EAST WATERWAY OPERABLE UNIT
SUPPLEMENTAL REMEDIAL INVESTIGATION/ FEASIBILITY STUDY
APPENDIX B: BASELINE HUMAN HEALTH RISK ASSESSMENT
FINAL

For submittal to:

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Region 10
Seattle, WA

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Table of Contents

List of Tables

List of Figures

List of Maps

Acronyms

ACRONYM	Definition
ABS	dermal absorption fraction
AF	adherence factor
ALM	Adult Lead Model
API	Asian and Pacific Islander
ATSDR	Agency for Toxic Substance and Disease Registry
BCA	bias-corrected accelerated bootstrap method
BEHP	bis(2-ethylhexyl) phthalate
BHC	benzene hexachloride
BPJ	best professional judgment
Cal EPA	California Environmental Protection Agency
CDC	Centers for Disease Control and Prevention
CDI	chronic daily intake
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CFR	Code of Federal Regulations
COC	chemical of concern
COPC	chemical of potential concern
cPAH	carcinogenic polycyclic aromatic hydrocarbon
cPAH TEQ	toxic equivalent of cPAHs to benzo(a)pyrene
CSM	conceptual site model
CSO	combined sewer overflow
CT	central tendency
DA_{event}	dermally absorbed dose per event
DDD	dichlorodiphenyldichloroethane
DDE	dichlorodiphenyldichloroethylene
DDT	dichlorodiphenyltrichloroethane
dioxin/furan TEQ	toxic equivalent of dioxins and furans to 2,3,7,8-TCDD
DMA	dimethyl arsenic acid
dw	dry weight
EISR	existing information summary report
EM	edible meat
EPA	US Environmental Protection Agency
EPC	exposure point concentration
ERA	ecological risk assessment
ESG	Environmental Solutions Group
EW	East Waterway
EWG	East Waterway Group
FS	feasibility study

ACRONYM	Definition
FI	fractional intake
GSD	geometric standard deviation
HEAST	Health Effects Assessment Summary Tables
HHRA	human health risk assessment
HI	hazard index
HPLC	high-performance liquid chromatography
HQ	hazard quotient
IEUBK	Integrated Exposure Uptake Biokinetic Model for Lead in Children
IR	ingestion rate
IRIS	Integrated Risk Information System
J-qualifier	estimated concentration
KC	King County
KM	Kaplan-Meier
LDW	Lower Duwamish Waterway
MIS	multi-increment sampling
MTCA	Model Toxics Control Act
N-qualifier	tentative identification
NCP	National Contingency Plan
NOAA	National Oceanic and Atmospheric Administration
NPL	National Priorities List
NRD	natural resource damage
NWR	national wildlife refuge
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PCB TEQ	toxic equivalent of dioxin-like PCBs to 2,3,7,8-TCDD
PCDD	polychlorinated dibenzo-*p*-dioxin
PCDF	polychlorinated dibenzofuran
PEF	potency equivalency factor
PPE	personal protective equipment
PPRTV	Provisional Peer-Reviewed Toxicity Values
PRG	preliminary remediation goal
PSAMP	Puget Sound Ambient Monitoring Program
QAPP	quality assurance project plan
QC	quality control
RAGS	Risk Assessment Guidance for Superfund
RAIS	risk assessment information system
RBC	risk-based concentration
RBTC	risk-based threshold concentration
RCRA	Resource Conservation and Recovery Act
RfD	reference dose

ACRONYM	Definition
RI	remedial investigation
RL	reporting limit
RME	reasonable maximum exposure
RPF	relative potency factor
RSL	regional screening level
Sd	standard deviation
SEM	simultaneously extracted metals
SF	slope factor
SMS	Washington State Sediment Management Standards
SRI	supplemental remedial investigation
SVOC	semivolatile organic compound
SWAC	spatially weighted average concentration
T-18	Terminal 18
T-30	Terminal 30
T-102	Terminal 102
TBT	tributyltin
TCDD	tetrachlorodibenzo-*p*-dioxin
TEF	toxic equivalency factor
TEQ	toxic equivalent
TPH	total petroleum hydrocarbons
U-qualifier	not detected at given concentration
U&A	Usual and Accustomed
UCL	upper confidence limit on the mean
USCG	US Coast Guard
WAC	Washington Administrative Code
WB	whole body
WDFW	Washington State Department of Fish and Wildlife
WHO	World Health Organization
Windward	Windward Environmental LLC
WQA	water quality assessment
WSDOH	Washington State Department of Health
WSOU	Waterway Sediment Operable Unit
ww	wet weight

Executive Summary

This document presents the baseline human health risk assessment (HHRA) that has been completed as part of the supplemental remedial investigation and feasibility study (SRI/FS) for the East Waterway (EW). Baseline risk assessments, as defined in US Environmental Protection Agency (EPA) guidance (EPA 1988a), "provide an evaluation of the potential threat to human health and the environment in the absence of any remedial action. They provide the basis for determining whether or not remedial action is necessary and the justification for performing remedial actions."

The baseline HHRA presents risk estimates for various scenarios whereby people could be exposed to chemicals of potential concern (COPCs) found in fish and shellfish tissue, sediment, and surface water in the EW (Map ES-1). To the extent possible, this HHRA is consistent with the approach and methods that were approved by EPA for use in the HHRA for the Lower Duwamish Waterway (LDW) (Windward 2007c), which is a Superfund site that is located upstream of and contiguous with the EW and has many physical and functional characteristics similar to those of the EW. In addition, this HHRA is consistent with the HHRA technical memorandum, which was approved by EPA (Windward 2010f).

Because knowledge of current and future site use is imperfect, the scenarios evaluated in this assessment have been selected in an attempt to not underestimate risks (i.e., to be health protective) and, as such, may overestimate risks for many site users. The dataset for the baseline HHRA consisted primarily of tissue, sediment, and surface water chemistry data collected from the EW as part of the EW SRI/FS sampling efforts, along with available historical data collected since 1995. The baseline HHRA includes sections on data evaluation, conceptual site model and exposure assessment, toxicity assessment, risk characterization, and uncertainty analysis, each of which is briefly summarized below.

ES.1 DATA EVALUATION

The data evaluation section of the HHRA includes a description of what data were available, a determination of how the data were used in the HHRA, and a discussion of the suitability of the data for risk assessment purposes.

Tissue chemistry data for evaluating exposures from seafood consumption were available for English sole, perch (shiner surfperch and striped perch), brown rockfish, crabs, clams (including geoducks), and mussels collected from within the EW. The consumption rates used in this HHRA were based on recent regional consumption surveys for areas in and around Puget Sound but were not specific to the consumption of seafood from only the EW. Tissue chemistry data from the EW were not available for some of the species reported as being consumed in these regional studies. However, the species collected from the EW were considered representative of all trophic groups of seafood that could be consumed (e.g., English sole are considered representative of other benthic fish such as speckled sanddab) and were thus used as surrogates as necessary. It should be noted that human health risk estimates for the EW did not include the consumption of salmon, despite the fact that of all of the fish species caught in the EW for seafood consumption, salmon are one of the most highly preferred and consumed species. The exposure of salmon to chemicals in EW sediment is not anticipated to significantly influence the concentrations in their tissue, primarily because of the very small portion of their lives spent in the EW (i.e., the vast majority of their lives is spent in Puget Sound and the Pacific Ocean). An analysis presented by O'Neill et al. (1998) indicated that less than 1% of the PCB body burden of adult salmon migrating through the LDW was obtained from prey items consumed in the LDW. Similarly, contributions to the salmon PCB body burden attributable to the EW would be expected to be minimal. This approach was consistent with the LDW HHRA (Windward 2007c).

Sediment chemistry data consisted of subtidal surface sediment samples (collected from 0 to 10 cm) and intertidal multi-increment sampling surface sediment samples (collected from 0 to 25 cm). To evaluate exposure to chemicals in surface water, data collected from 1 m below the water surface were used to estimate risks.

Tissue, sediment, and surface water data collected from the EW are considered to be representative of chemical concentrations throughout the EW and the expected human exposure at the site. However, it should be noted that any uncertainties in these data (e.g., laboratory qualification of data or representativeness of tissue concentrations as compared with what individuals are actually consuming) or in toxicity values may impact the risk estimates.

ES.2 CONCEPTUAL SITE MODEL AND EXPOSURE ASSESSMENT

The conceptual site model describes scenarios in which people could be exposed to COPCs from the EW in seafood tissue, sediment, or surface water. Exposure scenarios were selected for consistency with the LDW HHRA (Windward 2007c) and through input from EPA and various site users, including the Muckleshoot and Suquamish Tribes. Exposure pathways consisted of exposure through the consumption of seafood from the EW; direct contact with sediment during commercial netfishing, habitat restoration, and clam harvesting in the EW; and direct exposure to surface water while swimming in the EW.

A risk-based screening was performed using EPA guidance to identify the COPCs to be evaluated in the baseline HHRA. For the seafood consumption scenarios, a total of 54 chemicals or chemical groups (26 of which were never detected) were identified as COPCs. For the direct sediment contact scenarios, 12 chemicals or chemical groups (2 of which were never detected) were identified as COPCs. Lastly, for the surface water exposure scenarios, 15 chemicals or chemical groups (9 of which were never detected) were identified as COPCs.[1] Those COPCs that were not detected were evaluated as part of the uncertainty analysis.

Several levels of exposure were evaluated in the baseline HHRA to describe different intensities (e.g., frequency and duration) of site use or seafood consumption. These exposure levels include reasonable maximum exposure (RME) scenarios, high-end exposure scenarios, central tendency (CT) exposure scenarios, unit of exposure scenarios, and previously developed regional exposure scenarios. The following bullets describe how each was used in the risk assessment:

- **RME scenarios** – RME is the highest exposure that is reasonably expected to occur at a site. EPA generally uses RME scenarios to evaluate remedial actions at a site (EPA 1989). RME, by definition, likely overestimates exposure for many individuals. With regard to the adult tribal seafood consumption scenarios, application of EPA's tribal seafood consumption framework (EPA 2007b) has resulted in the use of Tulalip seafood consumption survey data to characterize adult tribal RME seafood consumption (although the EW is not part of the Tulalip Tribes' Usual and Accustomed [U&A] fishing area).
- **High-end scenario** – An additional tribal scenario is also evaluated based on Suquamish seafood consumption survey data. This scenario represents a high-end risk for the EW site (EPA 2005a).
- **CT scenarios** – In characterizing uncertainty in exposure and risk, it is useful to examine CT exposures (National Research Council 1994). CT risk estimates are intended to reflect average exposures. Average exposure estimates are not favored in decision-making because they will underestimate exposure for a substantial number of individuals (EPA 1989).
- **Unit of exposure scenarios** – Another method of examining exposure is to identify a unit of exposure that a member of the public can use to assess risks associated with their individual behavior. This approach was used to characterize seafood consumption exposure and direct contact exposures from clamming. The unit of exposure evaluated was one meal per month for seafood consumption and 7days per year for clamming. Rather than describing a behavior that is specific to the EW, these scenarios are intended to serve as a basis on which individuals can evaluate their own exposure using a method that is readily scaled to various seafood consumption levels or frequency of clamming (i.e., a change in the rate of consumption or clamming frequency to higher or lower amounts results in proportional change in the amount of

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[1] Detected COPCs identified for the seafood consumption, direct sediment exposure, and/or swimming scenarios were antimony, arsenic, cadmium, chromium, cobalt, copper, lead, mercury, molybdenum, selenium, vanadium, zinc, dibutyltin as ion, tributyltin as ion, cPAH TEQ, naphthalene, 1,4-dichlorobenzene, pentachlorophenol, total PCBs, PCB TEQ, alpha-benzene hexachloride (BHC), beta-BHC, total chlordane, total DDTs, dieldrin, heptachlor, heptachlor epoxide, mirex, and dioxin/furan TEQ.

chemical exposure and risk). This approach is not intended to represent a measured or established consumption rate or recreational clamming frequency for the EW. For example, older surveys on seafood consumption in Puget Sound suggest that seafood consumption by recreational anglers is greater than one meal per month (PSEP 1988).

- **Regional exposure scenarios** – Another method for evaluating risk is to use levels of exposure developed for nearby areas, as was done for estimating risks based on exposure to surface water while swimming. The three levels of exposure (high, medium, and low) that were previously developed for Elliott Bay and the Duwamish River (King County 1999a) were applied to the EW to provide a range of possible risk estimates. However, in this case, this approach likely resulted in a significant overestimation of swimming exposure levels for the EW, given that these levels of exposure were developed for areas that include larger numbers of recreational access points (e.g., Elliott Bay) than does the EW and do not have the EW's high concentration of large ship and tug boat traffic.

The exposure assessment presents the equations and parameters used to quantify exposures to COPCs in each scenario. The quantification of exposure consists of an estimate of the chemical concentrations to which people might be exposed. This estimate is calculated from the concentration data for each COPC and health-protective assumptions regarding intake rates of seafood, sediment, and surface water and the frequency and duration of the intake. When possible, exposure parameters were consistent with those in the LDW HHRA (Windward 2007c).

As with the LDW HHRA (Windward 2007c), no seafood consumption surveys that focus solely on the consumption of seafood from the EW were available for individuals (e.g., recreational anglers, tribal members, or other communities) who either currently consume seafood or may consume seafood from this resource in the future. Therefore, the rates of seafood ingestion assumed for the seafood consumption scenarios were selected by EPA based on data collected from several regional surveys (Toy et al. 1996; EPA 1999a; Suquamish Tribe 2000).[2] Seafood harvest and consumption in the vicinity of Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) sites are likely to be suppressed. Because of this, EPA Region 10 believes it is appropriate to use seafood consumption rates derived from surveys that cover areas with levels of contamination that are lower than those at CERCLA sites (EPA 2011a). Such an approach is needed to characterize risks based on reasonable future use following the remediation of chemical contamination. In evaluating risks and exposures from smaller sites (such as the EW) within larger bodies of water (such as Puget Sound),

[2] The Tulalip Tribes' survey (Toy et al. 1996), which was used to develop the tribal RME consumption scenario for the EW, did not include seafood obtained from the EW. Although the API survey (EPA 1999a) focused on King County, specific seafood harvest locations were not reported. The EW is a very small portion of the much larger fishing area that was included in the Suquamish Tribe's survey. Thus, the applicability of the overall consumption rate from these surveys to the EW alone is uncertain.

EPA Region 10 believes that using a holistic approach is appropriate and thus using consumption rates associated with larger water bodies is necessary (EPA 2011a). This will support remedial actions that result in the uniform protection of public health throughout the larger water body.

EPA Region 10 developed a framework to promote internal consistency in Puget Sound tribal seafood consumption risk assessments (EPA 2007b). In this framework, EPA selected Puget Sound tribal seafood consumption information to develop RME tribal seafood consumption rates for risk assessment for Resource Conservation and Recovery Act (RCRA) and CERCLA sites in the Puget Sound region. EPA Region 10 made the policy decision to use the quantity of current or potential high-quality shellfish habitat of a site to determine which tribal seafood consumption dataset would be the most appropriate for the site. For sites in the vicinity of large quantities of current or potential high-quality shellfish habitat, EPA advocates the use of the Suquamish Tribe's seafood consumption rate to characterize risk. In general, EPA advocates the use of the Tulalip Tribes' consumption rate in other cases.

For the EW, given the limited quantity of current or potential shellfish habitat (particularly high-quality habitat), the Tulalip Tribes' rate was selected, as approved by EPA (Windward 2010f), to characterize the RME seafood consumption risks in the EW. Inasmuch as the EW is within the U&A fishing area of the Suquamish Tribe, and the Suquamish Tribe has requested that their seafood consumption data be used to characterize risk, the EW HHRA also evaluates risk using Suquamish Tribe consumption rates. Although, as stated above, the framework supports consistency in internal EPA policy regarding tribal seafood consumption risk assessment, the recommendations of the framework (EPA 2007b) do not replace or supersede the need for consultation between EPA and the tribes to develop site-specific risk assessments. Discussions between EPA and the Suquamish Tribe did not result in tribal concurrence regarding the use of the Tulalip tribal consumption rates as the RME scenario for the EW HHRA.[3] Rather, the use of the Tulalip rates represents an EPA policy decision. However, the Muckleshoot and Suquamish Tribes recognize that sediment cleanup levels for bioaccumulative risk driver contaminants based on seafood consumption risks will likely be below background, regardless of whether Tulalip or Suquamish consumption rates are used to develop cleanup levels. For this reason, the tribes have not pursued their disagreement with EPA more vigorously regarding the selection of the Tulalip Tribes' rate to characterize RME seafood consumption risks for the EW. The tribes regard the EW seafood consumption rate decision to be site-specific and do not regard it as being precedent-setting.

Specifically, for representing RME seafood consumption by adult tribal members, EPA developed a seafood consumption scenario using survey data for adult Tulalip tribal members. This scenario (the adult tribal RME seafood consumption scenario based on

[3] The Suquamish Tribe requested that the tribal RME scenario be represented as a range of exposures based on the Tulalip and Suquamish consumption rates.

Tulalip data) includes a consumption rate of 97.5 g [4] of seafood per day (three meals per week, assuming 227-g [8-oz] meals (2000d)) based on a Tulalip tribal study on the consumption of resident species of fish and shellfish from the Puget Sound region (i.e., not including salmon). This consumption rate was assumed to be applicable to the ingestion of seafood caught in the EW and was further divided into seafood categories as follows:

- 8.1 g/day for pelagic fish (includes perch and rockfish)
- 7.5 g/day for benthic fish
- 34.4 g/day for crab
- 47.5 g/day for other shellfish (includes clams, mussels, and geoduck)

In the absence of a seafood consumption survey of tribal members that relates specifically to consumption of seafood from only the EW, it is not known whether tribal members currently consume seafood from the EW at the rates assumed or if they may do so under future conditions. There is uncertainty regarding the application of these rates to the EW, and it is likely that the current consumption rates of seafood from the EW are lower than those documented in the Tulalip tribal study because of existing seafood consumption advisories. EPA's Superfund risk assessment guidance requires that exposure estimates be protective of future uses (EPA 1989). Tribes with treaty rights to obtain seafood from the EW may increase their consumption rate in the future as conditions in the EW improve with regard to chemical contamination. Consumption rates also reflect cultural practices and traditions that differ between tribes, and future use scenarios should reflect the tribes' desire to be able to harvest resources throughout their U&A fishing areas.[5] Consequently, the seafood consumption rates evaluated for the RME scenarios in this HHRA are intended to be protective of both current and future uses.

In addition to the adult tribal seafood consumption RME scenario based on Tulalip data, risks associated with the consumption of resident fish and shellfish were also quantified for the following scenarios:[6]

- Asian and Pacific Islander (API) RME and CT scenarios (total ingestion rates of 51.6 and 5.3 g/day, respectively)
- Adult tribal CT scenario based on Tulalip data (total ingestion rate of 15.0 g/day)

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[4] Rate does not include the consumption of anadromous fish. The total consumption rate, including anadromous fish, is 194 g/day (EPA 2006a).

[5] Although the amount and type of seafood consumed from a particular area may be affected by the habitat present there, it is not assumed that the overall tribal consumption rates are affected by the EW site.

[6] For some population groups (adult and child tribal individuals based on Tulalip data and adult API), two scenarios were evaluated – one corresponding to an RME and one corresponding to a CT. Rates do not include the consumption of anadromous fish.

- Child tribal RME and CT scenarios based on Tulalip data (total ingestion rates of 39.0 and 6.0 g/day, respectively)
- Adult tribal scenario based on a Suquamish data (total ingestion rate of 583.5 g/day)
- Adult one-meal-per-month consumption scenario (total ingestion rate of 7.5 g/day)

As noted above, the ingestion rates presented here include the consumption of only resident fish (i.e., salmon are not included). The tribal seafood consumption scenario based on Suquamish data were included at the request of the Suquamish and Muckleshoot Tribes to assist in characterizing the range of potential seafood consumption risks, and because the EW is within the U&A of the Suquamish Tribe. The seafood consumption rates for the Suquamish Tribe are much higher than those for the Tulalip Tribes, with higher consumption rates across consumption categories (particularly for shellfish) as a result of cultural and locational differences. As acknowledged by EPA with regard to the LDW (EPA 2005a), the EW also lacks extensive high-quality intertidal shellfish habitat that would be necessary to sustain the higher shellfish consumption rates from the Suquamish study.

Exposure scenarios for the tribal children based on Tulalip data, adult tribal members based on Suquamish data, and API adults included a combination of the seafood categories listed above for the adult tribal RME scenario based on Tulalip data (i.e., a market basket approach). For the adult one-meal-per-month scenario, risks were evaluated based on the consumption of one meal per month of pelagic fish (perch or rockfish), benthic fish fillets, crab edible meat, or clams. Consistent with EPA risk assessment guidance, all assumptions regarding the amounts of seafood ingested in the RME scenarios were selected to be health-protective to avoid underestimating risks. Consequently, individual risk estimates may be overestimates but are unlikely to be underestimates for most chemicals.

The direct sediment exposure scenarios evaluated in this HHRA include netfishing, habitat restoration worker, and clamming scenarios. As in the LDW HHRA (Windward 2007c), exposure frequency and duration assumptions for the evaluation of direct sediment exposure under the commercial netfishing scenario were based on site use information collected from the Muckleshoot Indian Tribe, which conducts commercial netfishing for adult salmon in the Duwamish River, including the EW. Exposure parameter values for the clam harvesting scenarios were also consistent with the LDW HHRA (Windward 2007c) and based primarily on direction from EPA (e.g., 2007c), default EPA exposure parameters (e.g., 1997a), and best professional judgment where site-specific data on exposure frequency and duration were not available. The exposure parameter values for the habitat restoration worker scenario were slightly different from those in the LDW HHRA (Windward 2007c) and were based on default EPA exposure parameters and best professional judgment for frequency and duration assumptions.

Exposure to surface water in the EW was assessed for a swimming scenario, for which the exposure parameters were generally based on the adult swimming scenarios presented in the *King County Combined Sewer Overflow Water Quality Assessment for the Duwamish River and Elliott Bay* (King County 1999a). As was done in the King County assessment, three levels of exposure (high, medium, and low) were evaluated. However, none of these were designated as an RME for the EW. These levels of exposure are likely significant overestimates of swimming exposure levels for the EW, given that they were developed for areas that include a larger number of recreational access points (e.g., Elliott Bay) than does the EW and do not have the EW's high concentration of large ship and tug boat traffic.

Exposure point concentrations (EPCs) are the concentrations of COPCs in seafood tissue, sediment, and surface water collected from the EW that were used in the exposure equations to calculate COPC intake. The EPC was either the 95% upper confidence limit on the mean (95% UCL) if there were sufficient detected values in the dataset[7] or the maximum concentration of a COPC and was intended to represent a long-term exposure concentration. In some cases, the EPC was set equal to one-half the maximum RL if this value was higher than the maximum detected concentration or there were no detected concentrations. EPCs for the direct sediment exposure scenarios (i.e., netfishing, habitat restoration, and clam harvesting) were calculated for the sediment area over which the exposure could potentially occur.

EPCs for the seafood consumption scenarios were calculated separately for various types of seafood, referred to as consumption categories. Ten consumption categories were developed based on seafood tissue types available in the EW: fillets of benthic fish, whole bodies of benthic fish, perch (both fillets and whole body), whole bodies of rockfish, edible meat of crab, whole bodies of crab, clams, edible meat of geoduck, whole bodies of geoduck, and mussels. In some cases, chemistry data for more than one species were combined within a single consumption category (e.g., edible meat for Dungeness crab and red rock crab were combined in the crab edible meat category). A COPC intake rate was then calculated for each consumption category using the COPC tissue dataset and the consumption rate for each category. The chemical intakes for each consumption category were then summed within each seafood consumption scenario (except the adult one-meal-per-month scenario) to yield an overall COPC intake for that scenario.

The netfishing scenario assumed that people who engage in commercial netfishing could be exposed to both intertidal and subtidal sediment that might adhere to their nets. For both the habitat restoration and clam harvesting scenarios, it was assumed that individuals would only come into contact with intertidal sediment. The intertidal area was further divided such that all accessible (i.e., not under piers and accessible by boat

[7] Data management rules for calculating EPCs, as presented in Section B.3.3.4, considered the detection frequency and the number of samples. When EPCs were based on a UCL on the mean, the 95th percentile or higher UCL on the mean was selected, as recommended by the ProUCL software.

or shoreline access) intertidal sediment was used to assess risks for the habitat restoration worker and tribal clamming scenarios, while only the intertidal area sediment to which the general public has access was used to assess risks to the 7-day-per-year clamming scenario (which is intended to be more representative of a recreational clammer than is the tribal clamming scenario). For the swimming scenario, EPCs were calculated assuming site-wide exposure to chemicals in EW surface water.

ES.3 TOXICITY ASSESSMENT

EPA toxicity values (i.e., slope factors [SFs] for evaluation of carcinogenic risks or reference doses [RfDs] for evaluation of effects other than cancer) were identified for all COPCs. Toxicity values for each COPC have been established by EPA and other agencies and are based on either laboratory experiments that used animals or epidemiological studies of human populations who were unintentionally exposed in the workplace or in the environment. The SFs provide a health-protective means to evaluate risks because they represent upper-bound estimates of carcinogenic potency. Similarly, non-cancer toxicity values (i.e., RfDs) are health-protective because they are typically based on the most sensitive endpoint and population or test organism for which adequate data are available and include uncertainty factors or extrapolations to account for sensitive sub-populations or other limitations of the toxicity study data on which they are based.

ES.4 RISK CHARACTERIZATION AND UNCERTAINTY ANALYSIS

Carcinogenic risks and non-carcinogenic health effects were evaluated separately in the HHRA because of fundamental differences in assumptions about the mechanism of these toxic effects (EPA 1989). Carcinogenic risk estimates were calculated by multiplying the estimated chemical intake by the SF. Cancer risk estimates were compared with EPA's acceptable risk range of 10^{-6} to 10^{-4} established in the National Contingency Plan for Superfund sites (40 CFR 300). The lifetime risk of developing cancer in the US population is one in two (i.e., 5×10^{-1}) for men and one in three (i.e., 3×10^{-1}) for women (American Cancer Society 2006). A 1×10^{-6} excess cancer risk[8] represents an additional one-in-one-million probability that an individual may develop cancer over a 70-year lifetime as a result of exposure to chemicals in EW sediment and surface water (either through direct exposure or indirect exposure through the consumption of seafood).

Chemicals with non-carcinogenic health effects are generally not toxic below a certain threshold; a critical chemical dose must be exceeded before adverse health effects are observed. The potential for non-carcinogenic health effects is represented by the ratio of the estimated chemical intake to the critical chemical dose (an RfD), and is expressed as a hazard quotient (HQ). Exposures resulting in an HQ less than or equal to 1 are

[8] Excess cancer risk is defined as the additional probability (i.e., the probability above the lifetime cancer risk) of an individual developing cancer based on exposure to contaminants in the EW.

unlikely to result in non-cancer adverse health effects. Chemicals that affect the same organ or physiological function (called "toxicity endpoints") may have additive effects. For those chemicals, the HQs for the same endpoint may be summed as a hazard index (HI).

Concentrations of hazardous substances that arise from natural or anthropogenic background conditions unrelated to specific EW contaminant sources, may contribute to contaminant concentrations in EW sediment, tissue, and surface water and therefore represent a portion of the calculated risks. Background data are discussed in this HHRA, as allowed by CERCLA. However, this HHRA does not provide evaluations of background data, including the selection of appropriate datasets and their statistical analysis, for the purpose of selecting cleanup levels under CERCLA. Where evaluations of background data are presented in this HHRA, they are intended to provide only additional information relevant to exposure and risk estimates. Additional evaluations of background data to support CERCLA determinations of cleanup levels are provided, or will be provided, in the SRI and/or FS reports.

ES.4.1 Seafood consumption scenarios

Estimated excess cancer risks were highest for the seafood consumption scenarios (Table ES-1). The cumulative risk for all carcinogenic chemicals was 1×10^{-3} for the adult tribal RME seafood consumption scenario based on Tulalip data. Polychlorinated biphenyls (PCBs) were identified as the primary contributor, with excess cancer risks estimates equal to 1×10^{-3}. Other chemicals with excess cancer risks greater than 1×10^{-6} that contributed more than 5% to the total excess cancer risks were carcinogenic polycyclic aromatic hydrocarbon (cPAH) toxic equivalent (TEQ)[9] (1×10^{-4}), inorganic arsenic (2×10^{-4}), and dioxin/furan TEQ (1×10^{-4}). The relative contribution to the total excess cancer risk was generally similar for other scenarios, with these four chemicals together contributing 95% or more of the total excess cancer risk for all scenarios. Total excess cancer risks for the two other RME scenarios were lower than those for the adult tribal RME scenario based on Tulalip data, equal to 4×10^{-4} and 6×10^{-4} for the child tribal RME scenario based on Tulalip data and the adult API RME scenario, respectively. The risks for the adult tribal scenario based on Suquamish data were approximately 5 times higher than risks for the adult tribal RME scenario based on Tulalip data (the total excess cancer risk was equal to 1×10^{-2} for the adult tribal scenario based on Suquamish data), reflecting the much higher seafood consumption rate (almost three meals per day) used in the adult tribal scenario based on Suquamish data.

[9] TEQs were used for totaling certain groups of chemicals (cPAHs, PCBs, and dioxins/furans) relative to the most toxic component of the group: benzo(a)pyrene for cPAHs and 2,3,7,8- tetrachlorodibenzo-*p*-dioxin (TCDD) for dioxin-like PCBs and for dioxins/furans.

Table ES-1. Summary of seafood consumption scenario parameters and risks

Scenario	Ingestion Rate (g/day)								Meals per Month[d]	Exposure Frequency (days/yr)	Exposure Duration (years)	Body Weight (kg)	Total Excess Cancer Risk[e]	Maximum Non-Cancer HI[f]
	Benthic Fish[a]	Crab[b]	Mussel	Clam	Geoduck[b]	Perch[c]	Rockfish[c]	Total						
Adult tribal RME (Tulalip data)	7.5	34.4	0.8	39.3	7.4	7.1	1.0	97.5	13.1	365	70	81.8	1×10^{-3}	28
Adult tribal CT (Tulalip data)	1.2	5.3	0.1	6.0	1.1	1.1	0.2	15	2.0	365	30	81.8	7×10^{-5}	3
Child tribal RME (Tulalip data)	3.0	13.7	0.3	15.7	3.0	2.8	0.4	39.0	13.1	365	6	15.2	4×10^{-4}	59
Child tribal CT (Tulalip data)	0.48	2.1	0.04	2.4	0.44	0.44	0.08	6.0	2.0	365	6	15.2	4×10^{-5}	6
Adult tribal (Suquamish data)	25.9	49.8	5.0	393.7	49.8	0.6	55.4	583.5	78	365	70	79	1×10^{-2}	219
Adult API RME	2.4	10.6	4.6	29.1	–	0.5	4.4	51.5	6.9	365	30	63	6×10^{-4}	25
Adult API CT	0.24	1.1	0.5	3.0	–	0.05	0.45	5.3	0.7	365	9	63	1×10^{-5}	1
Adult one-meal-per-month[g]														
benthic fish	7.5	–	–	–	–	–	–	7.5	1.0	365	30	71.8	2×10^{-4}	13
clam	–	–	–	7.5	–	–	–	7.5	1.0	365	30	71.8	3×10^{-5}	0.5
crab	–	7.5	–	–	–	–	–	7.5	1.0	365	30	71.8	2×10^{-5}	0.9
pelagic fish, rockfish	–	–	–	–	–	–	7.5	7.5	1.0	365	30	71.8	4×10^{-4}	21
pelagic fish, perch	–	–	–	–	–	7.5	–	7.5	1.0	365	30	71.8	1×10^{-4}	8

[a] Includes both fillet and whole-body consumption.

[b] Includes both edible-meat and whole-body consumption.

[c] Both perch (fillet and whole body) and rockfish (whole body) were classified as pelagic fish in this risk assessment.

[d] It was assumed that one adult meal was equal to 227g (8 ounces). Child consumption rates were based on 40% of adult rates (EPA 2007b). For the purpose of calculating meals per month for children, this 40% conversion is assumed to represent a smaller meal size (40% of adults, which is equal to 91 g or 3.2 ounces).

[e] Total excess cancer risk is the higher of the two sums (i.e., either excluding PCB TEQ or total PCBs).

[f] The sum of non-cancer HQs across all chemicals is not directly interpretable for risk assessment because some hazard quotients may relate to different toxic effects (i.e., endpoints) that are not additive. Thus, the maximum non-cancer HI for any endpoint is presented here. For all scenarios, this maximum is for either the immunological endpoint, neurological endpoint, or integumentary endpoint, all of which include total PCBs in the sum.

[g] The adult one-meal-per-month scenarios are presented for informational purposes only, and are not used by EPA for risk management decisions.

