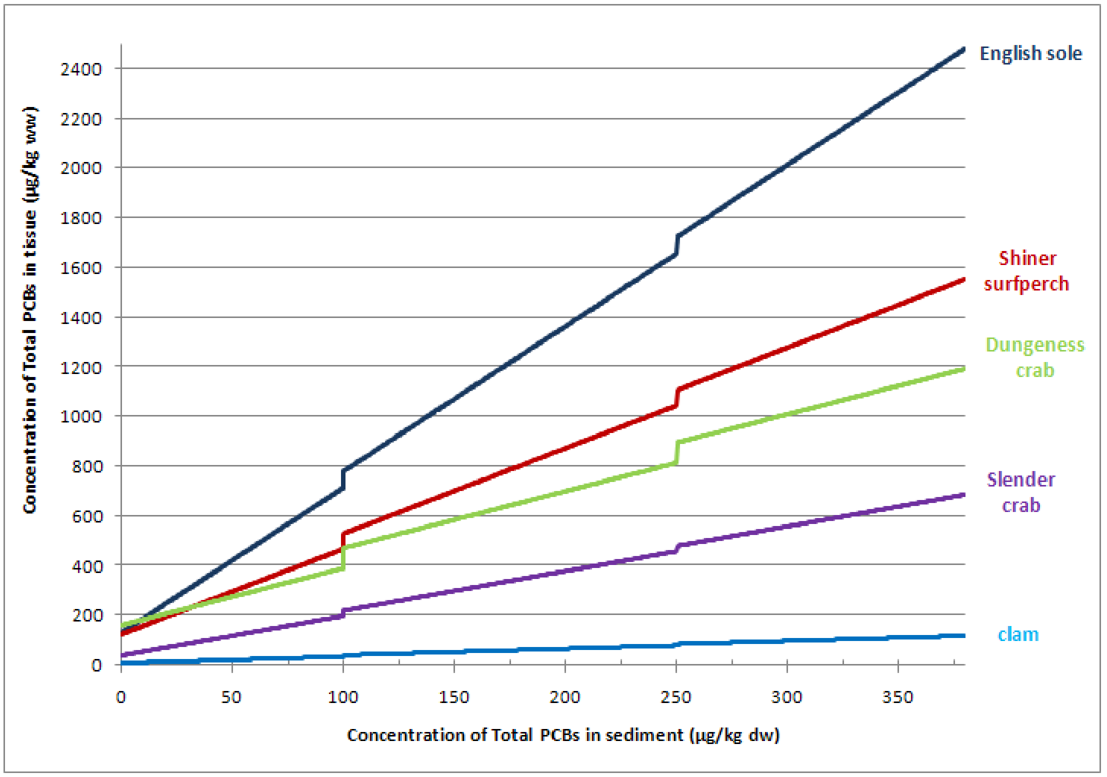


Figure D.9-1. Total PCB concentrations in whole-body tissues of seafood species as a function of total PCB concentrations in sediment

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.9-2 and D.9-3 present the results for Dungeness crabs and English sole, respectively, as examples. 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.

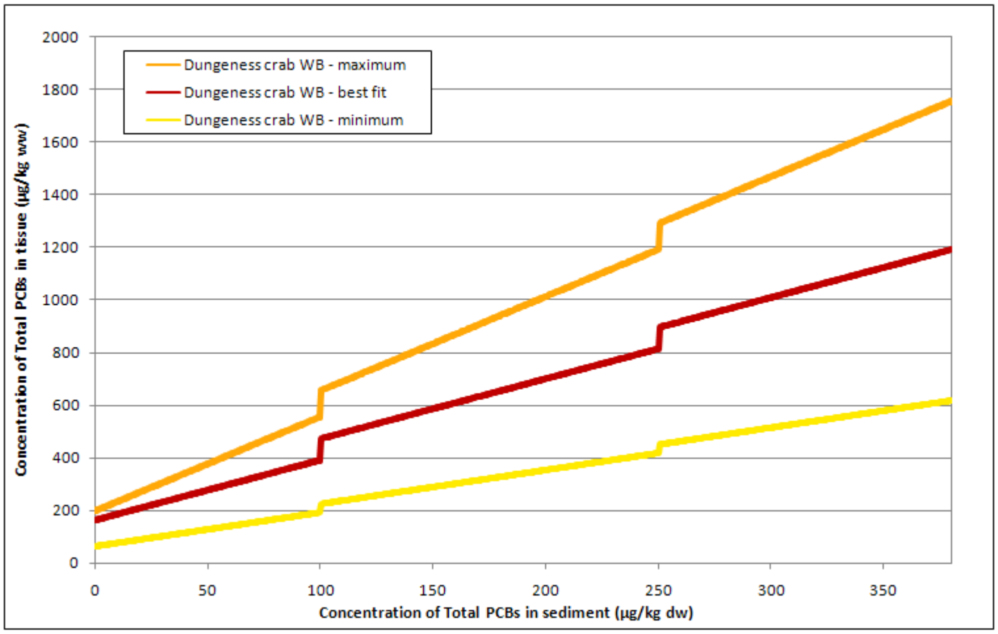


Figure D.9-2. Estimated total PCB concentrations in whole-body Dungeness crab using best-fit, maximum, or minimum parameter sets as a function of total PCB concentration in sediment

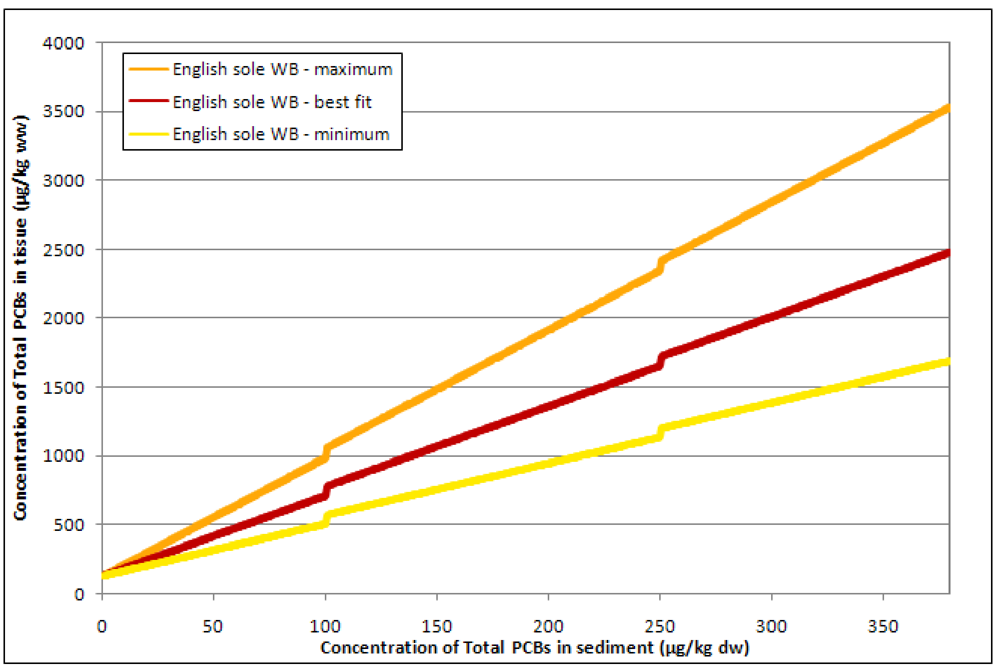


Figure D.9-3. Estimated total PCB concentrations in whole-body English sole using best-fit, maximum, or minimum parameter sets as a function of total PCB concentration in sediment

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). The upper- and lower-bound estimates are not upper and lower confidence intervals and do not reflect a statistical measure of uncertainty. Instead, the upper and lower bounds reflect some of the variability in FWM estimates, which was constrained by the model performance SPAF of ≤ 2. The upper and lower bounds do not include consideration of sediment variance (or uncertainty in the SWAC) because the sediment concentration was considered a decision variable (see Section D.4.1.1). Analyses of model sensitivity and uncertainty associated with the SWAC were presented in Section D.6.3.

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

The estimated total PCB concentrations in tissue for the modeled species,[41] 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

[41] 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 (crabs) 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 crabs, 0.139 for Dungeness crabs, and 0.526 for English sole. These conversion factors were based on the ratio of whole-body to edible-meat concentrations detected in individual LDW fish tissue samples and detected in composite crab edible meat and hepatopancreas samples collected as part of the LDW RI.

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. Uncertainties associated with the risk assumptions 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., 1×10^{-4}, 1×10^{-5}, and 1×10^{-6}). Thus, the risk estimates described in Step 3 were compiled in a table to facilitate the identification of the total PCB concentration in sediment corresponding to a selected excess cancer risk threshold (1×10^{-4}, 1×10^{-5}, or 1×10^{-6}) or a non-cancer hazard (hazard quotient = 1) for each of the exposure scenarios.

Table D.9-2 demonstrates the manner in which sediment RBTCs were identified for two of the seafood consumption scenarios. The full 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.9-2 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 (1×10^{-4} for the adult Tulalip reasonable maximum exposure [RME] and 1×10^{-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 5 µg/kg dw total PCBs was associated with the 1×10^{-4} excess risk level; for the adult tribal CT scenario based on Tulalip data, a sediment RBTC of 24 µg/kg dw total PCBs was associated with the 1×10^{-5} excess risk level. Sediment RBTCs for other risk scenarios and risk thresholds are presented in Section 8 in the main body of the RI.

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 8-7 in Section 8 in the main body of the RI.

At extremely low sediment PCB concentrations, the PCB concentration in water alone is sufficient to result in estimates of tissue concentrations that correspond to excess cancer risk estimates greater than 1×10^{-5} for human seafood consumers (for all RME consumption scenarios). Thus, it was not possible to calculate a sediment RBTC at the lower risk threshold levels, such as 1×10^{-6} and 1×10^{-5}. This exercise indicates that the assumption implicit in RBTC calculations that tissue concentrations (and therefore risk estimates) are predictable functions of PCB sediment concentrations, may be tenuous, particularly at very low sediment concentrations.

Table D.9-2. Excess risk levels for two seafood consumption scenarios corresponding to total PCB concentrations in sediment

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	11	63	43	13	164	23	117	126	137	72	9.0×10^{-5}	6.3×10^{-6}
5	0.6	12	67	51	15	174	24	127	141	163	86	$\mathbf{1.0 \times 10^{-4}}$	7.0×10^{-6}
10	0.6	13	72	58	17	185	26	139	158	191	101	1.1×10^{-4}	7.8×10^{-6}
20	0.6	16	81	74	22	208	29	161	192	248	130	1.3×10^{-4}	9.4×10^{-6}
24	0.6	17	84	80	23	216	30	170	204	270	142	1.4×10^{-4}	$\mathbf{1.0 \times 10^{-5}}$
30	0.6	18	90	89	26	231	32	185	226	306	161	1.6×10^{-4}	1.1×10^{-5}
40	0.6	21	99	106	31	254	35	208	261	365	192	1.8×10^{-4}	1.3×10^{-5}
50	0.6	23	108	121	36	277	38	232	295	423	223	2.0×10^{-4}	1.4×10^{-5}
70	0.6	28	126	153	45	322	45	278	363	539	284	2.5×10^{-4}	1.8×10^{-5}
90	0.6	34	144	185	55	368	51	325	432	656	345	3.0×10^{-4}	2.1×10^{-5}
100	0.6	36	153	201	59	391	54	348	467	715	376	3.2×10^{-4}	2.2×10^{-5}
150	0.9	54	230	301	89	587	82	523	700	1,072	564	4.8×10^{-4}	3.4×10^{-5}
200	0.9	67	276	380	112	700	97	638	870	1,361	716	5.9×10^{-4}	4.2×10^{-5}
250	0.9	80	321	460	136	815	113	756	1,044	1,655	871	7.1×10^{-4}	5.0×10^{-5}
300	1.2	98	398	561	165	1,011	141	930	1,277	2,012	1,059	8.7×10^{-4}	6.1×10^{-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.10 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 of 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 in the main body of the RI.

D.11 References

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.

CDFG. 2002. Marine Life Protection Act Initiative: Dungeness crab (*Cancer magister*) [online]. California Department of Fish and Game, Sacramento, CA. Updated 9/23/02. [Cited 7/15/08.] Available from: http://www.dfg.ca.gov/mlpa/response/crab.pdf

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.

Curtis DL, Jensen EK, McGaw IJ. 2007. Behavioral influences on the physiological responses of *Cancer gracilis*, the graceful crab, during hyposaline exposure. Biol Bull 212:222-231.

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

deBruyn AMH, Ikonomou MG, Gobas FAPC. 2004. Magnification and toxicity of PCBs, PCDDs, and PCDFs in upriver-migrating pacific salmon. Environ Sci Technol 38:6217-6224.

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 Technol 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. 1977. Stomach contents of northern California Dungeness crabs, *Cancer magister*. Cal Fish Game 63(1):43-51.

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 Technol 22:382-387.

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

Jones AC. 1962. The biology of the euryhaline fish *Leptocottus armatus armatus*. University of California Publications in Zoology 67:321-367.

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.

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, Wingert RC, Borton SF. 1975. Ecological survey of demersal fishes in the Duwamish River and at West Point 1974. Prepared for Municipality of

Metropolitan Seattle. Report no. FRI-UW-7509. Fisheries Research Institute, University of Washington, 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.

Morrow JE. 1980. The freshwater fishes of Alaska. Animal Resources Ecology Library, University of British Columbia, Vancouver, BC.

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.

Nairn B. 2009. Memorandum dated November 30, 2009 to J Stern and D Williston, KCDNRP: EFDC calibration process for predicting PCB water concentrations in LDW. King County Department of Natural Resources and Parks, Seattle, WA.

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.