API – Asian and Pacific Islander
CT – central tendency
EPA – US Environmental Protection Agency
HI – hazard index (a sum of the HQs for individual chemicals)
RME – reasonable maximum exposure
TEQ – toxic equivalent

Excess cancer risks from PCBs were calculated in two ways: as the sum of detected Aroclors (referred to herein as total PCBs) and as a TEQ.[10] In general, the risk from total PCBs calculated as a sum of detected Aroclors was equal to or up to approximately two times higher than the risk calculated from the PCB TEQ. Because of this difference, the total risk (i.e., the sum of individual chemical risk estimates for each exposure scenario) was calculated two ways, first by including total PCBs and excluding PCB TEQ and then by including PCB TEQ and excluding total PCBs. The higher of these two summation approaches for the total excess cancer risk is presented in Table ES-1. The total TEQ risk (i.e., sum of PCB TEQ and dioxin/furan TEQ) was also calculated for each scenario because of the shared mode of toxicity of the two chemical groups. For all seafood consumption scenarios, PCB TEQ contributed the majority (over 63%) to the total TEQ risk.

In the evaluation of non-cancer hazards, total PCB HQs were greater than 1 for all three RME scenarios, and the cadmium HQ was greater than 1 for the child tribal RME scenario based on Tulalip data, indicating some potential for adverse effects other than cancer. For the non-RME scenarios, total PCB HQs were greater than 1 for all scenarios, except the adult API CT scenario and the crab and clam adult one-meal-per-month scenarios. In addition to total PCBs, arsenic, cadmium, cobalt, mercury, and tributyltin all had HQs greater than 1 for the adult tribal scenario based on Suquamish data. Different chemicals can have different toxic effects (e.g., may adversely affect different endpoints such as immunological, kidney, liver, or neurological) that are not additive. Thus, Table ES-1 presents the maximum non-cancer HI for any endpoint under each exposure scenario, rather than a sum of HQs across all COPCs. The maximum HIs for all scenarios were either for the immunological endpoint, neurological endpoint, or integumentary endpoint. Total PCBs accounted for over 80% of these non-cancer HIs.

Risk estimates presented in this HHRA indicate that elevated risks result from exposures to a small number of chemicals, as demonstrated by the distribution of total excess cancer risks by chemical in Figure ES-1 for all seafood consumption scenarios. Although some variability exists regarding their percent contribution, arsenic, cPAH TEQ, total PCBs, and dioxin/furan TEQ together contribute 95% or more of the total excess cancer risk for the seafood consumption scenarios.

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[10] PCB TEQ is calculated using toxic equivalency factors, which relate the toxicity of the co-planar PCB congeners (i.e., those with dioxin-like properties) to the toxicity of 2,3,7,8-TCDD.

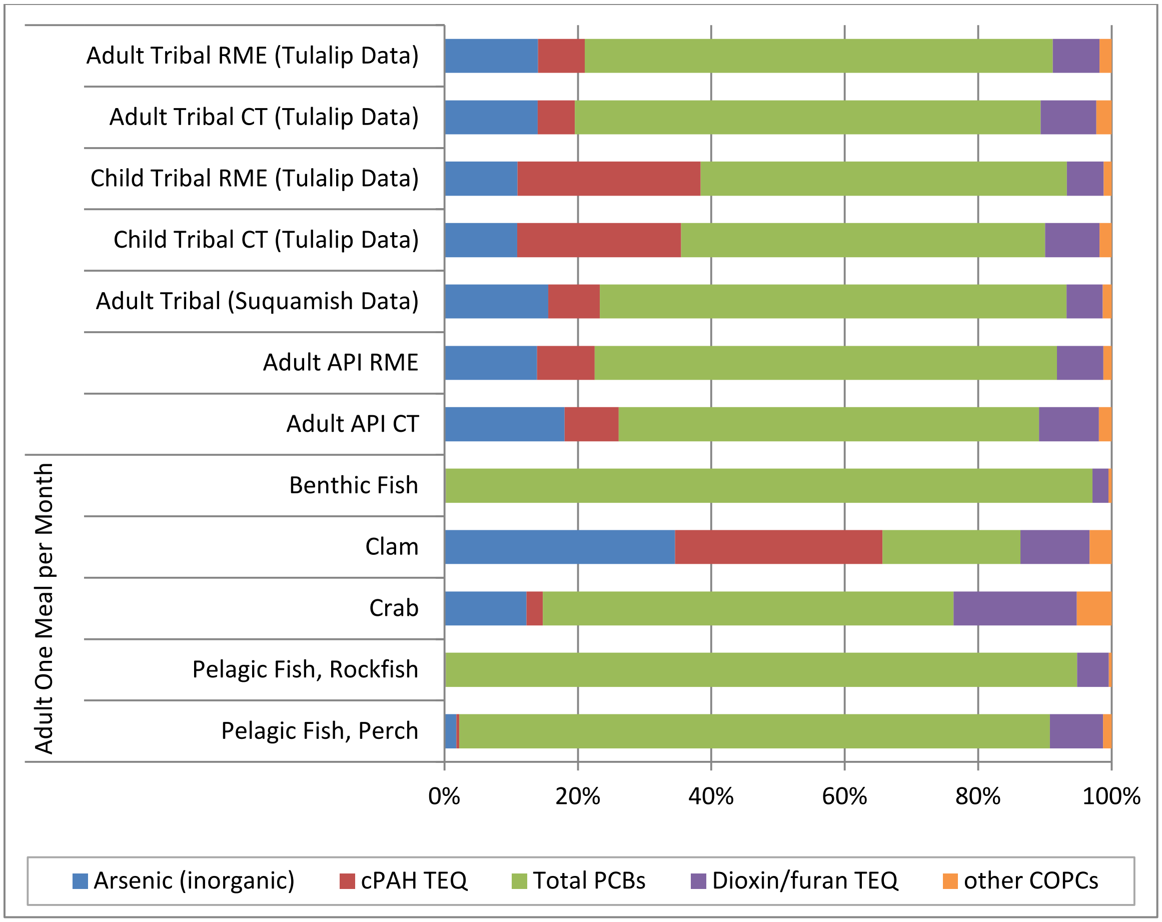

Figure ES-1. Percent contribution of COPCs to the total excess cancer risk for the seafood consumption scenarios

Of these four chemicals, tissue concentrations of arsenic in the EW were similar to those from background areas in Puget Sound. This comparison to background tissue concentrations is preliminary and will be revisited as part of the RI and/or FS, along with evaluations of background concentrations for other COCs.

It is also helpful to consider the risk estimates in terms of the proportions of the risk estimates related to the consumption of individual seafood types. Elevated risk estimates associated with inorganic arsenic and cPAH TEQ in seafood are largely attributed to clams for all adult seafood consumption scenarios (Figure ES-2). In contrast, the seafood consumption categories that contribute the majority of the risk for the other risk drivers (PCBs and dioxin/furan TEQ) are more variable by scenario. For PCBs, risks are primarily attributable to benthic fish fillet, perch, and/or rockfish. For dioxin/furan TEQ, risks are primarily attributable to clams, crab (both edible meat and whole body), and/or rockfish. It should be noted that although Figure ES-2 shows these proportions for only three of the adult seafood consumption scenarios evaluated in the HHRA, these figures capture the variability across all tribal and API consumption scenarios evaluated in this HHRA.

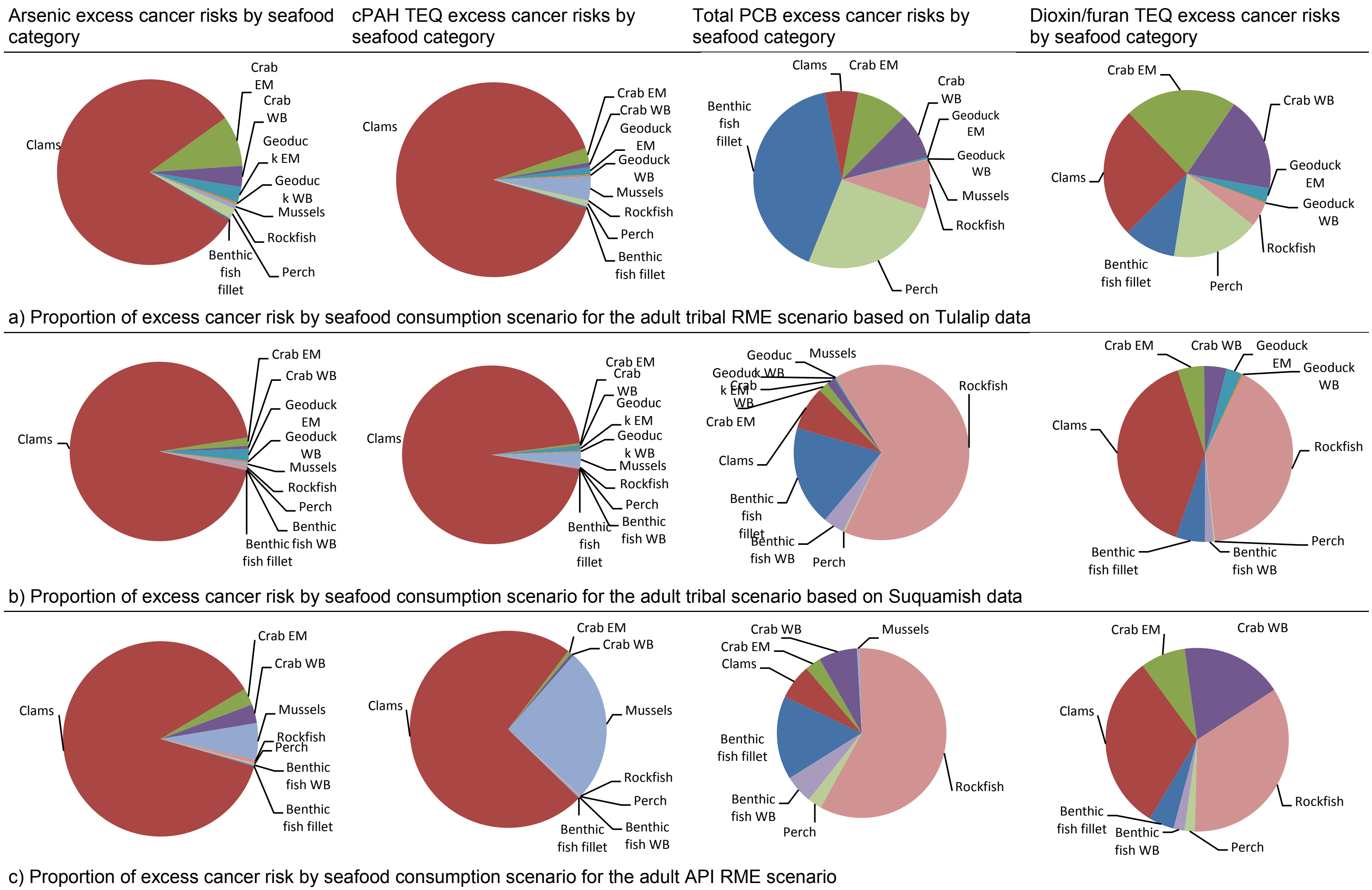

Figure ES-2. Proportion of cancer risks by seafood category for the adult seafood consumption scenarios

ES.4.2 Direct sediment exposure scenarios

Excess cancer risks for the direct sediment exposure scenarios were much lower than those for the seafood consumption scenarios (Table ES-2). Excess cancer risks for all scenarios were less than the upper end of EPA's risk range (1×10^{-4}). Risks were equal to 7×10^{-6} for the netfishing RME scenario and equal to 3×10^{-5} for the tribal clamming RME scenario. Risks were greater than the excess cancer risk threshold of 1×10^{-6} for arsenic (3×10^{-6}) and cPAH TEQ (3×10^{-6}) for the netfishing RME scenario and for arsenic (1×10^{-5}), cPAH TEQ (2×10^{-5}), and total PCBs (3×10^{-6}) for the tribal clamming RME scenario. In addition, for the tribal clamming RME scenario, the excess cancer risk for the total TEQ sum (PCB TEQ and dioxin/furan TEQ) was equal to 2×10^{-6}, although neither the PCB TEQ nor dioxin/furan TEQ risks independently were greater than 1×10^{-6}.

No chemicals had excess cancer risks greater than 1×10^{-6} for the netfishing CT scenario, clamming CT scenario, habitat restoration worker scenario, or 7-day-per-year clamming scenario. The total excess cancer risk for the tribal clamming 183-days-per-year scenario was greater than 1×10^{-6} (equal to 6×10^{-5}), with arsenic, cPAH TEQ, total PCB, PCB TEQ, and dioxin/furan TEQ risks all greater than the 1×10^{-6} excess cancer risk threshold. Non-cancer hazards are not expected for direct sediment exposures; no chemicals had HQs greater than 1 for any direct contact scenario.

Table ES-2. Summary of direct sediment exposure scenario parameters and risks

Scenario	Exposure Area	Age Class	Incidental Sediment IR (g/day)	Exposure Frequency (days/yr)	Exposure Duration (years)	Skin Surface Area Exposed (cm^2)	Body Weight (kg)	Total Excess Cancer Risk[a]
Netfishing RME	all subtidal and intertidal areas	adult	0.050	119	44	3,600[b]	81.8	7×10^{-6}
Netfishing CT	all subtidal and intertidal areas	adult	0.050	63	29	3,600[b]	81.8	1×10^{-6}
Habitat restoration worker	all accessible intertidal area	adult	0.1	15	20	6,040[c]	71.8	1×10^{-6}
Tribal clamming RME	all accessible intertidal area	adult	0.1	120	64	6,040[c]	81.8	3×10^{-5}
Tribal clamming 183-day-per-year	all accessible intertidal area	adult	0.1	183	70	6,040[c]	81.8	6×10^{-5}
Clamming 7-day-per-year	public-access intertidal area only	adult	0.1	7	30	6,040[c]	71.8	1×10^{-6}

Note: Non-cancer HQs did not exceed 1 for any chemical and are therefore not shown in this table.

a Total excess cancer risk is the higher of the two sums presented in Section B.5 (excluding PCB TEQ or total PCBs).

b Recommended skin surface area value for commercial/industrial worker. Assumes that head, hands, and forearms are uncovered (i.e., exposed).

c Assumes that 39% of the total adult body surface area is exposed.

CT – central tendency
HQ – hazard quotient
IR – ingestion rate
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent

As with the seafood consumption scenarios, the percent contribution of each COPC to the risk estimates was examined for the direct sediment exposure scenarios with total excess cancer risks greater than 1 × 10^{-6}. Cancer risks were highest for arsenic and cPAH TEQ, which together accounted for over 84% of the total excess cancer risk (Figure ES-3). PCBs and dioxin/furan TEQ were lesser contributors. Note that this figure includes all COPCs with greater than 0.1% contribution to the total excess cancer risk; however, not all these COPCs had excess cancer risk estimates greater than 1 × 10^{-6} for the RME scenarios, and hence those COPCs were not identified as chemicals of concern (COCs) for the indicated scenarios.

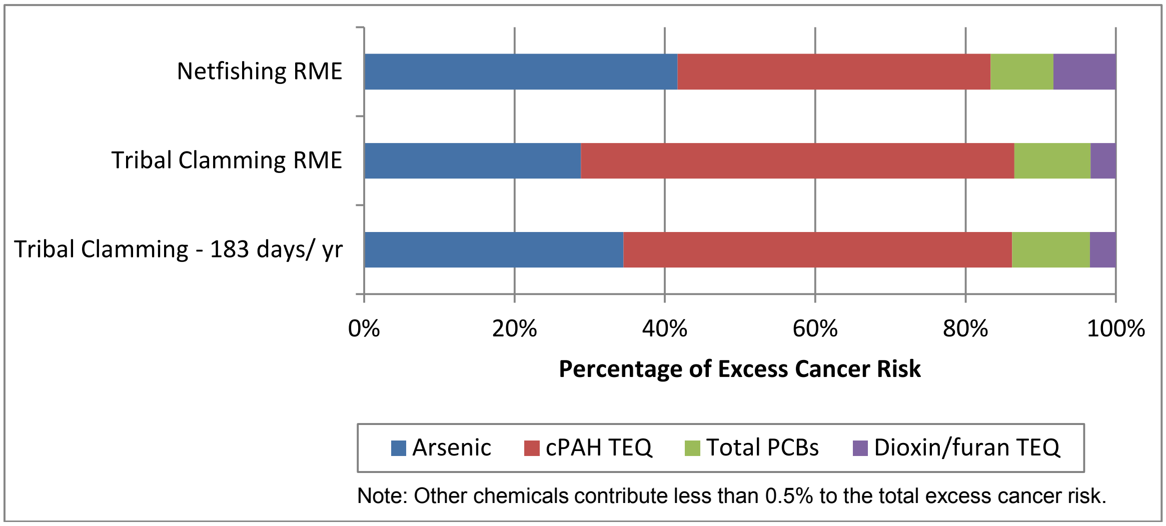

Figure ES-3. Percent contribution of COPCs to the total excess cancer risk for direct sediment exposure scenarios with total excess cancer risks greater than 1 × 10^{-6}

ES.4.3 Surface water exposure scenarios

In addition to the seafood consumption and direct sediment contact scenarios, three levels of swimming exposure were evaluated to assess risks based on exposure to surface water in the EW.[11] The only excess cancer risks that were greater than the 1 × 10^{-6} threshold were for PCB TEQ for both the high level of exposure (which assumed 2.4 hours of swimming, 24 days per year) and the medium level of exposure (which assumed 1 hour of swimming, 12 days per year)(equal to 9 × 10^{-6} and 2 × 10^{-6}, respectively). The total excess cancer risks (which includes all COPCs) for this scenario were also equal to 9 × 10^{-6} and 2 × 10^{-6}, respectively. No other COPCs (including total PCBs) had excess cancer risks greater than 1 × 10^{-6} or non-cancer HQs greater than 1 for any COPC-exposure level combination.

[11] The three levels of exposure evaluated for swimming were high (which assumed a 2.6-hour swim, 24 days per year for 70 years), medium (which assumed a 1-hour swim, 12 days per year for 30 years), and low (which assumed a 10-minute swim [0.17 hours], 2 days per year for 9 years).

ES.4.4 Uncertainties associated with risk estimates

There are uncertainties associated with the risk estimates for each exposure scenario in this HHRA. For example, the RME exposure assumptions were developed to result in high-end estimates of the risks associated with the EW. To be health-protective of potentially exposed populations, these risk estimates are intended to not underestimate risk and thus are likely to overestimate risks for most individuals for the chemicals that were evaluated.

Risk estimates were highest for the seafood consumption scenarios, but the uncertainties associated with those risk estimates are also very high. The tribal and API seafood consumption rates that were used in this HHRA, although based on well-designed consumption surveys, were not specific to populations who primarily fish in the EW. Although the collection and consumption of seafood from the EW are known to occur (a creel study by King County identified the Spokane Street Bridge on the EW as one of the more popular fishing locations along the shores of the Duwamish River and Elliott Bay (1999a)), it is uncertain how well they represent the behavior of people who eat fish and shellfish primarily from the EW, either now or in the future. All of the available recreational angler surveys have methodological and interpretation uncertainties that create difficulty in making conclusive observations about recreational seafood consumption. However, it is possible that recreational consumption rates for some anglers could be as high as the Tulalip Tribes' consumption rates. Given the lack of EW-specific seafood consumption rate estimates, the risk per unit consumption for various seafood categories may be used by individual seafood consumers to better understand their risks.

Another important uncertainty is in the methods used to characterize the cancer risks associated with exposures to PCBs, which are a group of chlorinated organic compounds with similar chemical properties. Two methods were used in this HHRA to assess risks associated with this group of chemicals:

- **Arithmetic sum of PCBs –** Exposures to total PCBs based on the arithmetic sum of Aroclors were evaluated using the cancer SF provided by EPA for total PCBs.
- **Toxicity-weighted sum of dioxin-like PCBs –** Data for PCB congeners that are thought to have toxic effects similar to those of dioxins/furans were weighted based on their toxicity relative to dioxins/furans. This weighed sum is called PCB TEQ. PCB TEQ exposures were evaluated using the cancer SF for dioxins/furans.

Because total PCB risk estimation methodology includes, to some degree, the risks posed by dioxin-like PCB congeners, the cancer risk estimates from these two methods were not summed in estimating cumulative risks in order to avoid double-counting cancer risks posed by dioxin-like PCBs. Hence, the risk estimates for the two methods are presented separately in this baseline HHRA. Although this approach avoids the double-counting of dioxin-like PCB cancer risks, it is possible that each method for quantifying PCB cancer risks on its own underestimates the overall PCB health risk. For example, differential bioaccumulation of more highly toxic PCB congeners in environmental mixtures relative to industrial Aroclor mixtures may lead to higher risks than those computed using total Aroclors or dioxin TEQ risks individually. The issues associated with assessing risks posed by environmental PCB mixtures, various approaches for addressing double-counting, and quantitative risk estimates derived using these approaches are discussed in the uncertainty analysis.

Additional uncertainties are associated with the chemistry data, exposure assumptions, and toxicities of the COPCs. Taking into account the uncertainties, the assessment tended to overestimate risks more than underestimate them, consistent with the health-protective nature of risk assessment. Thus, despite the uncertainties, the baseline characterization of RME risks for the EW is considered to be health-protective and sufficient to support risk management decisions.

ES.4.5 Chemicals of concern and risk drivers

Another purpose of this HHRA is to identify COCs and ultimately to identify risk drivers, which are the focus of the FS. The first step was to identify COCs, which are defined as COPCs with excess cancer risk estimates greater than 1×10^{-6} or an HQ greater than 1 for any RME exposure scenario. Next, risk drivers were identified from the COC list based on several considerations, including: 1) risk magnitude relative to acceptable risk thresholds (including a consideration of background concentrations, if applicable), 2) percent contribution to the total risk estimate, 3) detection frequency, and 4) other data quality or uncertainty considerations. Table ES-3 summarizes the COCs and risk drivers for the seafood consumption and direct sediment exposure scenarios. It should be noted that no RME level of exposure was defined for the swimming scenario, and thus no COCs or risk drivers were identified for that scenario.

Table ES-3. Summary of COCs and risk drivers based on RME scenarios

Scenario Type	COCs	Risk Drivers
Seafood consumption RME scenarios	arsenic, cadmium, cPAH TEQ, pentachlorophenol, total PCBs, PCB TEQ,[a] alpha-BHC, dieldrin, total chlordane, heptachlor epoxide, mirex, and dioxin/furan TEQ[a]	cPAH TEQ, PCBs,[a, b] and dioxin/furan TEQ[a]
Direct sediment exposure RME scenarios	arsenic, cPAH TEQ, total PCBs, and total TEQ[c]	arsenic and cPAH TEQ

a Chemical lists for COCs and risk drivers do not include total TEQ (i.e., the sum of PCB TEQ and dioxin/furan TEQ) because the two components of the sum qualified individually as COCs or risk drivers.

b Consideration of PCBs as a risk driver is intended to account for both total PCBs and PCB TEQ. It should be noted that risks for total PCBs were higher than those for PCB TEQ for all scenarios.

c For the direct sediment exposure clamming scenario, total TEQ is listed because neither PCB TEQ nor dioxin/furan TEQ independently qualified as a COC, yet the total TEQ did qualify as a COC. \

BHC –benzene hexachloride

COC – chemical of concern

cPAH – carcinogenic polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

RME – reasonable maximum exposure

TEQ – toxic equivalent

A subset of the COCs were identified as risk drivers (Table ES-3), as summarized briefly in the following bullets:

- **Seafood consumption scenarios –** Of the 12 chemicals or chemical groups that were identified as COCs, 3 were identified as risk drivers (PCBs,[12] cPAH TEQ, and dioxin/furan TEQ).
- **Direct sediment exposure scenarios** – Of the four chemicals or chemical groups that were identified as COCs, two were identified as risk drivers (arsenic and cPAH TEQ). All COCs, including those identified in the ERA (Appendix A to the SRI report) will be mapped and discussed in the RI. Risk drivers will be the focus of remedial alternatives analyses in the FS. In addition, in consultation with EPA and consistent with the evaluation of non-risk drivers in the LDW, COCs not selected as risk drivers in this HHRA will be evaluated qualitatively in the EW FS. This evaluation will include a follow-up check for the non-risk driver COCs to ensure that sediment with elevated levels of these COCs will be included in the remedial footprint of the remedial alternatives evaluated in the FS. Furthermore, all COCs will be included in the long-term monitoring plan for the EW.

[12] As indicated in Table ES-3, the consideration of PCBs as a risk driver is intended to account for both total PCBs and PCB TEQ. It should be noted that risks for total PCBs were higher than those for PCB TEQ for all scenarios.

B.1 Introduction

This document presents the baseline human health risk assessment (HHRA) that has been completed as part of the supplemental remedial investigation and feasibility study (SRI/FS) for the East Waterway (EW). The EW is an operable unit of the Harbor Island Superfund site, which was added to the US Environmental Protection Agency's (EPA's) National Priorities List (NPL) under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), also known as Superfund.

As described in EPA's Superfund regulations (1988a), EPA requires that an RI/FS be conducted for each site listed on the NPL. An RI evaluates the nature and extent of chemical contamination, estimates baseline human health and ecological risks under existing conditions, and is used by risk managers to identify areas that should be remediated because they pose an unacceptable risk to human health or the environment. An FS proposes alternative approaches to remediating the areas with unacceptable risk and analyzes and compares these alternatives. A cleanup plan is then established by EPA in a Record of Decision.

Under the oversight of EPA, the EW SRI/FS is being conducted by the East Waterway Group (EWG), which consists of the Port of Seattle, the City of Seattle, and King County. The Port of Seattle signed the Agreement and Order of Consent with EPA in October 2006, and subsequently signed a Memorandum of Agreement with the City of Seattle and King County to conduct the SRI/FS. To the extent possible, this HHRA is consistent with the approach and methods that were approved by EPA for use in the HHRA for the Lower Duwamish Waterway (LDW) (Windward 2007c), which is a Superfund site located upstream of and contiguous with the EW and has many physical and functional characteristics similar to those of the EW. However, there are also important differences between these waterways (e.g., the EW has less intertidal area, more deep-water habitat, fewer public access areas, and more large-vessel traffic as compared with the LDW), which resulted in some differences between the LDW and EW HHRA exposure scenarios.

The draft HHRA has been written in accordance with the HHRA technical memorandum (Windward 2010f), which provided much of the approach and technical basis for this HHRA and was developed in conjunction with EPA to be consistent with the LDW HHRA. Baseline risk assessments, as defined by EPA RI/FS guidance (1988a), "provide an evaluation of the potential threat to human health and the environment in the absence of any remedial action. They provide the basis for determining whether or not remedial action is necessary and the justification for performing remedial actions." The baseline HHRA presents risk estimates for various scenarios under which people may be exposed to chemicals of potential concern (COPCs) found in relevant environmental media of the EW, which are sediment, surface water, and fish and shellfish tissues. Because knowledge of current and future site use is imperfect, the scenarios evaluated in this assessment have been selected to be health protective in order to not underestimate risks, and, therefore, may overestimate actual risks for many site users. As a health-protective approach, risks are assessed using a reasonable maximum exposure (RME), which is intended to estimate risks and hazards in the upper range of those possible. Such risks and hazards are intentionally higher than those that might be experienced by most people that use the EW. Remedial actions based on average exposures would leave a substantial number of individuals potentially impacted by contamination (EPA 1989).

The conceptual site model (CSM)[13] for the EW was developed in the *Final Conceptual Site Model and Data Gaps Analysis Report* (Anchor et al. 2008) and provides the basis for the development of exposure scenarios in this HHRA. The CSM is detailed further in Section B.3.1 in the development of exposure scenarios. The majority of the EW is maintained as a federal navigation channel. The EW shoreline is highly developed and primarily composed of over-water piers (aprons), riprap slopes, constructed seawalls, and bulkheads for industrial and commercial use (Map B.1-1). Throughout the entire length of the EW, approximately 61% of the shoreline consists of over-water piers above riprap and/or sheetpile (Map B.1-2). Another 30% consists of exposed shoreline armored with riprap (including the entire area south of the Spokane Street corridor). The exposed riprap portion includes a small waterfront access area (Jack Perry Memorial Shoreline Public Access), which is covered with riprap and cobble. The remaining 9% consists of steel sheetpile bulkheads. There are no residential neighborhoods within a few blocks of or directly along the EW (the closest residential neighborhood is approximately a half-mile away from the EW) (Map B.1-1). In addition to industrial activities, the EW supports the collection of seafood (fishing, crabbing, shrimping, and clamming) by tribal members and others (e.g., recreational fishers or individuals collecting seafood to supplement their diet). It should be noted that tribal rights include the collection of fish, clam, crab, or shrimp from the EW. Because of the industrial nature of the waterway, there are currently a limited number of public access points along the shoreline, no actual beaches, and limited intertidal areas (Windward 2010f). However, the Spokane Street Bridge, which is located at the head of the EW, includes a public fishing pier (former transportation bridge) that is popular for fishing and crabbing. The other main locations at which the public can access the EW from land include one designated public access area (the Jack Perry Memorial Shoreline Public Access) located near the mouth of the waterway and the street end under bridge areas of the Spokane Street corridor.

The baseline HHRA has been developed in accordance with both national and regional EPA guidance (EPA 1989, 1991a, 1996a, 2001c, 1999c, 2004b, 2005d). The results of this baseline HHRA, along with the results of the EW baseline ecological risk assessment (ERA) and the EW SRI, will be considered when identifying cleanup levels in the EW FS. In addition, it should be noted that the LDW RI (Windward 2010g) and FS (in progress) will be used to inform the cleanup plan for the EW. The risk estimates for the

[13] The CSM is a graphical representation of exposure media, transport mechanisms, exposure pathways, exposure routes, and potentially exposed human populations.

selected receptors and exposure scenarios presented in the risk assessments will be used in the SRI to calculate risk-based threshold concentrations (RBTCs) for chemicals of concern (COCs) that were identified as risk drivers. RBTCs are chemical concentrations in sediment or tissue associated with acceptable risk thresholds for specific human or ecological exposure scenarios.

As discussed above, this baseline HHRA identifies risks associated with a wide variety of human exposure scenarios in the EW (e.g., several different seafood consumption scenarios, netfishing, clam digging, habitat restoration, swimming). The breadth of the human exposure scenarios evaluated informs risk managers and other interested parties of the potential range of health risks to humans who might be exposed to contaminated media from the EW on a regular basis.

This baseline HHRA includes the following sections:

Section B.2 – Data Evaluation

Section B.3 – Exposure Assessment

Section B.4 – Toxicity Assessment

Section B.5 – Risk Characterization

Section B.6 – Uncertainty Analysis

Section B.7 – Identification of Risk Drivers

Section B.8 – Conclusions

Section B.9 – References

Details on site background, previous investigations, and the environmental setting of the EW have been provided in the work plan (Anchor and Windward 2007) and the CSM and data gaps analysis report (Anchor et al. 2008). These reports are referenced in Section B.3, which discusses the development of human health exposure scenarios for the EW.

B.2 Data Evaluation

Recent chemistry data are available for various media (sediment, water, and fish and shellfish tissue) collected from the EW. These data and other kinds of data (e.g., site access surveys) are used to evaluate risks to people who may be exposed to various media from the EW. Figure B.2-1 presents the various data compilation and calculation steps that are described in this document. In addition, this figure provides a roadmap for the HHRA, showing how the development of the dataset leads into scenario development, calculation of risk estimates, analysis of uncertainty, and finally the identification of risk drivers. The figure also references the section in which each step in the process is discussed. The approach is the same as that used for the LDW HHRA (Windward 2007c).

People may be exposed to chemicals in the EW either through direct exposure to sediment and surface water or indirectly through the consumption of fish and shellfish collected from the EW (Section B.3.1). Accordingly, sediment, surface water, and tissue chemistry data from the EW are relevant for this HHRA. The following subsections describe data availability (Section B.2.1), data reduction (Section B.2.2), and the suitability of data for risk assessment purposes (Section B.2.3). Details on data aggregation and calculations are provided in Section B.2.3 and in applicable sections of the exposure assessment (Section B.3) where these calculations are used.

B.2.1 Data Availability and Selection

Many environmental investigations conducted within the EW have included the collection of chemistry data from samples of sediment, fish and shellfish tissue, or water. The sources for sediment, tissue, and water data are summarized below.

B.2.1.1 Surface sediment chemistry

The following considerations were made in selecting existing surface sediment data for the HHRA dataset:

- **Depth of sample –** For subtidal sediment, only grab samples that were collected from 0 to 10 cm were included. Intertidal multi-increment sampling (MIS) samples were collected from 0 to 25 cm.
- **Sampling date –** Only data that had been collected since 1995 were included.
- **Dredging activities –** Only data that were collected from locations that were not subsequently dredged were included.
- **Data quality –** Only data that were validated and considered acceptable for risk assessment under CERCLA were included (historical datasets were reviewed in the existing information summary report [EISR] (Anchor and Windward 2008)).

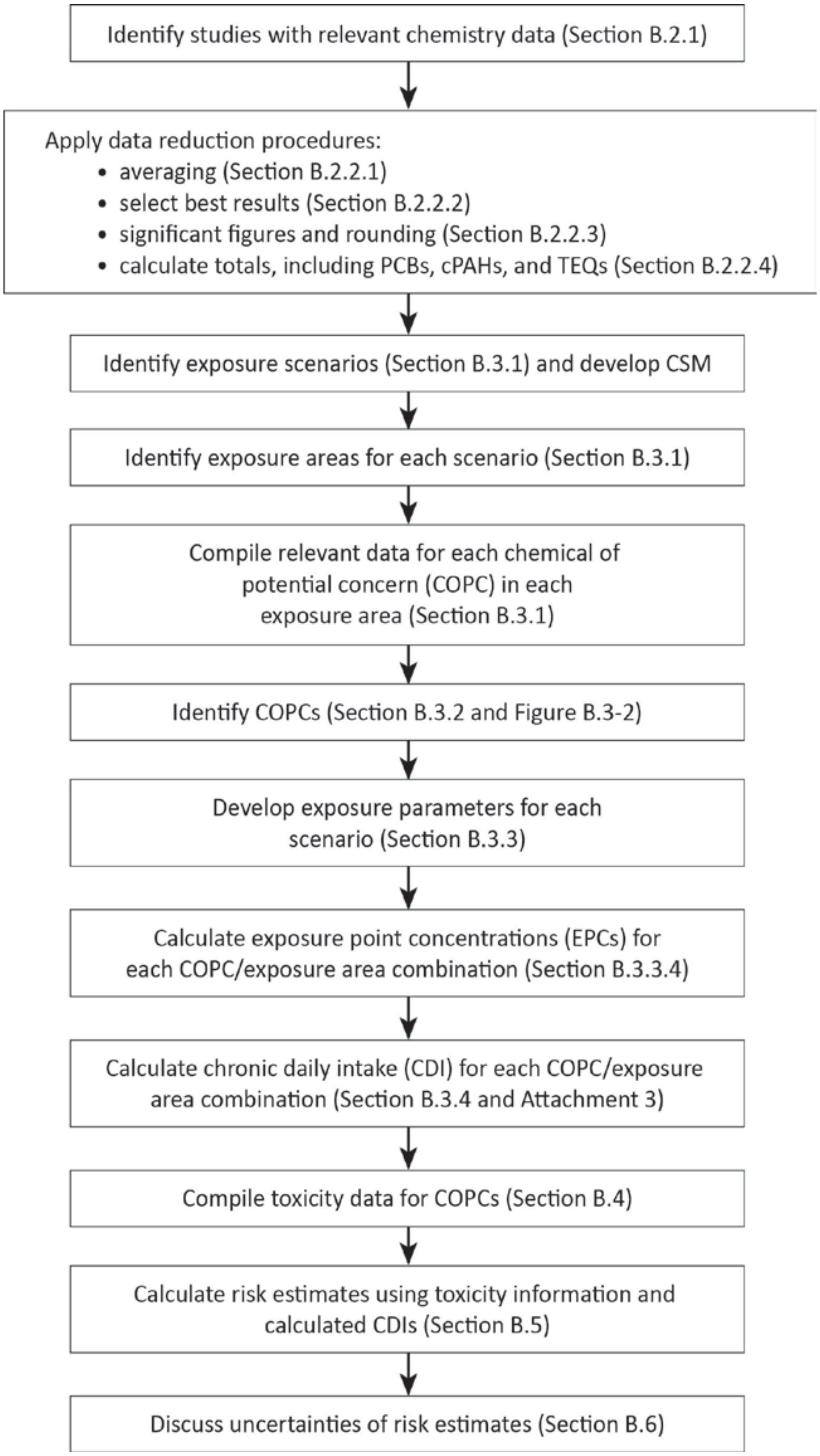

Figure B.2-1. HHRA flow chart

- As reported in the EISR (Anchor and Windward 2008), numerous investigations since 1995 involved the collection of surface sediment samples in the EW. Many of these sediment samples were collected as part of post-dredge monitoring or nature and extent of contamination investigations. For this HHRA, subtidal sediment samples collected from a sediment depth of 0 to 10 cm were included; these samples should reasonably characterize exposures during netfishing (nets will not likely be pulled through sediment deeper than 10 cm). Deeper samples (0 to 25 cm) were collected from the intertidal area because individuals could be exposed to somewhat deeper sediment as a part of clamming or habitat restoration activities (e.g., digging). The use of the 25-cm depth in the intertidal areas was based on site-specific clam burrowing depths for clam species collected in the EW (less than 30 cm for butter clams, less than 10 cm for littleneck clams, and approximately 10 cm for cockles), consistent with Pacific Northwest-specific information (Kozloff 1973). The adjacent LDW Superfund site used a depth of compliance of 45 cm for the Feasibility Study, largely because of the presence of Eastern soft-shell clams. Eastern soft-shell clams are found in a very limited portion of the upstream part of the EW, and thus it was decided that Eastern soft-shell clam burrowing depths should not be a factor when assessing direct contact contaminant exposure while clamming throughout the EW. Thus, 25 cm provides a good estimate of the average depth to which individuals might dig to collect intertidal clams. However, it should be noted that the exposure depth of 25 cm used in this HHRA is not intended to define the point of compliance for the EW. The depth of compliance will be determined in consultation with EPA as part of the FS.

In addition to the substantial number of surface sediment samples that have been collected in the EW since 1995, a large number of sediment samples were also collected as part of the EW SRI. The sampling design for the 2009 EW SRI data collection event was presented in the surface sediment quality assurance project plan (QAPP) (Windward 2009e). The sampling plan was designed to supplement existing acceptable sediment data in order to provide good overall spatial coverage for the EW. Nearly all of the surface sediment sampling locations from the older and more recent data collection efforts are considered to be relevant and representative of current conditions (reasons for the exclusion of some samples are listed in the bullets below). Table B.2-1 lists all of the sampling events for which data were reviewed and found to be acceptable for use in the HHRA. They include subtidal surface sediment samples collected at depths of 10 cm or less. However, some samples were excluded from the HHRA dataset (Table B.2-2) for one or more of the following reasons:[14]

- Samples were collected from outside the study area.
- Samples were superseded by more recent sampling events (samples located within 10 ft of earlier samples were considered to be resampled locations).

[14] Note that data, such as the co-located clam and benthic invertebrate sediment grabs, excluded from the HHRA may be used to evaluate sediment-tissue relationships, which will be explored in detail in the RI.

- Sampling location was dredged subsequent to sample collection.
- Sampling location was covered by a sand cap subsequent to sample collection. Because both composite and grab samples were collected at some locations, some composite samples were excluded to avoid mixing composite and grab samples for calculation of subtidal exposure estimates.[15]
- In two studies, multiple sediment depth intervals from the same location were analyzed, from which only the samples from the 0-to-10-cm interval were selected for the EW HHRA dataset. As part of the Phase 1 recontamination monitoring study conducted by Windward Environmental LLC (Windward) (2008c), 2 samples were collected at each of 10 sampling locations. The first sample was collected from a depth interval ranging from 0 to between 3 and 8 cm, and the second sample was collected from a depth interval of 0 to 10 cm. The sampling was designed to determine if samples of recently deposited material from the shallower depth intervals had substantially different concentrations than the 0 to 10 cm samples. No consistent differences were observed. In another study conducted by Hart Crowser (2005), samples were analyzed from the 0-to-1-cm and 0-to-10-cm intervals from two locations in Slip 36. In both cases, only the samples from the 0-to-10-cm interval were selected for the EW HHRA dataset. Further discussion of these samples is presented in Section 4.3.3 of the monitoring study report (2008c).

Table B.2-1. Summary of available surface sediment data used in the EW HHRA

Sampling Date	Sampling Event	No. of Samples	Analyses	Source
2009	EW Subtidal Surface Sediment Composites	13	PCB congeners, dioxin/furan congeners	Windward (2010d)
2009	EW intertidal MIS composites	4	metals, organometals, SVOCs, Aroclors, PCB congeners, dioxin/furan congeners, pesticides, conventionals	Windward (2010d)
2009	EW Surface Sediment Round 1	54	metals, organometals, SVOCs, Aroclors, pesticides, grain size, conventionals	Windward (2010d)
2009	EW Surface Sediment Round 2	44	metals, organometals, SVOCs, Aroclors, pesticides, grain size, conventionals	Windward (2010d)
2009	EW T30 Post-Dredge Monitoring 2009	17	metals, SVOCs, Aroclors, grain size, conventionals	Windward (2010d)
2008	EW Recontamination Monitoring 2008	12	metals, SVOCs, Aroclors, grain size, conventionals	Windward (2008c)
2007	East Waterway – Slip 27	7	metals, organometals, SVOCs, Aroclors, pesticides, grain size, conventionals	Windward (2007a)

[15] See Section B.3.3.4.2 for the discussion of how data were treated for the calculation of exposure concentrations.

Table B.2-2. Summary of available surface sediment data excluded from the EW HHRA (cont.)

Sampling Date	Sampling Event	No. of Samples	Analyses	Source
2007	Recontamination Monitoring 2007	24	metals, SVOCs, Aroclors, pesticides, grain size, conventionals	Windward (2008b)
2006	Recontamination Monitoring 2006	21	Metals, SVOCs, Aroclors, pesticides, grain size, conventionals	Windward (2007b)
2005	Post-Dredge Monitoring 2005 Phase2	9	metals, SVOCs, Aroclors, pesticides, grain size, conventionals	Anchor and Windward (2005a)
2005	USCG Pier 36 Post-Dredge Sediment Characterization	11	metals, SVOCs, Aroclors, grain size, conventionals	Hart Crowser (2005)
2001	EW/HI Nature and Extent Phase 1	54	metals, SVOCs, Aroclors, pesticides, grain size, conventionals	Windward (2002)
2001	EW/HI Nature and Extent Phase 2	21	metals, SVOCs, Aroclors, pesticides, grain size, conventionals	Windward (2002)
2000	T-18 Post-Dredge Monitoring	11	metals, organometals, PAHs, Aroclors, pesticides, conventionals	Windward (2001)
1996	KC CSO 96	4	metals, SVOCs, Aroclors, grain size, conventionals	King County (1996)
1996	Pier 36-underpier	2	metals, SVOCs, Aroclors, pesticides, conventionals	Tetra Tech (1996)
1995	HI Remedial Investigation 95	3	metals, SEM metals, organometals, SVOCs, Aroclors, pesticides, grain size, conventionals	EVS (1996a, b)
1995	KC CSO 95	6	metals, SVOCs, Aroclors, pesticides, TPH, grain size, conventionals	King County (1995)

CSO – combined sewer overflow
EW – East Waterway
HHRA – human health risk assessment
HI – Harbor Island
KC – King County
MIS – multi-increment sampling
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SEM – simultaneously extracted metals
SVOC – semivolatile organic compound
T-18 – Terminal 18
TPH – total petroleum hydrocarbons
USCG – US Coast Guard

Table B.2-2. Summary of available surface sediment data excluded from the EW HHRA

Sampling Date	Sampling Event[a]	No. of Samples	Reason for Exclusion	Source
2009	EW Surface Sediment Round 1	3	These samples were collected outside the study area.	Windward (2010d)
2009	EW Surface Sediment Round 2	13	These samples were intertidal grabs and not used in the HHRA (intertidal exposures were evaluated with MIS samples) or collected outside the study area.	Windward (2010d)
2009	EW Benthic Tissue 09	8	These subtidal composite samples were collected specifically for use in evaluating the bioaccumulation potential of benthic invertebrates. In addition, only grab sample data were used to characterize subtidal exposures.[b]	Windward (2009a)
2008	EW Benthic Tissue 08	13	Composite subtidal sediment samples were excluded because only grab sample data were used to characterize subtidal exposures for all chemicals except PCB congeners and dioxin/furan congeners. These composite samples were not analyzed for PCB congeners and dioxin/furan congeners.	Windward (2009a)
2008	EW Clam Survey	5	These intertidal grab samples were collected specifically for use in the evaluation of the bioaccumulation potential of clams. In addition, grab samples were not used to characterize intertidal exposures in the HHRA (which were evaluated with MIS samples), and the depth interval for these samples (0 to 10 cm) was inconsistent with the expected depth for clamming exposure (0 to 25 cm).	Windward (2010b)
2008	EW Recontamination Monitoring	10	Samples from the 0- to-10-cm interval were selected over the samples from the overlying material intervals (0 to 3 cm and 0 to 8 cm).	Windward (2008c)
2008	EW T-30 Post-Dredge Monitoring 2008	5	Data were superseded by 2009 event.	Windward (2010d)
2005	EW Pre-Sand Placement Monitoring	37	Samples no longer represent surface.	Anchor and Windward (2005a)
2005	Post-Dredge Monitoring-2005	6	The post-dredge monitoring data were collected prior to the placement of the sand layer in this area. The current surface sediment in this area is represented by the recontamination monitoring dataset listed in Table B.2-1.	Anchor and Windward (2005a)
2005	USCG Pier 36 Post-Dredge Sediment Characterization	2	Samples from the 0-to-10-cm interval were selected over those from the 0-to-1-cm interval for two locations.	Hart Crowser (2005)
2001	EW/HI Nature and Extent Phase 2	2	Location was dredged.	Windward (2002)
1997	Pier 36/37 – surface sediment	3	Location was dredged.	Tetra Tech (1997)

Table B.2-2. Summary of available surface sediment data excluded from the EW HHRA (cont.)

Sampling Date	Sampling Event[a]	No. of Samples	Reason for Exclusion	Source
1996	Pier 36-underpier	1	This was an intertidal grab sample, which is inconsistent with the sample types used for characterizing intertidal exposure (only MIS samples were used for intertidal exposure).	Tetra Tech (1996)
1996	KC CSO 96	2	Location was dredged.	King County (1996)
1995	KC CSO 95	1	Location was dredged.	King County (1995)
1995	HI Remedial Investigation 95	15	Location was dredged.	EVS (1996a, b)

a Data excluded from the HHRA may be used to evaluate sediment-tissue relationships, which will be explored in detail as part of the RI.

b Subtidal composite samples were analyzed using PCB congener and dioxin/furan TEQ analyses, but these samples were not analyzed for these chemicals.

CSO – combined sewer overflow
EW – East Waterway
HHRA – human health risk assessment
HI – Harbor Island
KC – King County
MIS – multi-increment sampling
PCB – polychlorinated biphenyl
RI – remedial investigation
T-30 – Terminal 30
TEQ – toxic equivalent
USCG – US Coast Guard

The SRI sampling events included subtidal grab samples, subtidal grab composite samples, and intertidal MIS samples (Table B.2-1). Three of the four MIS samples represented study area-wide intertidal areas. The remaining MIS sample represented only intertidal areas that were publically accessible from the shore. Because the intertidal MIS was designed specifically to characterize human intertidal exposure (Windward 2009e), the MIS data were used to estimate intertidal exposures, and intertidal grab sample data were excluded (Table B.2-2). The SRI surface sediment sampling locations for subtidal grab samples are shown on Map B.2-1, and the SRI surface sediment sampling locations for intertidal MIS and subtidal composite samples are shown on and Map B.2-2.

All subtidal grab samples collected from 1995 to 2009 that were determined to be appropriate based on the project data rules were used to estimate subtidal exposures for all chemicals except dioxins/furans and polychlorinated biphenyl (PCB) congeners (Table B.2-1; Map B.2-1). The 13 subtidal composite samples from 2009 listed in Table B.2-1 and the benthic subtidal sediment composite samples collected in 2008 and 2009 listed in Table B.2-2 were not used to estimate subtidal exposures to these chemicals to avoid mixing composite and grab sample data. In addition, these samples were intended to characterize exposure in subtidal sediment. The only subtidal composite samples used in the HHRA were the 13 samples from 2009, as listed in Table B.2-1 (shown on Map B.2-2), which were used to estimate subtidal exposures for PCB congeners and dioxin/furan congeners. The locations and collection methods for these samples were designed to represent EW-wide subtidal concentrations of these chemicals (Windward 2009e).

B.2.1.2 Tissue chemistry

Study area tissue chemistry data for several different tissue types were available from several sampling events conducted since 1995. Site-specific tissue chemistry data are available for the following species that may potentially be consumed by people: English sole, brown rockfish, shiner surfperch, striped perch, Dungeness crab, red rock crab, mussels, butter clams, littleneck clams, cockles, soft-shell clams, and geoduck clams.[16] Although other species in the EW may also be consumed by people, the broad range of fish and shellfish data available from the EW are expected to be representative of all consumed species.[17] The list of fish and invertebrate species collected from the EW during sampling efforts is provided in the EW ERA (Appendix A).

The tissue dataset for the EW HHRA is summarized in Table B.2-3. The SRI fish tissue sampling locations for samples used in the HHRA are shown on Map B.2-3. The SRI shellfish tissue sampling locations for samples used in the HHRA are shown on Map B.2-4. It should be noted that after extensive surveying of the EW, intertidal clams (greater than or equal to 4 cm in length) and geoduck were collected from all areas where they were found (Windward 2010b). All tissue data collected since 1995 for resident species thought to be consumed by people were included in the EW HHRA dataset. As reported in the EISR (Anchor and Windward 2008) and indicated in Table B.2-3, five studies conducted outside of the EW SRI/FS process (i.e., all events conducted between 1995 and the 2008 SRI sampling) have reported tissue chemical concentrations for fish and shellfish collected throughout the EW. English sole were analyzed by EVS Environment Consultants (Battelle 1996); transplanted mussels were collected by King County in 1996 and 1997 (1999a); red rock crab and striped perch were collected by Environmental Solutions Group (ESG) (1999), and English sole, shiner surfperch, and rock fish were collected by Windward (2005d). PCBs, mercury, and tributyltin (TBT) were the most frequently analyzed chemicals in tissue samples. King County (1999a) conducted the only study prior to the SRI with an extensive analytical list that included metals, organometals, semivolatile organic compounds (SVOCs), PCBs, and pesticides, but they are available only for mussels. The largest dataset quantifying chemicals in fish and shellfish from the EW is the 2008 SRI sampling (Anchor and Windward 2008), which included the collection of English sole, brown rockfish, shiner surfperch, crabs, mussels, geoducks, and clams.

In developing data subsets for different tissue types for this HHRA, all tissue chemistry data for the same tissue type from all years (from 1995 to 2008) were combined because there was no reason to expect that these samples could not be used together (i.e., environmental conditions and chemical contamination were not expected to have changed substantially over the years of collection).

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[16] One shrimp composite sample is available but was not used in this HHRA because of the low abundance of shrimp, as discussed in Table B.2-3 and B.3-5.

[17] Salmon were not included in the seafood consumption scenarios, as discussed in Section B.3.3.1.

Table B.2-3. Summary of available tissue data used in the EW HHRA

Species[a]	Sampling Event	Year of Sample Collection	No. of Samples	No of Individuals per Sample	Sample Type	Analytes	Source
English sole	EW-Fish Collection 2008	2008	11	5	whole body	PCB Aroclors, pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins/furans (subset of samples), PCB congeners (subset of samples)	Windward (2010a, c)
			11	5	skin-on fillet		
	EW-Fish Collection 2005	2005	2	5	skin-on fillet and remainder[b]	PCB Aroclors, mercury, lipids	Windward (2006b)
	EVS 95	1995	3	6 to 8	skinless fillet	PCB Aroclors and subset of PCB congeners, butyltins, mercury, methylmercury, lipids	Battelle (1996) and Frontier GeoSciences (1996)
Brown rockfish	EW-Fish Collection 2008	2008	13	1	whole body	PCB Aroclors, pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins/furans (subset of samples), PCB congeners (subset of samples)	Windward (2010a, c)
	EW-Fish Collection 2005	2005	2	1		PCB Aroclors, mercury, lipids	Windward (2006b)
Shiner surfperch	EW-Fish Collection 2008	2008	8	10	whole body	PCB Aroclors, pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins/furans (subset of samples), PCB congeners (subset of samples)	Windward (2010a, c)
	EW-Fish Collection 2005	2005	3	6 to 8		PCB Aroclors, mercury, lipids	Windward (2006b)
Striped perch	WSOU	1998	3	2 to 8	skinless fillet	PCB Aroclors, mercury, TBT, lipids	ESG (1999)
	WSOU	1998	3	2 to 8	skin-on fillet	PCB Aroclors, mercury, TBT, lipids	ESG (1999)
Dungeness crab[c]	EW-Fish Collection 2008	2008	1	7	edible meat	PCB Aroclors, pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins/furans (subset of samples), PCB congeners (subset of samples)	Windward (2010a, c)
	EW-Fish Collection 2008				hepato-pancreas	PCB Aroclors, pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins/furans (subset of samples), PCB congeners (subset of samples)	Windward (2010a, c)
Red rock crab[c]	EW-Fish Collection 2008	2008	8	7	edible meat	PCB Aroclors, pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins/furans (subset of samples), PCB congeners (subset of samples)	Windward (2010a, c)

Table B.2-3. Summary of available tissue data used in the EW HHRA (cont.)

Species[a]	Sampling Event	Year of Sample Collection	No. of Samples	No of Individuals per Sample	Sample Type	Analytes	Source
	WSOU	1998	3	5		PCB Aroclors, mercury, TBT	ESG (1999)
	EW-Fish Collection 2008	2008	8	7	hepato-pancreas	PCB Aroclors, pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins/furans (subset of samples), PCB congeners (subset of samples)	Windward (2010a, c)
Mussels	EW-Fish Collection 2008	2008	11	89 to 101	soft tissue	PCB Aroclors, pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins/furans (subset of samples)	Windward (2010a, c)
	KC WQA	1997	3	50 to 100		PCB Aroclors, SVOCs, pesticides, metals, butyltins, lipids, solids	King County (1999a)
	KC WQA	1996	3	50 to 100		PCB Aroclors, SVOCs, pesticides, metals, butyltins, lipids, solids	King County (1999a)
Clams – butter	EW-Clam Survey	2008	7	6 to 15	soft tissue	PCB Aroclors, SVOCs, pesticides, metals, butyltins, lipids, solids, dioxins/furans (in two samples), PCB congeners (in two samples)	Windward (2010a, b)
Clams – littleneck	EW-Clam Survey	2008	2	4 to 9	soft tissue	PCB Aroclors, pesticides, metals, butyltins, lipids, solids	Windward (2010a, b)
Clams – cockle	EW-Clam Survey	2008	2	13 to 17	soft tissue	PCB Aroclors, SVOCs, pesticides, metals, butyltins, lipids, solids, dioxins/furans (in one sample), PCB congeners (in one sample)	Windward (2010a, b)
Clams – soft-shell	EW-Clam Survey	2008	1	15	soft tissue	PCB Aroclors SVOCs, pesticides, metals, butyltins, lipids, solids	Windward (2010a, b)
Geoduck[d]	EW-Clam Survey	2008	5	1	edible meat	PCB Aroclors, SVOCs, pesticides, metals, butyltins, lipids, solids, dioxins/furans (subset of samples), PCB congeners (subset of samples)	Windward (2010a, b)
	EW-Clam Survey	2008	3	3	gut ball	PCB Aroclors, SVOCs, pesticides, metals, butyltins, lipids, solids, dioxins/furans (subset of samples), PCB congeners (subset of samples)	

[a] During the EW sampling efforts, 16 traps intended for the collection of shrimp were deployed, and 26 shrimp were collected. This small number of shrimp indicates that there are likely insufficient shrimp of harvestable size present in the EW to constitute a significant portion of seafood consumption for the scenarios evaluated in this HHRA. Thus, as discussed further in B.3.3.1.1, shrimp were not included in the HHRA tissue dataset.

[b] The results for the fillet composite samples and the remainder composite samples were weighted based on the fraction of the whole-body mass represented by each sample in order to calculate whole-body results (Windward 2006b).

Table B.2-3. Summary of available tissue data used in the EW HHRA (cont.)

[c] Data from hepatopancreas composite samples were mathematically combined with data from edible-meat composite samples to form composite samples of edible meat plus hepatopancreas. Whole-body (i.e., edible meat plus hepatopancreas) crab chemical concentrations were calculated using the relative weights and concentrations of the edible meat and hepatopancreas.

[d] Data from gut ball composite samples were mathematically combined with data from edible-meat composite samples to form composite samples of edible meat plus gut ball. Whole-body (i.e., edible meat plus gut ball) geoduck chemical concentrations were calculated using the relative weights and concentrations of the edible meat and gut ball.

ESG – Environmental Solutions Group

EW – East Waterway

HHRA – human health risk assessment

KC – King County

PCB – polychlorinated biphenyl

SVOC – semivolatile organic compound

TBT – tributyltin

WQA – water quality assessment

WSOU – Waterway Sediment Operable Unit

The majority of the samples for all tissue types (except striped perch) were collected as part of the SRI sampling, specifically for the purposes of the HHRA and ERA. Although data collected prior to 2008 are limited, a discussion of changes in tissue concentrations over time will be presented in the RI.

The specific details of how the tissue samples were grouped to describe exposure (i.e., the consumption categories for which exposure point concentrations were developed) are discussed in Section 3.3.4.

B.2.1.3 Surface water chemistry

Surface water data were collected in 2008-2009, specifically for the EW SRI/FS (Anchor and Windward 2008), and were intended to represent the EW spatially and temporally (Table B.2-4; Map B.2-5). Additional surface water data from the EW were available from three previous collection efforts, as presented in the EISR (Anchor and Windward 2008) and the human health technical memorandum (Windward 2010f). However, based on discussions with EPA, these other datasets were not included in the HHRA water dataset. Although large, the King County water quality assessment (WQA) dataset (1999a) was not included in the HHRA dataset for three reasons: limited spatial scope (collected from three locations along one transect), detections primarily for metals, and elevated analytical reporting limits (RLs) for many SVOCs.[18] Data from two post-dredge monitoring events (Anchor and Windward 2005b; SEA 2000) were not included because they were not considered to add sufficient value to the spatial and temporal representation of the dataset. Hence, only the EW SRI/FS (Anchor and Windward 2008) water data were used in risk characterization for this HHRA (Table B.2-4). The potential impacts of the inclusion of other data were evaluated in the uncertainty analysis.

Table B.2-4. Summary of available surface water data used in the EW HHRA

Year of Sample Collection	Sampling Event	No. of Samples Analyzed	Analytes	Source
2008-2009	SRI/FS	49	metals (filtered and unfiltered), PCBs (congeners), SVOCs, TBT, conventionals	Windward (2009f)

EW – East Waterway
FS – feasibility study
HHRA – human health risk assessment
PCB – polychlorinated biphenyl
SRI – supplemental remedial investigation
SVOC – semivolatile organic compound
TBT – tributyltin

[18] The King County water samples were not analyzed for PCBs because PCBs were not detected in a pilot study that used the PCB Aroclor analytical method.

B.2.2 Data Reduction

Data reduction refers to computational methods used to aggregate data. The methods used for data reduction for this HHRA were consistent with those used in the LDW HHRA. Data that were selected from those datasets presented in Tables B.2-1, B.2-3, and B.2-4 for determining exposures of humans were expressed on a dry-weight basis for sediment chemistry, on a wet-weight basis for tissue chemistry, and on a mass-per-unit-volume basis for water chemistry. All concentrations qualified as estimated (i.e., J- qualified data) were assumed to indicate the positive identification of the chemical and were used without modification in subsequent calculations (less than 15% of the dataset was J-qualified). Less than 1% of the data were rejected by data validators for quality issues and flagged with an R-qualifier. R-qualified data were not used at all in the risk assessment because the data validator determined these results to be unusable.

The most significant use of aggregated data was for the calculation of exposure point concentrations (EPCs), which are used to estimate long-term exposures in the HHRA. The EPC computation methods are described in detail in the exposure assessment (Section B.3.3.4).

Additional procedures related to averaging, the selection of the best data points when multiple data are available, the selection of significant figures and rounding procedures, and the calculation of totals for chemical groupings (i.e., PCBs, carcinogenic polycyclic aromatic hydrocarbons [cPAHs], dichlorodiphenyltrichloroethane [DDTs] and dioxins/furans) are described in the following subsections.

B.2.2.1 Averaging duplicate or replicate samples

Chemical concentrations obtained from the analysis of laboratory duplicates or replicates (i.e., two or more analyses performed on the same sample) were averaged for a closer representation of the “true” concentration compared with the results of a single analysis. Averaging rules were dependent on whether the individual chemical concentrations were detected or undetected. If all concentrations for a given chemical were detected, the values were averaged arithmetically. If all concentrations for a given chemical were undetected, the minimum RL was reported. If the results were a combination of detected and undetected concentrations, any two or more detected concentrations were averaged arithmetically, and the undetected concentrations were excluded. If the combined concentrations consisted of a single detected concentration and one or more undetected concentrations, the detected concentration was reported. The latter two rules were applied regardless of whether the RL was higher or lower than the detected concentration.

Identical averaging rules were applied in situations in which multiple sediment samples were collected from the same location at the same time, such as field duplicate samples. In these instances, a single “average” result for each chemical was generated for that sediment sampling location.

B.2.2.2 Selection of best results

In some instances, the laboratory generates more than one result for a chemical for a given sample. Multiple results can occur for several reasons, including: 1) the original result did not meet the laboratory's internal quality control (QC) guidelines, and a reanalysis was performed; 2) the original result did not meet other project data quality objectives, such as a sufficiently low RL, and a reanalysis was performed; or 3) two different analytical methods were used for that chemical. In each case, a single best result was selected for use. The procedures for selecting the best result differed depending on whether a single or multiple analytical methods were used for a given chemical. For the same analytical method, if the results were:

- Detected and not qualified, the result from the lowest dilution was selected, unless multiple results from the same dilution were available, in which case, the result with the highest concentration was selected.
- A combination of estimated and unqualified detected results, the unqualified result was selected. This situation most commonly occurred when the original result was outside of calibration range, thus requiring a dilution. No results outside the calibration range were used in the HHRA.
- All estimated, then the "best result" was selected using best professional judgment in consideration of the rationale for qualification. For example, a result qualified based on laboratory replicate results outside of QC objectives for precision would be preferred to a qualified result that was outside the calibration range.
- A combination of detected and undetected results, the detected result was selected. If there was more than one detected result, the applicable rules for multiple results (as discussed above) were followed.
- All undetected results, the lowest RL was selected.

If the multiple results were from different analytical methods, the result from the preferred method specified in the QAPP or based on the consensus of the professional opinions of project chemists was selected. Attachment 1 provides a detailed discussion of the samples and analytes with multiple results.

B.2.2.3 Significant figures and rounding

Analytical laboratories reported results with various numbers of significant figures depending on the QAPP instructions, the instrument, the parameter, and the concentration relative to the RL. The reported (or assessed) precision of each observation was explicitly stored in the project database by recording the number of significant figures assigned by the laboratory. Tracking of significant figures becomes important when calculating averages and performing other data summaries.

When a calculation involves addition, such as totaling PCBs or polycyclic aromatic hydrocarbons (PAHs), the calculation can be only as precise as the least precise number that went into the calculation. For example (assuming two significant figures):

210 + 19 = 229 would be reported as 230 because 19 is reported only to 2 significant digits, and the enhanced precision of the trailing zero in the number 210 is not significant. 210 + 19.0 = 229 would also be reported as 230.

When a calculation involves multiplication or division, such as carbon normalization, the original figures for each value are carried through the calculation (i.e., individual values are not adjusted to a standard number of significant figures, instead the appropriate adjustment is made to the resultant value at the end of the calculation). The result is rounded at the end of the calculation to reflect the value used in the calculation with the fewest significant figures. For example:

59.9 × 1.2 = 71.88 would be reported as 72 because there are two significant figures in the number 1.2.

59.9 × 1.0 = 59.9 would be reported as 60 because there are two significant figures. Note that 59.9 × 1 would also be reported as 60, although in that case, the 0 would not be considered significant.

When rounding, if the number following the last significant figure is less than 5, the digit is left unchanged. If the number following the last significant figure is equal to or greater than 5, the digit is increased by 1.