O'Neill S, Myers M, Moser M, Quinnell S, Anulacion B, West JE. 2005. Use of acoustic tagging to study home range and migration of English sole (*Parophrys vetulus*) in Puget Sound: application to management of contaminated sediments. Puget Sound Research Conference, Seattle, WA.

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.

Pauley GB, Armstrong DA, Heun TW. 1986. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (Pacific Northwest): Dungeness crab. Biological Report 82 (11.63). TR EL-82-4. Coastal Ecology Group, US Army Corps of Engineers, Vicksburg, MS and National Wetlands Research Center, US Fish and Wildlife Service, Slidell, LA.

PSDDA. 1988. PSDDA Reports - Management plan report: Unconfined open-water disposal of dredged material, phase I (Central Puget Sound). Prepared for Puget Sound Dredged Disposal Authority. US Army Corps of Engineers, Seattle District; US Environmental Protection Agency, Region 10, Seattle, WA; Washington Department of Ecology; and Washington Department of Natural Resources.

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 Biol 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 Technol 33:2390-2394.

Shannon J. 2006. Personal communication (e-mails to Matt Luxon, Windward Environmental, regarding Duwamish-Elliott beach seine data collected by Taylor Associates in 1998, 2000, and 2002-2003. Biologist, Taylor Associates, Seattle, WA. March 30, 2006.

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 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.

Tasto RN. 1975. Aspects of the biology of Pacific staghorn sculpin, *Leptocottus armatus* Girard, in Anaheim Bay. In: Lane DE, Hill CW, eds, The marine resources of Anaheim Bay. Vol Fish. Bull. 165. pp 123-135.

Thibodeaux LJ, Bierman VJ. 2003. The bioturbation-driven chemical release process. Environ Sci Technol 37(13):233A-264A.

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.

WDFW. 2002a. Puget Sound Ambient Monitoring Program (PSAMP): Dungeness crab (*Cancer magister*) [online]. Washington Department of Fish and Wildlife, Olympia, WA. Updated 2002. [Cited July 15, 2008.] Available from: http://wdfw.wa.gov/fish/psamp/dungenesscrab.htm.

WDFW. 2002b. Puget Sound Ambient Monitoring Program (PSAMP): Staghorn sculpin (*Leptocottus armatus*) [online]. Washington Department of Fish and Wildlife, Olympia, WA. Updated 2002. [Cited July 15, 2008.] Available from: http://wdfw.wa.gov/fish/psamp/staghornsculpin.htm.

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

Williston D. 2008. In prep. Technical memo: March-June 2007 Duwamish River /Green River water column PCB congener survey transmittal of data and quality assurance documentation. Water Quality Planner, King County Department of Natural Resources and Parks, Seattle, WA.

Windward. 2004a. Lower Duwamish Waterway remedial investigation. Crab and shrimp pilot study and quarterly surveys data report. Prepared for Lower Duwamish Waterway Group. Windward Environmental LLC, Seattle, WA.

Windward. 2004b. 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. 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.

Windward. 2009. Lower Duwamish Waterway remedial investigation. Data report: chemical analyses of fish, crab, and clam tissue samples collected in 2007. Prepared for Lower Duwamish Waterway Group. 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[42] in benthic invertebrate tissue and co-located sediment. Although the log-log relationship provided the closest fit to the data, a simple linear regression was selected because of the uncertainty associated with adjusting for variance when back-transforming predictions from a log-log regression model. The selected linear model has a reasonable fit with homogeneous residuals (Figure 1), 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 for the regression when the two influential values were included was 0.72. Without these two influential values, the regression continued to provide a good fit to the data in the range for which total PCB concentrations will be estimated in tissue. Exclusion of the two influential high values is warranted if the use of the model is to provide the closest fit to the data for the range across which the model will be applied. Because the FWM will be used to calculate tissue concentrations less than or equal to the current SWAC of 380 µg/kg dw, removal of the two high values that influenced the regression slope was justified. The R^2 value with the two high values removed was 0.57. The regression parameters were estimated with one half the reporting limit for the two non-detect samples.[43] The equation for the line with outliers removed is presented as Equation 1.

$$C_{BI} = (0.34 \times C_S) + 75 \qquad \textbf{Equation 1}$$

Where:

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

C_S = total PCB concentration in sediment (µg/kg dw)

---

[42] 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.

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

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.

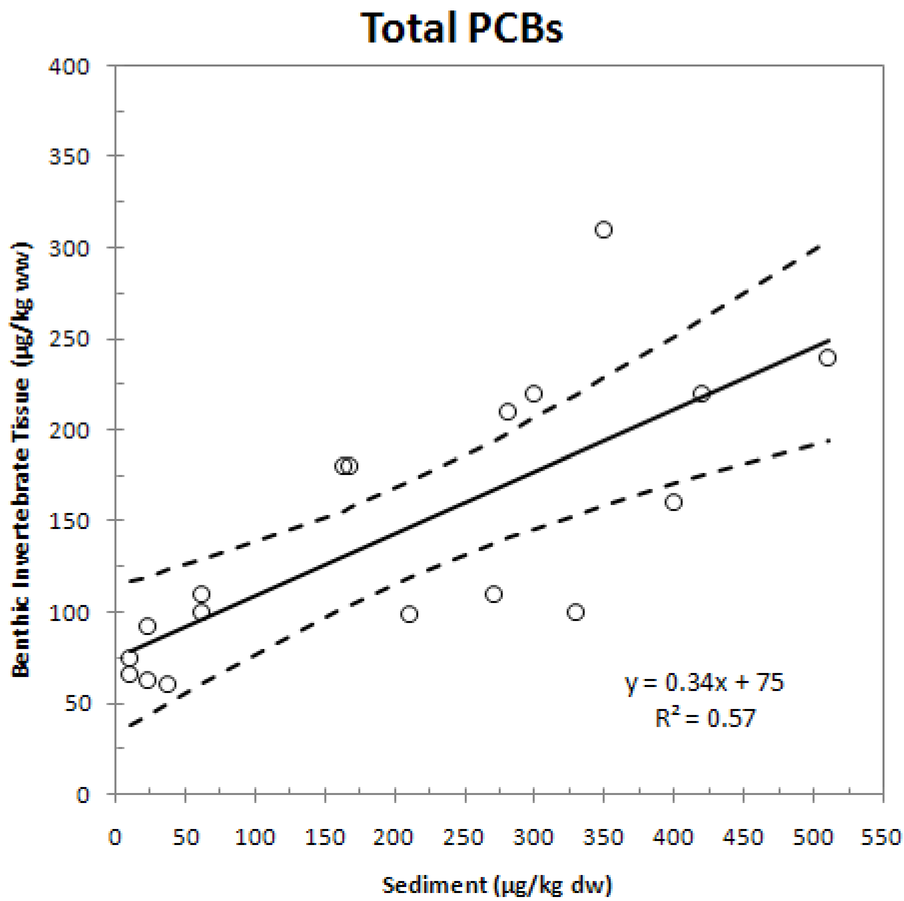


Figure 1. Linear regression fit to total PCB concentration in benthic invertebrate tissue as a function of total PCB concentration in sediment

Attachment 2 Statistics for the Calibrated Food Web Model

Parameter Description	Unit	Passed Model Performance Filter					Top 10 Runs (Ranked by average SPAF across species)			
		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
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}
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}
Mean water column temperature	°C	9.9	12.5	11.2	2.5	11.0	10.8	11.9	11.2	1.1
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
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}
Concentration of PCBs in sediment	µg/kg dw	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%
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
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
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}

Attachment 2, cont. Statistics for the calibrated food web model

Parameter Description	Unit	Passed Model Performance Filter					Top 10 Runs (Ranked by average SPAF across species)			
		Min	Max	Mean	Range	Best Fit	Min	Max	Mean	Range
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
Density of lipids	kg/L	0.8	1.0	0.9	0.2	0.9	0.8	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%
Water content of organism	%	93.7%	97.2%	95.6%	3.5%	95.7%	94.8%	96.7%	95.5%	2.0%
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}
Lipid content	%	0.2%	2.3%	1.2%	2.1%	1.4%	0.7%	1.7%	1.3%	1.0%
Water content of organism	%	85%	96%	90%	10%	92%	88%	92%	91%	4%
Dietary absorption efficiency of lipids (ε_L)	%	55%	85%	71%	30%	66%	61%	81%	71%	20%
Dietary absorption efficiency of NLOM (ε_N)	%	55%	85%	71%	29%	72%	58%	83%	70%	25%
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}
Lipid content	%	0.69%	1.05%	0.86%	0.35%	0.83%	0.69%	0.90%	0.80%	0.21%
Water content of organism	%	71%	87%	81%	15%	82%	76%	85%	83%	10%
Relative fraction of porewater ventilated	unitless	0.050	0.247	0.142	0.197	0.134	0.059	0.22	0.13	0.161
Dietary absorption efficiency of lipids (ε_L)	%	16%	95%	61%	80%	30%	30%	89%	59%	58%
Dietary absorption efficiency of NLOM (ε_N)	%	17%	93%	52%	77%	56%	18%	76%	43%	58%
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}
Lipid content	%	0.6%	4.6%	2.4%	4.0%	1.5%	1.1%	3.1%	1.9%	2.0%

Attachment 2, cont. **Statistics for the calibrated food web model**

Parameter Description	Unit	Passed Model Performance Filter					Top 10 Runs (Ranked by average SPAF across species)			
		Min	Max	Mean	Range	Best Fit	Min	Max	Mean	Range
Water content of organism	%	65.9%	82.0%	74.0%	16.1%	74.3%	69.8%	76.3%	73.2%	6.5%
Relative fraction of porewater ventilated	unitless	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%	92%	91%	93%	92%	2%
Dietary absorption efficiency of NLOM (ε_N)	%	50%	65%	58%	15%	54%	54%	61%	58%	7%
Slender Crab										
Organism weight	kg	0.152	0.180	0.167	0.028	0.165	0.163	0.175	0.167	0.012
Lipid content	%	0.9%	1.2%	1.1%	0.3%	1.1%	1.0%	1.2%	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%
Relative fraction of porewater ventilated	unitless	0.01	0.03	0.02	0.02	0.03	0.01	0.03	0.02	0.02
Dietary absorption efficiency of lipids (ε_L)	%	16%	95%	62%	79%	75%	39%	90%	68%	51%
Dietary absorption efficiency of NLOM (ε_N)	%	16%	95%	62%	79%	76%	39%	89%	68%	51%
Dungeness Crab										
Organism weight	kg	0.328	0.719	0.527	0.391	0.653	0.431	0.653	0.570	0.222
Lipid content	%	1.1%	4.2%	2.6%	3.1%	3.4%	2.3%	3.4%	2.8%	1.1%
Water content of organism	%	79%	84%	82%	5%	81%	81%	83%	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
Dietary absorption efficiency of lipids (ε_L)	%	16%	96%	61%	79%	71%	47%	82%	66%	35%
Dietary absorption efficiency of NLOM (ε_N)	%	18%	95%	62%	77%	59%	48%	82%	65%	34%
Pacific Staghorn Sculpin										
Organism weight	kg	0.062	0.089	0.077	0.026	0.075	0.065	0.078	0.074	0.013

Attachment 2, cont. **Statistics for the calibrated food web model**

Parameter Description	Unit	Passed Model Performance Filter					Top 10 Runs (Ranked by average SPAF across species)			
		Min	Max	Mean	Range	Best Fit	Min	Max	Mean	Range
Lipid content	%	1.9%	2.3%	2.1%	0.4%	2.1%	2.0%	2.2%	2.1%	0.2%
Water content of organism	%	79%	79%	79%	1%	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
Dietary absorption efficiency of lipids (ε_L)	%	90%	95%	92%	5%	93%	90%	94%	92%	3%
Dietary absorption efficiency of NLOM (ε_N)	%	50%	65%	58%	14%	50%	50%	62%	58%	12%
Shiner Surfperch										
Organism weight	kg	0.017	0.021	0.019	0.003	0.019	0.018	0.020	0.019	0.001
Lipid content	%	3.9%	5.3%	4.6%	1.3%	4.6%	4.2%	4.9%	4.6%	0.7%
Water content of organism	%	72.8%	75.2%	73.9%	2.4%	74.0%	73.3%	74.4%	73.9%	1.0%
Relative fraction of porewater ventilated	unitless	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%	94%	91%	94%	92%	3%
Dietary absorption efficiency of NLOM (ε_N)	%	50%	65%	58%	15%	56%	52%	63%	58%	11%
English Sole										
Organism weight	kg	0.212	0.282	0.247	0.070	0.246	0.231	0.258	0.246	0.027
Lipid content	%	4.7%	6.2%	5.5%	1.4%	5.5%	5.2%	6.0%	5.5%	0.7%
Water content of organism	%	74.0%	76.0%	75.0%	2.0%	75.0%	74.4%	75.3%	75.0%	1.0%
Relative fraction of porewater ventilated	unitless	0.01	0.20	0.10	0.19	0.15	0.04	0.1	0.09	0.11
Dietary absorption efficiency of lipids (ε_L)	%	90%	95%	92%	5%	92%	91%	94%	93%	3%
Dietary absorption efficiency of NLOM (ε_N)	%	50%	65%	58%	15%	59%	54%	63%	59%	9%

Attachment 2, cont. Statistics for the calibrated food web model

Parameter Description	Unit	Passed Model Performance Filter					Top 10 Runs (Ranked by average SPAF across species)			
		Min	Max	Mean	Range	Best Fit	Min	Max	Mean	Range
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
Phytoplankton	fraction	0.07	0.21	0.15	0.14	0.18	0.10	0.18	0.16	0.08
Zooplankton	fraction	0.01	0.17	0.08	0.16	0.12	0.04	0.12	0.08	0.08
Juvenile Fish										
Sediment	fraction	0.00	0.01	0.00	0.01	0.00	0.00	0.01	0.00	0.01
Zooplankton	fraction	0.35	0.81	0.57	0.47	0.53	0.51	0.78	0.60	0.27
Benthic invertebrate	fraction	0.18	0.65	0.42	0.46	0.47	0.22	0.49	0.39	0.27
Slender Crab										
Sediment	fraction	0.000	0.049	0.015	0.048	0.021	0.00	0.02	0.01	0.018
Zooplankton	fraction	0.004	0.118	0.076	0.114	0.094	0.016	0.115	0.075	0.099
Benthic invertebrate	fraction	0.860	0.976	0.899	0.115	0.876	0.863	0.968	0.903	0.105
Juvenile fish	fraction	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
Zooplankton	fraction	0.01	0.59	0.33	0.57	0.37	0.07	0.39	0.31	0.32
Benthic invertebrate	fraction	0.16	0.73	0.33	0.57	0.24	0.18	0.53	0.31	0.34
Juvenile fish	fraction	0.16	0.58	0.32	0.41	0.39	0.28	0.42	0.37	0.14
Pacific Staghorn Sculpin										
Sediment	fraction	0.00	0.05	0.02	0.05	0.00	0.00	0.02	0.01	0.02
Zooplankton	fraction	0.01	0.50	0.24	0.49	0.22	0.22	0.40	0.32	0.18
Benthic invertebrate	fraction	0.073	0.744	0.415	0.671	0.543	0.27	0.54	0.39	0.277
Juvenile fish	fraction	0.172	0.661	0.325	0.489	0.236	0.176	0.335	0.280	0.159

Attachment 2, cont. **Statistics for the calibrated food web model**

Parameter Description	Unit	Passed Model Performance Filter					Top 10 Runs (Ranked by average SPAF across species)			
		Min	Max	Mean	Range	Best Fit	Min	Max	Mean	Range
Shiner Surfperch										
Sediment	fraction	0.00	0.01	0.01	0.01	0.00	0.004	0.009	0.007	0.01
Zooplankton	fraction	0.188	0.689	0.403	0.501	0.230	0.23	0.37	0.32	0.137
Benthic invertebrate	fraction	0.304	0.803	0.591	0.499	0.765	0.629	0.765	0.677	0.135
English Sole										
Sediment	fraction	0.00	0.08	0.02	0.08	0.04	0.003	0.037	0.022	0.03
Phytoplankton	fraction	0.05	0.10	0.07	0.05	0.05	0.05	0.08	0.07	0.03
Zooplankton	fraction	0.00	0.08	0.04	0.08	0.05	0.01	0.06	0.04	0.05
Benthic invertebrate	fraction	0.86	0.90	0.88	0.04	0.86	0.86	0.89	0.87	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
Estimated total PCB concentration in zooplankton tissue	µg/kg ww	7	130	45	120	45	15	76	49	61
Estimated total PCB concentration in benthic invertebrate tissue	µg/kg ww	230	400	360	170	300	270	350	310	77
Estimated total PCB concentration in juvenile fish tissue	µg/kg ww	230	1200	700	940	470	410	680	530	270
Estimated total PCB concentration in slender crab tissue	µg/kg ww	340	1,300	670	980	690	570	740	660	170
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
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
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
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

Attachment 2, cont. Statistics for the calibrated food web model

Parameter Description	Unit	Passed Model Performance Filter					Top 10 Runs (Ranked by average SPAF across species)			
		Min	Max	Mean	Range	Best Fit	Min	Max	Mean	Range
Species Predictive Accuracy Factor (SPAF)										
Benthic invertebrate SPAF	unitless	1	2	2	1	1.5	1.4	1.7	1.5	0.4
Slender crab SPAF	unitless	1.0	2.0	1.2	1.0	1.0	1.0	1.2	1.1	0.2
Dungeness crab SPAF	unitless	1.0	2.0	1.3	1.0	1.1	1.0	1.2	1.1	0.2
Pacific staghorn sculpin SPAF	unitless	1.0	2.0	1.7	1.0	1.2	1.2	1.4	1.3	0.2
Shiner surfperch SPAF	unitless	1.0	2.0	1.2	1.0	1.2	1.1	1.5	1.2	0.4
English sole SPAF	unitless	1.0	1.7	1.2	0.7	1.1	1.0	1.2	1.1	0.2
Average SPAF	unitless	1.2	1.7	1.4	0.5	1.18	1.2	1.2	1.2	0.05

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

Attachment 3 EFDC Calibration Process for Predicting PCB Water Concentrations in Lower Duwamish Waterway

King County
Department of Natural Resources and Parks
Wastewater Treatment Division
King Street Center, KSC-NR-0500
201 South Jackson Street
Seattle, WA 98104-3855

MEMO

Date: November 30, 2009

TO: Jeff Stern, Debra Williston

FM: Bruce Nairn

RE: EFDC Calibration Process for predicting PCB water concentrations in Lower Duwamish Waterway

1.0 Introduction

A three-dimensional hydrodynamic model of the Duwamish River and Elliott Bay was developed for the King County *Water Quality Assessment of the Duwamish River and Elliott Bay* (WQA Study) (King County 1999a). This model was created using the Environmental Fluid Dynamics Code (EFDC) and included Elliott Bay, the East and West Waterways, and the Lower Duwamish Waterway (LDW) (Figure 1). The model was used to simulate hydrodynamic, sediment transport and chemical fate processes within the modeled waterbodies. Calibration and validation results indicate that the model (referred to as the King County model) simulates hydrodynamic processes in the Duwamish River and Elliott Bay with reasonable accuracy (King County 1999b). The model also preformed well for the sediment transport and chemical fate processes. However, at the time there were limited polychlorinated biphenyl (PCB) water data available for model calibration. Therefore, the purpose of this memorandum is to document the modifications made to the King County model to improve upon the water column predictions of PCBs for the Lower Duwamish Waterway. These water column predictions were used to help calibrate the PCB food web model used for the LDW Remedial Investigation (RI). The remainder of this memo discusses the model modifications and calibration process and the resulting outcomes to PCB water column predictions. Many of these modifications were made in consultation with the national and regional Environmental Protection Agency staff and regional NOAA staff.

2.0 EFDC Model Modifications and Calibration

The EFDC model was based on the model used in the WQA study with some modifications. The source code was updated with code revisions that had been released before April 19, 2004. EFDC was configured to simulate wetting and drying of the model cells and the model cell depths were updated with the 2004 bathymetric survey. The model grid was expanded to include slips along the Waterway. Sediment concentrations were updated with recent sampling data, and partitioning constants were recalculated based on recent PCB congener data. The sections below provide more detail regarding the model modifications and calibration updates.

2.1 Grid Cells

The King County model as used in the WQA Study generally had three horizontal grid cells typically being used to represent lateral variations in the LDW[1]. The basic structure of the original hydrodynamic model developed by King County was not

[1] The area east of Kellogg Island had four horizontal grid cells because of the greater width of the LDW in this area.

altered for this application. However, the numerical horizontal grid for the LDW was slightly refined (Figure 2). Horizontal model cells were added to provide a coarse representation of the slips along the LDW. The horizontal boundary of the cells throughout the LDW was adjusted to better align with the physical shoreline. Figure 2 shows a comparison of the revised grid and the previous grid structure for the LDW. Rectangular cells were used to replace the triangular cells to the west of Kellogg Island. The revised model contains a total of 521 horizontal cells (up from 512 in the WQA King County model), of which 115 cells represented the LDW. Three lateral grid cells represented the LDW at most locations.

Each horizontal cell contains 10 vertical cells, each vertical cell is 1/10 of the water column depth. No modifications were made to the number of vertical cells. The cell depths were updated to reflect bathymetric data collected in the LDW during 2003 (Windward and DEA 2004). The wetting-drying option in EFDC was activated for these simulations to account for the wetting and drying of shallow areas resulting from variable tidal and flow conditions.

The original EFDC grid did not include model cells for the slips because the width of the slips is smaller than the length of a model grid cell. However, for this application grid cells were added to provide a coarse representation of the slips. It was thought this would be beneficial to account for the generally higher PCB sediment concentrations that are found in some of the slips. For this purpose, one or two grid cells were added to the appropriate side of the 3-cell wide LDW model at the location closest to each slip. The cells were sized so that the surface area of the cell (or cells) was approximately equal to the surface area of the corresponding slip. If the slip was represented by multiple cells, it was separated into sections that corresponded to the surface area of the model cells. The sediment grain sizes and PCB concentrations of each slip were calculated for the area represented by each model cell and applied to the model. The cells representing the slips are handled identically to other cells in the LDW by the EFDC model. The model calculates the transport flux between the cells that represent the LDW channel and those that represent the slips. As the model is not constructed at a sufficiently fine horizontal scale to accurately represent the physical dimensions of the slips, the level of uncertainty associated with predictions within the slips should be considered to be much higher than the rest of the model.

2.2 Sediment Conditions

The model was configured with four sediment layers, each 2 cm thick. A high sediment diffusivity was used so these layers act as one 8cm thick layer. The previous WQA model used one layer of infinite thickness. The sediment/water column flux was represented by a flux velocity instead of a diffusion parameter.

Sediment characteristics within the model were updated to include historic and LDW RI Phase 2 surface sediment data [2](sand/silt/clay composition and total PCBs). Sediment conditions reflecting a post Diagonal/Duwamish Combined Sewer Overflow/storm drain (CSO/SD) site remediation in 2004 and 2005 (both cap and perimeter conditions prior to placement of thin sand layer cover in 2005) were also included (Figure 3). Pre- and post- remediation samples at perimeter stations showed a change in both sediment grain size and PCB concentrations. As a result, only post remediation samples were used in both the remediation area and in a buffer area surrounding the remediation area. Details of the sediment concentrations and sample locations are provided in Appendix A. Multiple data points within a model cell were averaged; values in cells lacking any data were determined by linear interpolation from neighboring cells. Any data points outside of the grid (due to shoreline irregularities, for instance) were included in the nearest grid cell average.

One sample about 430 ft to the northwest of the Norfolk CSO/SD outfall was removed from the dataset (Figure 4). This sample is located near the upper bank and has a total PCB value of 220,000 µg/kg dw. When this value was included, it elevated the PCB concentrations in that model cell in excess of what was thought to be reasonable when all other data for that cell were considered (mean 660 µg/kg dw, n = 56). This is because this value was much higher than all other data. This approach was discussed and agreed to with EPA and NOAA.

2.3 Boundary conditions

The hydrodynamic model requires three types of boundary conditions:

1) water surface (tidal) elevation, salinity, and PCB concentrations along the open boundary in Elliott Bay,
2) surface wind velocity and direction, and
3) freshwater inflow and PCB concentrations from the Lower Green River at the upstream boundary.

At the Elliott Bay boundary, the tidal forcing consists of six tidal harmonic constituents (i.e., M2, S2, N2, K1, O1, P1), as discussed in the WQA modeling appendix (King County 1999a). This is the same parameterization of the tidal height as in the WQA configuration. Salinity of the incoming water from Elliott Bay was updated from 1996-97 data to reflect the actual observations for 2004 and 2005. CTD measurements taken by King County at station LTED04[3] were used to represent this open boundary. PCB water concentrations were set based on 2005 low-level PCB congener sampling (King County, 2006). PCBs were measured in August, September, November and December 2005 from a depth of 20 m below the water surface at Station LTED04 in Elliott Bay

[2] At the time of the model update, only Round 1 and 2 surface sediment data from Phase 2 data collection efforts of the LDW RI were available (Windward 2005a, b).
[3] Data can be obtained from King County at http://dnr.metrokc.gov/wlr/waterres/marine/DownloadData.aspx.

(Figure 5). An average total PCB concentration of 65 pg/L was used to set the Elliott Bay boundary condition for PCBs as the data showed no seasonal trend.

Wind forcing was assumed to be spatially uniform and was based on observations recorded at Boeing Field, and was updated for 2004-2005.

Upstream boundary conditions for PCBs were also based on the 2005 low-level PCB congener sampling. Samples for PCBs were collected in the Green River by Fort Dent Park (Station TGS/1) during the same time as those for Elliott Bay (Figure 5). An average total PCB concentration of 100 pg/L was used to set the Green River boundary condition for PCBs. TSS was calculated at this boundary using the same regression equation as was used in the WQA configuration (Equations 3-1 and 3-2 in King County 1999).

For the fine sands/coarse silts sediment class:

$$SS\left[{}^{mg}\!/\!{}_{l}\right] = 0.964Q^{1.09} \text{ where Q is flow } (m^3/s)$$

and for both the silt and the clay sediment classes:

$$SS\left[{}^{mg}\!/\!{}_{l}\right] = 0.654Q^{1.09} \text{ where Q is flow } (m^3/s)$$

PCB concentrations in lateral inflows into the LDW were set at zero, following the approach used in the original King County model. This is largely due to a lack of water column data with detected results for PCBs for lateral inflows. During the WQA study, CSOs were sampled for PCBs but no PCBs were detected. At the time the model was configured, there had been no CSO or stormwater samples collected for PCBs. Based on this approach, the source of PCBs in the water column of the LDW is largely due to flux from the sediments as well as some inputs from the boundary conditions. Current sediment transport modeling of the LDW indicates that the greatest solids input to the LDW is from the Green River with lateral inflows only contributing approximately 0.6% of the sediment load (QEA 2007-STM Report). This may result in a slightly overestimated flux being predicted from the sediment.

2.4 Partition Coefficients for PCBs

Updated solids partitioning coefficients (Kd) for total PCBs in both the sediment and water column were estimated based on a PCB congener weighted octonal-water partition coefficients (Kow). The congener weighted Kow for sediments was calculated in the same method described in the Food Web Model Memorandum 3 (Windward, 2006). The congener weighted Kow was based on the PCB congener data for benthic invertebrates because a full congener analysis was not available for sediments in the LDW. It was agreed with EPA and NOAA that using the benthic invertebrate congener data was the closest approximation to sediment congener results for the purposes of developing a weighted Kow for sediments. Therefore, within the sediment bed, log Kow of 6.6 was used as a representative value based on the benthic invertebrate PCB congener data (Windward, 2006).