B.2.2.4 Calculating totals

Concentrations for several chemical sums were calculated as follows:

- Total PCBs were calculated using only detected concentrations for seven Aroclor mixtures (1016, 1221, 1232, 1242, 1248, 1254, and 1260)[19] in accordance with Washington State Sediment Management Standards (SMS) (Washington Administrative Code [WAC] 173-204). For individual samples in which none of the seven Aroclor mixtures was detected, total PCBs were given a value equal to the highest RL of the seven Aroclors. An alternative approach for computing total PCBs has been used for other HHRAs in EPA Region 10 and is evaluated in Section B.6.1.1.3.
- Toxic equivalents (TEQs) were used for totaling certain groups of chemicals, specifically dioxins and furans, dioxin-like PCBs, and cPAHs. The 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) toxic equivalency factors (TEFs) for coplanar PCBs and certain polychlorinated dibenzo-*p*-dioxin (PCDD) or polychlorinated dibenzofuran (PCDF) (dioxin/furan) congeners are presented in Table B.2-5. The TEFs relate the toxicity of the co-planar PCB congeners and

[19] For several sediment samples, Aroclors 1262 and 1268 were also included in the total PCB calculation, although these Aroclors are rarely quantified.

certain dioxin/furan congeners to the toxicity of 2,3,7,8-TCDD. Similarly, potency equivalency factors (PEFs) relate the toxicity of certain PAH compounds to that of benzo(a)pyrene. PEFs for cPAHs are also presented in Table B.2-5. PCB TEQ, dioxin/furan TEQ, and cPAH totals were calculated for each sample by summing the products of the concentrations of each individual congener or compound and its specific TEF or PEF for each group (PCB TEQ, dioxin/furan TEQ, and cPAHs, respectively). Congeners or compounds that were undetected for a given sample were assigned a value equal to one-half the sample-specific RL for use in the TEQ calculation. The use of other assumptions (i.e., full RL or 0) for non-detected values is further explored in Section B.6.1.6.6).

- Total DDTs were calculated from detected concentrations of three to six of the following: 2,4′-dichlorodiphenyldichloroethane (DDD), 2,4′-dichlorodiphenyldichloroethylene (DDE), 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, and 4,4′-DDT. For samples in which all individual DDDs, DDEs, and DDTs were undetected, the single highest RL for that sample was assigned to represent the total DDT sum.

Table B.2-5. Toxic equivalency and potency equivalency factors for dioxins/furans, PCB congeners, and cPAHs

Compound	Toxic Equivalency or Potency Equivalency Factor
Dioxins/Furans[a]	
2,3,7,8-tetrachlorodibenzo-*p*-dioxin	1
1,2,3,7,8-pentachlorodibenzo-*p*-dioxin	1
1,2,3,6,7,8-hexachlorodibenzo-*p*-dioxin	0.1
1,2,3,4,7,8-hexachlorodibenzo-*p*-dioxin	0.1
1,2,3,7,8,9-hexachlorodibenzo-*p*-dioxin	0.1
1,2,3,4,6,7,8-heptachlorodibenzo-*p*-dioxin	0.01
Octachlorodibenzo-*p*-dioxin	0.0003
2,3,7,8-tetrachlorodibenzofuran	0.1
1,2,3,7,8-pentachlorodibenzofuran	0.03
2,3,4,7,8-pentachlorodibenzofuran	0.3
1,2,3,6,7,8-hexachlorodibenzofuran	0.1
1,2,3,7,8,9-hexachlorodibenzofuran	0.1
1,2,3,4,7,8-hexachlorodibenzofuran	0.1
2,3,4,6,7,8-hexachlorodibenzofuran	0.1
1,2,3,4,6,7,8-heptachlorodibenzofuran	0.01
1,2,3,4,7,8,9-heptachlorodibenzofuran	0.01
Octachlorodibenzofuran	0.0003
PCBs[a]	
PCB 77	0.0001
PCB 81	0.0003
PCB 105	0.00003
PCB 114	0.00003

Table B.2-5. Toxic equivalency and potency equivalency factors for dioxins/furans, PCB congeners, and cPAHs (cont.)

Compound	Toxic Equivalency or Potency Equivalency Factor
PCB 118	0.00003
PCB 123	0.00003
PCB 126	0.1
PCB 156	0.00003
PCB 157	0.00003
PCB 167	0.00003
PCB 169	0.03
PCB 189	0.00003
cPAHs[b]	
Benzo[a]pyrene	1
Benz[a]anthracene	0.1
Benzo[b]fluoranthene	0.1
Benzo[k]fluoranthene	0.1
Chrysene	0.01
Dibenz[a,h]anthracene[c]	0.4
Indeno[1,2,3-cd]pyrene	0.1

[a] TEFs for dioxins/furans and PCB congeners are from the World Health Organization (Van den Berg et al. 2006).

[b] PEFs for cPAHs were defined by the Cal EPA, Office of Environmental Health Hazard Assessment (California EPA 1994). PEFs are available for PAHs that were not analyzed in EW sediments. The PEFs for these compounds are not shown here and were not used in this risk assessment.

[c] The PEF was determined by the Cal EPA by dividing the inhalation unit risk factor for this compound by the inhalation unit risk factor for benzo[a]pyrene.

Cal EPA – California Environmental Protection Agency
cPAH – carcinogenic polycyclic aromatic hydrocarbon
EW – East Waterway
PCB – polychlorinated biphenyl
PEF – potency equivalency factor
TEF – toxic equivalency factor

B.2.3 Suitability of Data for Risk Assessment

Several factors need to be considered when assessing the suitability of environmental data for risk assessments (EPA 1989, 1992b). Of primary importance is the degree to which the data adequately represent site-related contamination and the expected human exposures at the site. Also important to consider are the data quality criteria goals and the source, documentation, analytical methods, RLs, and level of review associated with the data. Because data from many different investigations were available for the EW, the factors described above were evaluated for each dataset to determine whether it was reasonable to combine all data for use in this HHRA.

B.2.3.1 Representativeness of data to site-related contamination

B.2.3.1.1 Sediment

Sediment studies within the EW have been designed for both reconnaissance investigations (e.g., EW/Harbor Island Nature and Extent – Phases 1 and 2) and focused investigations of areas of concern (e.g., Slip 27). In addition, a substantial amount of sediment chemistry data has been collected as part of this SRI. Most of the events outside of the EW SRI/FS process focused primarily on subtidal sediments. The representativeness of the existing dataset was evaluated during the design of the SRI surface sediment sampling conducted in 2009 (Windward 2010d). The combined dataset (i.e., acceptable existing data identified in Table B.2-1 and data collected for the SRI) was designed to be representative of surface sediment throughout EW.

MIS samples were collected to measure contaminant concentrations for assessing direct exposure to contaminants in sediment. The two exposure areas associated with these MIS samples are described below:

- **Study area-wide intertidal exposure area –** For the analysis of risks based on direct exposure to all EW intertidal sediment, a total of 108 individual grab samples were collected in a spatially representative way and divided between the three MIS samples so that each MIS sample provided a representative estimate of contaminant concentrations in the intertidal area (Windward 2010d). Grab samples were collected in all intertidal areas from which sediment could be collected (i.e., areas not covered in riprap).
- **Public access intertidal exposure area –** For the analysis of risks based on direct exposure to publicly accessible EW intertidal sediment, a total of 32 individual grab samples were collected and analyzed as a single MIS sample to provide an estimate of the contaminant concentrations in this exposure area (Windward 2010d). Grab samples were collected in all publicly accessible intertidal areas from which sediment could be collected (i.e., areas not covered in riprap).

As is discussed further in the RI, the percent difference between the samples and the mean was generally less than 50% for all samples, with the exception of PAHs. The high variance in the PAH MIS samples indicates the presence of a small area of higher concentrations that resulted in more skewed results. The uncertainty associated with the MIS samples is discussed in Section B.6.1.1.6.

B.2.3.1.2 Tissue

Within the EW, samples of crab, shrimp, clams, mussels, brown rockfish, shiner surfperch, and English sole were collected from multiple locations throughout the waterway. Most of the samples used in this HHRA were collected during the 2008 tissue sample collection efforts specifically for this SRI (Windward 2009e, 2008e). Given the size of the EW, the spatial distribution of sampling locations, and the number of tissue chemistry samples, the available tissue chemistry data are considered adequate to represent site-related exposures with respect to the EW seafood consumption exposure route.

B.2.3.1.3 Water

Surface water chemistry data have been collected in the EW as part of the SRI (Windward 2009b). These water data were collected from four locations distributed throughout the EW and represent dry-season and wet-season conditions and three flow regimes (e.g., slack tide, ebb tide, and a wet-weather flood event). These sampling regimes were specifically designed to characterize surface water chemistry throughout the EW and throughout the year.

B.2.3.2 Representativeness of data to expected human exposure

B.2.3.2.1 Sediment

People may come in contact with EW sediment through various activities, with known or possible exposures related to netfishing,[20] clamming, and habitat restoration, as described in Section B.3.1. The clamming and restoration activities may take place in intertidal areas, whereas netfishing occurs throughout the EW in both subtidal and intertidal areas. Data are available from several sediment sampling events that involved the collection of sediment from throughout the entire EW. Therefore, the overall distribution of sediment samples is considered to adequately reflect the expected human exposure to sediment throughout the EW.

The human access survey for the EW (Windward 2008a) included the identification and characterization of areas with public access to the intertidal sediment of the waterway. This survey was used to develop the intertidal surface sediment sampling plan for public access areas (Windward 2009d), from which sediment samples were collected. Consequently, sediment chemistry data exist for intertidal areas most likely to be visited by the general public as well for all intertidal areas that could possibly be accessed by individuals for specific activities (e.g., by tribal members with fishing and clamming rights or individuals performing habitat restoration).

B.2.3.2.2 Tissue

The representativeness of the tissue data for estimating potential human exposure to contaminants from seafood was evaluated based on: 1) which species are typically or known to be consumed by humans, 2) the time of sample collection (Does it coincide with a time during which harvest normally occurs?), 3) the size range of the organisms collected and analyzed (Does the size range reflect the size of fish normally consumed by people?), and 4) which species are resident to the EW. An extensive review conducted by ESG (1999) of existing seafood consumption surveys for Puget Sound

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[20] Netfishing in this context refers to gill nets deployed from boats by the Muckleshoot Indian Tribe for salmon fishing. Incidental sediment ingestion and dermal exposure to sediment may occur during net retrieval, as discussed in Section B.3.2.3.

indicated that all of the species listed in Table B.2-3 are potentially consumed by anglers in the LDW, EW, and Elliott Bay. A creel survey by King County (1999a) also indicated that the species listed in Table B.2-3 are consumed by local anglers. Flatfish and perch may be consumed year-round, although these species are not favored during seasons when adult salmon can be legally harvested (Landolt et al. 1985). Although the tissue sample data collected since 1995 (Table B.2 3) were collected between March and December, the vast majority were collected during the late summer or early fall when weather and tides for fishing and shellfish collection are most favorable, which is particularly important for recreational consumers. However, it should also be noted that year-round seafood harvesting is conducted by tribal members or other individuals who consume seafood at a higher rate. Thus, although tissue concentrations vary seasonally, individuals who consume seafood throughout the year will be exposed to an average annual concentration (i.e., an individual will not consume seafood exclusively at the highest or lowest concentration throughout the year). It is assumed that the tissue data represent average annual concentrations, although there is some uncertainty associated with this assumption, as lipophilic concentrations can vary annually and seasonally (as lipid content increases, the concentrations of many contaminants also increases).

The Washington State Department of Health (WSDOH) has issued fish consumption advisories for the Duwamish River (which includes the LDW, the EW, and the West Waterway). Currently, WSDOH recommends that no crab, shellfish, or fish (other than salmon) be consumed from the Duwamish River, which includes the EW. Despite these existing consumption advisories for select EW species (WSDOH 2010), some people consume resident fish and shellfish from the EW (1999a). The Washington State Department of Fish and Wildlife (WDFW) is responsible for enforcing fishing regulations; however, with respect to the consumption advisory, their role is to remind anglers of the advisory and to point out warning signs, such as those at Jack Perry Memorial Shoreline Public Access and on the Spokane Street Bridge (i.e., the low bridge below the West Seattle Bridge that is commonly used for fishing in the EW). WDFW has no enforcement responsibility pertaining to citations or other penalties.

Regardless of whether WSDOH issues consumption advisories, the tribes (which are also co-managers of the state's natural resources with WSDOH) maintain treaty rights to harvest seafood from the EW (or elsewhere in the state). This was guaranteed by the 1974 Boldt Decision, in which the US District Court ruled that treaty rights entitled the Tribes to half of the harvestable fish running in their traditional waters, a right that was later affirmed to include shellfish and other natural resources.

The objective of this risk assessment is to examine what the risks to consumers might be given the patterns of seafood consumption that could exist in the absence of chemical contamination. The fish and crab specimens included in the composite samples in the datasets identified for this HHRA are considered to be representative of sizes typically consumed by people fishing and crabbing the EW. In addition, because these larger specimens usually have higher levels of chemical contamination than do younger/smaller fish and shellfish, the use of larger specimens of the size typically collected for consumption in the HHRA is considered to be a health protective approach. The smallest specimens included in the composite samples were 20 cm for English sole, 8 cm for shiner surfperch, and 9 cm (carapace width) for Dungeness and red rock crabs (Windward 2010c).

B.2.3.3 Quality assurance/quality control results

All datasets used in the HHRA have been acceptably validated by the authors of the individual studies or by outside third parties. Summaries of the data validation reviews that have been conducted are presented in the EISR (Anchor and Windward 2008). Data validation reports for samples collected by EWG for the SRI are included in the data reports (Windward 2010b, c, d, 2009b). No additional data collection or validation is planned for this HHRA.

B.2.3.4 Documentation of field and laboratory practices

Documenting field and laboratory procedures makes it possible to assess the impact of any deviation from these procedures on data usability. As described in the EISR (Anchor and Windward 2008), such procedures were documented during the verification process that was conducted on historical data during database construction. A thorough review of the documentation (e.g., method descriptions, QC results) that was provided for the existing studies did not reveal any issues that would adversely affect the usability of the data for risk assessment purposes. Data collected for the SRI followed field and laboratory procedures that were approved by EPA and that were similar to those used during historical sampling events. Consistency in the field and laboratory procedures ensures compatibility among the various historical datasets and the data collected for the SRI and ensures their applicability for risk assessment.

B.3 Exposure Assessment

This exposure assessment describes scenarios in which people may come in contact with EW sediment-associated COPCs and provides equations and parameters so that potential exposures can be quantified. This section presents a summary of the CSM for the HHRA, the process for chemical screening and evaluation, including EPC calculation and COPC selection, details of the human health exposure scenarios, and the process for the calculation of chronic daily intake (CDI) values.

B.3.1 Conceptual Site Model and Selection of Scenarios for Quantification

A CSM is a graphical representation of exposure media, transport mechanisms, exposure pathways, exposure routes, and potentially exposed human populations. It provides the basis for developing exposure scenarios to be evaluated in the exposure assessment component of the HHRA.

B.3.1.1 Conceptual site model

A detailed version of the CSM for this HHRA was previously presented in the CSM and data gaps memorandum for the EW SRI (Anchor et al. 2008), which was approved by EPA. A summary is provided here to give context to the exposure scenarios described in Section B.3.3. A similar summary was also provided in the human health technical memorandum (Windward 2010f). The exposure assessment focuses only on scenarios that include a direct (i.e., ingestion or dermal contact) or indirect (i.e., consumption of fish or shellfish) pathway of exposure to chemicals in sediment, surface water, or biota in the EW (Figure B.3-1). Details on exposures scenarios, including the identification of receptors (e.g., children, adults, residents, workers) are provided in Section B.3.1.2. Section B.3.1.2 also presents the rationale for the selection or exclusion of specific scenarios for evaluation in the risk characterization. Full details of the specific scenarios developed for evaluation in the risk characterization are provided in Section B.3.3.

For each exposure pathway and media combination in the EW CSM, a determination was made in the CSM report (Anchor et al. 2008) as to whether the pathway is complete or incomplete. A complete exposure pathway includes the following components: an exposure medium, an exposure point, a potentially exposed population, and an exposure route. Pathways that do not include all four components are incomplete. Incomplete pathways should not be evaluated quantitatively in the risk assessment because without exposure, there is no risk. An example of an incomplete pathway for the EW is surface water as a source of drinking water for people. The saline conditions of the EW prevent this from being a complete pathway. Surface water exposure pathways are indirectly linked to sediment via flux from sediment to the water (Figure B.3-1). For simplicity, the inhalation pathway is not shown in Figure B.3-1 because it is considered insignificant (Anchor et al. 2008).

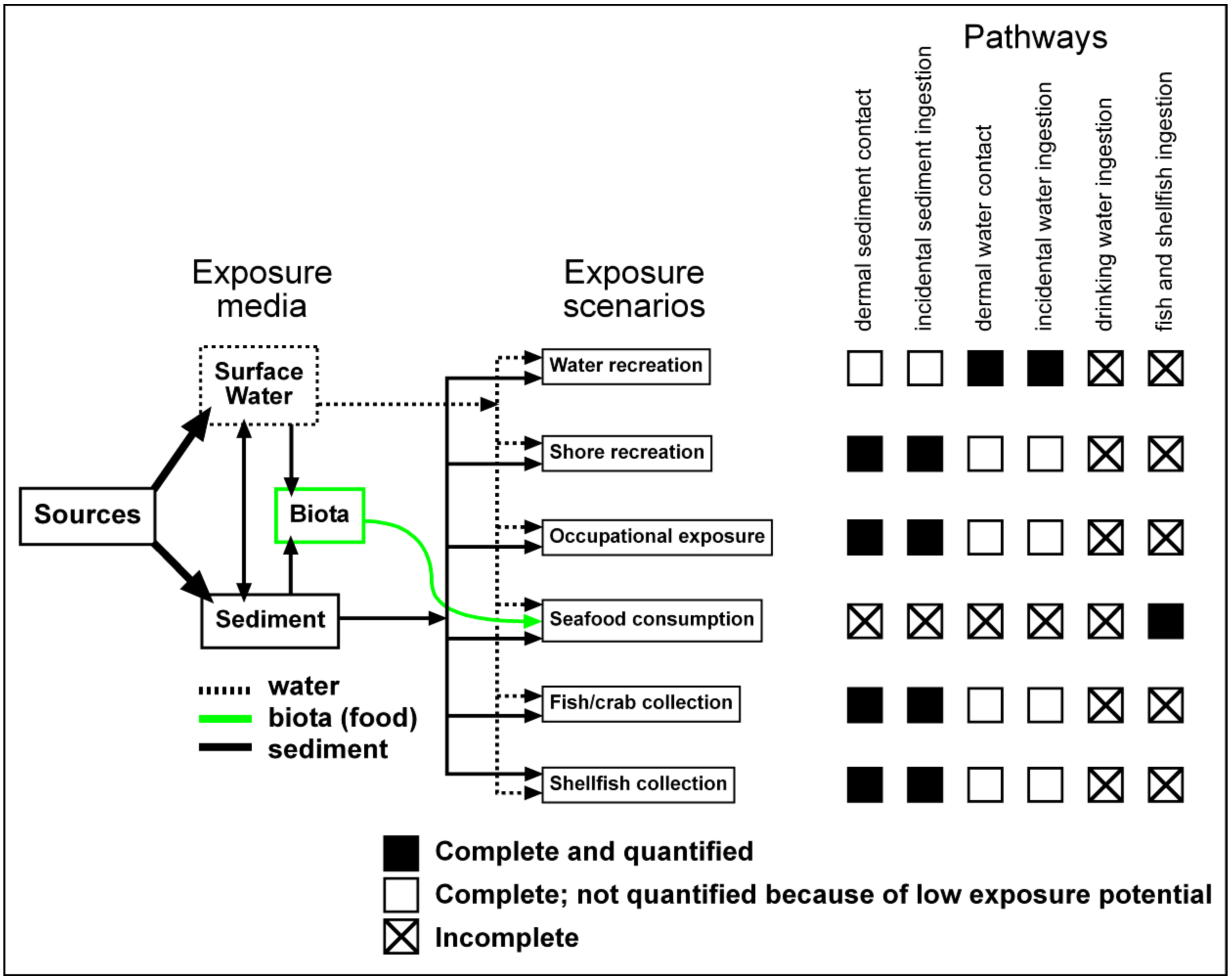

Figure B.3-1. Conceptual site model for the EW HHRA

The combined risk associated with all complete exposure pathways that have the greatest exposure potential have been evaluated in the HHRA. Some pathways identified as complete but with low exposure and risk potential relative to other evaluated pathways (e.g., exposure to water during shore clamming) are discussed qualitatively in the uncertainty analysis for risk communication purposes. The qualitative assessment of pathways with low exposure potential is appropriate because such pathways have minimal potential for causing excess risk or adverse health effects.

Six general exposure scenarios are presented in the HHRA CSM (Figure B.3-1). Detailed exposure parameters for each scenario evaluated are presented in Section B.3.3. Each exposure scenario involves at least one potential exposure pathway to contaminated sediment (e.g., dermal contact with sediment or incidental ingestion of sediment) or water and a potential exposure route through which contaminants can enter the body of an exposed individual (e.g., dermal absorption of contaminants through exposed skin surfaces or gastrointestinal absorption of ingested contaminants). However, the importance of some pathway and route combinations may be minor (i.e., low exposure potential), or the pathways may be incomplete. The scenarios presented are not mutually exclusive, and combinations of different pathways have been considered in this HHRA.

Several levels of exposure scenarios were used in the risk assessment to describe different intensities (e.g., frequency, magnitude, and duration) of site use or seafood consumption. The different levels of exposure provide a range of exposure and risk information for a given exposure scenario for use by the risk manager. This risk assessment includes four different levels of exposure. The RME level describes exposures well above the average but still within the range of possible exposure levels. EPA generally uses RME scenarios to evaluate the need for remedial actions at a site (EPA 1989). RME by definition likely overestimates exposure for many individuals (EPA 1989). Central tendency (CT) risk estimates are intended to reflect risk associated with average exposures.[21] Average exposure estimates are not favored in decision making because they will underestimate exposure for a substantial number of individuals (EPA 1989). A third level of exposure is the high-end level, which is likely to fall at or above the highest exposures that could occur. As determined through consultation between the tribes and EPA and for consistency with the LDW HHRA (Windward 2007c), the Suquamish seafood consumption scenario is presented in this context.[22] In addition, it should be noted that this scenario is presented because the EW is within the Usual and Accustomed (U&A) fishing areas of the Suquamish Tribe, and the Suquamish Tribe requested that their seafood consumption data be used to characterize risk in the HHRA. Finally, the unit exposure level was used to describe a baseline from which people can develop estimates of their own individual exposure potential. Unit exposures (e.g., one meal of seafood per month) are not intended to characterize any specific receptor and are presented for informational purposes. The 7-days-per-year clamming scenario may be used to characterize risk on a time frame that can be scaled to allow individuals to assess exposure and risk based on their unique behaviors.

For some pathways, both RME scenarios and CT scenarios were developed to describe some of the range of possible exposures and risks. The exposure parameters for each scenario and exposure level (i.e., RME, CT, high-end exposure, and unit exposure) are discussed in detail in Section B.3.3, and risk estimates for scenarios associated with each of these levels are presented in the risk characterization portion of the risk assessment.

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[21] In this HHRA, CT risk estimates were only evaluated for specific exposure populations (e.g., Asian and Pacific Islander consumption of seafood) as described in Sections B.3.3.1 and B.3.3.2. This HHRA did not attempt to provide CT risk estimates for the general population.

[22] Per the EPA tribal framework guidance for assessing tribal risks (EPA 2007b), the selection of seafood consumption scenarios is determined through consultation between the tribes and EPA. The selection of specific scenarios for evaluation at specific locations has no bearing on harvest treaty rights and has no implications regarding tribal harvest of seafood now or in the future.

B.3.1.2 Selection of exposure scenarios for quantification

Specific exposure assumptions were developed to quantify the complete pathways with significant exposure potential that are identified in Figure B.3-1. A complete exposure pathway includes an exposure medium, exposure point, a potentially exposed population (including age category [i.e., adult versus child]), and an exposure route. Separate scenarios for current and future land use were not evaluated for the following reasons:

- Future land use within the EW is not expected to differ greatly from current land use (Port of Seattle 2007). The use of the EW for commercial and industrial purposes is expected to continue into the foreseeable future, although certain recreational activities that are consistent with these land uses may be more common in the future as habitat improves.
- Because site-specific parameters based on current land use practices are not always available, reasonable maximum values were selected and were generally based on estimates of the potential or possible use of the area. Thus these parameters are also intended to account for potential future use; therefore, these values will overestimate current exposure but will provide information to risk managers to enable them to evaluate risk assuming increased site exposure in the future.
- Tribal harvest of seafood, as a treaty-reserved right, is now and will continue to be unrestricted.

Summing risks from multiple exposure pathways is reasonable if multiple pathways are relevant to the same person or group of people. EPA (1989) suggests that summing risks from multiple RME scenarios that do not occur simultaneously could be overly conservative. Several summed scenarios were assessed in the risk characterization (e.g., clamming and seafood consumption). Although CT scenarios for netfishing and seafood consumption are available, the netfishing RME scenario was summed with a seafood consumption RME scenario when risks across different exposure pathways were evaluated because these activities are not mutually exclusive, and both could be practiced by some individuals.

Details of each exposure pathway were provided in the CSM and data gaps report (Anchor et al. 2008) and the human health technical memorandum (Windward 2010f). Table B.3-1 summarizes the decision process for selecting exposure pathways for quantification. Details of each scenario evaluated in the HHRA are provided in Section B.3.3.

Table B.3-1. Rationale for the selection or exclusion of exposure pathways by exposure scenario

Exposure Point	Exposure Medium	Exposed Population	Age Category	Exposure Route	Type of Analysis	Rationale for Selection or Exclusion of Exposure Pathway
Water Recreation (e.g., swimming)						
Water recreation areas in the EW	sediment	resident	adult	dermal, ingestion[a]	qualitative	Exposure to sediment via swimming is lower than exposure via other pathways.
			child	dermal, ingestion[a]	qualitative	Exposure to sediment via swimming is lower than exposure via other pathways.
	surface water	resident	adult	dermal, ingestion[b]	numeric	The extent of swimming in the EW is unknown but expected to be low (King County 1999a). Potential exposure to surface water from swimming was evaluated using recently collected surface water data and exposure parameters developed by King County (1999a).
			child	dermal, ingestion[b]	qualitative	Swimming was evaluated only for adults. Children ages 6 and under are not expected to swim or be exposed to surface water in the EW, and thus exposure for children was not evaluated.
Shore Recreation (e.g., kayaking)						
Exposed EW intertidal areas	sediment	resident	adult	dermal, ingestion[a]	qualitative	There are no residential areas adjacent to or within a few blocks of EW public access areas (the closest residential area is approximately a half-mile away from the EW), and areas of tidally exposed sediment at public access locations are relatively small. The non-tribal clamming scenario is expected to be protective of any shore recreation activities.
			child	dermal, ingestion[a]	qualitative	There are no residential areas adjacent to or within a few blocks of EW public access areas, and areas of tidally exposed sediment at public access locations are relatively small.
	surface water	resident	adult	dermal, ingestion[b]	qualitative	Exposure attributable to resuspended sediment in the water column is insignificant compared with that from direct contact with bedded sediment. Exposure to dissolved chemicals in surface water while swimming is expected to be protective of exposure to surface water during shoreline recreation (e.g., kayaking).
			child	dermal, ingestion[b]	qualitative	Exposure attributable to resuspended sediment in the water column is insignificant compared with that from bedded sediment. Exposure is expected to be much lower than that evaluated for water recreation exposures.

Table B.3-1. Rationale for the selection or exclusion of exposure pathways by exposure scenario (cont.)

Exposure Point	Exposure Medium	Exposed Population	Age Category	Exposure Route	Type of Analysis	Rationale for Selection or Exclusion of Exposure Pathway
Occupational Exposure (e.g., habitat restoration)						
Industrial facilities adjacent to the EW	sediment	worker	adult	dermal, ingestion[a]	qualitative	Exposure is expected to be much lower than that evaluated in the habitat restoration sediment exposure scenario.
	surface water	worker	adult	dermal, ingestion[b]	qualitative	Exposure is expected to be much less than that evaluated for water recreation exposures.
Habitat restoration and EW cleanup locations	sediment	worker	adult	dermal, ingestion[a]	numeric	Workers engaged in habitat restoration or site cleanup projects (e.g., the proposed Bluefields project on the west side of the EW near the Spokane Street Bridge[c]) may come in contact with sediment. Risk estimates will help to identify what level of PPE is appropriate for these workers.
	surface water	worker	adult	dermal, ingestion[b]	qualitative	Exposure is expected to be much less than that evaluated for water recreation exposures.
Fish and Crab Collection						
Commercial netfishing locations in the EW, which potentially include all EW sediment	sediment	worker	adult	dermal, ingestion[a]	numeric	Commercial fishers are active at the site throughout the fishing season; nets contact the sediment.
	surface water	worker	adult	dermal, ingestion[b]	qualitative	Exposure attributable to surface water is insignificant compared with that from bedded sediment.
Fishing locations in the EW	sediment	resident	adult	dermal, ingestion[a]	qualitative	Exposure is difficult to quantify, and likely to be lower than occupational exposure. Incidental exposure during finfishing and crabbing is insignificant given the methods that are used.
	surface water	resident	adult	dermal, ingestion[b]	qualitative	Incidental exposure is insignificant.
Shellfish Collection						
Exposed EW intertidal areas	sediment	resident	adult	dermal, ingestion[a]	numeric	Clamming exposure scenarios were evaluated in the EW HHRA. Tribal members and members of the general public may participate in intertidal clamming activities in the EW now or in the future. The exposure area for non-tribal clamming was limited to areas with intertidal sediment that the public can access by foot. The exposure area for tribal clamming included all intertidal areas with exposed sediment.

Table B.3-1. Rationale for the selection or exclusion of exposure pathways by exposure scenario (cont.)

Exposure Point	Exposure Medium	Exposed Population	Age Category	Exposure Route	Type of Analysis	Rationale for Selection or Exclusion of Exposure Pathway
	surface water	resident	adult	dermal, ingestion[b]	qualitative	Exposure attributable to surface water is insignificant compared with that from bedded sediment. Exposure is expected to be much lower than that evaluated for water recreation exposures because clam collection occurs during low tides (and thus water contact is minimal).
Human Consumption of Resident Seafood						
Site-wide	resident fish and shellfish tissue (biota)	resident[d]	adult, child	ingestion	numeric	Tribal fish and shellfish consumption was evaluated based on consultation between EPA and the tribes. An API consumption scenario was evaluated using consumption rates derived from a recent survey of the API community (EPA 1999a; Kissinger 2005). Because information is not available to develop a recreational seafood consumption scenario, a one-meal-per-month consumption scenario was evaluated to provide individuals with a scalable tool to assess risks associated with their consumption habits.

[a] Incidental sediment ingestion associated with dermal contact.

[b] Incidental water ingestion associated with dermal contact.

[c] Bluefield Holdings is a Seattle-based ecological development company that specializes in NRD remediation. Bluefield habitat restoration projects on City-owned property along the Duwamish River and EW are expected to generate NRD credits that can be used to satisfy NRD liabilities for various entities. One of the proposed habitat restoration projects is located on the west side of the EW under the West Seattle Bridge, which would provide off-channel mudflat and marsh habitat, along with riparian vegetation. The project would also involve removal of debris and creosote structures from the shoreline areas. However, it should be noted that the Bluefield projects have no affect on the direct sediment exposure scenario assumptions.

[d] Resident may include APIs, tribal members, transients, or recreational fishers. Some of the different fish-consuming populations are described in Section B.3.3.1.

API – Asian and Pacific Islander

EPC – exposure point concentration

EPA – US Environmental Protection Agency

EW – East Waterway

HHRA – human health risk assessment

NRD – natural resource damage

PPE – personal protective equipment

For some scenarios, certain exposure pathways were not designated for quantitative analysis (e.g., sediment exposure for swimmers or surface water exposure for habitat restoration workers). In these cases, it was assumed that the exposure levels for these pathways under other scenarios were higher. Example calculations are presented in the uncertainty analysis to demonstrate how these exposures relate to scenarios that were evaluated quantitatively (Section B.6). However, a semi-quantitative evaluation of risk estimates was not performed for scenarios deemed not applicable to the EW (e.g., recreational activities such as dog walking along the shore).

B.3.2 CHEMICAL SCREENING AND EVALUATION

In order to focus the risk assessment on EW COPCs, a screening step was performed using general exposure pathways based on the CSM. A comprehensive set of chemicals was analyzed in sediment, water, and tissue collected from the EW, as described in Section B.2. In accordance with EPA guidelines (1996a), a risk-based screening was conducted on these data to determine which chemicals should be quantitatively evaluated in the HHRA. The screening process resulted in the identification of COPCs to be evaluated in the risk assessment. Exposures associated with COPCs were evaluated quantitatively for the scenarios described in Section B.3.3.

The decision process for identifying COPCs is shown in Figure B.3-2. This is similar to the process that was used in the LDW HHRA (Windward 2007c) for sediment and tissue. For detected chemicals with regional screening levels (RSLs) developed by EPA for Superfund sites (EPA 2010b),[23] the maximum detected concentration was compared with the applicable RSL (Step 3a). RLs were also compared with RSLs for chemicals with non-detected values that had maximum detected concentrations that did not exceed the RSLs, as shown in Figure B.3-2 (Steps 4a and 4b). If a chemical was detected in more than 10% of the samples, and those detected values never exceeded the RSL, the chemical was excluded from further analysis regardless of whether the RL for the non-detects exceeded the RSL. For those chemicals with a detection frequency less than 10%, the number of times the RL exceeded the RSL was determined (the right side of Figure B.3-2; Step 4b). If RLs exceeded the RSL with a frequency greater than 10% (Step 4b), the chemical was retained as a COPC because of uncertainty that the RSL may be exceeded. Risks related to COPCs identified based on RLs greater than RSLs alone were evaluated in the uncertainty analysis of the risk assessment. Chemicals without RSLs were not screened or quantitatively evaluated but are discussed in the uncertainty analysis (Section B.6.2.1). Details on the selection of RSLs by media are discussed in Sections B.3.2.1 through B.3.2.3.