A different partition coefficient for PCBs in the water column of the LDW was assigned to the model. This log Kow value was based on weighted PCB congener data for PCBs in whole water samples collected in 2005 in the LDW. Two locations with two depths each were collected for low-level PCB congener analysis (Figure 5). These data were collected during the same months the low-level PCB congener samples in the Green River and Elliott Bay were collected. The Kows for each congener were based on data provided in Hawker and Connell (1988). This is the same data as was used in the LDWG's Food Web Model Memorandum 3 (Windward, 2006). Based on these PCB sampling data, log Kow of 5.8 was selected as a representative value for PCB's in the water column.

The EFDC model was configured to use a solids partitioning coefficient, similar to the approach used in the WQA study. The partitioning coefficient (Kd) is the ratio of the solid to dissolved concentration, C_s(mg/kg)/C_d(mg/L). To estimate a solids partitioning coefficient (Kd), the approach in Schwarzenbach, Gschend, and Imboden (Environmental Organic Chemistry, 1993) was used. Binding sites on the solids is assumed to be dominated by organic carbon, so the solids partitioning coefficient is estimated as the product of the organic carbon partition coefficient (Koc) and the fraction of organic carbon in the sediment. Log Kd was estimated based on the following equation:

$$\log Kd = \log (2\ foc) + 0.88 \log Kow - 0.27$$

Where foc is the fraction of organic carbon to sediment mass, and Kow is based on the weighted PCB congener results as described above. Organic carbon of the sediments was assumed to be 2% based on the overall average total organic carbon data for the LDW. The organic carbon content of the suspended solids was set at 4% based on TOC and TSS data collected in 2005 by King County in the LDW. In this approach to partitioning, these organic carbon values do not vary temporally or spatially. The partitioning coefficients calculated in this manner are Kd = 1.4×10^{-2} L/mg for the sediment bed and Kd = 5.5×10^{-3} L/mg for the water column.

The use of a single partitioning coefficient means that the EFDC model does not simulate the effect of temperature on the physical properties of PCB compounds. The temperature in the LDW surface water varied between 4°C and 17°C, and the near-bottom water ranged between 7°C and 14°C based on historical King County data.

2.5 Calibration of Total PCB Water Column Predictions

The model was previously calibrated during King County's WQA study for tidal elevation, salinity, velocity, suspended solids, metal and organic chemicals, and bacterial parameters. While minor modifications were made to the model grid, the existing calibration was used without modification. The effect of the grid changes on the calibration were not quantified, although qualitative comparison indicated the model produced similar results.

The calibration objective for this modeling was to obtain total PCB predictions that would be within a factor of two when averaged LDW-wide on a monthly timescale. The diffusion flux across the sediment/water interface is the primary calibration parameter. This flux rate was adjusted so that the model predictions would align with the low level PCB congener water data collected in the LDW by King County in 2005 (King County, 2006) (Figure 5). Water samples were collected during both the dry warmer summer period and wet, cooler fall season. The flux rate was set at 1.0e-6 m/s for this simulation. This flux rate is constant in time, and the model does not include any simulation of biological activity or time varying estimates of the sediment flux rate. No information is available on the relative amount of biological activity or the influence of that activity on PCB flux rates.

The model predictions and empirical total PCB water column data are shown in Figure 6 for the Spokane Street Bridge (LTKE03) station and in Figure 7 for 16th Avenue Bridge (LTUM03) station. Samples at these locations were collected at depths of 1 meter below the surface and 1 meter above the bottom. Model predictions are shown for the vertical model layers 2 and 9, which correspond to the sample water depths. For example, at mean sea level (MSL) model layer 2 spans the distance from 0.73 to 1.47 meters above the bottom at 16th Avenue Bridge sample location (LTUM03), which corresponds with the 1 meter above the bottom water sampling location. Model layer 9 corresponds to 0.73 to 1.47 meters below the surface at 16th Avenue Bridge sample location (LTUM03), which corresponds with the 1 meter below the surface water sampling location. Finally, model layer 2 and 9 at Spokane Street Bridge sample location (LTKE03) spans the distance of 0.93 to 1.86 meters above the bottom and below the surface, respectively, which corresponds to each surface and bottom water sampling location. As the model layers are a fraction of the water depth, the depths will vary with the tidal elevation. The sampling locations remain within the indicated model layers for most tidal conditions. Therefore, the model was calibrated to corresponding sample water depths. As a conservative step, total PCB concentrations were not adjusted for potential laboratory blank contamination for specific congeners detected in laboratory blank samples.

The model predictions and observations are in qualitative agreement. At the Spokane Street Bridge station (LTKE03), the model predicts surface water concentrations to be higher at the surface than at depth, in agreement with the empirical observations (Figure 6). This is consistent with the overall circulation of the LDW. At the downstream portion of the LDW (closer to Elliott Bay), relatively clean Elliott Bay water is expected to enter the LDW at depth, moving upriver until it is entrained into the outgoing surface water. This is reflected in the model predictions as well as the empirical observations where PCB concentrations tend to be lower at depth.

At the 16th Avenue Bridge station (LTUM03), the model predicts bottom water concentrations to be similar or slightly higher than near surface, corresponding with the trend in the empirical observations (Figure 7). Located towards the upstream portion of the LDW, the PCB concentration in bottom water has increased in concentration due to PCB flux from the sediments. Surface concentrations are lower due to the lower concentrations observed in the freshwater input from the Green River.

As previously noted, the EFDC model does not include effects of temperature or biological activity on the physical parameters of PCBs or on the exchange rate at the sediment-water interface. The calibration data was obtained between August and December, and this period of calibration should encompass variations resulting from changes in temperature and biological activity. The predicted concentrations tend to increase in late summer and decline afterwards, following the same trend as the observations (Figures 6, 7). While consideration of these processes could lead to potential improvements in model calibration, the existing level of calibration suggests that the impact is likely to be small.

3.0 Model Simulation and Results for Total PCBs

A 1 year "spin-up" period (2004) was simulated to remove the influence of the initial model conditions. This is typically done in model simulations when the exact conditions throughout the domain are unknown. It allows the model to evolve to a condition that is reflective of the boundary conditions, which are typically better known than conditions throughout the domain. The model was then run for an additional year (2005) to provide predictions of total PCB water column concentrations for the LDW. These predictions were used in the LDW food web model calibration process. For purposes of the food web model, the model output was processed to calculate monthly average concentrations, spatially averaged over the entire LDW, and separately, over each of the four modeling sub areas (Figure 8). This was averaged vertically over the entire water column, and separately, over the bottom three model layers (bottom 30% of the water column). These results are provided in Tables 1 and 2.

The modeling predicted PCB concentrations on a monthly basis ranging from 0.99 ng/L to 1.78 ng/L for the LDW as a whole when the entire water column was included (Table 1). The ranges for the bottom three model cells only were very similar to monthly averages for the entire water column (Table 2). PCB concentrations did show some differences on a modeling area scale with area 3 having the highest predicted concentrations. The modeling area results for the bottom three cells showed some slight increases for areas 3 and 4 when compared to the entire water column but areas 1 and 2 were similar. These results are likely due to the influence of Elliott Bay flows in the bottom layer in areas 1 and 2 and the differences in sediment concentrations between modeling areas.

The model output was also post-processed to examine the distribution of PCB concentrations at the two sampling stations within the LDW (LTKE03, LTUM03) at both the bottom and surface sampling depths. This was to provide a bounds for the long-term averages used in the LDW Food Web Model. The values are shown graphically on Figures 6 and 7, with a minimum of 0.13 ng/L and a maximum of 5.3 ng/L.

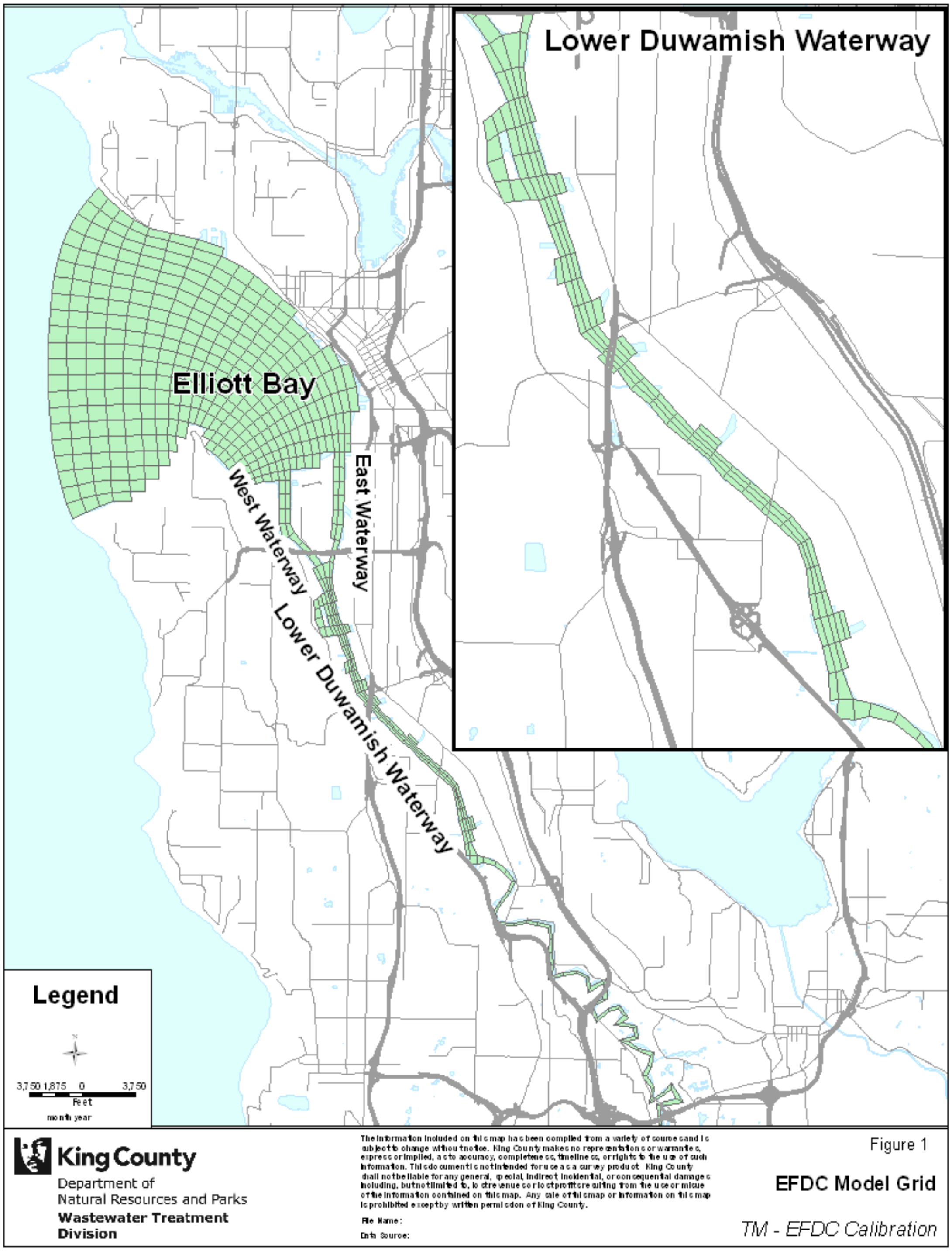


Figure 1. EFDC model grid.

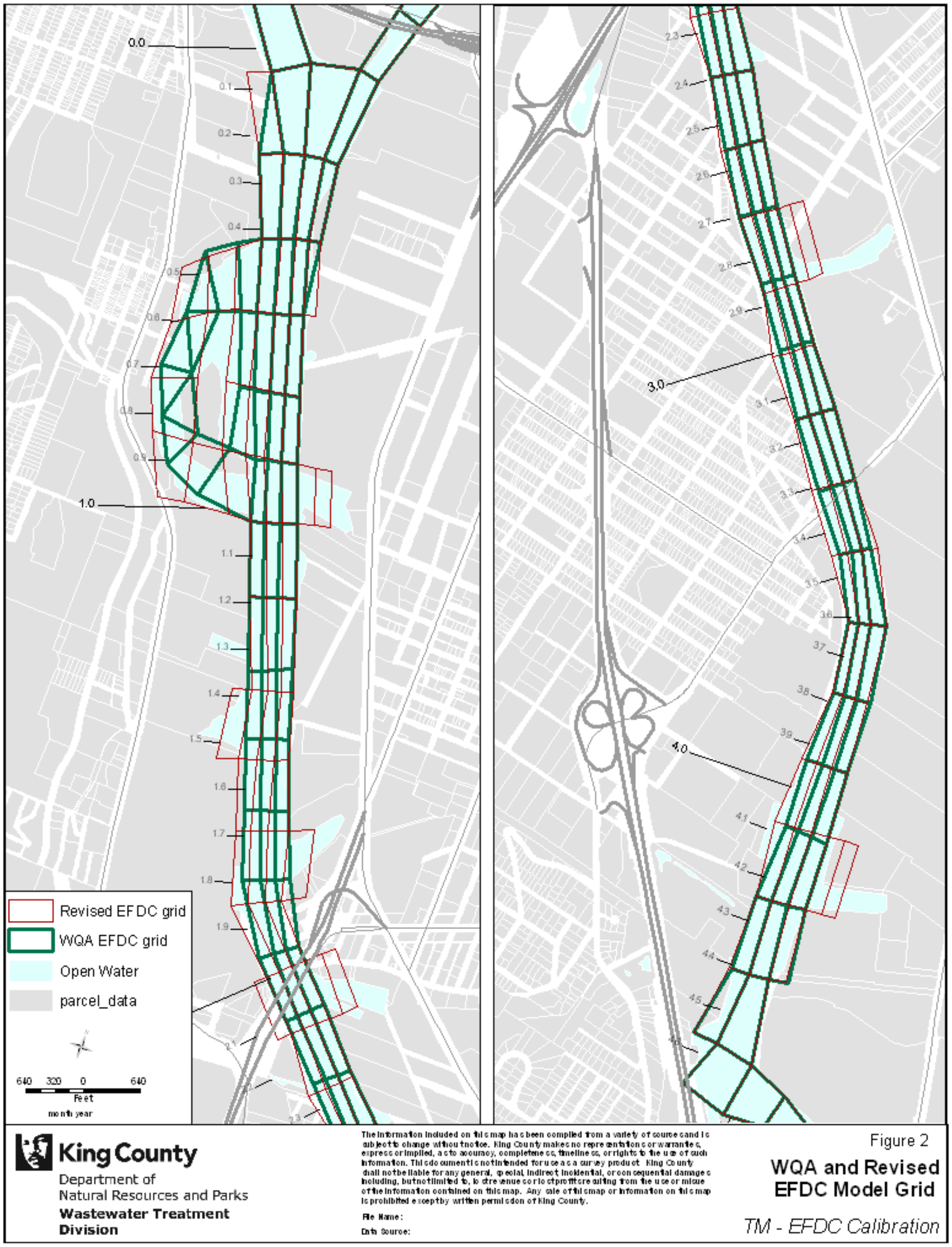


Figure 2. Comparison of original WQA model grid and revised model grid. Cells representing slips were sized to match surface area and depth of the slips.