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[23] The LDW HHRA was completed prior to the creation of EPA's RSLs (EPA 2010b). For the LDW HHRA, regional preliminary remediation goals (PRGs) were used to screen COIs. EPA's RSLs include many of the regional PRGs. The differences between the EW screening process is discussed for sediment in Section B.3.2.1 and for tissue in Section B.3.2.2.

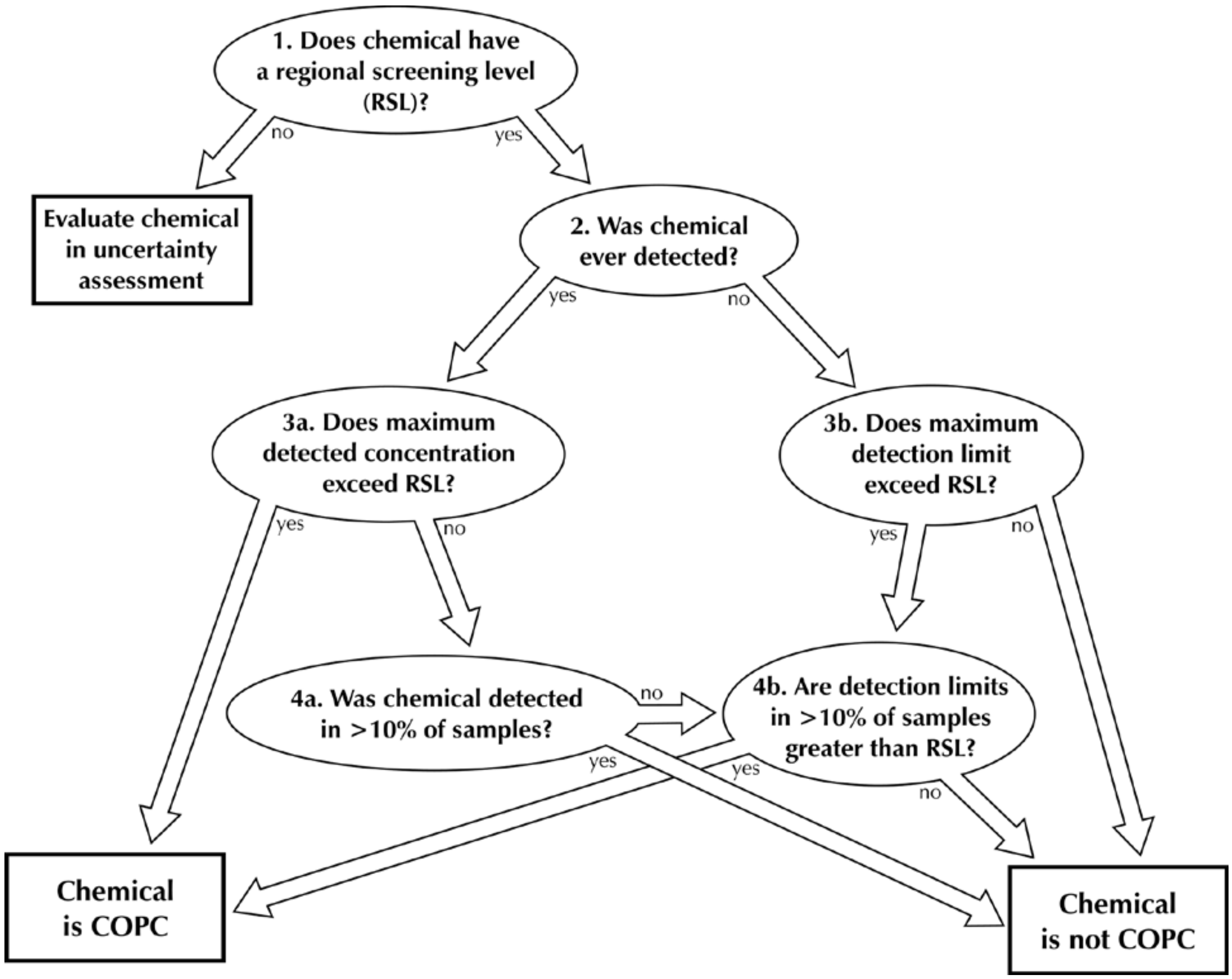

Figure B.3-2. COPC identification flow chart

Some chemicals (e.g., cPAHs and PCDDs/PCDFs) were evaluated in the COPC screening process as groups, rather than as individual compounds, using the TEQ approaches described in Section B.2.2.4. Screening was conducted separately for sediment, seafood consumption, and water exposure scenarios. Tables describing the occurrence and selection of COPCs are provided in Attachment 2. Human health COPCs identified for the EW are also compared with those identified in the LDW HHRA in Table 8 of Attachment 2.

B.3.2.1 Screening sediment data

In addition to the use of different RSLs, the exposure areas evaluated for each of the sediment exposure scenarios are also different. The entire EW was assumed to be the exposure area for the netfishing scenario, but the exposure area for the habitat restoration worker and intertidal clamming scenarios was limited in extent to the accessible intertidal portion of the EW (i.e., intertidal area not under piers and thus accessible by boat or from the shore). Thus, scenario-specific COPC lists were generated for the sediment exposure scenarios.

Typically, the COPC screen is a comparison of the maximum empirical value with the appropriate RSL. For intertidal sediment exposure areas, the screening was slightly more complex. As discussed in Section B.3.3.4.2, MIS provides an efficient method for estimating the mean of a population but does not allow for the estimation of the extremes of the dataset. Therefore, for COPC screening associated with the intertidal exposure areas, 95% upper confidence limits on the mean (UCLs) were calculated based on the three area-wide intertidal MIS samples (habitat restoration worker and tribal clamming scenarios) or the one public access intertidal MIS sample (7-days-per -year clamming scenario) for all chemicals. The method for calculating 95% UCLs for the MIS samples, as agreed upon with EPA (Kissinger 2010; Windward 2010h), is presented in Section B.3.3.4.2. When there are few samples, the estimated 95% UCL may exceed the maximum measured concentration, as was the case for most of the chemicals in the MIS samples. To ensure that the COPC screen was sufficiently health protective, the highest value of the maximum empirical concentration, the maximum reporting limit, or the MIS 95% UCL was compared with the appropriate RSL for each chemical.

Tables 1 through 4 in Attachment 2 compare the maximum sediment concentrations for each chemical (or the MIS 95% UCL, if higher than the maximum concentration or maximum reporting limit) with the applicable RSL and include summary statistics, such as detection frequency, minimum detected concentration, and range of RLs. For the netfishing scenario, data for subtidal and intertidal sediment were compared in the screening because nets may come into contact with sediment at both water depths. Only intertidal data sediment data were used for screening for the habitat restoration worker and clamming scenarios because contact with subtidal sediments is considered to be unlikely for those scenarios. Note that the three area-wide intertidal MIS samples were used for the habitat restoration worker and the tribal clamming scenarios, whereas only the one public access MIS sample was used for the 7-days-per-year clamming scenario. The COPCs for the direct sediment exposure scenarios are identified in Table B.3-2, which summarizes the results of Attachment 2.

Based on detected concentrations, nine chemicals or chemical groups were identified as COPCs for the netfishing scenario, five chemicals or chemical groups were identified as COPCs for the habitat restoration worker scenario, eight chemicals or chemical groups were identified as COPCs for the tribal clamming scenario, and seven chemicals or chemical groups were identified as COPCs for the 7-days-per-year clamming scenario. The different numbers of COPCs for the various sediment exposure scenarios was the result of the different RSLs that were used (i.e., industrial RSLs were used for the netfishing and habitat restoration worker scenarios while residential RSLs were used for the clamming scenarios) and the different exposure areas (i.e., site-wide for netfishing, all intertidal areas for the habitat restoration worker and tribal clamming scenarios, and only the public access intertidal areas for the 7-days-per-year clamming scenario).

Table B.3-2. Identification of COPCs for sediment exposure scenarios in the EW

Chemical	Netfishing		Habitat Restoration Worker		Tribal Clamming		Clamming – 7 Days per Year (Public Access only)	
	COPC?	Rationale	COPC?	Rationale	COPC?	Rationale	COPC?	Rationale
Detected Chemicals								
Antimony[a]	yes	maximum detect > RSL	nd		nd		nd	
Arsenic	yes	maximum detect > RSL	yes	MIS 95% UCL > RSL	yes	MIS 95% UCL > RSL	yes	MIS 95% UCL > RSL
Cobalt	no		no		yes	MIS 95% UCL > RSL	yes	MIS 95% UCL > RSL
Lead	yes	maximum detect > RSL	no		yes	MIS 95% UCL > RSL	no	
Vanadium	yes	maximum detect > RSL	yes	MIS 95% UCL > RSL	yes	MIS 95% UCL > RSL	yes	MIS 95% UCL > RSL
cPAH TEQ	yes	maximum detect > RSL	yes	MIS 95% UCL > RSL	yes	MIS 95% UCL > RSL	yes	MIS 95% UCL > RSL
1,4-Dichlorobenzene	yes	maximum detect > RSL	no		no		no	
Total PCBs	yes	maximum detect > RSL	yes	MIS 95% UCL > RSL	yes	MIS 95% UCL > RSL	yes	MIS 95% UCL > RSL
PCB TEQ	yes	for consistency with total PCBs[b]	yes	for consistency with total PCBs[b]	yes	MIS 95% UCL > RSL	yes	for consistency with total PCBs[b]
Dioxin/furan TEQ	yes	maximum detect > RSL	no		yes	MIS 95% UCL > RSL	yes	MIS 95% UCL > RSL
Undetected Chemicals								
Antimony[a]	na		no		yes	3 of 3 RLs > RSL	yes	1 of 1 RL > RSL
Toxaphene	no		no		yes	MIS 95% UCL[c] > RSL	yes	MIS 95% UCL[c] > RSL
n-Nitroso-dimethylamine	no		no		yes	MIS 95% UCL[c] > RSL	yes	MIS 95% UCL[c] > RSL

a Antimony is listed in this table twice because it was detected in subtidal sediment (part of the netfishing exposure area) but not in intertidal sediment (the extent of the habitat restoration worker and clamming scenarios).

b PCB TEQ did not screen in for these scenarios, but because total PCBs did screen in, PCB TEQ risks were evaluated in risk characterization.

c MIS 95% UCL based on non-detected concentrations.

COPC – chemical of potential concern
cPAH – carcinogenic polycyclic aromatic hydrocarbons
EPC – exposure point concentration
EW – East Waterway
MIS – multi-increment sampling
nd – not detected in this exposure area
PCB – polychlorinated biphenyl
RL – reporting limit
TEQ – toxic equivalent

In addition to the COPCs identified based on detected concentrations, three chemicals were identified as COPCs based on reporting limits for the clamming scenarios (i.e., more than 10% of reporting limits were greater than the RSL). No chemicals were identified as COPCs based on reporting limits for the netfishing or habitat restoration worker scenarios. Risks for chemicals that were identified as COPCs based on non-detected concentrations were evaluated in the uncertainty analysis (Section B.6).

The full list of comparisons of sediment concentrations with RSLs is provided in Tables 1 through 4 of Attachment 2. Many of the chemicals in these tables were not selected as COPCs for sediment exposure because of the lack of RSLs. These chemicals are qualitatively evaluated in the uncertainty analysis (Section B.6).

B.3.2.2 Screening tissue data

COPCs in fish and shellfish tissue were identified by comparing EW concentrations with RSLs for fish tissue developed using EPA's RSL calculator (EPA 2009d). Default exposure factors for fish RSLs include: target HQ = 1, target excess cancer risk = 10^{-6}, body weight = 70 kg, exposure frequency = 350 days per year, exposure duration = 30 years, and fish ingestion rate = 54 g/day (EPA 2009d). These exposure factors are consistent with Region 10 guidance for performing risk assessments (EPA 1996a), with the exception of the target HQ. Region 10 recommends a target HQ of 0.1 to account for cumulative effects from multiple chemicals and pathways. RSLs for chemicals with non-carcinogenic effects were therefore decreased by a factor of 10 (using the option of the RSL calculator to change default values) to be consistent with guidance from EPA Region 10.

In addition to the modification described above for target HQ, the RSLs for both carcinogenic and non-carcinogenic endpoints were modified using the RSL calculator (EPA 2009d) to account for site-specific tribal exposure assumption differences in consumption rate (97.5 g/day vs. 54 g/day; see Section B.3.3), exposure frequency (365 days vs. 350 days), body weight (81.8 kg vs. 70 kg), and exposure duration (70 years vs. 30 years).[24]

Table 5 in Attachment 2 compares the maximum tissue concentrations for each chemical with the applicable RSL and includes summary statistics, such as detection frequency, minimum detected concentration, and range of RLs. COPC screening was performed using the combined tissue dataset summarized in Table 5 of Attachment 2 rather than by tissue type (e.g., benthic fish fillets, crab whole body). In other words, a single detected concentration in any tissue sample greater than the applicable RSL could result in a chemical being designated as a COPC, regardless of tissue type. The COPCs for the seafood consumption scenarios are identified in Table B.3-3, which summarizes the results of Attachment 2.

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[24] The fish tissue RSLs are an update of the Region 3 risk-based concentrations (RBCs) for fish tissue (EPA 2005b). The Region 3 RBCs were used for screening the LDW fish tissue (Windward 2007c) with the same adjustments to HQ, exposure frequency, exposure duration, and body weight.

Table B.3-3. Identification of COPCs for seafood consumption scenarios in the EW

Chemical	Rationale
Detected Chemicals	
Antimony	maximum detect > RSL
Arsenic (inorganic)[a]	maximum detect > RSL
Cadmium	maximum detect > RSL
Chromium	maximum detect > RSL
Cobalt	maximum detect > RSL
Copper	maximum detect > RSL
Lead[b]	maximum detect > RSL
Mercury	maximum detect > RSL
Molybdenum	maximum detect > RSL
Selenium	maximum detect > RSL
Vanadium	maximum detect > RSL
Zinc	maximum detect > RSL
Dibutyltin as ion	maximum detect > RSL
Tributyltin as ion	maximum detect > RSL
cPAH TEQ	maximum detect > RSL
1,4-Dichlorobenzene	maximum detect > RSL
Pentachlorophenol	maximum detect > RSL
Total PCBs	maximum detect > RSL
PCB TEQ	maximum detect > RSL
Total DDTs	maximum detect > RSL
Dieldrin	maximum detect > RSL
alpha-BHC	maximum detect > RSL
beta-BHC	maximum detect > RSL
Total chlordane	maximum detect > RSL
Heptachlor[c]	22 of 23 reporting limits > RSL
Heptachlor epoxide	maximum detect > RSL
Mirex	maximum detect > RSL
Dioxin/furan TEQ	maximum detect > RSL
Undetected Chemicals	
BEHP	21 of 57 reporting limits > RSL
Butyl benzyl phthalate	21 of 98 reporting limits > RSL
1,2,4-Trichlorobenzene	92 of 98 reporting limits > RSL
1,2-Diphenylhydrazine	6 of 6 reporting limits > RSL
2,4,6-Trichlorophenol	98 of 98 reporting limits > RSL
2,4-Dichlorophenol	90 of 98 reporting limits > RSL
2,4-Dinitrophenol	92 of 98 reporting limits > RSL
2,4-Dinitrotoluene	98 of 98 reporting limits > RSL
2,6-Dinitrotoluene	92 of 98 reporting limits > RSL

Table B.3-3. Identification of COPCs for seafood consumption scenarios in the EW (cont.)

Chemical	Rationale
2-Nitroaniline	74 of 98 reporting limits > RSL
4,6-Dinitro-o-cresol	98 of 98 reporting limits > RSL
4-Chloroaniline	88 of 91 reporting limits > RSL
4-Nitroaniline	94 of 97 reporting limits > RSL
Aniline	70 of 97 reporting limits > RSL
Bis(2-chloroethoxy)methane	43 of 98 reporting limits > RSL
Bis(2-chloroethyl)ether	98 of 98 reporting limits > RSL
Hexachlorobenzene	98 of 98 reporting limits > RSL
Hexachlorobutadiene	18 of 98 reporting limits > RSL
Hexachlorocyclopentadiene	83 of 98 reporting limits > RSL
Hexachloroethane	81 of 98 reporting limits > RSL
n-Nitroso-di-n-propylamine	98 of 98 reporting limits > RSL
n-Nitrosodimethylamine	88 of 98 reporting limits > RSL
n-Nitrosodiphenylamine	73 of 98 reporting limits > RSL
Nitrobenzene	74 of 98 reporting limits > RSL
Aldrin	23 of 23 reporting limits > RSL
Toxaphene	95 of 95 reporting limits > RSL

[a] Both arsenic and inorganic arsenic maximum concentrations were greater than the RSL. Because inorganic arsenic is more toxic to humans and is the basis for the toxicity values, inorganic arsenic was evaluated in this HHRA. A discussion of arsenic and inorganic arsenic is presented in Section B.6.2.6.

[b] Unlike other COPCs, lead, after screening in as a COPC based on the comparison with the RSL, was evaluated using the lead models for adults and children. This approach is consistent with EPA recommendations and is discussed further in Section B.3.3.5.

[c] One sample of heptachlor was detected in one mussel sample collected from the EW at a concentration of 0.1 J µg/kg ww (less than the RSL of 0.186 µg/kg ww). However, 22 of 23 reporting limits were greater than the RSL, and thus heptachlor was identified as a COPC. The risks for heptachlor are discussed in the uncertainty analysis (Section B.6).

BEHP – bis(2-ethylhexyl) phthalate
BHC – benzene hexachloride
COPC – chemical of potential concern
cPAH – carcinogenic polycyclic aromatic hydrocarbon
EPA – US Environmental Protection Agency
EW – East Waterway
PCB – polychlorinated biphenyl
RSL – regional screening level
TEQ – toxic equivalent
ww – wet weight

A total of 28 chemicals or chemical groups were identified as COPCs for the seafood consumption scenarios based on detected concentrations, including 12 metals, 2 organometals, cPAH TEQ, 2 other SVOCs, PCBs, 8 pesticides, and dioxin/furan TEQ. In addition, a total of 26 undetected chemicals were identified as COPCs based on the RLs. Risks for chemicals identified as undetected COPCs are evaluated in the uncertainty analysis (Section B.6).

In addition to COPCs identified from the analysis of tissue samples from the EW, chemicals analyzed in sediment but never analyzed in tissue were also evaluated to determine whether they should be selected as COPCs for the seafood consumption scenarios. A total of 10 chemicals were analyzed in EW sediment but were never analyzed in tissue samples. These chemicals are listed in Table 6 of Attachment 2. Of these chemicals, all 10 were detected in sediment. However, none of the chemicals were identified as sediment COPCs, and none were identified by EPA (2000a) as potential bioaccumulative compounds. Thus, they would not be expected to be detected at elevated concentrations in seafood tissue from exposure to sediment in the EW. Therefore, none of the chemicals listed in Table 6 of Attachment 2 were identified as COPCs for tissue.

B.3.2.3 Screening surface water data

For surface water, data were screened against a modified version of EPA's RSLs for tap water (EPA 2010b). EPA's tap water RSLs are based on an assumed ingestion rated of 2 L/day. To account for the fact that water exposure through swimming would be lower than the assumed exposure used to develop EPA's tap water RSLs, these RSLs were multiplied by 10, as specified in the HHRA technical memorandum (Windward 2010f). These modified water RSLs were then used to screen EW water data.

Table 7 in Attachment 2 compares the maximum surface water concentrations for each chemical with the applicable RSL and includes summary statistics, such as detection frequency, minimum detected concentration, and range of RLs. The COPCs for the surface water exposure scenarios are identified in Table B.3-4, which summarizes the results of Attachment 2.

Table B.3-4. Identification of COPCs for the surface water exposure scenario in the EW

Chemical	Rationale
Detected Chemicals	
Arsenic (total)	maximum detect > RSL
Chromium (total)	maximum detect > RSL
Vanadium (total)	maximum detect > RSL
Naphthalene	maximum detect > RSL
Total PCBs	maximum detect > RSL
PCB TEQ	for consistency with total PCBs[a]
Undetected Chemicals	
Benzo(a)pyrene	4 of 28 reporting limits > RSL
2,4-Dinitrotoluene	28 of 28 reporting limits > RSL
3,3'-Dichlorobenzidine	28 of 28 reporting limits > RSL
4,6-Dinitro-o-cresol	28 of 28 reporting limits > RSL
4-Chloroaniline	28 of 28 reporting limits > RSL
Bis(2-chloroethyl)ether	28 of 28 reporting limits > RSL
Hexachlorobenzene	28 of 28 reporting limits > RSL
n-Nitroso-di-n-propylamine	28 of 28 reporting limits > RSL
n-Nitrosodimethylamine	24 of 24 reporting limits > RSL

[a] PCB TEQ did not screen in for this scenario; but because total PCBs did screen in, PCB TEQ risks were evaluated in the risk characterization.

COPC – chemical of potential concern
EW – East Waterway
PCB – polychlorinated biphenyl
RSL – regional screening level
TEQ – toxic equivalent

A total of six chemicals or chemical groups were identified as COPCs based on detected concentrations, including three trace elements, naphthalene, total PCBs, and PCB TEQ.

In addition to the COPCs identified based on detected concentrations, a total of nine chemicals were identified as COPCs based on reporting limits. Risks for chemicals that were identified as COPCs based on non-detected concentrations are evaluated in the uncertainty analysis (Section B.6).

B.3.3 Selection of Exposure Parameters and Calculation of Chronic Daily Intake

The exposure scenarios quantified in this HHRA are summarized in Sections B.3.3.1 (seafood ingestion), B.3.3.2 (sediment exposure), and B.3.3.3 (surface water exposure). Each section includes summary tables that contain key exposure parameters,[25] so that the scenarios can be compared with each other, and detailed tables in which all exposure parameters for each scenario are presented. EPCs were calculated as chemical concentrations in tissue, sediment, and surface water for the various exposure scenarios following the procedure outlined in Section B.3.3.4. The exposure parameters are used in conjunction with the EPCs calculated for each COPC to estimate CDIs.

CDIs represent the estimated daily chemical dose for an individual over the exposure duration for each scenario. Separate CDIs were calculated for chemicals with carcinogenic and non-carcinogenic effects because the averaging times over which the doses are calculated are different. The CDI results were used in the risk characterization and uncertainty analysis.

B.3.3.1 Seafood exposure scenarios and parameters

The EW is an important commercial salmon fishery. However, because of the migratory nature of salmon, bioaccumulative chemicals found in adult salmon tissue are largely the result of exposures from the Pacific Ocean and Puget Sound, where they spend most of their life, rather than chemicals present in the EW. The contribution of exposures to adult body burdens in the EW is likely to be insignificant because the large majority of a salmon's growth occurs in open marine waters (O'Neill et al. 1998). An analysis presented by O'Neill et al. (1998) indicated that less than 1% of the PCB body burden of adult salmon migrating through the LDW could have been obtained from prey items consumed in the LDW. Similarly, contributions to salmon PCB body burden attributable to the EW would be expected to be minimal. Therefore, salmon consumption was not included in this risk assessment, consistent with the HHRA for the LDW. The uncertainty analysis includes a discussion of uncertainties in the risk estimates resulting from the exclusion of the consumption of salmon.

[25] Summary tables are provided for seafood consumption and direct sediment exposure scenarios. No summary table is provided for swimming because only one scenario was evaluated.

Seafood in the EW is collected by tribal members and the general public. Survey information suggests that other populations with relatively high seafood consumption rates may use the EW for at least part of their seafood collection (EPA 1999a; King County 1999a). A total of eight scenarios, including RME and CT scenarios, were developed and parameterized to represent a range of potential exposures via the consumption of EW seafood by different groups.

EPA Region 10 has developed tribal seafood consumption scenarios for application to CERCLA and Resource Conservation and Recovery Act (RCRA) sites in Puget Sound and the Strait of Georgia based on seafood consumption studies of the Tulalip Tribes and the Suquamish Tribe (EPA 2007b). In this document, EPA specifies tribal consumption rates for each type of seafood (i.e., seafood category) that are to be used as a starting point for consultation with tribes and negotiations with PRPs. The adult tribal seafood consumption RME scenario follows the approach in the LDW HHRA (Windward 2007c), in which the Tulalip seafood consumption survey data were used to characterize adult tribal RME and child tribal RME seafood consumption. However, it should be noted that tribal concurrence regarding the use of the Tulalip consumption rates for this HHRA was not obtained.[26] Rather, the use of the Tulalip rates represents an EPA policy decision (neither the LDW nor the EW is part of the Tulalip Tribes' U&A fishing areas).

EPA Region 10 developed a framework to promote internal consistency in Puget Sound tribal seafood consumption risk assessments (EPA 2007b). The framework used available Puget Sound tribal seafood consumption information to develop RME tribal seafood consumption rates for risk assessment for RCRA and CERCLA sites in the Puget Sound region. EPA Region 10 made the policy decision to use the quantity of current or potential high-quality shellfish habitat to determine which tribal seafood consumption dataset was the most appropriate. For sites in the vicinity of large quantities of current or potential high-quality shellfish habitat, EPA advocates the use of the Suquamish Tribe's seafood consumption rate to characterize risk. In general, EPA advocates the use of the Tulalip tribal consumption rate in other cases. For the EW, given the limited quantity of current or potential shellfish habitat (particularly high-quality habitat), the aforementioned criterion leads to the use of the Tulalip tribal rate to characterize the RME seafood consumption risks in the EW, as approved by EPA (Windward 2010f).

The Suquamish and Muckleshoot Tribes, in consultation with EPA, currently recognize that background sediment concentrations for COCs, such as PCBs, will likely be above the risk-based threshold concentration for sediment (based on a 1×10^{-6} excess cancer risk) derived using Tulalip consumption rates for both the LDW and EW sites. The use

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[26] The Suquamish Tribe requested that the tribal RME scenario be represented as a range of exposure based on the Tulalip and Suquamish consumption rates. The Suquamish Tribe's acquiescence regarding the use of the Tulalip tribal consumption rates is site-specific. The Suquamish Tribe does not regard the use of the Tulalip tribal rates as precedent-setting.

of the higher Suquamish consumption rates would result in even lower, unattainable risk-based threshold concentrations that would have no measurable impact on cleanup levels when the risk-based threshold concentration is below background. Hence the Suquamish and Muckleshoot Tribes did not oppose the use of Tulalip consumption rates to characterize RME seafood consumption risks for the EW.

An additional tribal scenario was evaluated in this HHRA based on Suquamish seafood consumption survey data (Suquamish Tribe 2000) per requests from the Suquamish and Muckleshoot Tribes. This scenario is consistent with the LDW HHRA (Windward 2007c). This scenario provided a high-end exposure to characterize a range of tribal consumption rates.

As was done in the LDW HHRA (Windward 2007c), risk estimates for the Suquamish scenario are presented as quantitative estimates and discussed in the risk characterization section of the risk assessment. The Suquamish Tribe believes that the children's consumption rates presented in the Suquamish survey are valid and relevant to children of the Suquamish Tribe. For consistency with the LDW HHRA (Windward 2007c), the tribe agreed that estimates of risk to Suquamish children would be included in the uncertainty analysis section of the HHRA for the EW.

In summary, the following five tribal scenarios were developed for evaluation in the risk characterization for this HHRA: adult tribal scenarios (RME and CT) based on Tulalip data, child tribal scenarios (RME and CT) based on Tulalip data, and an adult tribal scenario based on Suquamish data, as summarized in Sections B.3.3.1.1 and B.3.3.1.2. A review and interpretation by EPA of the two tribal consumption studies provided the basis for the tribal scenarios presented here (EPA 2007b; Hiltner 2007). These are the same five tribal scenarios that were evaluated in the LDW HHRA. The selection of consumption scenarios for evaluation in the EW HHRA does not set a precedent for other sites, and consultation between EPA and the tribes will be necessary to select seafood consumption rates for other sites.

A seafood consumption survey prepared for the King County Water Quality Assessment (King County 1999a) verified that fish and crab were being harvested within and near the EW by the public. Specifically, seafood harvesting was reported to have occurred from the Spokane Street Bridge and Jack Perry Memorial Shoreline Public Access. The Spokane Street Bridge location was identified as the third most popular location for seafood harvest of the Elliott Bay and LDW locations included in the survey. Crabs were collected by more people than any other species. The number of individuals who collected sole was a third of the number of individuals who collected crabs. The King County survey also documented that a substantial fraction of Duwamish/Elliott Bay anglers are Asians and Pacific Islanders (APIs). Guidance for the development of API consumption scenarios (RME and CT) based on a King County survey (EPA 1999a) was also provided by EPA (Kissinger 2005), and details on these scenarios are presented in Section B.3.3.1.3.

Finally, to provide risk information for the general public and risk information on individual resource types, a seafood consumption scenario that considered the consumption of a single meal per month was developed of each of the following seafood categories: pelagic fish (both perch and rockfish), benthic fish, crabs, and clams (Section B.3.3.1.4). This scenario is not based on any specific fish consumption survey and is instead intended to provide additional information for less frequent (i.e., one meal per month) seafood consumers on a resource-by-resource basis. It can also be readily scaled to individual consumption rates.

This section provides a summary of the CDI calculation for COPCs for the ingestion of seafood as well as details on the exposure parameters used to evaluate each seafood consumption scenario. Equation 3-1 shows the generic CDI equation used for the ingestion of seafood. The scenario-specific equations using the seafood consumption categories relevant to each scenario are presented in Section B.3.3.1.5.

$$CDI_o = \frac{EPC \times IR \times FI \times EF \times ED \times CF}{BW \times AT}$$

Equation 3-1

Where:

CDI_o	=	chronic daily intake from oral exposure route (mg/kg-day)
EPC	=	chemical-specific exposure point concentration (mg/kg)
IR	=	seafood ingestion rate (g/day)
FI	=	fractional intake of media derived from contaminated source (unitless)
EF	=	exposure frequency (days per year)
ED	=	exposure duration (years)
CF	=	conversion factor (kg/g)
BW	=	body weight (kg)
AT	=	averaging time (days), equivalent to the exposure duration for non-carcinogenic COPCs and 70 years for carcinogenic COPCs

Detailed explanations of the scenarios and their development are provided in Sections B.3.3.1.1 through B.3.3.1.4, and tables with all exposure parameters needed to calculate CDIs are provided in Section B.3.3.1.5. For all seafood consumption scenarios, the exposure unit is assumed to be the entire EW study area (i.e., each scenario is based on the consumption of fish and shellfish caught or collected throughout the EW study area).

B.3.3.1.1 Adult tribal seafood consumption scenarios based on Tulalip and Suquamish data

The consumption rates in the tribal framework (EPA 2007b) are based on seafood consumption surveys of the Tulalip Tribes (Toy et al. 1996) and the Suquamish Tribe (Suquamish Tribe 2000). Briefly, the 95th percentile of total seafood consumption from Puget Sound was attributed to different seafood categories (anadromous, bottom feeding, and pelagic fish, as well as shellfish) assuming the proportion of consumption in each category calculated for average consumption (including both consumers and non-consumers) also applied to the 95th percentile consumption of Puget Sound seafood. For example, the average consumption of anadromous fish divided by the sum of the averages of consumption of all seafood categories was 49.7%. Thus, it was assumed that 49.7% of the 95th percentile of total seafood consumed from Puget Sound by Tulalip Tribal members (194 g/day) was anadromous fish (96.4 g/day) (EPA 2007b). The same approach was applied for estimating the consumption of different seafood categories for the adult Tulalip CT scenario using the 50th percentile of total seafood consumed from Puget Sound (Hiltner 2007). Total quantities of non-anadromous seafood consumed for the tribal adult scenario based on Tulalip data were 97.5 g/day and 15 g/day for the RME and CT scenarios, respectively. Total non-anadromous seafood consumed for the tribal adult scenario based on Suquamish data was 583.5 g/day.

Table B.3-5 presents the tribal seafood consumption rates used for different components of the market basket, which is defined as the specific quantities and types of seafood consumed. The last column discusses the presence and prevalence of each seafood group in the EW.

As agreed upon with EPA, consumption of anadromous fish was not included for EW tribal exposure and risk estimates (EPA 2005a) because the bulk of the body burden of bioaccumulative contaminants in adult salmon is not obtained from the EW. Because the site-related contaminant body burden is low, most risks associated with salmon consumption were deemed not to be site-related.