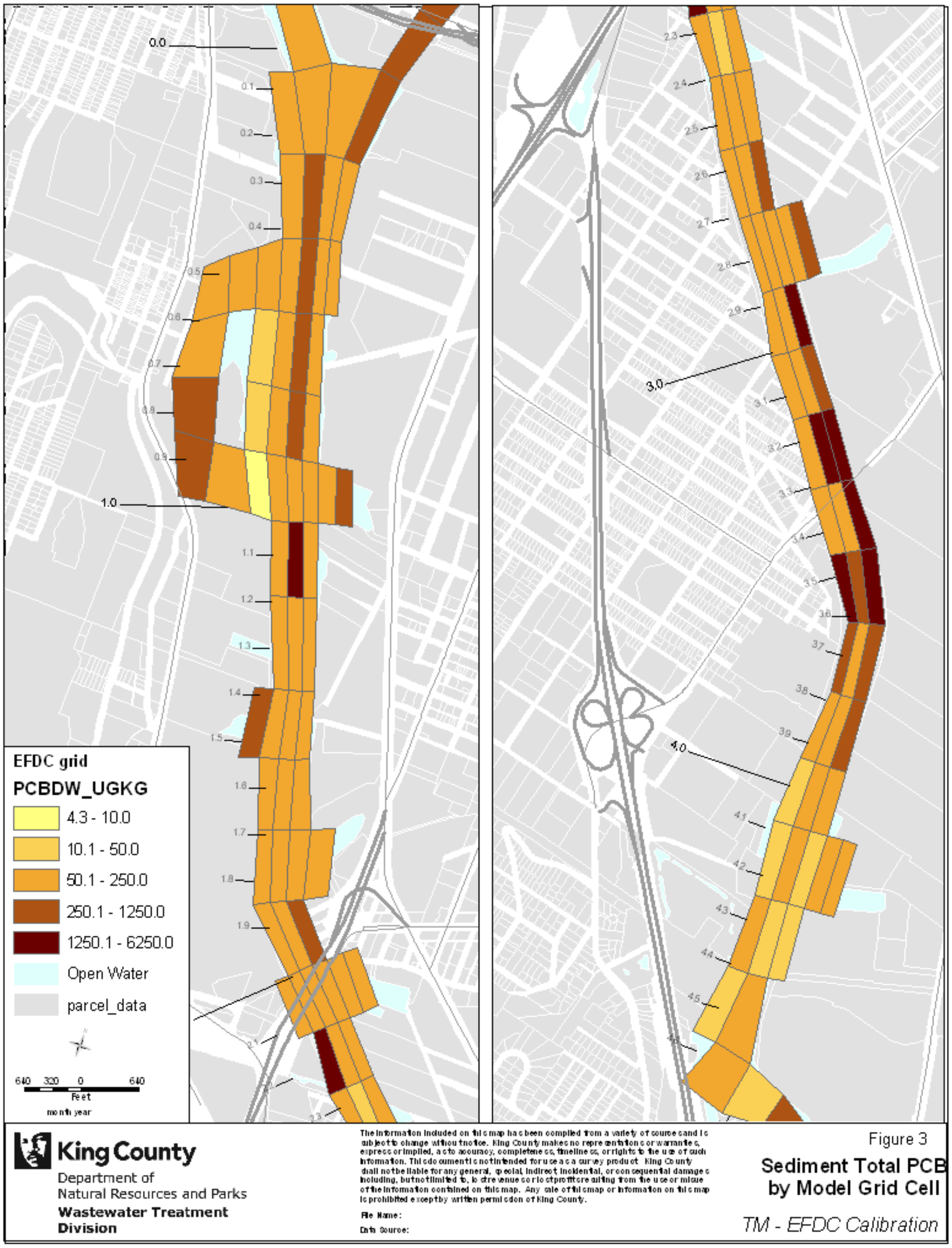


Figure 3. Sediment conditions reflecting a post Diagonal/Duwamish Combined Sewer Overflow/storm drain (CSO/SD) site remediation in 2005 (both cap and perimeter conditions prior to placement of thin-layer cap) were also included.

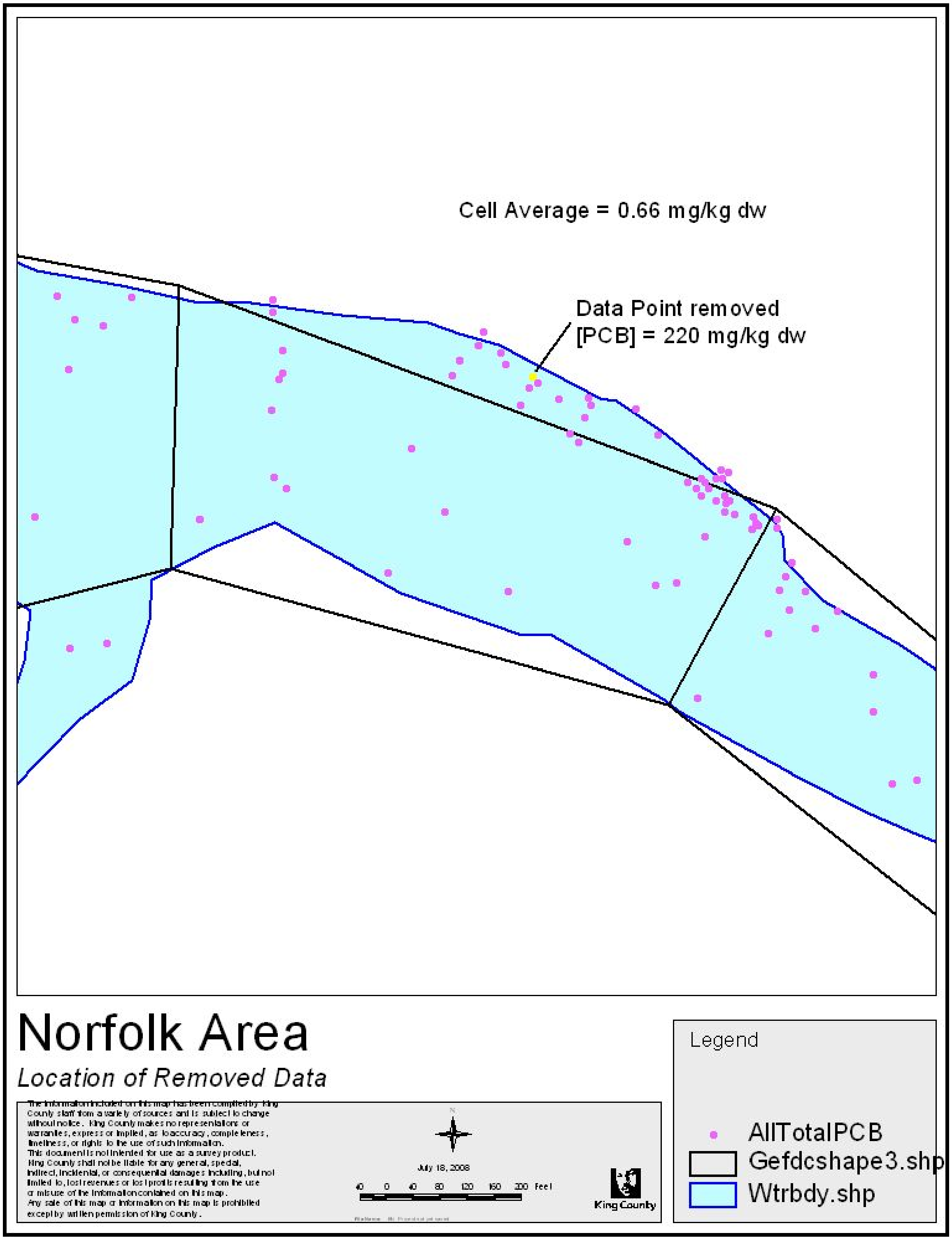


Figure 4. One sample about 430 ft to the northwest of the Norfolk CSO/SD outfall was removed from the dataset.

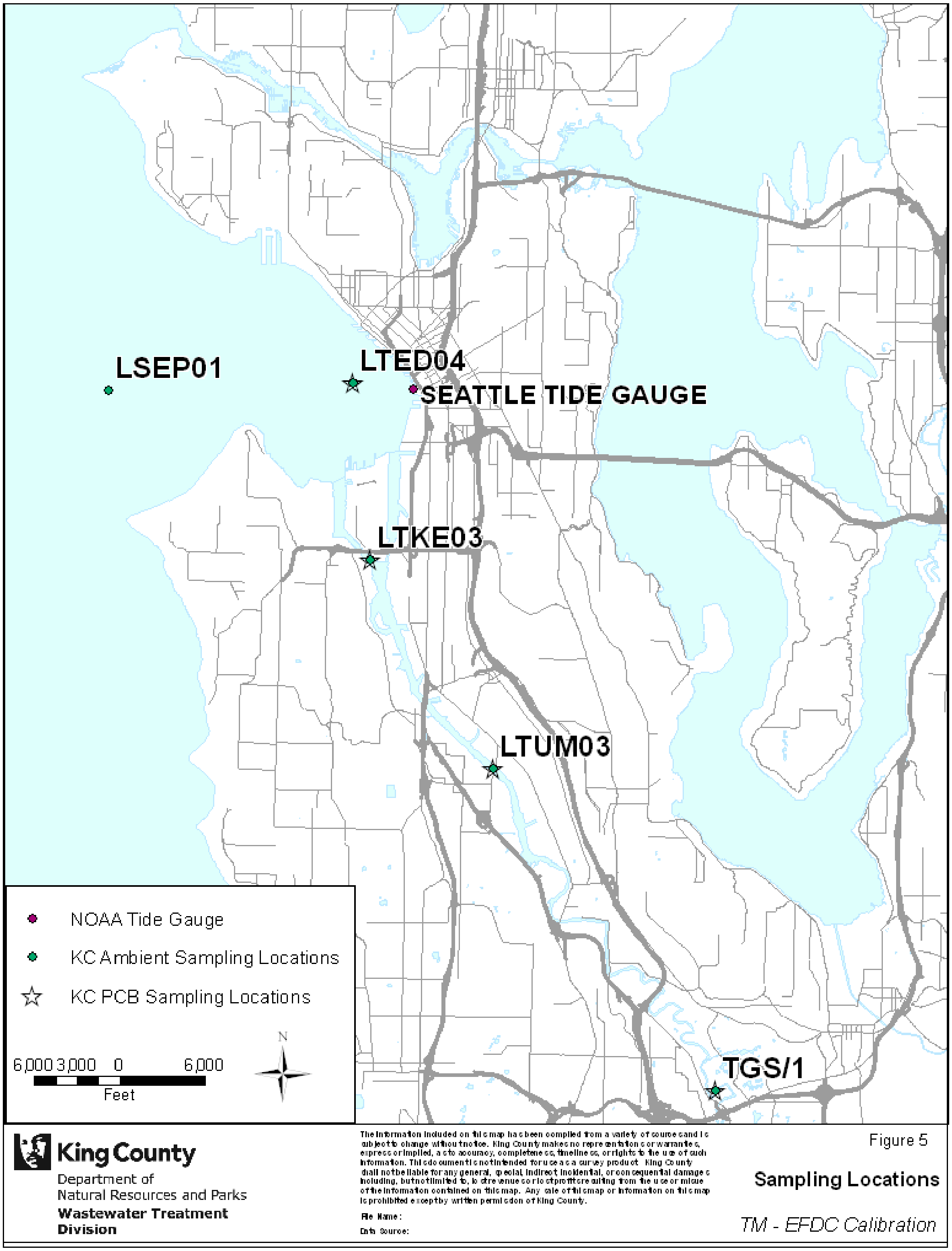


Figure 5. Sampling locations for ambient water properties and low level PCB congener data.

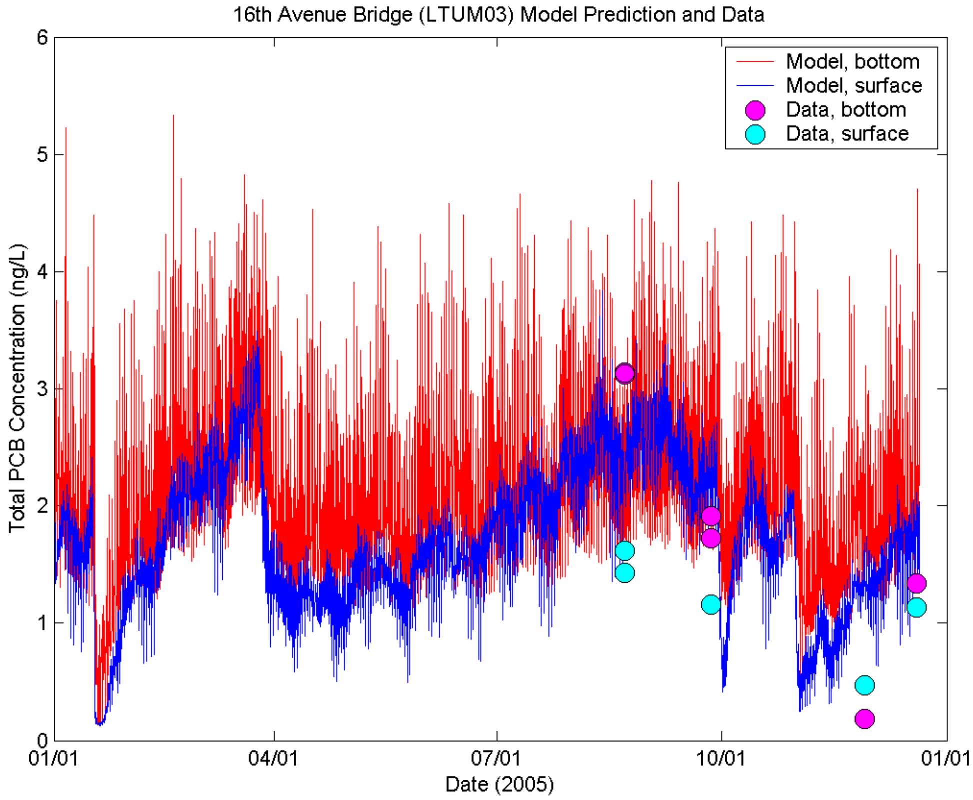


Figure 6. Model calibration for total PCBs at LTUM03 sampling location.

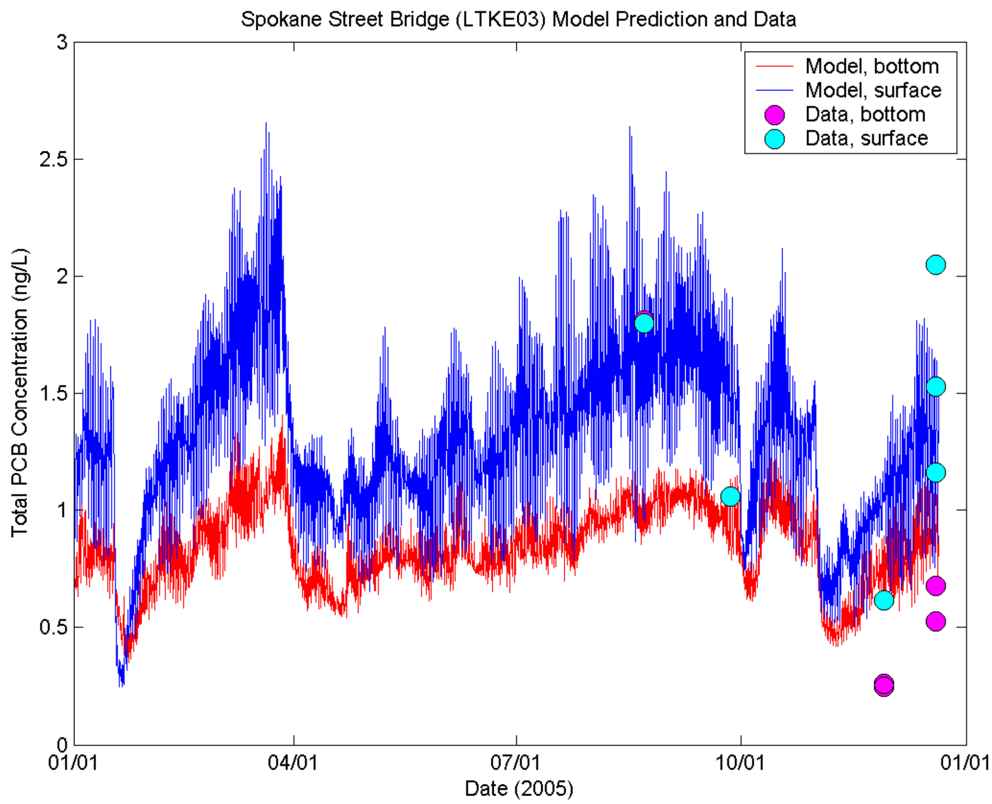


Figure 7. Model calibration for total PCBs at LTKE03 sampling location.

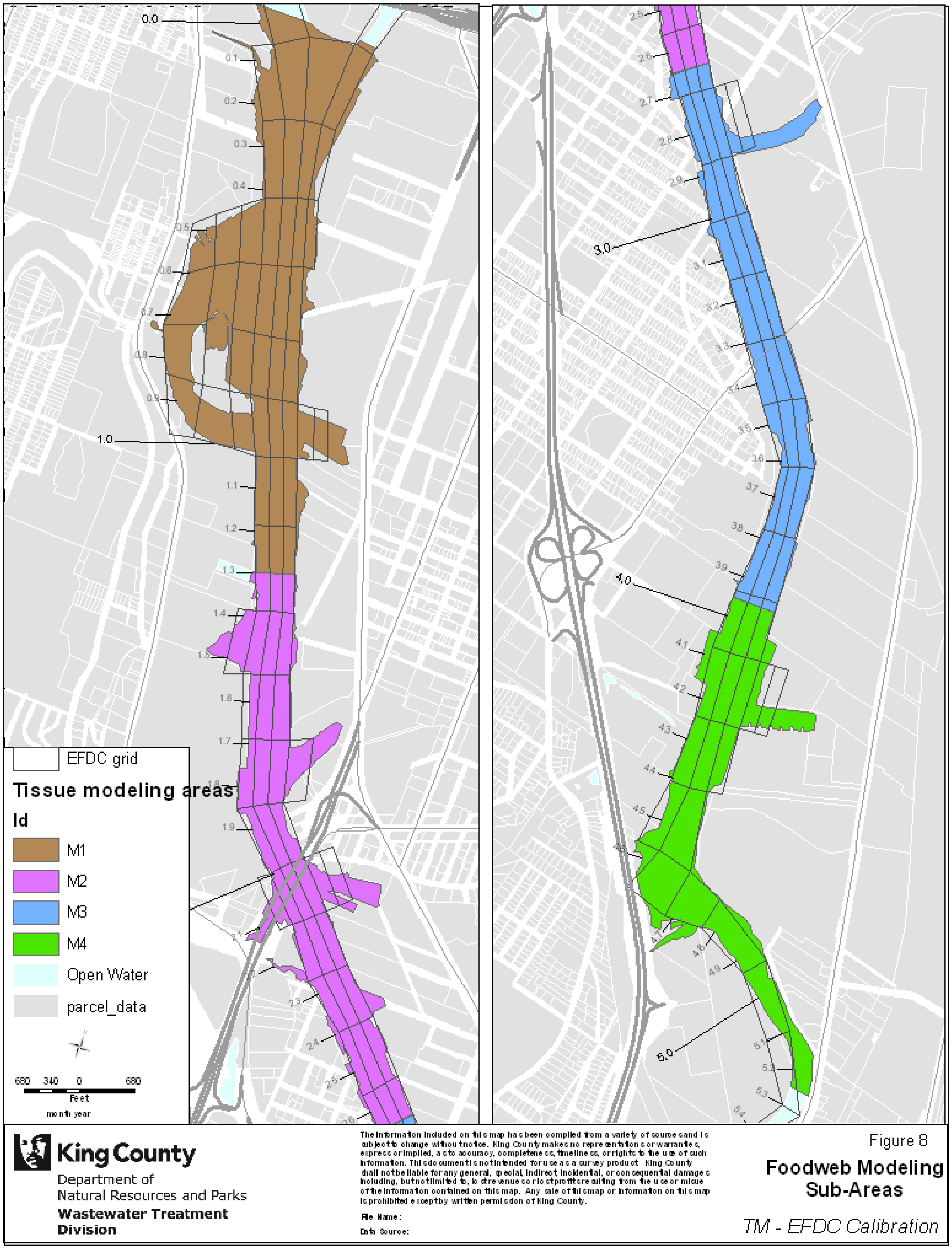


Figure 8. The EFDC grid and the four tissue modeling areas used in the LDW Food Web Model.

Table 1. EFDC Water Column Total PCB monthly averages (ng/L) for entire LDW and four modeling areas.

	EFDC Water Column Total PCB Averages (ng/L)				
	LDW	Area 1	Area 2	Area 3	Area 4
Jan	1.05	0.88	1.07	1.59	1.23
Feb	1.21	1.01	1.23	1.82	1.46
Mar	1.78	1.46	1.79	2.60	2.37
Apr	1.18	1.01	1.22	1.74	1.30
May	1.15	0.98	1.17	1.70	1.29
Jun	1.25	1.05	1.26	1.87	1.50
Jul	1.40	1.14	1.40	2.13	1.83
Aug	1.64	1.30	1.64	2.47	2.32
Sep	1.71	1.37	1.71	2.54	2.40
Oct	1.38	1.15	1.40	2.01	1.71
Nov	0.99	0.84	1.02	1.47	1.10
Dec	1.23	1.04	1.25	1.81	1.43

Table 2. EFDC Total PCB monthly averages (ng/L) within the bottom 30% of the water column for entire LDW and four modeling areas.

	Bottom 3 Model Layers Only EFDC Water Column Total PCB Averages (ng/L)				
	LDW	Area 1	Area 2	Area 3	Area 4
Jan	1.19	0.88	1.07	2.34	1.66
Feb	1.36	1.01	1.21	2.65	1.98
Mar	1.82	1.35	1.62	3.30	2.93
Apr	1.34	1.00	1.24	2.57	1.85
May	1.30	0.97	1.18	2.53	1.81
Jun	1.38	1.01	1.23	2.70	2.03
Jul	1.47	1.06	1.30	2.88	2.29
Aug	1.63	1.16	1.45	3.03	2.75
Sep	1.69	1.23	1.51	3.09	2.85
Oct	1.48	1.10	1.33	2.74	2.24
Nov	1.15	0.86	1.06	2.23	1.60
Dec	1.39	1.05	1.25	2.63	1.94

References

Arega F, Hayter EJ. 2004. Hydrodynamic and transport modeling in a highly stratified estuary. In: Lee J, Lam K, eds, Environmental Hydraulics and Sustainable Water Management, Proceedings of the 4th International Symposium on Environmental Hydraulics and the 14th Congress of Asia and Pacific Division, International Association of Hydraulic Engineering and Research. Hong Kong, China, December 15-18, 2004.

Hawker, D.W. and D.W. Connell, 1988. Octanol-Water Partition Coefficients of Polychlorinated Biphenyl Congeners, *Environ. Sci. Technol.* 22, 382-287.

King County. 1999a. King County combined sewer overflow water quality assessment for the Duwamish River and Elliott Bay. King County Department of Natural Resources, Seattle, WA.

King County. 1999b. King County combined sewer overflow water quality assessment for the Duwamish River and Elliott Bay. Vol 1, Appendix B1: Hydrodynamic and fate and transport numerical model. King County Department of Natural Resources, Seattle, WA.

King County. 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 and Parks, Seattle, WA.

Schwarzenbach, René P., Philip M. Gschwend, and Dieter M. Imboden. 1993. *Environmental organic chemistry*. Chichester: Wiley.

Windward, 2005a. Data Report: Round 1 Surface Sediment Sampling for Chemical Analysis and Toxicity Testing. October 2005. Prepared for Lower Duwamish Waterway Group. Windward Environmental LLC, Seattle, WA.

Windward, 2005b. Data Report: Round 2 Surface Sediment Sampling for Chemical Analysis and Toxicity Testing. December 2005. Prepared for Lower Duwamish Waterway Group. Windward Environmental LLC, Seattle, WA.

Windward, 2006. Food Web Model Memorandum 3: Preliminary Model Results, April 2006. Prepared for Lower Duwamish Waterway Group. Windward Environmental LLC, Seattle, WA.

Windward and QEA, 2007. Sediment Transport Analysis Report – Draft Final. June 2006. Prepared for Lower Duwamish Waterway Group. Windward Environmental LLC, Seattle, WA and Quantitative Environmental Analysis, Montvale, NJ.

Appendix A - Diagonal/Duwamish Sediments

Sediment remediation occurred in the vicinity of the Diagonal/Duwamish CSO/storm drain in 2004/2005. The dredging and capping at the Diagonal/Duwamish project took place between November 2003 and March 2004, with an additional sand placement occurring in February 2005. Water samples were collected for PCB analysis between August and December of 2005. To provide a more realistic simulation of PCB concentrations, the sediments in the vicinity of this remediation work were updated to reflect sediment conditions after capping/dredging but prior to placement of the sand-layer around the southern portion of the remediation area. These sediment concentrations were specified from samples taken on the cap in June 2004 (year 0) and the perimeter in January 31 - February 2, 2005 (year 1).

Following the placement of the thin-layer of sand in February 2005, samples were taken from the thin-layer placement and the original cap in March and April 2005, respectively. The perimeter, thin-layer placement and cap stations were not sampled again until March 2006. At the time the model was configured, the sediment data collected in March 2006 was not yet available. This March 2006 data is likely to be more reflective of sediment concentrations in the perimeter and thin-layer placement areas during the period in which water samples were collected (August - December 2005). It would appear that using the post thin-layer placement samples (March 2005) and the year 1 cap samples (April 2005) would more closely match the sediment concentrations during the period of time when water samples were collected (that were used for model calibration). A comparison of the sediment concentrations including the post thin-layer data collected in March and April 2005 (Figure A6) with the data used in the model (Figure A5) appears quite similar, and unlikely to make a noticeable difference in model predictions at the scale the model was used for.