The consumption of different types of shellfish within the shellfish seafood category for the adult tribal RME scenario based on Tulalip data and the adult tribal scenario based on Suquamish data was specified by EPA in the application of their framework to the LDW (EPA 2005a, 2009b). The same approach was applied for the EW. The species-specific information was used together with concentration data for that species (where available) in the market basket estimate. The same methodology was applied to develop the adult tribal CT scenario based on Tulalip data. Briefly, average consumption rates (for consumers and non-consumers) of clams, mussels, and crabs were calculated and used by EPA to develop concentration weighting factors that could be applied to the shellfish seafood category. Using the adult tribal RME clam consumption rate based on Tulalip data as an example, average clam consumption was 48% of the sum of averages of other shellfish consumed (clams, mussels, crabs, and geoduck). This percentage was applied to the adult tribal shellfish consumption rate (81.9 g/day, 95th percentile of Puget Sound shellfish consumption) to generate a clam consumption rate of 39.3 g/day for the adult tribal RME scenario based on Tulalip data. Similar procedures were used to develop consumption rates for the adult tribal CT scenario based on Tulalip data and for the adult tribal scenario based on Suquamish data. Table B.3-6 presents the concentration weighting factors (as percentages) for clams, mussels, and crabs and the calculated consumption of each of the adult tribal RME and CT scenarios based on Tulalip data and adult tribal scenario based on Suquamish data.

Table B.3-5. Seafood species consumed by Tulalip and Suquamish adults and EW species used to represent consumed species

Seafood Category	Members	Grams per Day: Adult Tulalip RME[a]	Grams per Day: Adult Tulalip CT[b]	Grams per Day: Adult Suquamish[c]	Rationale for Inclusion/Exclusion and Representative Species Present in the EW
Anadromous fish	salmon	96.4	14.9	183.5	Consumption rate was not used in this HHRA. Although adult salmon are common in EW, they were not included in the EW HHRA because of their migratory behavior (i.e., only a brief portion of their life is spent in the EW).
Pelagic fish	including cod, perch, and rockfish	8.1	1.3	56	Perch and rockfish are common in the EW.
Benthic/ demersal fish	halibut, sole, snappers	7.5	1.2	29.1	English sole are common in the EW.
Shellfish	bivalves,[d] snails, shrimp, crabs	81.9	12.5	498.4	Marine shellfish species (crabs, clams, and mussels) are present in the EW. As part of the 2008 sampling effort, only 26 shrimp were collected, which allowed for the analysis of only one composite shrimp sample. The small number of shrimp captured indicates that there are likely insufficient shrimp of harvestable size (i.e., most are too small to be captured by typical nets or cages) present in the EW to constitute a significant portion of seafood consumption for the scenarios evaluated in this HHRA.

a From Table B-1 of EPA (2007b), 95th percentile of the total seafood consumption rate from Puget Sound = 194 g/day.

b Provided by EPA (Hiltner 2007); 50th percentile of total seafood consumption rate from Puget Sound = 29.9 g/day.

c From Table B-2 of EPA (2007b); 95th percentile of the total seafood consumption rate from Puget Sound = 766.8 g/day.

d Bivalves include Manila/littleneck clams, horse clams, butter clams, cockles, oysters, and scallops.

CT – central tendency
EPA – US Environmental Protection Agency
EW –East Waterway
HHRA – human health risk assessment
RME – reasonable maximum exposure

The shellfish consumption rate was fractionated to develop ingestion rate-weighted concentrations when data on multiple species were available for the shellfish market basket fraction. Rates for individual shellfish market basket components should not be used outside of this context. For example, if risks associated with consumption of a particular resource, such as crabs, were of interest, development of a 95th percentile consumer only crab consumption rate would be appropriate (i.e., the crab consumption rate provided here is part of a market basket representing the 95th percentile of total seafood consumption, but does not represent the 95th percentile of crab consumption).

Table B.3-6. Adult tribal consumption of shellfish (crabs, clams, and mussels) based on Tulalip and Suquamish data

Shellfish Type	Percentage of Total Shellfish Consumption	RME or 95th Percentile Scenario Consumption Rate (g/day)[a]	CT Scenario Consumption Rate (g/day)
Adult Tribal RME Based on Tulalip Data[b]			
Crabs	42	34.4	5.3
Clams[c]	48	39.3	6.0
Mussels	1	0.8	0.1
Geoduck[d]	9	7.4	1.1
Adult Tribal Based on Suquamish Data[e]			
Crabs	10	49.8	na
Clams[c]	79	393.7	na
Mussels	1	5.0	na
Geoduck	10	49.8	na

a The adult consumption rate is the product of the percentage of total consumption and the overall shellfish consumption rate for the Tulalip and Suquamish Tribes, as applicable. The rate based on the Tulalip Tribes study (Toy et al. 1996) is defined as the adult tribal RME scenario, consistent with the LDW HHRA. The scenario based on Suquamish data is provided for the estimation of high-end risks and is not designated as an RME scenario.

b Tulalip Tribes 95th percentile total Puget Sound shellfish consumption = 81.9 g/day, consumption percentages provided to EWG by Lon Kissinger (December 12, 2008). The Tulalip Tribes CT scenario for total Puget Sound seafood consumption was based on an ingestion rate of 29.9 g/day (Hiltner 2007).

c Includes Manila/littleneck clams, horse clams, butter clams, cockles, oysters, and scallops (EPA 2005a).

d Geoduck consumption was not reported in the Tulalip Tribes survey (Toy et al. 1996). Therefore average geoduck consumption for the Tulalip Tribes-based scenario was assumed to occur at the average Suquamish Tribe geoduck consumption rate (Suquamish Tribe 2000) multiplied by the ratio of total Tulalip Tribes shellfish consumption divided by total Suquamish Tribe shellfish consumption.

e Suquamish Tribe 95th percentile total Puget Sound shellfish consumption = 498.4 g/day; consumption percentages from Table B-2 of EPA (2007b).

CT – central tendency
EPA – US Environmental Protection Agency
EWG – East Waterway Group
HHRA – human health risk assessment
LDW – Lower Duwamish Waterway
na – not applicable
RME – reasonable maximum exposure

An approach similar to the apportionment of total shellfish consumption was used for the apportionment of the pelagic fish consumption into perch and rockfish categories. This apportionment is illustrated in Table B.3-7. The Tulalip Tribes consumption study provided information about rockfish and perch consumption as part of the pelagic fish category in that study (Toy et al. 1996). Average consumption rates (for consumers and non-consumers) of perch and rockfish were calculated and used to develop concentration weighting factors that could be applied to the shellfish seafood category. In the Suquamish Tribe study, rockfish were included as part of the benthic fish category (Suquamish Tribe 2000). Perch were a part of the pelagic category in the Suquamish study but were only eaten by two respondents. Rockfish are considered to be pelagic in their lifestyle as discussed in the ecological risk assessment technical memorandum for the EW (Windward 2010e). For the adult tribal scenario based on Suquamish data, total reported pelagic fish consumption was allocated between perch and rockfish using percentages based on average perch consumption relative to average rockfish consumption, as shown in Table B.3-7.

Table B.3-7. Adult tribal consumption of pelagic fish (perch and rockfish) based on Tulalip and Suquamish data

Fish Type	Percentage of Total Pelagic Consumption	RME or 95th Percentile Scenario Consumption Rate (g/day)[a]	CT Scenario Consumption Rate (g/day)
Adult Tribal RME Based on Tulalip Data[b]			
Perch	88	7.1	1.1
Rockfish	12	1	0.2
Adult Tribal Based on Suquamish Data[c]			
Perch	1	0.6	na
Rockfish	99	55.4	na

a The adult consumption rate is the product of the percentage of total consumption and the overall pelagic fish consumption rate for the Tulalip and Suquamish Tribes, as applicable. The rate based on the Tulalip Tribes study (Toy et al. 1996) is defined as the adult tribal RME scenario, consistent with the LDW HHRA. The scenario based on Suquamish data is provided for the estimation of high-end risks and is not designated as an RME scenario.

b Percentage of each fish type calculated based on average perch and rockfish consumption provided by Lon Kissinger to EWG (December 12, 2008).

c Percentage of each fish type calculated based on reported average consumption of rockfish and perch (Suquamish Tribe 2000). Note that rockfish consumption was included in the total consumption for the benthic category in the Suquamish survey, but based on the rationale provided in Section B.3.3.1.1, they were considered part of the pelagic fish category for apportionment.

CT – central tendency
EPA – US Environmental Protection Agency
EWG – East Waterway Group
HHRA – human health risk assessment
LDW – Lower Duwamish Waterway
na – not applicable
RME – reasonable maximum exposure

The EPA tribal seafood consumption framework does not provide specific guidance on the portions of seafood consumed (e.g., whole body vs. filleted fish) within a specific seafood category. Quantification of these portions allows for the refinement of risk estimates and reduction of uncertainty. For pelagic fish, clams (other than geoduck), and mussels, only whole-body data are available (whole body, including the siphon but not the shell for mussels and clams) so it was not possible to consider the different types of tissue consumed for these seafood categories. For benthic fish from the EW, both whole-body and fillet chemical concentration data are available. Similarly, for EW crab, chemical concentration data for edible meat (i.e., muscle tissue) and estimates of whole-body chemical concentration data (based on edible meat and hepatopancreas) are also available. Geoduck consumption was also apportioned as edible meat or whole-body consumption. Geoduck whole body includes the edible meat and the gut ball portions. Information on the relative percentage of consumption of these seafood categories is available from the seafood consumptions surveys of the Tulalip Tribes

(Toy et al. 1996) and the Suquamish Tribe (2000). The percentages for the tissue categories and mean consumption rates for whole-body crabs, whole-body benthic fish, and whole-body geoduck were used to calculate the consumption rates for each of the seafood tissue categories, as presented in Table B.3-8.

Table B.3-8. Portions of benthic fish, crab, and geoduck consumed – adult tribal RME and CT scenarios based on Tulalip data and adult tribal scenario based on Suquamish data

Seafood Category	Percentage of Consumption	RME Scenario or 95th Percentile Consumption Rate (g/day)[a]	CT Scenario Consumption Rate (g/day)
Adult Tribal RME Scenario Based on Tulalip Data			
Crab, edible meat	76[b]	26.1	4.0
Crab, whole body	24[b]	8.3	1.3
Benthic fish, fillet	100[c]	7.5	1.2
Benthic fish, whole body	0[c]	0	0
Geoduck, edible meat	88[d]	6.5	1.0
Geoduck, whole body	12[d]	0.9	0.1
Adult Tribal Scenario Based on Suquamish Data			
Crab, edible meat	76[d]	37.8	na
Crab, whole body	24[d]	12.0	na
Benthic fish, fillet	89[d]	25.9	na
Benthic fish, whole body	11[d]	3.2	na
Geoduck, edible meat	88	43.8	na
Geoduck, whole body	12	6.0	na

[a] Product of percentage of consumption and the consumption rate for total crab, benthic fish, or geoduck from EPA framework (EPA 2005a); see Tables B.3-5 and B.3-6. The rate based on the Tulalip Tribes study (Toy et al. 1996) is defined as the adult tribal RME scenario, consistent with the LDW HHRA. The scenario based on Suquamish data is provided for the estimation of high-end risks and is not designated as an RME scenario.

[b] Portions of crab or geoduck consumed were not reported for Tulalip Tribes (Toy et al. 1996); values from the Suquamish Tribe (Suquamish Tribe 2000) were used as surrogates.

[c] No Tulalip Tribe respondents reported the consumption of benthic whole-body fish (Toy et al. 1996).

[d] Values from the Suquamish Tribe (Suquamish Tribe 2000).

CT – central tendency

EPA – US Environmental Protection Agency

HHRA – human health risk assessment

LDW – Lower Duwamish Waterway

na – not applicable

RME – reasonable maximum exposure

B.3.3.1.2 Child tribal seafood consumption based on Tulalip data

EPA noted in their initial framework document for selecting and using tribal fish and shellfish consumption rates for risk-based decisions (EPA 2007b) that child-specific rates appropriate for use in the framework are not available from the two Puget Sound studies (Toy et al. 1996; Suquamish Tribe 2000). The two consumption studies included adult-reported child seafood consumption for children under 5 years of age (Tulalip study, n = 21) and under 6 years of age (Suquamish study, n = 31). As discussed previously, the Tulalip Tribes study (Toy et al. 1996) was the basis for the RME seafood consumption scenario for the EW. Thus, the child tribal exposure scenarios were developed based on data from the Tulalip Tribes consumption study. EPA specified for the LDW HHRA that the total consumption rate for the child tribal RME scenario based on Tulalip data should be equal to 40% of the adult tribal RME consumption rate based on Tulalip data (EPA 2006b). The rationale provided by EPA (2007a) included concerns about the small number of children surveyed in the Tulalip Tribes study (i.e., low sample size) and the relatively low consumption rates reported as compared with other regional tribal fish and seafood consumption studies (CRITFC 1994; Toy et al. 1996) and national fish consumption studies (EPA 2002b). The 40% ratio is based on a comparison of child and adult fish and seafood consumption data from regional and national studies (EPA 2006b, 2007a). A child tribal CT scenario based on Tulalip data was also developed with a total seafood consumption rate equal to 40% of the adult tribal CT total seafood consumption rate based on Tulalip data (Hiltner 2007).

The limitations in sample size for estimating seafood consumption rates for children also limit these data for use in estimating the percentage of seafood categories consumed by children. Therefore, as was done for the LDW HHRA, the same percentages for consumption of the different seafood categories and portions used for the adult tribal scenario based on Tulalip data (Tables B.3-5 through B.3-8) were used for the EW child tribal scenarios (i.e., adult tribal RME and CT consumption rates based on Tulalip data for each seafood category and portion were multiplied by 40% to estimate child tribal RME and CT consumption rates based on Tulalip data) (Table B.3-9). Thus, no child-specific data from the Tulalip study, other than body weight, was used for the development of the child tribal exposure scenarios based on Tulalip data (Section B.3.3.1.5) (Toy et al. 1996). As with the adult tribal seafood consumption scenarios based on Tulalip data, consumption of anadromous fish was not included for EW child tribal exposures and risk estimates based on Tulalip data (EPA 2005a), which consider only the consumption of resident seafood organisms. The total non-anadromous seafood consumed in the tribal child scenario based on Tulalip data was 39.0 g/day and 6.0 g/day for the RME and CT scenarios, respectively.

Seafood consumption rates based on the 95th percentile of seafood consumption for children reported in the Tulalip Tribes study (Toy et al. 1996) and associated risk estimates for consumption of resident EW seafood are presented in the uncertainty analysis (Section B.6). As discussed in Section B.3.3.1, risk estimates for a child tribal scenario based on Suquamish data are presented in the uncertainty analysis (Section B.6).

Table B.3-9. Rates of child tribal (RME and CT) seafood consumption based on Tulalip data associated with different seafood categories

Seafood Category	Consumption Rate (g/day)	
	RME Scenario[a]	CT Scenario[b]
Anadromous fish[c]	38.6	6.0
Pelagic fish – rockfish	0.4	0.08
Pelagic fish – perch	2.8	0.44
Benthic fish, fillet	3.0	0.48
Benthic fish, whole body	0	0
Crab, edible meat	10.4	1.6
Crab, whole body	3.3	0.5
Clams	15.7	2.4
Mussels	0.3	0.04
Geoduck, edible meat	2.6	0.4
Geoduck, whole body	0.4	0.04

[a] Total consumption rate = 77.6 g/day. Total consumption rate and consumption rates for seafood categories calculated as 40% of the adult tribal RME consumption rates based on Tulalip data (Tables B.3-5 through B.3-8).

[b] Total consumption rate = 12 g/day. Total consumption rate and consumption rates for seafood categories calculated as 40% of the adult tribal CT consumption rates based on Tulalip data (Tables B.3-5 through B.3-8).

[c] Consumption rate was not used in this HHRA.

CT – central tendency

HHRA – human health risk assessment

RME – reasonable maximum exposure

B.3.3.1.3 Adult API seafood consumption rates

A specific scenario was also developed for adult API consumption of EW seafood. The API populations studied by EPA (1999a) may consume fish and shellfish collected from the EW, but the survey did not include geographic distinctions to determine the fishing frequency in the EW compared with other areas in King County over which the survey was based. However, information collected by WDFW enforcement personnel (Frame 2001) indicate that individuals of API ethnicity are more commonly encountered engaging in non-commercial fishing in the Duwamish River than any other ethnic group. Several Puget Sound seafood consumption studies have documented a substantial number of APIs fishing in urban embayments (Landolt et al. 1985; McCallum 1985; Landolt et al. 1987), including in the EW (King County 1999a; EPA 1999a). Although there is uncertainty regarding the degree of seafood consumption by any group within the EW, this HHRA provides an estimate for the API population; this population may consume more seafood than does the general public.

The EPA study included 202 adult men and women from 10 different ethnic groups (Cambodian, Chinese, Filipino, Hmong, Japanese, Korean, Laotian, Mien, Samoan, and Vietnamese) (EPA 1999a). As in the adult tribal consumption rates based on Tulalip data, EPA provided guidance on the application of data from this study for deriving fish and shellfish consumption rates for risk assessment (Kissinger 2005). An approach similar to that used for the development of tribal rates was used for API consumption rate development. The raw data were used to estimate the 95th percentile of consumption by individuals reporting consumption of seafood caught in King County.

However, unlike the tribal studies, in which each individual respondent was weighted equally, the respondents in the API study were weighted to reflect their ethnic group's population in King County relative to their representation in the consumption study. For example, 20 of the study participants were Cambodian, representing 10% of the survey respondents (20/202). However, Cambodians make up only 3.91% of the total King County population of the 10 ethnic groups included in the study (EPA 1999a). Thus, Cambodians were over-represented in the survey relative to the populations of the other nine API groups in King County. To account for this over-representation, consumption data from each Cambodian respondent was weighted specifically to adjust for this difference (Kissinger 2005). The same was done for each respondent based on their ethnicity and the representation of their ethnicity in the study relative to the representation of their ethnicity in the King County API population.

In EPA's 2005 reanalysis of the 1999 API data, only data for individuals consuming seafood from King County were included; weights based on all participants in the survey were not developed. Weighting factors for King County consumers for various ethnic groups were a function of the percentage of that ethnic group as determined in the 2000 United States census and the number of individuals in that ethnic group that consumed seafood from King County (Kissinger 2005). For example, the weighting factor for Cambodians was derived based on the fact that 11 out of 20 Cambodians consumed seafood harvested in King County, that the percentage of Cambodians in the 2000 US census for King County was 3.91%, and that there were 99 King County seafood consumers in the 1999 API study. The 95th percentile ingestion rate was developed from the consumer-only dataset of weighted ingestion rates.

The data were also adjusted to account for the fact that some shellfish consumption was reported on a cooked-weight, rather than on a raw-weight, basis. Consumption of the following shellfish was recorded in terms of cooked weight: butter clams, cockles, crabs, geoducks, horse clams, *Macoma* clams, Manila/little neck clams, moon snails, and mussels (EPA 1999a). Consumption of soft-shell clams (*Mya arenaria*) was not recorded. Two revised estimates of average (consumer and non-consumer) raw shellfish consumption were made by EPA, using 25 and 50% cooking loss correction factors for those shellfish species for which consumption was reported on a cooked-weight basis. The average of these two estimates was provided by EPA (Kissinger 2006a).[27] This approach for adjusting cooked weight is described in detail in Kissinger (2005). The recommended 95th percentile of total King County API seafood consumption in that document was 57.1 g/day (n = 99, demographically weighted).

[27] This calculation required access to the information beyond what was provided in the publicly available report (EPA 1999a).

To apportion the total seafood consumption rate of 57.1 g/day into the different seafood categories, EPA calculated demographically weighted mean ingestion rates for each seafood category for individuals who consumed seafood caught in King County. The demographically weighted mean ingestion rates were then used to derive the percentage of consumption of each seafood category (Table B.3-10). These percentages were then applied to the total consumption rate (57.1 g/day) to derive consumption rates for each seafood category (Table B.3-10). Anadromous fish were not included in the exposure scenario because of the lack of linkage between chemicals in EW sediment and those found in adult salmon tissues, consistent with the LDW HHRA and per EPA recommendation (EPA 2005a). To estimate the CT consumption rate for the API scenario, the 50th percentile of total King County API consumption (5.8 g/day) (Kissinger 2005) was multiplied by the percentage of consumption for the various seafood categories. Total non-anadromous seafood consumption for the API scenarios was 51.6 g/day and 5.3 g/day for the RME and CT scenarios, respectively.

Table B.3-10. Percentages and rates of adult API RME and CT seafood consumption associated with different seafood categories

Seafood Category	Percentage of Total Consumption[a]	Consumption Rate (g/day)	
		RME Scenario[b]	CT Scenario[b]
Anadromous fish[c]	9.6	5.5	0.56
Pelagic fish	8.6	4.9[d]	0.5
Benthic fish	4.2	2.4[d]	0.24
Shellfish	77.5	44.3[d]	4.6

a Calculated from average consumption rates by seafood category for consumers of King County species as provided by EPA (Kissinger 2006a).

b For the RME scenario, the 95th percentile of total King County API seafood consumption, 57.1 g/day (Kissinger 2005), was multiplied by the percentage of consumption for the various seafood categories. For the CT scenario, the 50th percentile of total King County API consumption, 5.8 g/day (Kissinger 2005), was multiplied by the percentage of consumption for the various seafood categories.

c Consumption rate was not used in this HHRA.

d Freshwater fish make up 8.3% of API seafood consumption. As requested by EPA, freshwater fish were apportioned into benthic fish, pelagic fish, and shellfish categories according to the respective consumption rates for those fish (EPA 2006b). This apportionment assumes that API consumers who catch and consume freshwater fish outside the EW would instead catch and consume more marine species inside the EW.

API – Asian and Pacific Islander
CT – central tendency
EPA – US Environmental Protection Agency
HHRA – human health risk assessment
RME – reasonable maximum exposure

To calculate the consumption of mussels, crabs, and clams for the API scenario, the same general approach followed for the tribal consumption calculations was used. The average demographically weighted consumption of clams, mussels, and crabs for the API consumers of these shellfish species self-harvested only from King County (n = 99) was provided by EPA (Kissinger 2006a) and used to calculate the percentage of each shellfish type consumed (Table B.3-11). This weighting factor was used with the estimate of the 95th percentile of King County API shellfish consumption (44.3 g/day, Table B.3-10) to calculate the consumption of clams, mussels, and crabs. Consumption of pelagic fish was apportioned based on reported consumption within these categories (Table B.3-12). As with the tribal consumption estimate, the crab consumption rates were apportioned between crab whole body and edible meat, and the benthic fish consumption rates were apportioned between benthic fish fillet and whole body (Table B.3-13) based on the reported consumption of these seafood tissue categories by API consumers.[28] This information was provided by EPA as demographically weighted average percentages of crab whole-body and crab edible-meat consumption by API members who consume at least some King County seafood (n = 96; 3 individuals did not consume any crab) (Kissinger 2007a). Similarly, EPA provided the average demographically weighted percentages of whole-body versus fillet consumption by API members who consume at least some King County seafood (n = 99) (Kissinger 2007a). This latter information was used to apportion benthic fish consumption into benthic whole-body and benthic fillet consumption.

Table B.3-11. Adult API RME and CT consumption of shellfish (crabs, clams, and mussels)

Shellfish Type	Percentage of Total Shellfish Consumption[a]	Consumption Rate (g/day)[b, c]	
		RME Scenario	CT Scenario
Crabs	24.0	10.6	1.1
Clams[d]	65.6	29.1	3.0
Mussels	10.4	4.6	0.5

a Calculated from average consumption rates provided by EPA for API consumers of King County species (Kissinger 2006b).

b Product of percentage of total shellfish consumption (for each shellfish type) and total shellfish consumption (Table B.3-10).

c Consumption includes freshwater fish.

d Includes Manila/littleneck clams, horse clams, butter clams, cockles, oysters, and scallops.

API – Asian and Pacific Islander

CT – central tendency

EPA – US Environmental Protection Agency

RME – reasonable maximum exposure

28 Because of the low sample size, both self-harvesters and non-self-harvesters were used to estimate portions of crab and benthic fish consumed.

Table B.3-12. Adult API RME and CT consumption of pelagic fish (perch and rockfish)

Pelagic Fish Type	Percentage of Total Pelagic Fish Consumption[a]	Consumption Rate (g/day)[b, c]	
		RME Scenario	CT Scenario
Perch	10	0.5	0.05
Rockfish	90	4.4	0.45

[a] Calculated from average consumption rates provided by EPA for API consumers of King County species (provided by Lon Kissinger (Kissinger 2008). Reported consumption of herring was used as a surrogate for consumption of perch, which was not reported.

[b] Product of percentage of total consumption and total pelagic fish consumption.

[c] Consumption includes freshwater fish.

API – Asian and Pacific Islander

CT – central tendency

EPA – US Environmental Protection Agency

RME – reasonable maximum exposure

Table B.3-13. Adult API RME and CT consumption of portions of benthic fish and crab

Seafood Category	Percentage of Consumption[a]	Consumption Rate (g/day)[b, c]	
		RME Scenario	CT Scenario
Crab, edible meat	53.3	5.7	0.6
Crab, whole body	46.7	4.9	0.5
Benthic fish, fillet	82.3	2.0	0.2
Benthic fish, whole body	17.7	0.4	0.04

[a] As provided by EPA for crab or fish (Kissinger 2007a) for API consumers of King County species.

[b] Percentage of consumption multiplied by total crab consumption (Table B.3-11) or total benthic fish consumption (Table B.3-10).

[c] Consumption includes freshwater fish.

API – Asian and Pacific Islander

CT – central tendency

EPA – US Environmental Protection Agency

RME – reasonable maximum exposure

Unlike the consumption scenarios based on Tribal data, the API seafood scenario does not include geoduck consumption as a portion of total shellfish consumption. The tribes that have U&A fishing rights in the EW commercially harvest geoducks and therefore have the equipment (i.e., scuba gear) needed to collect them. However, the API population does not have commercial harvesting rights to geoducks in the EW. Because of this and the fact that special equipment and training in its use are required to harvest geoducks, the API population was assumed not to consume geoducks from the EW.

B.3.3.1.4 Adult one-meal-per-month seafood consumption rates

Consumption rates for recreationally caught fish are not available for the EW. Although there have been some creel studies conducted in the LDW/EW area (Landolt et al. 1985; McCallum 1985), there has not been a comprehensive recreational fish consumption study for the EW site or nearby areas of similar quality as the recent tribal studies (Toy et al. 1996; Suquamish Tribe 2000) and API studies (EPA 1999a). All of the available recreational angler surveys have methodological and interpretation uncertainties that make it difficult to make conclusive observations about recreational seafood consumption. However, it is possible that recreational consumption rates for some anglers could be as high as the Tulalip Tribes' consumption rates.

Recreational fishing, particularly salmon angling, is known to occur on the EW despite the existence of fishing advisories (King County 1999a), but the magnitude is uncertain. It is expected that current recreational consumption of resident species is likely to be relatively low and potentially suppressed because of public awareness of chemical contamination in the EW and LDW and because of WSDOH seafood consumption advisories for the LDW and Elliott Bay (WSDOH 2010).

In an effort to provide information that would allow site users (e.g., recreational anglers) to evaluate the risks associated with seafood consumption from the EW, four hypothetical scenarios were developed. To evaluate risks associated with consumption of various resources independently (i.e., in addition to the market basket approach that was used for the tribal and API seafood consumption evaluation), the consumption of five different seafood categories were evaluated separately for benthic fish (fillets), pelagic fish (both for perch and rockfish), clams, and crabs (edible meat). Each of these five scenarios assumes that consumption would average approximately one meal (227 g, per EPA (2000d) guidance) per month of a given seafood category, which equates to 7.5 g/day. Totaling the risks from each of these four scenarios provides an estimate of risk associated with four meals per month, one of each seafood category. Note that EW data are lacking to support this quantity and pattern of recreational consumption for current or future use. The one-meal-per-month seafood consumption scenario and the associated risk estimates are intended to serve as a tool for risk communication and are not intended to directly reflect actual recreational seafood consumption because these rates are highly uncertain and may currently be suppressed as a result of consumption advisories.

One-meal-per-month scenarios include specific targeted species and seafood portions expected to reflect what individuals might choose to consume. The benthic fish one-meal-per-month scenario represents the consumption of English sole fillets. The pelagic one-meal-per-month scenario was evaluated for perch and rockfish separately (i.e., as two independent scenarios). The crab one-meal–per-month scenario evaluated only the consumption of crab edible meat. Finally, the clam one-meal-per-month scenario included clams collected from intertidal areas.[29] The one-meal-per-month scenarios provide a basis for individuals to evaluate their own exposure using a method that is readily scaled to various seafood consumption levels. For example, if someone eats two meals per month of EW crab and one meal per month of EW pelagic fish, he or

[29] Geoducks were not included in non-tribal scenarios because geoducks are harvested with scuba gear and other specialized tools.

she could multiply the one-meal-per-month crab risk estimate by two and add the product to the one-meal-per-month pelagic fish risk estimate to approximate the risk associated with his or her own EW seafood consumption. Graphical representations of seafood consumption rates versus risk estimates are presented by species to make scaling (e.g., one crab meal per month to one crab meal per week) easier for the public (Section B.5.6.4).

As mentioned above for the tribal and API scenarios and based on EPA recommendations, consumption of adult salmon from the EW was excluded from the HHRA (EPA 2005a). Thus, although salmon have been identified as the most commonly sought species for recreational fishers in the EW (King County 1999a), bioaccumulative chemical concentrations in adult salmon caught in the EW are largely attributable to uptake during the portion of a salmon's life spent in the Pacific Ocean and Puget Sound, and thus most of the risk associated with consumption of adult salmon is not related to EW sediment. Therefore, the adult one-meal-per-month exposure scenarios derived here do not address risks from the consumption of adult salmon from the EW.

B.3.3.1.5 Exposure parameters for the seafood consumption scenario

Table B.3-14 presents a summary of the key scenario-specific parameters used to calculate the CDI for seafood ingestion. The seafood ingestion rates in this table are same as those that were used for the LDW HHRA (Windward 2007c, 2009c),[30] with the exception of the inclusion of two additional species: geoduck and rockfish. Consumption of geoduck and rockfish[31] were not included in the LDW HHRA because these species are not found (geoduck) or are rare (rockfish) in the LDW (Windward 2007c). Tables B.3-15 through B.3-22 provide all of the exposure parameters for each seafood consumption scenario, including ingestion rates (as derived in Sections B.3.3.1.1 through B.3.3.1.4) and other exposure parameters used to calculate CDIs (e.g., exposure duration and body weight), which are the same as those used for the LDW HHRA (Windward 2007c).

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[30] Ingestion rates for crab and other shellfish for the adult and child tribal RME scenarios based on Tulalip data, as well as the adult and child tribal CT scenarios based on Tulalip data, were adjusted by EPA following the completion of the LDW HHRA (EPA 2009b) and thus differ slightly from the original LDW values. The 2009 errata to the LDW HHRA presents the LDW risks using these revised ingestion rates (Windward 2009c), which have been used in this HHRA.

[31] In the EW HHRA, pelagic fish consumption is comprised of both perch and rockfish. In the LDW HHRA, pelagic fish consumed only of perch because of the low abundance of rockfish in the LDW (Windward 2007c).

Table B.3-14. Summary of seafood ingestion scenarios

Scenario	Ingestion Rate (g/day)					Exposure Duration (years)	Location of Scenario-Specific Details
	Pelagic Fish	Benthic Fish	Crab	Other Shellfish	Total		
Adult tribal RME (Tulalip data)	8.1	7.5	34.4	47.5	97.5	70	Table B.3-15
Adult tribal CT (Tulalip data)	1.3	1.2	5.3	7.2	15.0	30	Table B.3-16
Child tribal RME (Tulalip data)	3.2	3.0	13.7	19.0	39.0[a]	6	Table B.3-17
Child tribal CT (Tulalip data)	0.52	0.48	2.1	2.9	6.0	6	Table B.3-18
Adult tribal (Suquamish data)	56	29.1	49.8	448.5	583.5[a]	70	Table B.3-19
Adult API – RME	4.9	2.4	10.6	33.7	51.6	30	Table B.3-20
Adult API – CT	0.5	0.24	1.1	3.5	5.3	9	Table B.3-21
Adult one meal per month[b]	7.5	7.5	7.5	7.5	na	30	Table B.3-22

[a] As the result of rounding, the total ingestion rate is equal to 0.1 g greater than the sum of the ingestion rates for the seafood categories.

[b] Adult one-meal-per-month consumption was evaluated by individual seafood categories independently to reflect different fishing and consumption practices.