The following methodology was used to update the sediment grain size and PCB concentrations.

A review of the results at the perimeter sampling locations indicated sizeable changes in sediment grain size and total PCBs from pre- to post- remediation (Table A1). The sand composition changed by more than 8% at all stations analyzed, and changed by more than 20% at stations DUD_8C, DUD_9C, DUD_12C, and DUD_20C. As a result, the sediment characteristics were updated in the area surrounding the remediation, as well as the remediation site. Determination of the area affected by the remediation work was done by best professional judgment, as samples are not available to delineate the affected region. Delineation was done by creating a simple polygon around the remediation site and surrounding sediment samples (Figure A1). Thiessen polygons were created from the LDWG baseline data set, and data points to be removed were identified based on the assumed affected area (Figure A2). The final data set is shown with thiessen polygons in Figure A3.

Sediment grain size and total PCB concentrations in the EFDC model were determined by a simple average of the data points within each grid cell. The sampling points and EFDC sediment concentrations by grid cell are shown using data prior to remediation (Figure A4) and post remediation (Figure A5).

The effect of the thin layer placement on sediment concentrations is shown in Figure A6 based on the samples collected in March and April 2005 on thin layer placement and dredged/capped areas. This may be more representative of sediment conditions in these areas at the time water samples were collected, however it was not used in the model. These sediment differences would not likely results in any appreciable differences in PCB water column concentration predictions at the resolution (or scale) at which the EFDC model predictions were used in the FWM.

Table A1. Sediment Characteristics at Perimeter stations Pre- and Post- Dredge/Capping

Pre-Remediation					Post-Dredging/Capping, Pre-Thin Layer Placement				
Locator	Sample Date		Value	units	Locator	Sample Date		Value	units
DUD_7C	10/20/2003	total PCB	428	ug/kg dw	DUD_7C	1/31/2005	total PCB	397	ug/kg dw
		Gravel	3.5	%			Gravel	3.7	%
		Sand	34.9	%			Sand	43	%
		Silt	51	%			Silt	33.8	%
		Clay	10.7	%			Clay	12.4	%
DUD_8C	10/21/2003	total PCB	5026	ug/kg dw	DUD_8C	2/1/2005	total PCB	809	ug/kg dw
		Gravel	0.85	%			Gravel	7	%
		Sand	25.6	%			Sand	66.1	%
		Silt	65.1	%			Silt	23.4	%
		Clay	8.5	%			Clay	9.8	%
DUD_9C	10/21/2003	total PCB	103	ug/kg dw	DUD_9C	1/31/2005	total PCB	137	ug/kg dw
		Gravel	2.8	%			Gravel	0.4	%
		Sand	79.3	%			Sand	59	%
		Silt	12.9	%			Silt	26.7	%
		Clay	4.9	%			Clay	11.2	%
DUD_10C	10/21/2003	total PCB	373	ug/kg dw	DUD_10C	2/1/2005	total PCB	328	ug/kg dw
		Gravel	3.6	%			Gravel	1.2	%
		Sand	54.7	%			Sand	62.9	%
		Silt	29.2	%			Silt	22.7	%
		Clay	12.7	%			Clay	9	%
DUD_12C	10/21/2003	total PCB	263	ug/kg dw	DUD_12C	2/2/2005	total PCB	334	ug/kg dw
		Gravel	1 J	%			Gravel	0.9 J	%
		Sand	45.4	%			Sand	69.2	%
		Silt	38.8	%			Silt	22.7	%
		Clay	14.9	%			Clay	9.1	%
EST231	9/19/1997	total PCB	230	ug/kg dw	DUD_20C	2/2/2005	total PCB	458	ug/kg dw
		Gravel	3	%	(30 ft East)		Gravel	0.3	%
		Sand	33	%			Sand	63.2	%
		Silt	40	%			Silt	24.7	%
		Clay	23	%			Clay	15	%
DR057	8/31/1998	total PCB	139	ug/kg dw	DUD_20C	2/2/2005	total PCB	458	ug/kg dw
		Gravel	0.38	%			Gravel	0.3	%
		Sand	38	%			Sand	63.2	%
		Silt	42	%			Silt	24.7	%
		Clay	20.7	%			Clay	15	%

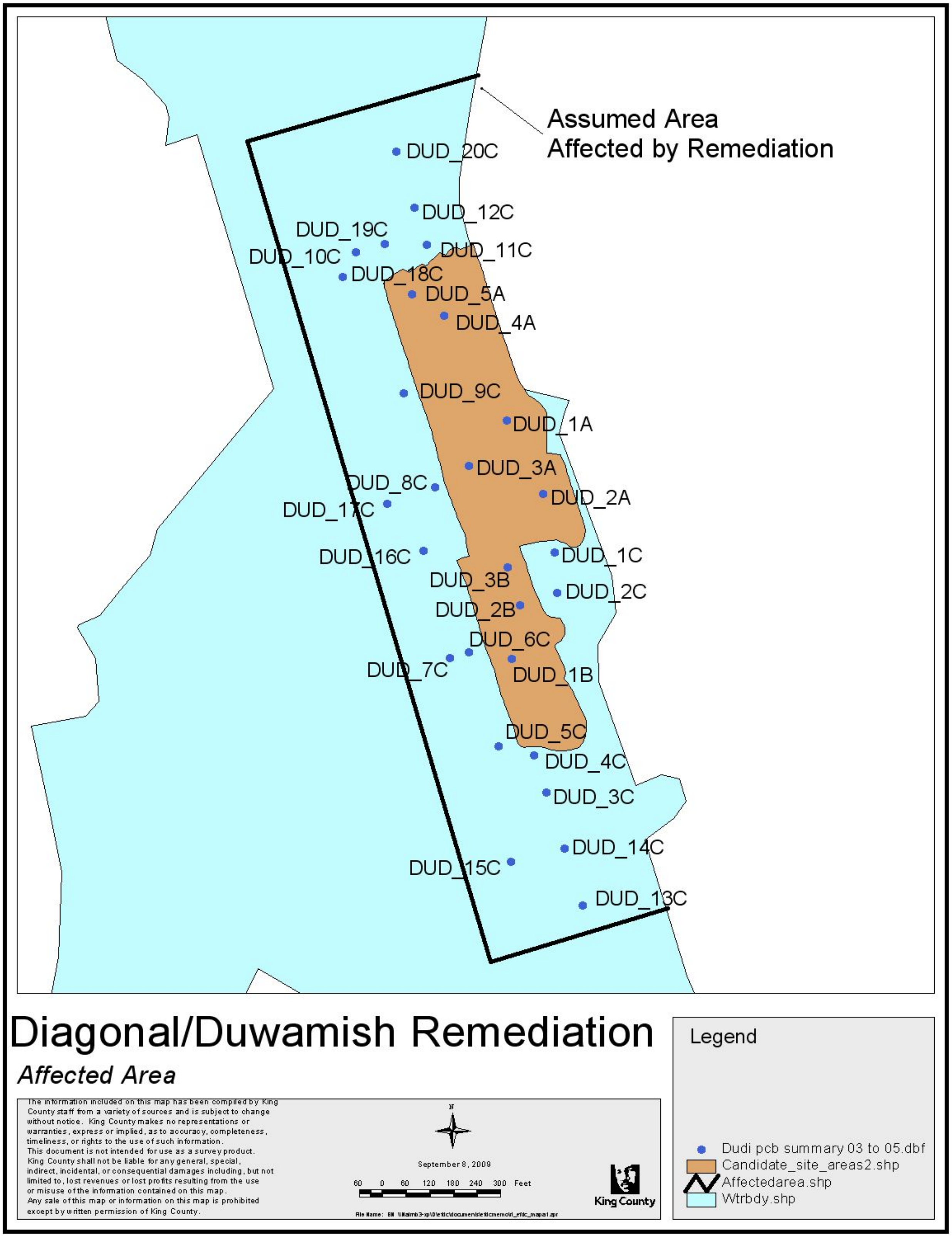


Figure A1. Area identified as potentially affected by sediment remediation project

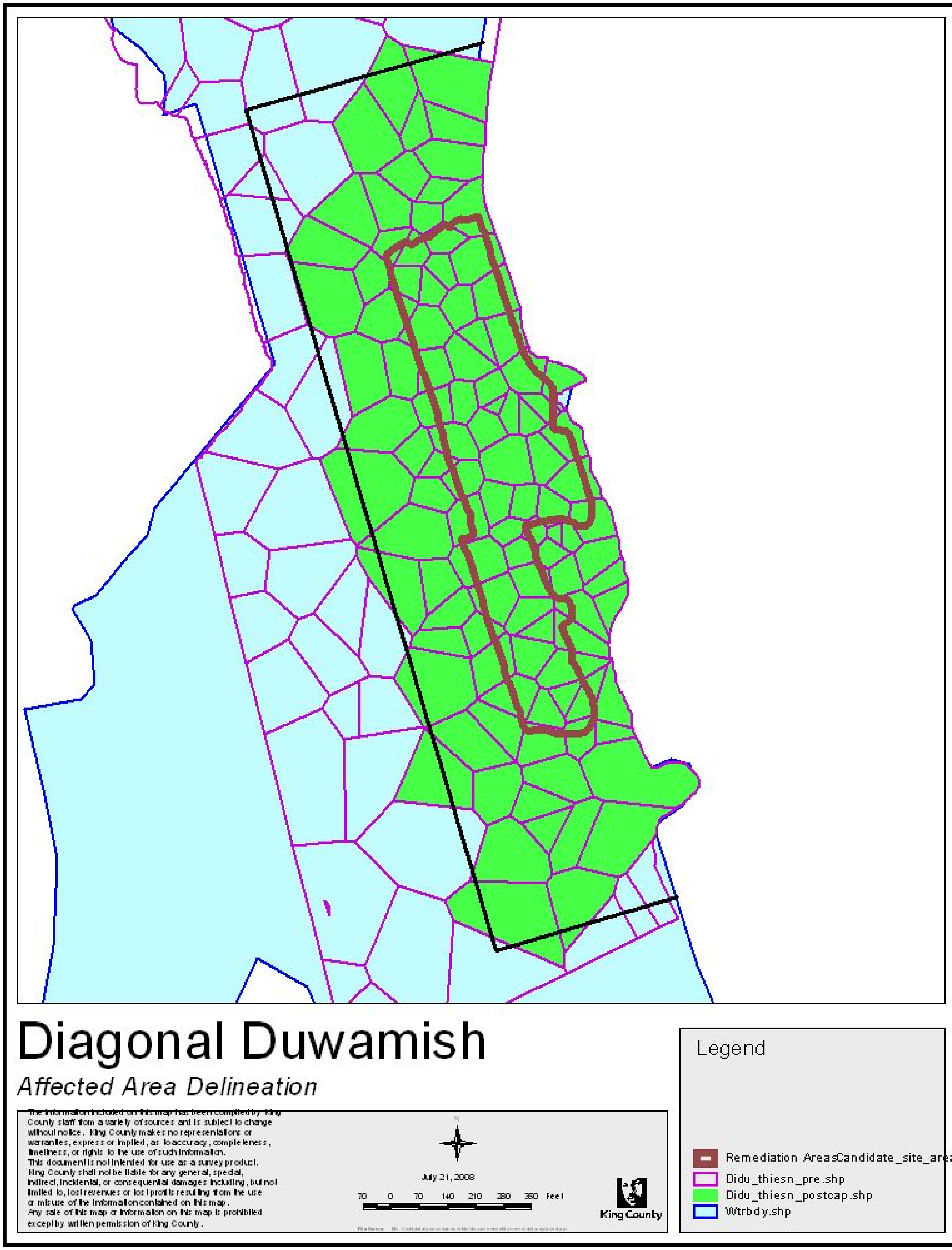


Figure A2. Data points as represented by Theissen polygons replaced by post-remediation sediment parameters