API – Asian and Pacific Islander

CT – central tendency

na – not applicable

RME – reasonable maximum exposure

Table B.3-15. Daily intake calculations – seafood ingestion, adult tribal RME scenario based on Tulalip data

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Exposure route: Ingestion
Equation for chronic daily intake (CDI) (mg/kg-day):

$$CDI = \frac{\left(\sum\left(EPC_s \times IR_s\right)\right) \times FI \times EF \times ED_a \times CF}{BW_a \times AT}$$

Where:

$$\sum\left(EPC_s \times IR_s\right) = \left(EPC_p \times IR_p\right) + \left(EPC_r \times IR_r\right) + \left(EPC_b \times IR_b\right) + \left(EPC_{bwb} \times IR_{bwb}\right) + \left(EPC_c \times IR_c\right) + \left(EPC_{cwb} \times IR_{cwb}\right) + \left(EPC_m \times IR_m\right) + \left(EPC_{cl} \times IR_{cl}\right) + \left(EPC_g \times IR_g\right) + \left(EPC_{gwb} \times IR_{gwb}\right)$$

Parameter	Parameter Definition	Unit	Value	Rationale/
EPC-p	exposure point concentration in pelagic fish, perch	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-r	exposure point concentration in pelagic fish, rockfish	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-b	exposure point concentration in benthic fish, fillet	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-bwb	exposure point concentration in benthic fish, whole body	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-c	exposure point concentration in crabs, edible meat	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-cwb	exposure point concentration in crabs, whole body	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-m	exposure point concentration in mussels	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-cl	exposure point concentration in clams	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-g	exposure point concentration in geoduck, edible meat	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-gwb	exposure point concentration in geoduck, whole body	mg/kg ww	Table B.3-42	Section B.3.3.4
IR-p	ingestion rate – pelagic fish, perch	g/day	7.1	Section B.3.3.1.1
IR-r	ingestion rate – pelagic fish, rockfish	g/day	1.0	Section B.3.3.1.1
IR-b	ingestion rate – benthic fish, fillet	g/day	7.5	Section B.3.3.1.1
IR-bwb	ingestion rate – benthic fish, whole body	g/day	0	Section B.3.3.1.1
IR-c	ingestion rate – crabs, edible meat	g/day	26.1	Section B.3.3.1.1
IR-cwb	ingestion rate – crabs, whole body	g/day	8.3	Section B.3.3.1.1
IR-m	ingestion rate – mussels	g/day	0.8	Section B.3.3.1.1
IR-cl	ingestion rate –clams	g/day	39.3	Section B.3.3.1.1
IR-g	ingestion rate – geoduck, edible meat	g/day	6.5	Section B.3.3.1.1
IR-gwb	ingestion rate – geoduck, whole body	g/day	0.9	Section B.3.3.1.1
FI	fractional intake derived from source	unitless	1[a]	EPA (2007b)
EF	exposure frequency	days/yr	365[b]	EPA (1991a)
ED-a	exposure duration – adult	years	70	EPA (2005a)
CF	conversion factor	kg/g	0.001	na
BW-a	body weight-adult	kg	81.8	Toy et al. (1996)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	25,550	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

[a] A value of 1 was used for the fractional intake derived from source as directed by EPA (2007b).

[b] Default exposure frequency of 350 days/yr modified to 365 days/yr to account for the fact that seafood consumption rate estimates are based on 365 days/yr.

EPA – US Environmental Protection Agency
na – not applicable
RME – reasonable maximum exposure
ww – wet weight

Table B.3-16. Daily intake calculations – seafood ingestion, adult tribal CT scenario based on Tulalip data

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Exposure route: Ingestion
Equation for chronic daily intake (CDI) (mg/kg-day):

$$CDI = \frac{\left(\sum\left(EPC_s \times IR_s\right)\right) \times FI \times EF \times ED_a \times CF}{BW_a \times AT}$$

Where:

$$\sum\left(EPC_s \times IR_s\right) = \left(EPC_p \times IR_p\right) + \left(EPC_r \times IR_r\right) + \left(EPC_b \times IR_b\right) + \left(EPC_{bwb} \times IR_{bwb}\right) + \left(EPC_c \times IR_c\right) + \left(EPC_{cwb} \times IR_{cwb}\right) + \left(EPC_m \times IR_m\right) + \left(EPC_{cl} \times IR_{cl}\right) + \left(EPC_g \times IR_g\right) + \left(EPC_{gwb} \times IR_{gwb}\right)$$

Parameter	Parameter Definition	Unit	Value	Rationale/
EPC-p	exposure point concentration in pelagic fish, perch	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-r	Exposure point concentration in pelagic fish, rockfish	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-b	exposure point concentration in benthic fish, fillet	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-bwb	exposure point concentration in benthic fish, whole body	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-c	exposure point concentration in crabs, edible meat	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-cwb	exposure point concentration in crabs, whole body	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-m	exposure point concentration in mussels	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-cl	exposure point concentration in clams	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-g	exposure point concentration in geoduck, edible meat	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-gwb	exposure point concentration in geoduck, whole body	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
IR-p	ingestion rate – pelagic fish, perch	g/day	1.1	Section B.3.3.1.1
IR-r	ingestion rate – pelagic fish, rockfish	g/day	0.2	Section B.3.3.1.1
IR-b	ingestion rate – benthic fish, fillet	g/day	1.2	Section B.3.3.1.1
IR-bwb	ingestion rate – benthic fish, whole body	g/day	0	Section B.3.3.1.1
IR-c	ingestion rate – crabs, edible meat	g/day	4.0	Section B.3.3.1.1
IR-cwb	ingestion rate – crabs, whole body	g/day	1.3	Section B.3.3.1.1
IR-m	ingestion rate – mussels	g/day	0.1	Section B.3.3.1.1
IR-cl	ingestion rate –clams	g/day	6.0	Section B.3.3.1.1
IR-g	ingestion rate – geoduck, edible meat	g/day	1.0	Section B.3.3.1.1
IR-gwb	ingestion rate – geoduck, whole body	g/day	0.1	Section B.3.3.1.1
FI	fractional intake derived from source	unitless	1[b]	EPA (2007b)
EF	exposure frequency	days/yr	365[c]	EPA (1991a)
ED-a	exposure duration – adult	years	30	EPA (1997a)
CF	conversion factor	kg/g	0.001	na
BW-a	body weight – adult	kg	81.8	Toy et al. (1996)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	10,950	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

[a] EPCs for CT scenarios are based on mean concentrations, in contrast to the EPCs for the RME scenarios, which are based on 95% UCLs on mean concentrations.

[b] A value of 1 was used for the fractional intake derived from source as directed by EPA (2007b).

[c] Default exposure frequency of 350 days/yr modified to 365 days/yr to account for the fact that seafood consumption rate estimates are based on 365 days/yr.

CT – central tendency
EPA – US Environmental Protection Agency
na – not applicable
RME – reasonable maximum exposure
UCL – upper confidence limit on the mean
ww – wet weight

Table B.3-17. Daily intake calculations – seafood ingestion, child tribal RME scenario based on Tulalip data

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Exposure route: Ingestion
Equation for chronic Daily Intake (CDI) (mg/kg-day):

$$CDI = \frac{\left(\sum (EPC_s \times IR_s)\right) \times FI \times EF \times ED_a \times CF}{BW_a \times AT}$$

Where:

$$\sum (EPC_s \times IR_s) = (EPC_p \times IR_p) + (EPC_r \times IR_r) + (EPC_b \times IR_b) + (EPC_{bwb} \times IR_{bwb}) + (EPC_c \times IR_c) + (EPC_{cwb} \times IR_{cwb}) + (EPC_m \times IR_m) + (EPC_{cl} \times IR_{cl}) + (EPC_g \times IR_g) + (EPC_{gwb} \times IR_{gwb})$$

Parameter	Parameter Definition	Unit	Value	Rationale/
EPC-p	exposure point concentration in pelagic fish, perch	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-r	exposure point concentration in pelagic fish, rockfish	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-b	exposure point concentration in benthic fish, fillet	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-bwb	exposure point concentration in benthic fish, whole body	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-c	exposure point concentration in crabs, edible meat	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-cwb	exposure point concentration in crabs, whole body	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-m	exposure point concentration in mussels	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-cl	exposure point concentration in clams	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-g	exposure point concentration in geoduck, edible meat	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-gwb	exposure point concentration in geoduck, whole body	mg/kg ww	Table B.3-42	Section B.3.3.4
IR-p	ingestion rate – pelagic fish, perch	g/day	2.8	Section B.3.3.1.2
IR-r	ingestion rate – pelagic fish, rockfish	g/day	0.4	Section B.3.3.1.2
IR-b	ingestion rate – benthic fish, fillet	g/day	3.0	Section B.3.3.1.2
IR-bwb	ingestion rate – benthic fish, whole body	g/day	0	Section B.3.3.1.2
IR-c	ingestion rate – crabs, edible meat	g/day	10.4	Section B.3.3.1.2
IR-cwb	ingestion rate – crabs, whole body	g/day	3.3	Section B.3.3.1.2
IR-m	ingestion rate – mussels	g/day	0.3	Section B.3.3.1.2
IR-cl	ingestion rate –clams	g/day	15.7	Section B.3.3.1.2
IR-g	ingestion rate – geoduck, edible meat	g/day	2.6	Section B.3.3.1.2
IR-gwb	ingestion rate – geoduck, whole body	g/day	0.4	Section B.3.3.1.2
FI	fractional intake derived from source	unitless	1[b]	EPA (2007b)
EF	exposure frequency	days/yr	365[a]	EPA (1991a)
ED-c	exposure duration – child	years	6	EPA (1991a)
CF	conversion factor	kg/g	0.001	na
BW-ct	body weight – child Tulalip	kg	15.2	Toy et al. (1996)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	2,190	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

[a] Default exposure frequency of 350 days/yr modified to 365 days/yr to account for the fact that seafood consumption rate estimates are based on 365 days/yr.

[b] A value of 1 was used for the fractional intake derived from source as directed by EPA (2007b).

EPA – US Environmental Protection Agency
na – not applicable
RME – reasonable maximum exposure
ww – wet weight

Table B.3-18. Daily intake calculations – seafood ingestion, child tribal CT scenario based on Tulalip data

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Exposure route: Ingestion
Equation for chronic Daily Intake (CDI) (mg/kg-day):

$$CDI = \frac{\left(\sum\left(EPC_s \times IR_s\right)\right) \times FI \times EF \times ED_a \times CF}{BW_a \times AT}$$

Where:

$$\sum\left(EPC_s \times IR_s\right) = \left(EPC_p \times IR_p\right) + \left(EPC_r \times IR_r\right) + \left(EPC_b \times IR_b\right) + \left(EPC_{bwb} \times IR_{bwb}\right) + \left(EPC_c \times IR_c\right) + \left(EPC_{cwb} \times IR_{cwb}\right) + \left(EPC_m \times IR_m\right) + \left(EPC_{cl} \times IR_{cl}\right) + \left(EPC_g \times IR_g\right) + \left(EPC_{gwb} \times IR_{gwb}\right)$$

Parameter	Parameter Definition	Unit	Value	Rationale/
EPC-p	exposure point concentration in pelagic fish, perch	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-r	exposure point concentration in pelagic fish, rockfish	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-b	exposure point concentration in benthic fish, fillet	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-bwb	exposure point concentration in benthic fish, whole body	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-c	exposure point concentration in crabs, edible meat	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-cwb	exposure point concentration in crabs, whole body	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-m	exposure point concentration in mussels	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-cl	exposure point concentration in clams	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-g	exposure point concentration in geoduck, edible meat	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-gwb	exposure point concentration in geoduck, whole body	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
IR-p	ingestion rate – pelagic fish, perch	g/day	0.44	Section B.3.3.1.2
IR-r	ingestion rate – pelagic fish, rockfish	g/day	0.08	Section B.3.3.1.2
IR-b	ingestion rate – benthic fish, fillet	g/day	0.48	Section B.3.3.1.2
IR-bwb	ingestion rate – benthic fish, whole body	g/day	0	Section B.3.3.1.2
IR-c	ingestion rate – crabs, edible meat	g/day	1.6	Section B.3.3.1.2
IR-cwb	ingestion rate – crabs, whole body	g/day	0.5	Section B.3.3.1.2
IR-m	ingestion rate – mussels	g/day	0.04	Section B.3.3.1.2
IR-cl	ingestion rate –clams	g/day	2.4	Section B.3.3.1.2
IR-g	ingestion rate – geoduck, edible meat	g/day	0.4	Section B.3.3.1.2
IR-gwb	ingestion rate – geoduck, whole body	g/day	0.04	Section B.3.3.1.2
FI	fractional intake derived from source	unitless	1[b]	EPA (2007b)[b]
EF	exposure frequency	days/yr	365[c]	EPA (1991a)
ED-c	exposure duration – child	years	6	EPA (1991a)
CF	conversion factor	kg/g	0.001	na
BW-ct	body weight – child Tulalip	kg	15.2	Toy et al. (1996)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	2,190	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

[a] EPCs for CT scenarios are based on mean concentrations, in contrast to the EPCs for the RME scenarios, which are based on 95% UCLs on mean concentrations.

[b] A value of 1 was used for the fractional intake derived from source as directed by EPA (2007b).

[c] Default exposure frequency of 350 days/yr modified to 365 days/yr to account for the fact that seafood consumption rate estimates are based on 365 days/yr.

CT – central tendency
EPA – US Environmental Protection Agency
na – not applicable
UCL – upper confidence limit on the mean
ww – wet weight

Table B.3-19. Daily intake calculations – seafood ingestion, adult tribal scenario based on Suquamish data

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Exposure route: Ingestion
Equation for chronic daily intake (CDI) (mg/kg-day):

$$CDI = \frac{\left(\sum \left(EPC_s \times IR_s\right)\right) \times FI \times EF \times ED_a \times CF}{BW_a \times AT}$$

Where:

$$\sum \left(EPC_s \times IR_s\right) = \left(EPC_p \times IR_p\right) + \left(EPC_r \times IR_r\right) + \left(EPC_b \times IR_b\right) + \left(EPC_{bwb} \times IR_{bwb}\right) + \left(EPC_c \times IR_c\right) + \left(EPC_{cwb} \times IR_{cwb}\right) + \left(EPC_m \times IR_m\right) + \left(EPC_{cl} \times IR_{cl}\right) + \left(EPC_g \times IR_g\right) + \left(EPC_{gwb} \times IR_{gwb}\right)$$

Parameter Code	Parameter Definition	Unit	Value	Rationale/ Reference
EPC-p	exposure point concentration in pelagic fish, perch	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-r	exposure point concentration in pelagic fish, rockfish	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-b	exposure point concentration in benthic fish, fillet	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-bwb	exposure point concentration in benthic fish, whole body	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-c	exposure point concentration in crabs, edible meat	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-cwb	exposure point concentration in crabs, whole body	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-m	exposure point concentration in mussels	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-cl	exposure point concentration in clams	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-g	exposure point concentration in geoduck, edible meat	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-gwb	exposure point concentration in geoduck, whole body	mg/kg ww	Table B.3-42	Section B.3.3.4
IR-p	ingestion rate – pelagic fish, perch	g/day	0.6	Section B.3.3.1.1
IR-r	ingestion rate – pelagic fish, rockfish	g/day	55.4	Section B.3.3.1.1
IR-b	ingestion rate – benthic fish, fillet	g/day	25.9	Section B.3.3.1.1
IR-bwb	ingestion rate – benthic fish, whole body	g/day	3.2	Section B.3.3.1.1
IR-c	ingestion rate – crabs, edible meat	g/day	37.8	Section B.3.3.1.1
IR-cwb	ingestion rate – crabs, whole body	g/day	12.0	Section B.3.3.1.1
IR-m	ingestion rate – mussels	g/day	5.0	Section B.3.3.1.1
IR-cl	ingestion rate –clams	g/day	393.7	Section B.3.3.1.1
IR-g	ingestion rate – geoduck, edible meat	g/day	43.8	Section B.3.3.1.1
IR-gwb	ingestion rate – geoduck, whole body	g/day	6.0	Section B.3.3.1.1
FI	fractional intake derived from source	unitless	1[b]	EPA (2007b)
EF	exposure frequency	days/yr	365[a]	EPA (1991a)
ED-a	exposure duration – adult	years	70	EPA (2005a)
CF	conversion factor	kg/g	0.001	na
BW-a	body weight – adult	kg	79[c]	Suquamish Tribe (2000)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	25,550	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

[a] Default exposure frequency of 350 days/yr modified to 365 days/yr to account for the fact that seafood consumption rate estimates are based on 365 days/yr.

[b] A value of 1 was used for the fractional intake derived from source as directed by EPA (2007b).

[c] Average body weight based on information provided by the Suquamish Tribe.

EPA – US Environmental Protection Agency

ww – wet weight

na – not applicable

Table B.3-20. Daily intake calculations – seafood ingestion, adult API RME scenario

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Exposure route: Ingestion
Equation for chronic daily intake (CDI) (mg/kg-day):

$$CDI = \frac{\left(\sum\left(EPC_s \times IR_s\right)\right) \times FI \times EF \times ED_a \times CF}{BW_a \times AT}$$

Where:

$$\sum\left(EPC_s \times IR_s\right) = \left(EPC_p \times IR_p\right) + \left(EPC_r \times IR_r\right) + \left(EPC_b \times IR_b\right) + \left(EPC_{bwb} \times IR_{bwb}\right) + \left(EPC_c \times IR_c\right) + \left(EPC_{cwb} \times IR_{cwb}\right) + \left(EPC_m \times IR_m\right) + \left(EPC_{cl} \times IR_{cl}\right) + \left(EPC_g \times IR_g\right) + \left(EPC_{gwb} \times IR_{gwb}\right)$$

Parameter	Parameter Definition	Unit	Value	Rationale/
EPC-p	exposure point concentration in pelagic fish, perch	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-r	exposure point concentration in pelagic fish, rockfish	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-b	exposure point concentration in benthic fish, fillet	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-bwb	exposure point concentration in benthic fish, whole body	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-c	exposure point concentration in crabs, edible meat	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-cwb	exposure point concentration in crabs, whole body	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-m	exposure point concentration in mussels	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-cl	exposure point concentration in clams	mg/kg ww	Table B.3-42	Section B.3.3.4
IR-p	ingestion rate – pelagic fish, perch	g/day	0.5	Section B.3.3.1.3
IR-r	ingestion rate – pelagic fish, rockfish	g/day	4.4	Section B.3.3.1.3
IR-b	ingestion rate – benthic fish, fillet	g/day	2.0	Section B.3.3.1.3
IR-bwb	ingestion rate – benthic fish, whole body	g/day	0.4	Section B.3.3.1.3
IR-c	ingestion rate – crabs, edible meat	g/day	5.7	Section B.3.3.1.3
IR-cwb	ingestion rate – crabs, whole body	g/day	4.9	Section B.3.3.1.3
IR-m	ingestion rate – mussels	g/day	4.6	Section B.3.3.1.3
IR-cl	ingestion rate –clams	g/day	29.1	Section B.3.3.1.3
FI	fractional intake derived from source	unitless	1	Kissinger (2005)
EF	exposure frequency	days/yr	365[b]	EPA (1991a)
ED-a	exposure duration – adult	years	30	EPA (1989)
CF	conversion factor	kg/g	0.001	na
BW-a	body weight – adult	kg	63[c]	EPA (1999a)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	10,950	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

[a] A value of 1 was used for the fractional intake derived from source as directed by EPA (Kissinger 2005).

[b] Default exposure frequency of 350 days/yr modified to 365 days/yr to account for the fact that seafood consumption rate estimates are based on 365 days/yr.

[c] Average body weight for all surveyed individuals in API seafood consumption study in King County, as reported in EPA (1999a).

API – Asian and Pacific Islander
EPA – US Environmental Protection Agency
na – not applicable
RME – reasonable maximum exposure
ww – wet weight

Table B.3-21. Daily intake calculations – seafood ingestion, adult API CT scenario

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Exposure route: Ingestion
Equation for chronic daily intake (CDI) (mg/kg-day):

$$CDI = \frac{\left(\sum(EPC_s \times IR_s)\right) \times FI \times EF \times ED_a \times CF}{BW_a \times AT}$$

Where:

$$\sum(EPC_s \times IR_s) = (EPC_p \times IR_p) + (EPC_r \times IR_r) + (EPC_b \times IR_b) + (EPC_{bwb} \times IR_{bwb}) + (EPC_c \times IR_c) + (EPC_{cwb} \times IR_{cwb}) + (EPC_m \times IR_m) + (EPC_{cl} \times IR_{cl}) + (EPC_g \times IR_g) + (EPC_{gwb} \times IR_{gwb})$$

Parameter	Parameter Definition	Unit	Value	Rationale/
EPC-p	exposure point concentration in pelagic fish, perch	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-p	exposure point concentration in pelagic fish, rockfish	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-b	exposure point concentration in benthic fish, fillet	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-bwb	exposure point concentration in benthic fish, whole body	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-c	exposure point concentration in crabs, edible meat	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-cwb	exposure point concentration in crabs, whole body	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-m	exposure point concentration in mussels	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
EPC-cl	exposure point concentration in clams	mg/kg ww	Table B.3-42[a]	Section B.3.3.4
IR-p	ingestion rate – pelagic fish, perch	g/day	0.05	Section B.3.3.1.3
IR-r	ingestion rate – pelagic fish, rockfish	g/day	0.45	Section B.3.3.1.3
IR-b	ingestion rate – benthic fish, fillet	g/day	0.2	Section B.3.3.1.3
IR-bwb	ingestion rate – benthic fish, whole body	g/day	0.04	Section B.3.3.1.3
IR-c	ingestion rate – crabs, edible meat	g/day	0.6	Section B.3.3.1.3
IR-cwb	ingestion rate – crabs, whole body	g/day	0.5	Section B.3.3.1.3
IR-m	ingestion rate – mussels	g/day	0.5	Section B.3.3.1.3
IR-cl	ingestion rate –clams	g/day	3.0	Section B.3.3.1.3
FI	fractional intake derived from source	unitless	1[b]	Kissinger (2005)
EF	exposure frequency	days/yr	365[c]	EPA (1991a)
ED-a	exposure duration – adult	years	9	EPA (1989)
CF	conversion factor	kg/g	0.001	na
BW-a	body weight – adult	kg	63[d]	EPA (1999a)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	3,285	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

a EPCs for CT scenarios are based on mean concentrations, in contrast to the EPCs for the RME scenarios, which are based on 95% UCLs on mean concentrations.

b value of 1 was used for the fractional intake derived from source as directed by EPA (Kissinger 2005).

c Default exposure frequency of 350 days/yr modified to 365 days/yr to account for the fact that seafood consumption rate estimates are based on 365 days/yr.

d Average body weight for all surveyed individuals in API seafood consumption study in King County, as reported in EPA (1999a).

API – Asian and Pacific Islander
CT – central tendency
EPA – US Environmental Protection Agency
na – not applicable
UCL – upper confidence limit on the mean
ww – wet weight

Table B.3-22. Daily intake calculations – seafood ingestion, adult one-meal-per-month scenario

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Exposure route: Ingestion
Equations for chronic daily intake (CDI) (mg/kg-day):

$$CDI = \frac{(EPC_p \times IR_p) \times FI \times EF \times ED_a \times CF}{BW_a \times AT}$$

$$CDI = \frac{(EPC_c \times IR_c) \times FI \times EF \times ED_a \times CF}{BW_a \times AT}$$

$$CDI = \frac{(EPC_r \times IR_r) \times FI \times EF \times ED_a \times CF}{BW_a \times AT}$$

$$CDI = \frac{(EPC_{cl} \times IR_{cl}) \times FI \times EF \times ED_a \times CF}{BW_a \times AT}$$

$$CDI = \frac{(EPC_b \times IR_b) \times FI \times EF \times ED_a \times CF}{BW_a \times AT}$$

Parameter Code	Parameter Definition	Unit	Value	Rationale/ Reference
EPC-p	exposure point concentration in pelagic fish, perch[a]	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-r	exposure point concentration in pelagic fish, rockfish[a]	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-b	exposure point concentration in benthic fish, fillet	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-c	exposure point concentration in crabs, edible meat	mg/kg ww	Table B.3-42	Section B.3.3.4
EPC-cl	exposure point concentration in intertidal clams	mg/kg ww	Table B.3-42	Section B.3.3.4
IR-p	ingestion rate – pelagic fish, perch	g/day	7.5[b]	Section B.3.3.1.4
IR-r	ingestion rate – pelagic fish, rockfish	g/day	7.5[b]	Section B.3.3.1.4
IR-b	ingestion rate – benthic fish, fillet	g/day	7.5[b]	Section B.3.3.1.4
IR-c	ingestion rate – crabs, edible meat	g/day	7.5[b]	Section B.3.3.1.4
IR-cl	ingestion rate – clams	g/day	7.5[b]	Section B.3.3.1.4
FI	fractional intake derived from source	unitless	1	na
EF	exposure frequency	days/yr	365[c]	EPA (1991a)
ED-a	exposure duration – adult	years	30	EPA (1989)
CF	conversion factor	kg/g	0.001	na
BW-a	body weight – adult	kg	71.8	EPA (1997a)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	10,950	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

[a] The adult one-meal-per-month pelagic fish consumption scenario was evaluated using both perch and rockfish.

[b] Adult one-meal-per-month consumption was evaluated by individual seafood categories independently to provide information to the public and risk managers on consumption of various potential types of fish and shellfish. Risks from adult one-meal-per-month consumption were divided into five scenarios that address risks individually for each of the four main seafood consumption categories (i.e., benthic fish, rockfish, perch, clams, and crabs). Each scenario assumes that one 227 gram (8 oz.) meal is consumed per month, which equates to 7.5 g/day. Consumption of anadromous fish (e.g., salmon) was not considered based on the EPA recommendation that the site-related concentration term for salmon is zero for bioaccumulative contaminants (EPA 2005a).

[c] Default exposure frequency of 350 days/yr modified to 365 days/yr to account for the fact that seafood consumption rate estimates are based on 365 days/yr.

EPA – US Environmental Protection Agency

na – not applicable

ww – wet weight

B.3.3.2 Sediment exposure scenarios and parameters

As indicated in the CSM for this HHRA, direct exposure to sediment may occur through occupational (e.g., netfishing), clam harvesting, or recreational activities. Several scenarios were evaluated for the EW in an effort to capture the range of potential exposure magnitude (i.e., the amount of skin exposure to sediment and the amount of sediment incidentally ingested), frequency of exposure, and exposure areas within the EW. Workers involved in tribal commercial netfishing in the EW may come in contact with sediment. The gillnet lead lines typically come in contact with sediment during normal operations. The netfishers may contact this sediment incidentally upon net retrieval and may then also have incidental contact with sediment suspended in surface water. People conducting intertidal habitat restoration routinely come in contact with sediment, though the frequency and duration of exposure would be expected to be less than that for tribal netfishing. Finally, tribal members and the general public may choose to collect clams in intertidal areas of the EW.[32]

B.3.3.2.1 Summary of human access survey results

As discussed in the human access survey report for the EW (Windward 2008a), there are currently only three areas in the EW where the general public can access the shoreline from upland areas: Jack Perry Memorial Shoreline Public Access, Terminal 102 (T-102), and the shoreline below the bridge complex (West Seattle Bridge, Spokane Street Bridge, and railroad bridge) (Map B.3-1). There are a few intertidal areas that may change from restricted to public access areas in the future (e.g., the bank north of the Spokane Street Bridge on the east side of the EW). Restrictions on intertidal shoreline areas presented in the survey results apply to the general public. Members of the Suquamish and Muckleshoot Tribes have U&A fishing rights throughout the EW, including areas where intertidal sediment is present, and do not have access restrictions. Tribal members therefore have access to all available intertidal areas of the EW shoreline.

B.3.3.2.2 Exposure areas and parameters

Two commercial netfishing scenarios were evaluated for adult exposures: a CT scenario that assumes a typical frequency and duration of netfishing activity, as recommended by EPA, and an RME scenario that assumes more frequent and longer-term netfishing. The exposure areas for netfishing were assumed to cover the entire EW. Data from sediment samples taken from throughout the waterway, including intertidal and subtidal areas, were included. For tribal clamming, an RME (120 days per year) and a

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[32] Tribal members may also collect geoducks subtidally. However, risks associated with dermal sediment exposure are unlikely because individuals engaged in geoduck collection must wear scuba gear, (e.g., wet- or dry-suits, face masks, and gloves), which would insulate them from the cold water as well as protect them from sediment exposure. Thus, an exposure scenario specific to geoduck collection (i.e., a subtidal sediment exposure specific to clamming) will not be evaluated in the EW HHRA. However, exposure to subtidal sediment will be addressed in the fish collection (netfishing) scenario, which includes exposure to all surface sediment in the EW, both intertidal and subtidal.

183-day-per-year scenario were evaluated. One habitat restoration scenario was evaluated. For the tribal clamming and habitat restoration scenarios, the exposure areas included sediment samples from all intertidal areas not covered by overhanging docks. Exposure units for the intertidal exposure scenarios (i.e., clamming and habitat restoration) are indicated on Map B.3-1. The netfishing and clamming scenarios for the EW used the same exposure parameters as those used for the LDW HHRA (Windward 2007c). The habitat restoration worker scenario used most of the same parameters as those in the LDW HHRA; however, this scenario is discussed in the risk characterization section in this HHRA rather than in the uncertainty analysis, as was done in the LDW HHRA.[33]

For non-tribal clamming, a 7-day-per-year scenario was evaluated. This exposure frequency was assumed to be once per month during months when there was a minus tide during daylight hours, based on National Oceanic and Atmospheric Administration (NOAA) tidal information (NOAA 2006) from 2004 through 2006. This is consistent with the clamming 7-day-per-year scenario evaluated in the LDW HHRA. As previously discussed (Section B.3.3.2.1.), based on findings from the human access survey (Windward 2008a), there are currently only a few areas in the EW where the general public can access the shoreline from upland: Jack Perry Memorial Shoreline Public Access, T-102, and the shoreline below the West Seattle Bridge, Spokane Street Bridge, and railroad bridge. However, at T-102, the intertidal area is primarily composed of gravel, cobble, and boulders with no exposed intertidal sediment, so clamming is not possible. Therefore, the exposed sediment at the other areas comprised the exposure area for the 7-day-per-year non-tribal clamming scenario (Map B.3-1).

This section provides a summary of the CDI calculations for COPCs for sediment exposure as well as details on the exposure parameters used to evaluate each sediment exposure scenario. Exposure to COPCs in sediment is expressed as the CDI, which is the estimated daily chemical dose for an individual that occurs over the exposure duration for each scenario. Two routes of exposure are relevant: ingestion and dermal contact. The CDI for ingestion (which is calculated the same way as for the ingestion of seafood) is calculated as:

$$CDI_o = \frac{EPC \times IR \times FI \times EF \times ED \times CF}{BW \times AT}$$

Equation 3-2

Where:

CDI_o = chronic daily intake from oral exposure route (mg/kg-day)

EPC = chemical-specific exposure point concentration (mg/kg)

[33] The inclusion of the habitat restoration worker scenario in the risk characterization section of the HHRA (as compared with its inclusion in the uncertainty analysis of the LDW HHRA) was a result of stakeholder requests. The EW HHRA does not include the evaluation of the child beach play scenario because of the lack of suitable exposure areas, and thus the habitat restoration worker scenario provides an additional measure of the risks associated with direct sediment exposure for the EW.

IR = sediment ingestion rate (g/day)
FI = fractional intake of media derived from contaminated source (unitless)
EF = exposure frequency (days per year)
ED = exposure duration (years)
CF = conversion factor (kg/g)
BW = body weight (kg)
AT = averaging time (days), equivalent to the ED for non-carcinogenic COPCs and 70 years for carcinogenic COPCs

The CDI for dermal exposure[34] is calculated as:

$$CDI_d = \frac{EPC \times ABS \times SA \times AF \times FI \times EF \times ED \times CF}{BW \times AT}$$ Equation 3-3

Where:

CDI_d = chronic daily intake from dermal exposure route (mg/kg-day)
EPC = chemical-specific exposure point concentration (mg/kg)
ABS = dermal absorption fraction (unitless)
SA = skin surface area exposed (cm^2)
AF = sediment to skin adherence factor by event (mg/cm^2-event)
FI = fractional intake of media derived from contaminated source (unitless)
EF = exposure frequency (events/year)
ED = exposure duration (years)
CF = conversion factor (kg/mg)
BW = body weight (kg)
AT = averaging time (days), equivalent to the ED for non-carcinogenic COPCs and 70 years for carcinogenic COPCs

Two parameters that warrant additional discussion, dermal adherence factor (AF) and dermal absorption fraction (ABS), are discussed in Sections B.3.3.2.3 and B.3.3.2.4.

Sediment exposure scenarios are summarized in Table B.3-23 for netfishing, habitat restoration, and clamming. All scenarios include exposures from dermal contact and incidental ingestion of sediment. Key exposure parameters relative to these exposure routes that highlight the differences across scenarios are provided in Tables B.3-24 to B.3-35.

[34] Although chronic daily intake technically refers only to oral exposure, this term is used in this HHRA to refer to dermal exposure, which is technically an absorbed dose. For this HHRA, the difference in internal exposures between orally administered doses and dermally absorbed doses was made by adjusting the oral toxicological benchmarks, as appropriate, according to EPA guidance (2004b).