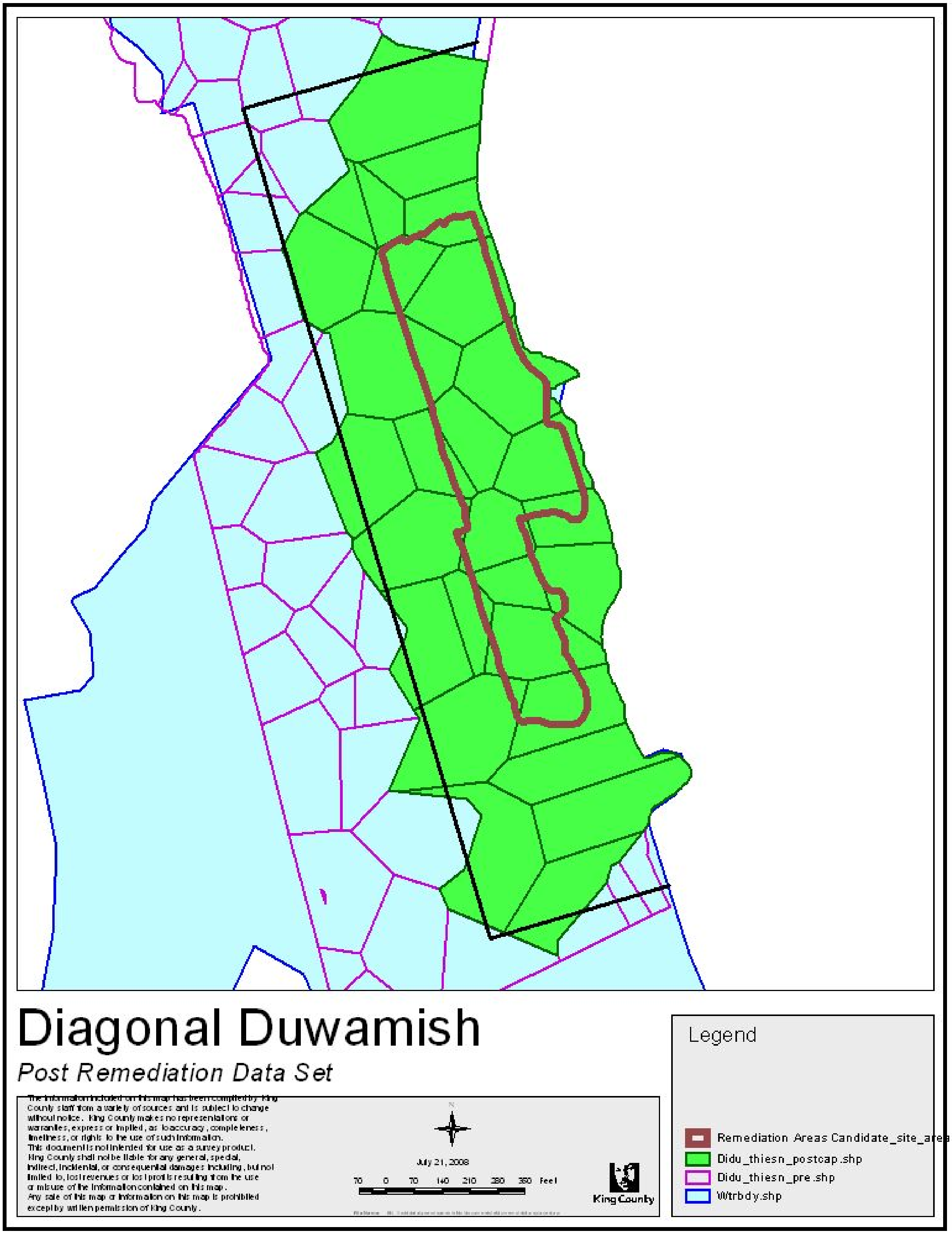


Figure A3. Data points as represented by Theissen polygons used to characterize post-remediation sediment parameters

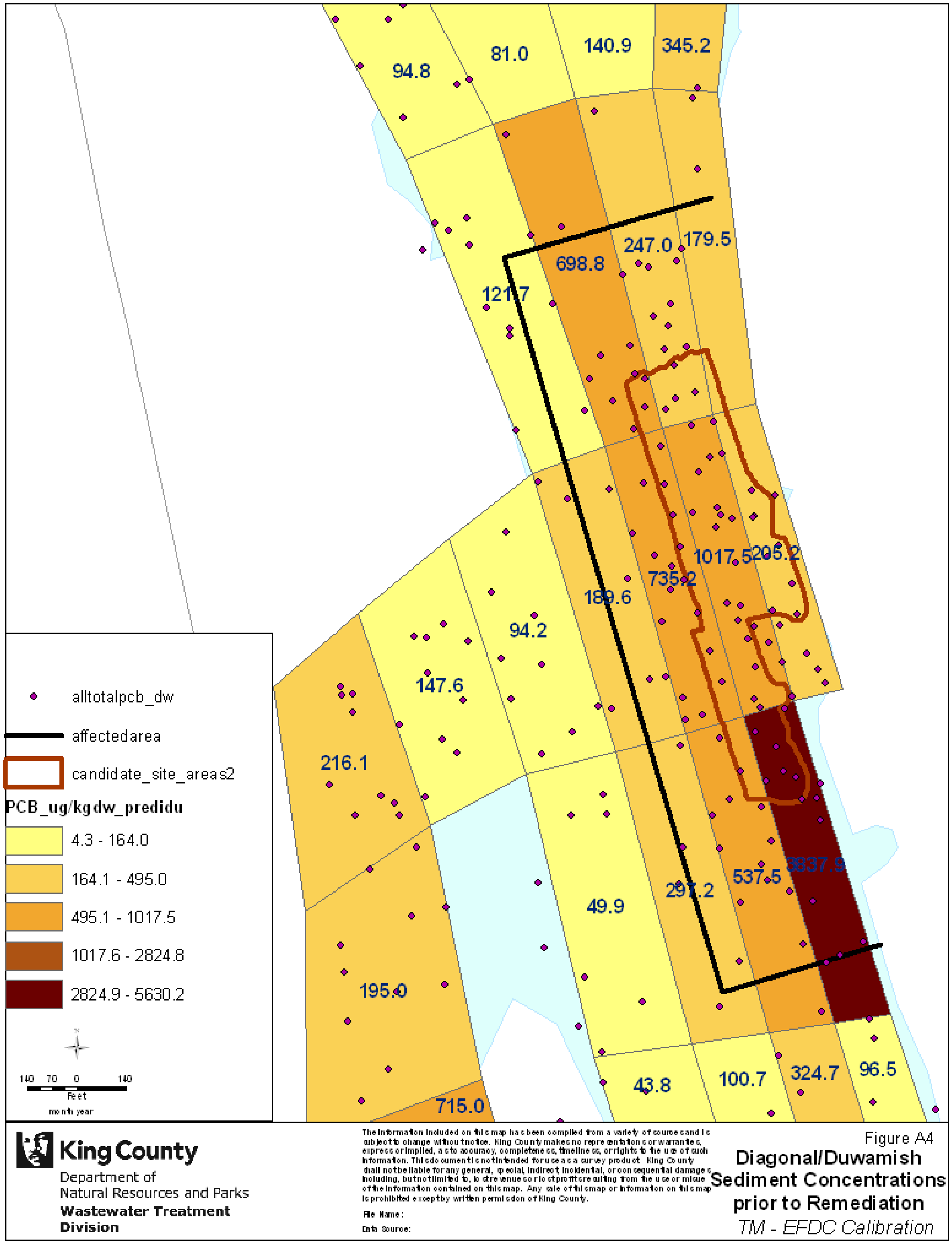


Figure A4. PCB sediment concentrations by EFDC grid cell using pre-remediation values.

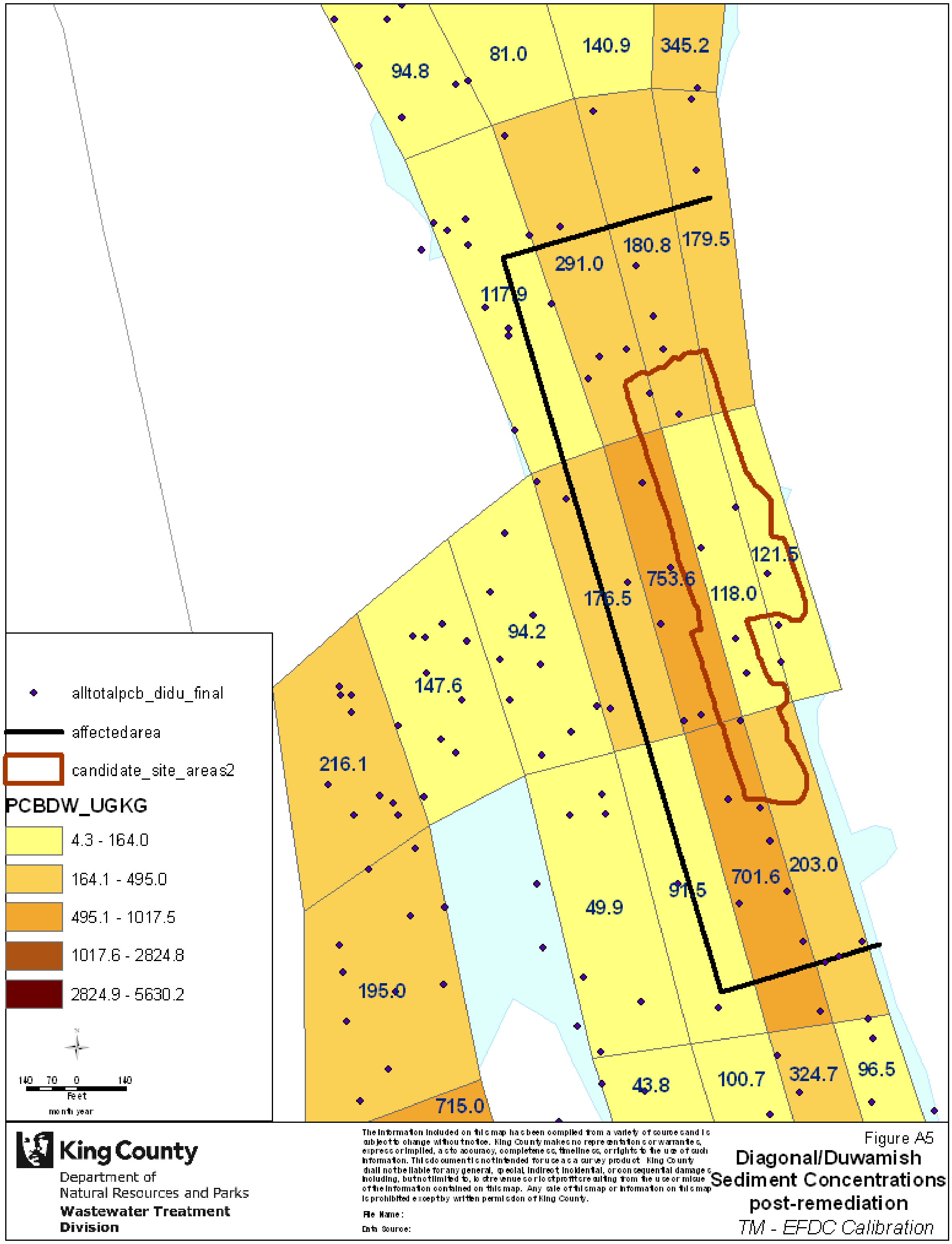


Figure A5. PCB sediment concentrations by EFDC grid cell using post-dredging/capping but pre-thin layer placement values.

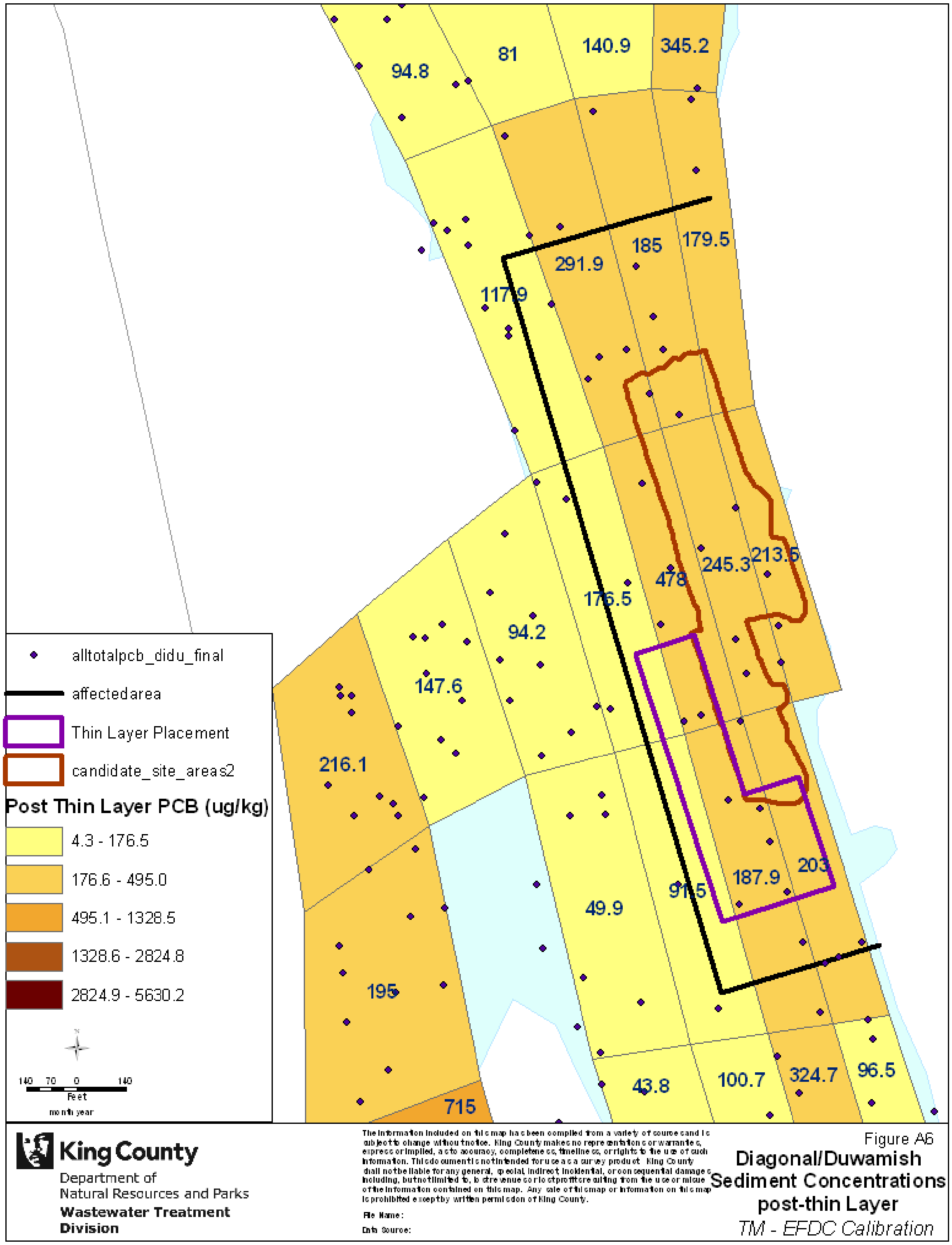


Figure A6. PCB sediment concentrations by EFDC grid cell using post-dredging/capping and post-thin layer placement values based on 2005 samples. These values were not used in the model.