Table B.3-23. Summary of sediment exposure scenarios

Scenario	Incidental Sediment IR (g/day)	Exposure Frequency (days/yr)	Exposure Duration (years)	Skin Surface Area Exposed (cm^2)	Location of Scenario-Specific Details	
					Incidental Ingestion	Dermal Absorption
Netfishing RME	0.050	119	44	3,600	Table B.3-24	Table B.3-25
Netfishing CT	0.050	63	29	3,600	Table B.3-26	Table B.3-27
Habitat restoration worker	0.1	15	20	6,040	Table B.3-28	Table B.3-29
Tribal clamming RME	0.1	120	64	6,040	Table B.3-30	Table B.3-31
Tribal clamming – 183 days per year	0.1	183	70	6,040	Table B.3-32	Table B.3-33
Clamming – 7 days per year	0.1	7	30	6,040	Table B.3-34	Table B.3-35

CT – central tendency
IR – ingestion rate
RME – reasonable maximum exposure

Table B.3-24. Daily intake calculations – incidental sediment ingestion during netfishing, adult tribal RME scenario

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Exposure route: Ingestion (incidental)
Equation for chronic daily intake (CDI) (mg/kg-day):

$$CDI = \frac{EPC \times IR_s \times FI \times EF \times ED \times CF}{BW_a \times AT}$$

Parameter Code	Parameter Definition	Unit	Value	Rationale/ Reference
EPC	exposure point concentration in sediment	mg/kg dw	Table B.3-46	Section B.3.3.4
IR-s	incidental ingestion rate	g/day	0.050	EPA (1991a)
FI	fractional intake derived from source	unitless	1[a]	na
EF	exposure frequency	days/yr	119[b]	na
ED	exposure duration	years	44[b]	na
CF	conversion factor	kg/g	0.001	na
BW-a	body weight – adult	kg	81.8	Toy et al. (1996)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	16,060	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

[a] Fractional intake of 1 was used to be consistent with EPA direction for seafood consumption scenarios.

[b] Value recommended by EPA based on the length of the 2001 salmon season and on conversations with Muckleshoot Indian Tribe Assistant Harvest Manager regarding fishing frequency. This approach assumes that a fisher is present for each day of the fishing season. See Subappendix B.3 in Windward (2003) for more details on the derivation of this value.

dw – dry weight

EPA – US Environmental Protection Agency

na – not applicable

RME – reasonable maximum exposure

Table B.3-25. Daily intake calculations – dermal contact with sediment during netfishing, adult tribal RME scenario

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Exposure route: Dermal
Equation for chronic daily intake (CDI) (mg/kg-day):

$$CDI = \frac{EPC \times ABS \times SA \times AF \times FI \times EF \times ED \times CF}{BW_a \times AT}$$

Parameter Code	Parameter Definition	Unit	Value	Rationale/ Reference
EPC	exposure point concentration in sediment	mg/kg dw	Table B.3-46	Section B.3.3.4
ABS	dermal absorption fraction	unitless	Table B.3-36	Section B.3.3.2.4
SA	skin surface area exposed	cm^2	3,600[a]	EPA (1997a)
AF	adherence factor by event	mg/cm^2-event	0.2	EPA (1999c)
FI	fractional intake derived from source	unitless	1[b]	na
EF	exposure frequency	events/yr	119[c]	na
ED	exposure duration	years	44[c]	na
CF	conversion factor	kg/mg	0.000001	na
BW-a	body weight – adult	kg	81.8	Toy et al. (1996)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	16,060	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

a Recommended surface area for commercial/industrial worker. Assumes that head, hands, and forearms are exposed. Selected value represents sum of 50th percentile surface areas for men (most netfishers are men) for these body parts; taken from Table 6-2 in EPA (1997a). Given the higher body weight of individuals surveyed in Toy et al. (1996) compared with the general US population, the surface area values selected here for commercial/industrial workers may underestimate the surface area of tribal fishermen body parts. However, no conversion data are available at the present time.

b Fractional intake of 1 was used to be consistent with EPA direction for seafood consumption scenarios.

c Value recommended by EPA based on conversation with Muckleshoot Indian Tribe Assistant Harvest Manager. See Subappendix B.3 in Windward (2003) for more details on the derivation of this value.

dw – dry weight

EPA – US Environmental Protection Agency

na – not applicable

RME – reasonable maximum exposure

Table B.3-26. Daily intake calculations – incidental sediment ingestion during netfishing, adult tribal CT scenario

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Exposure route: Ingestion (incidental)
Equation for chronic daily intake (CDI) (mg/kg-day):

$$CDI = \frac{EPC \times IR_s \times FI \times EF \times ED \times CF}{BW_a \times AT}$$

Parameter Code	Parameter Definition	Unit	Value	Rationale/ Reference
EPC	exposure point concentration in sediment	mg/kg dw	Table B.3-46	Section B.3.3.4
IR-s	incidental ingestion rate	g/day	0.050	EPA (1991a)
FI	fractional intake derived from source	unitless	1[b]	na
EF	exposure frequency	days/yr	63[c]	na
ED	exposure duration	years	29[d]	na
CF	conversion factor	kg/g	0.001	na
BW-a	body weight – adult	kg	81.8	Toy et al. (1996)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	10,585	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

[a] EPCs for CT scenarios were based on mean concentrations, in contrast to the EPCs for the RME scenarios, which were based on 95% UCLs on mean concentrations.

[b] Fractional intake of 1 was used to be consistent with EPA direction for seafood consumption scenarios.

[c] Value recommended by EPA based on conversation with Muckleshoot Indian Tribe Assistant Harvest Manager. Selected value is duration of coho fishing season (most individual fish for coho). See Appendix B, Section B.3, in Windward (2003) for more details on the derivation of this value.

[d] Value recommended by EPA based on conversation with Muckleshoot Indian Tribe Assistant Harvest Manager. Selected value is EPA's best professional judgment assuming that fishing starts at age 16 and ends at age 45.

CT – central tendency
dw – dry weight
EPA – US Environmental Protection Agency
na – not applicable
UCL – upper confidence limit on the mean

Table B.3-27. Daily intake calculations – dermal contact with sediment during netfishing, adult tribal CT scenario

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Exposure route: Dermal
Equation for chronic daily intake (CDI) (mg/kg-day):

$$CDI = \frac{EPC \times ABS \times SA \times AF \times FI \times EF \times ED \times CF}{BW_a \times AT}$$

Parameter Code	Parameter Definition	Unit	Value	Rationale/ Reference
EPC	exposure point concentration in sediment	mg/kg dw	Table B.3-46	Section B.3.3.4
ABS	dermal absorption fraction	unitless	Table B.3-36	Section B.3.3.2.4
SA	skin surface area exposed	cm^2	3,600[b]	EPA (1997a)
AF	adherence factor by event	mg/cm^2-event	0.02[c]	EPA (2004b)
FI	fractional intake derived from source	unitless	1[d]	na
EF	exposure frequency	event/year	63[e]	na
ED	exposure duration	years	29[f]	na
CF	conversion factor	kg/mg	0.000001	na
BW-a	body weight – adult	kg	81.8	Toy et al. (1996)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	10,585	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

[a] EPCs for CT scenarios are based on mean concentrations, in contrast to the EPCs for the RME scenarios, which are based on 95% UCLs on mean concentrations.

[b] Recommended surface area for commercial/industrial worker. Assumes that head, hands, and forearms are exposed. Selected value represents sum of 50th percentile surface areas for men (most netfishers are men) for these body parts; taken from Table 6-2 in EPA (1997a). Given the higher body weight of individuals surveyed in Toy et al. (1996) compared to the general US population, the surface area values selected here for commercial/industrial workers may underestimate the surface area of tribal fishermen body parts. However, no conversion data are available at the present time.

[c] Default value for CT industrial workers in the Risk Assessment Guidance for Superfund, Part E (EPA 2004b).

[d] Fractional intake of 1 was used to be consistent with EPA direction for seafood consumption scenarios.

[e] Value recommended by EPA based on conversation with Muckleshoot Indian Tribe Assistant Harvest Manager. Selected value is duration of coho fishing season (most individual fish for coho). See Subappendix B.3 in Windward (2003) for more details on the derivation of this value.

[f] Value recommended by EPA based on conversation with Muckleshoot Indian Tribe Assistant Harvest Manager. Selected value is EPA's best professional judgment assuming that fishing starts at age 16 and ends at age 45.

CT – central tendency
dw – dry weight
EPA – US Environmental Protection Agency
na – not applicable
UCL – upper confidence limit on the mean

Table B.3-28. Daily intake calculations – incidental sediment ingestion during habitat restoration

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Exposure route: Ingestion (incidental)
Equation for chronic daily intake (CDI) (mg/kg-day):

$$CDI = \frac{EPC \times IR_s \times FI \times EF \times ED \times CF}{BW_a \times AT}$$

Parameter Code	Parameter Definition	Unit	Value	Rationale/ Reference
EPC	exposure point concentration in sediment	mg/kg dw	Table B.3-46	Section B.3.3.4
IR-s	incidental ingestion rate	g/day	0.1[a]	EPA (1997a)
FI	fractional intake derived from source	unitless	1[b]	na
EF	exposure frequency	days/yr	15[c]	na
ED	exposure duration	years	20[d]	EPA (1989)
CF	conversion factor	kg/g	0.001	na
BW-a	body weight – adult	kg	71.8[e]	EPA (1997a)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	7,300	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

[a] Default for agricultural and residential exposure (EPA 1997a).

[b] Fractional intake of 1 was used to be consistent with EPA direction for seafood consumption scenarios.

[c] Assume biologist only on site during a restoration activity. This is consistent with value used in LDW HHRA (Windward 2007c).

[d] Accounts for a reasonably long career in the same position, but assumes that the most senior scientists will spend very little time in the field.

[e] Mean body weight for male and female adults from Table 7-2 in EPA (1997a).

dw – dry weight

EPA – US Environmental Protection Agency

na – not applicable

Table B.3-29. Daily intake calculations – dermal contact with sediment during habitat restoration

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Exposure route: Dermal
Equation for chronic daily intake (CDI) (mg/kg-day):

$$CDI = \frac{EPC \times ABS \times SA \times AF \times FI \times EF \times ED \times CF}{BW_a \times AT}$$

Parameter Code	Parameter Definition	Unit	Value	Rationale/ Reference
EPC	exposure point concentration in sediment	mg/kg dw	Table B.3-46	Section B.3.3.4
ABS	dermal absorption fraction	unitless	Table B.3-36	Section B.3.3.2.4
SA	skin surface area exposed	cm^2	6,040[a]	EPA (1997a)
AF	adherence factor by event	mg/cm^2-event	0.2[b]	EPA (2004b)
FI	fractional intake derived from source	unitless	1[c]	na
EF	exposure frequency	events/yr	15[d]	na
ED_i	exposure duration	years	20[e]	EPA (1989)
CF	conversion factor	kg/mg	0.000001	na
BW-a	body weight – adult	kg	71.8[f]	EPA (1997a)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	7,300	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

[a] Skin surface area used for adult clamming scenario in this HHRA.

[b] Default health-protective factor for exposures of children and adults to moist soil recommended by EPA (2004b).

[c] Fractional intake of 1 was used to be consistent with EPA direction for seafood consumption scenarios.

[d] Assume biologist only on site during a restoration activity. This is consistent with value used in LDW HHRA (Windward 2007c).

[e] Accounts for a reasonably long career in the same position, but assumes that the most senior scientists will spend very little time in the field.

[f] Mean body weight for male and female adults from Table 7-2 in EPA (1997a).

dw – dry weight

EPA – US Environmental Protection Agency

na – not applicable

Table B.3-30. Daily intake calculations – incidental sediment ingestion during tribal clamming RME scenario

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Exposure route: Ingestion (incidental)
Equation for chronic daily intake (CDI) (mg/kg-day):

$$CDI = \frac{EPC \times IR_s \times FI \times EF \times ED \times CF}{BW_a \times AT}$$

Parameter Code	Parameter Definition	Unit	Value	Rationale/ Reference
EPC	exposure point concentration in sediment	mg/kg dw	Table B.3-46	Section B.3.3.4
IR-s	Incidental ingestion rate	g/day	0.1	EPA (1997a)
FI	fractional intake derived from source	unitless	1[a]	na
EF	exposure frequency	days/yr	120[b]	Kissinger (2007c)
ED	exposure duration	years	64[c]	Kissinger (2007c)
CF	conversion factor	kg/g	0.001	na
BW-a	body weight – adult	kg	81.8	Toy et al. (1996)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	23,360	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

[a] Fractional intake of 1 was used to be consistent with EPA direction for seafood consumption scenarios.

[b] Exposure frequency determined by EPA to reflect tribal clamming patterns (Kissinger 2007c).

[c] Exposure duration determined by EPA to reflect tribal clamming patterns (Kissinger 2007c).

dw – dry weight

EPA – US Environmental Protection Agency

na – not applicable

Table B.3-31. Daily intake calculations – dermal contact with sediment during tribal clamming RME scenario

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Exposure route: Dermal
Equation for chronic daily intake (CDI) (mg/kg-day):

$$CDI = \frac{EPC \times ABS \times SA \times AF \times FI \times EF \times ED \times CF}{BW_a \times AT}$$

Parameter Code	Parameter Definition	Unit	Value	Rationale/ Reference
EPC	exposure point concentration in sediment	mg/kg dw	Table B.3-46	Section B.3.3.4
ABS	dermal absorption fraction	unitless	Table B.3-36	Section B.3.3.2.4
SA	skin surface area exposed	cm^2	6,040[a]	EPA (1997a)
AF	adherence factor by event	mg/cm^2-event	0.2	EPA (2004b)
FI	fractional intake derived from source	unitless	1[b]	na
EF	exposure frequency	events/yr	120	Kissinger (2007c)
ED_i	exposure duration	years	64	Kissinger (2007c)
CF	conversion factor	kg/mg	0.000001	na
BW-a	body weight – adult	kg	81.8	Toy et al. (1996)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	23,360	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

[a] Assumes that 39% of the total body surface area is exposed, roughly corresponding to a barefoot individual wearing a short-sleeve shirt and short pants (EPA 1992a). Body surface area data taken from Tables 6-2, 6-3 and 6-4 in EPA (1997a) and corresponds to head, lower arms, hands, lower legs, and feet.

[b] Fractional intake of 1 was used to be consistent with EPA direction for seafood consumption scenarios.

[c] Exposure frequency determined by EPA to reflect tribal clamming patterns (Kissinger 2007c).

[d] Exposure duration determined by EPA to reflect tribal clamming patterns (Kissinger 2007c).

dw – dry weight

EPA – US Environmental Protection Agency

na – not applicable

Table B.3-32. Daily intake calculations – incidental sediment ingestion during tribal clamming, 183-day-per-year scenario

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Exposure route: Ingestion (incidental)
Equation for chronic daily intake (CDI) (mg/kg-day):

$$CDI = \frac{EPC \times IR_s \times FI \times EF \times ED \times CF}{BW_a \times AT}$$

Parameter Code	Parameter Definition	Unit	Value	Rationale/ Reference
EPC	exposure point concentration in sediment	mg/kg dw	Table B.3-46	Section B.3.3.4
IR-s	Incidental ingestion rate	g/day	0.1	EPA (1997a)
FI	fractional intake derived from source	unitless	1[a]	na
EF	exposure frequency	days/yr	183[b]	Kissinger (2007c)
ED	exposure duration	years	70[c]	Kissinger (2007c)
CF	conversion factor	kg/g	0.001	na
BW-a	body weight – adult	kg	81.8	Toy et al. (1996)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	25,550	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

[a] Fractional intake of 1 was used to be consistent with EPA direction for seafood consumption scenarios.

[b] Exposure frequency requested by Muckleshoot and Suquamish Tribes (Kissinger 2007c).

[c] Exposure duration requested by Muckleshoot and Suquamish Tribes (Kissinger 2007c).

dw – dry weight
EPA – US Environmental Protection Agency
na – not applicable

Table B.3-33. Daily intake calculations – dermal contact with sediment during tribal clamming, 183-day-per-year scenario

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Exposure route: Dermal
Equation for chronic daily intake (CDI) (mg/kg-day):

$$CDI = \frac{EPC \times ABS \times SA \times AF \times FI \times EF \times ED \times CF}{BW_a \times AT}$$

Parameter Code	Parameter Definition	Unit	Value	Rationale/ Reference
EPC	exposure point concentration in sediment	mg/kg dw	Table B.3-46	Section B.3.3.4
ABS	dermal absorption fraction	unitless	Table B.3-36	Section B.3.3.2.4
SA	skin surface area exposed	cm^2	6,040[a]	EPA (1997a)
AF	adherence factor by event	mg/cm^2-event	0.2	EPA (2004b)
FI	fractional intake derived from source	unitless	1[b]	na
EF	exposure frequency	events/yr	183[c]	Kissinger (2007c)
ED_i	exposure duration	years	70[d]	Kissinger (2007c)
CF	conversion factor	kg/mg	0.000001	na
BW-a	body weight – adult	kg	81.8	Toy et al. (1996)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	25,550	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

a Assumes that 39% of the total body surface area is exposed, roughly corresponding to a barefoot individual wearing a short-sleeve shirt and short pants (EPA 1992a). Body surface area data taken from Tables 6-2, 6-3 and 6-4 in EPA (1997a) and corresponds to head, lower arms, hands, lower legs, and feet.

b Fractional intake of 1 was used to be consistent with EPA direction for seafood consumption scenarios.

c Exposure frequency requested by Muckleshoot and Suquamish Tribes (Kissinger 2007c).

d Exposure duration requested by Muckleshoot and Suquamish Tribes (Kissinger 2007c).

dw – dry weight

EPA – US Environmental Protection Agency

na – not applicable

Table B.3-34. Daily intake calculations – incidental sediment ingestion during clamming, 7-day-per-year scenario

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Exposure route: Ingestion (incidental)
Equation for chronic daily intake (CDI) (mg/kg-day):

$$CDI = \frac{EPC \times IR_s \times FI \times EF \times ED \times CF}{BW_a \times AT}$$

Parameter Code	Parameter Definition	Unit	Value	Rationale/ Reference
EPC	exposure point concentration in sediment	mg/kg dw	Table B.3-46	Section B.3.3.4
IR-s	incidental ingestion rate	g/day	0.1	EPA (1997a)
FI	fractional intake derived from source	unitless	1[a]	na
EF	exposure frequency	days/yr	7[b]	na
ED	exposure duration	years	30	EPA (1989)
CF	conversion factor	kg/g	0.001	na
BW-a	body weight – adult	kg	71.8[c]	EPA (1997a)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	10,950	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

[a] Fractional intake of 1 was used to be consistent with EPA direction for seafood consumption scenarios.

[b] Exposure frequency was assumed to be once per month during months when there is a daylight minus tide, based on NOAA tidal information (NOAA 2006) from 2004 through 2006.

[c] Mean body weight for male and female adults from Table 7-2 in EPA (1997a).

dw – dry weight

EPA – US Environmental Protection Agency

na – not applicable

Table B.3-35. Daily intake calculations – dermal contact with sediment during clamming, 7-day-per-year scenario

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Exposure route: Dermal
Equation for chronic daily intake (CDI) (mg/kg-day):

$$CDI = \frac{EPC \times ABS \times SA \times AF \times FI \times EF \times ED \times CF}{BW_a \times AT}$$

Parameter Code	Parameter Definition	Unit	Value	Rationale/ Reference
EPC	exposure point concentration in sediment	mg/kg dw	Table B.3-46	Section B.3.3.4
ABS	dermal absorption fraction	unitless	Table B.3-36	Section B.3.3.2.4
SA	skin surface area exposed	cm^2	6,040[a]	EPA (1997a)
AF	adherence factor by event	mg/cm^2-event	0.2	EPA (2004b)
FI	fractional intake derived from source	unitless	1[b]	na
EF	exposure frequency	events/yr	7[c]	na
ED_i	exposure duration	years	30	EPA (1989)
CF	conversion factor	kg/mg	0.000001	na
BW-a	body weight – adult	kg	71.8[d]	EPA (1997a)
AT-C	averaging time – cancer	days	25,550	EPA (1989)
AT-N	averaging time – non-cancer	days	10,950	EPA (1989)

Source: Standard Table 4 in EPA (2001c)

a Assumes that 39% of the total body surface area is exposed, roughly corresponding to a barefoot individual wearing a short-sleeve shirt and short pants (EPA 1992a). Body surface area data taken from Tables 6-2, 6-3 and 6-4 in EPA (1997a) and corresponds to head, lower arms, hands, lower legs, and feet.

b Fractional intake of 1 was used to be consistent with EPA direction for seafood consumption scenarios.

c Exposure frequency was assumed to be once per month during months when there is a daylight minus tide, based on NOAA tidal information (NOAA 2006) from 2004 through 2006.

d Mean body weight for male and female adults from Table 7-2 in EPA (1997a).

dw – dry weight
EPA – US Environmental Protection Agency
na – not applicable
NOAA – National Oceanic and Atmospheric Administration

B.3.3.2.3 Dermal adherence factor

The potential for sediment to adhere to skin has not been well characterized. Data for AFs for marine sediment, such as that found in the EW, are extremely limited. A range of adherence factors exists for various soil conditions, including wet soil. Kissel et al. (1996) showed that soil adherence typically increases with increasing moisture content. Although EPA (2004b) guidelines address the increase in soil adherence factors associated with moisture present in soil or sediment, more recent research suggests that the actual marine sediment adherence factors may be higher than those derived by EPA for wet soil (Shoaf et al. 2005a, b). The level of adherence directly affects dermal exposure estimates. As sediment loading increases, the fraction of chemical that adheres to the skin and is available to be absorbed will remain constant until all of the skin is covered by a thin layer of soil (known as the mono-layer) (Duff and Kissel 1996). Once this mono-layer threshold is crossed, the fraction of chemical that can be absorbed will decrease, inasmuch as not all of the soil is in constant, direct contact with skin. Both the amount of soil required to form the mono-layer and the associated adherence capability of the soil depend on grain size. Generally, larger particles have a lower adherence factor than do smaller particles. However, as previously mentioned, wet marine sediment is generally expected to have higher adherence capabilities than similarly composed dry soil. For the purposes of this risk assessment, a value of 0.2 mg/cm^2-event (EPA 2004b) was used in all risk calculations for the clamming scenarios, the habitat restoration scenario, and the RME netfishing scenario. A lower adherence factor (0.02 mg/cm^2-event) (EPA 2004b) was used for the netfishing CT scenario. These are same values as those used in the LDW HHRA (Windward 2007c).

B.3.3.2.4 Dermal absorption fraction

The dermal absorption fraction (ABS) refers to the fraction of the chemical in sediment applied to the skin surface that is absorbed into the bloodstream. Many studies have focused on this topic, but there is considerable uncertainty regarding chemical-specific values (EPA 1992a). EPA (2004b) has developed supplemental guidance for dermal risk assessment that provides ABS values for many organic chemicals but provides ABS values for only one trace element COPC, arsenic (Table B.3-36). The guidance document states that speciation of inorganic substances is crucial to estimating dermal absorption and data are insufficient to derive default values for other inorganic substances. Older EPA guidance (EPA 2001b) on dermal absorption provided a general value of 0.01 for all metals, reflecting a generally low dermal absorption of metals. Because specific absorption values are not provided, the dermal absorption pathway was not evaluated quantitatively for metals without dermal absorption fractions. This approach is suggested in EPA's Risk Assessment Guidance for Superfund (RAGS), Part E: Supplemental Guidance for Dermal Risk Assessment (2004b), with values supplied in Exhibit 3-4 of that document.

Table B.3-36. Dermal absorption fractions for COPCs

Chemical	ABS (unitless)	Oral Absorption Adjustment[a]
Antimony	none	RfD × 0.15
Arsenic	0.03	none
Cobalt	none	none
Lead	none	none
Vanadium	none	RfD × 0.026
cPAH TEQ	0.13	none
1,4-Dichlorobenzene[b]	0.1	none
n-Nitrosodimethylamine[b]	0.1	none
Total PCBs	0.14	none
PCB TEQ	0.14	none
Toxaphene[c]	0.1	none
Dioxin/furan TEQ	0.03	none

Source: RAGS Part E (EPA 2004b)

[a] The oral adjustment values are presented in Exhibit 4-1 of EPA (2004b).

[b] The ABS value for semivolatile organic compounds is 0.1, as recommended in EPA (2004b).

[c] The ABS value for these organochlorine pesticides is the default value for semivolatile organic compounds, as recommended in EPA (2004b).

ABS – dermal absorption fraction
COPC – chemical of potential concern
cPAH – carcinogenic polycyclic aromatic hydrocarbon
EPA – US Environmental Protection Agency
PCB – polychlorinated biphenyl
RAGS – Risk Assessment Guidance for Superfund
RfD – reference dose
TEQ – toxic equivalent

The toxicological benchmarks discussed in Section B.4 are based on orally administered doses, which are not necessarily equivalent to dermally absorbed doses because of incomplete oral or dermal absorption. Although a summary of gastrointestinal absorption data for many chemicals is provided in Exhibit 4-1 of EPA's RAGS Part E: Supplemental Guidance for Dermal Risk Assessment (2004b), data are not available for all chemicals evaluated. In the case of organic chemicals evaluated in this HHRA, absorption via the oral route is greater than 50%. In these instances, EPA (2004b) recommends that no conversion of the oral toxicity value is needed. Thus, for this HHRA, a gastrointestinal absorption factor of 1 was used for organic chemicals (i.e., oral toxicological benchmarks were applied without modification).

Reference doses (RfDs) for dermal exposures are adjusted downward to reflect higher internal exposure from an absorbed dose compared with that from an ingested dose. The adjustment to the oral RfD (see Table B.3-36) is intended to reflect the internal dose that results from the dermally absorbed exposure. The adjustment for RfDs for dermal exposure is oral RfD × gastrointestinal fraction absorbed. Currently, EPA does not recommend an absorption adjustment for any chemical with a carcinogenic mode of action. In addition, for metals that lack an ABS factor (i.e., all EW metal COPCs for sediment except arsenic), no dermal absorption was assumed for the risk characterization; therefore, the RfD adjustment is not relevant. The approach presented in this section is consistent with that used in the LDW HHRA (Windward 2007c).

B.3.3.3 Surface water exposure parameters

As per the HHRA CSM, risks to humans were assessed based on exposure to chemical concentrations in surface water while swimming in the EW. As discussed in Section B.3.1.1, because of the limited public access and high shipping traffic in the EW now and expected in the future, opportunities for swimming in the EW are and will be limited. The swimming exposure scenario evaluated includes dermal absorption and incidental ingestion of surface water, as might occur when swimming from a boat or jumping or falling off a dock. This exposure scenario was evaluated only for adults because children (6 years of age or younger) would not be expected to swim in the EW. The exposure area for this scenario is the entire EW study area.

This section discusses the methods that were used to calculate the CDI rates associated with this exposure pathway and presents the values used to parameterize this scenario. The scenario parameters are generally based on the adult swimming from boat scenarios presented in the *King County Combined Sewer Overflow Water Quality Assessment for the Duwamish River and Elliott Bay* (King County 1999a), and as agreed upon for application to the EW in the QAPP for the surface water data collection effort (Windward 2009f). Results from the King County WQA were also used to approximate swimming exposures and risks in the LDW HHRA (Windward 2007c).

B.3.3.3.1 Dermal exposure to surface water

This section discusses the calculation of dermal exposure to chemical concentrations in surface water (e.g., while swimming or wading). These risks were calculated using Equation 3-4 (EPA 2004b):

$$CDI = \frac{DA_{event} \times EV \times EF \times ED \times SA_{w}}{BW \times AT}$$

Equation 3-4

Where:

CDI	=	chronic daily intake rate, dermally absorbed dose (mg/kg-day)
DA_{event}	=	absorbed dose per event (mg/cm^2-event)
EV	=	event frequency (events/day)
EF	=	exposure frequency (days/year)
ED	=	exposure duration (years)
SA_w	=	skin surface area (cm^2)
BW	=	body weight (kg)
AT	=	averaging time (cancer or non-cancer) (days)

The absorbed dose per event (DA_{event}) is calculated differently for organic and inorganic compounds based on the different absorption properties of these chemicals. Equation 3-5 presents the approach for calculating the absorbed dose per event for inorganic chemicals (EPA 2004b):

$$DA_{event} = K_p \times C_w \times t_{event}$$ **Equation 3-5**

Where:

DA_{event}	=	absorbed dose per event (mg/cm²-event)
K_p	=	chemical-specific dermal permeability coefficient of compound in water (cm/hour)
C_w	=	chemical concentration in water (mg/cm³)
t_{event}	=	event duration (hours/event)

For organic chemicals, depending on whether the time needed to reach steady state with regard to absorption through the skin is greater or less than the event duration, a different equation is required to calculate dose per event. Equations **3-**6 and **3-**7 present the two approaches for calculating the absorbed dose per event for organic chemicals (EPA 2004b):

If $t_{event} \leq t^*$, then Equation 3-6 should be used:

$$DA_{event} = 2 \times FA \times K_p \times C_w \times \sqrt{\frac{6 \times \tau_{event} \times t_{event}}{\pi}}$$ **Equation 3-6**

If $t_{event} > t^*$, then Equation 3-**7** should be used:

$$DA_{event} = FA \times K_p \times C_w \times \left[\frac{t_{event}}{1+B} + 2 \times \tau_{event} \times \left(\frac{1+3B+3B^2}{(1+B)^2}\right)\right]$$ **Equation 3-7**

Where:

t_{event}	=	event duration (hours/event)
t^*	=	time to reach steady-state (hours)
DA_{event}	=	absorbed dose per event (mg/cm²-event)
FA	=	chemical-specific fraction absorbed from water (unitless)
K_p	=	chemical-specific dermal permeability coefficient of compound in water (cm/hour)
C_w	=	chemical concentration in water (mg/cm³)
τ_{event}	=	chemical-specific lag time per event (hr/event)
B	=	ratio of the permeability coefficient of a compound through the stratum corneum (one of two skin layers) relative to its permeability coefficient across the variable epidermis (one of two skin layers) (unitless)

B.3.3.3.2 *Incidental ingestion of surface water*

Exposure to chemical concentrations via the incidental ingestion of surface water while swimming is calculated as shown in Equation 3-8 (EPA 1989):

$$CDI = \frac{C_w \times IR_w \times t_{event} \times EV \times EF \times ED \times CF}{BW \times AT}$$

Equation 3-8

Where:

CDI	=	chronic daily intake rate, dermally absorbed dose (mg/kg-day)
C_w	=	chemical concentration in water (mg/cm^2)
IR_w	=	incidental water ingestion rate (ml/hr)
t_{event}	=	event duration (hrs/event)
EV	=	event frequency (events/day)
EF	=	exposure frequency (assume one event per day) (days/year)
ED	=	exposure duration (years)
CF	=	conversion factor (1×10^{-3}) (L/ml)
BW	=	body weight (kg)
AT	=	averaging time (cancer or non-cancer) (days)

B.3.3.3.3 *Exposure parameters*

The swimming scenario was parameterized using the values presented in the King County WQA (King County 1999a). Table B.3-37 presents the chemical-specific values, and Table B.3-38 presents a summary of the site-specific parameters for the swimming scenario. The parameterization of the low, medium, and high exposure levels evaluated for the EW was generally consistent with the low, medium, and high exposure levels evaluated for the King County WQA. When parameterizing the swimming scenarios, King County considered both EPA Region 10 guidance and qualitatively considered the low water temperatures (King County 1999a, b). For reference, it should be noted that the water temperatures in the EW are generally quite cold. In samples collected in 2008 and 2009, water temperatures ranged from 5.1 to 14.1°C (41 to 57°F), with a mean of approximately 10°C (50°F). At these low temperatures, hypothermia (and even death by hypothermia) may become an issue after one hour (USSARTF 2012).

Table B.3-37. Summary of chemical-specific swimming exposure scenario values

Exposure Parameter	Symbol	Unit	Chemical-Specific Values[s]	Source
Chemical-specific dermal permeability coefficient of compound in water	K_p	cm/hour	arsenic: 0.001 chromium: 0.001 vanadium: 0.001 naphthalene: 0.047 total PCBs: 0.75 PCB TEQ: 0.81	EPA (2004b), Exhibit B-3 and B-4
Chemical-specific lag time per event (applicable only for organic compounds)	τ_{event}	hr/event	naphthalene: 0.56 total PCBs: 4.63 PCB TEQ: 6.82	EPA (2004b), Exhibit B-3
Time to reach steady-state (applicable only for organic compounds)	t*	hours	naphthalene: 1.34 total PCBs: 20.27 PCB TEQ: 30.09	EPA (2004b), Exhibit B-3
Chemical-specific fraction absorbed from water (applicable only for organic compounds)	FA	unitless	naphthalene: 1 total PCBs: 0.6 PCB TEQ: 0.5	EPA (2004b), Exhibit B-3
Ratio of the permeability coefficient of a compound through the stratum corneum[b] relative to its permeability coefficient across the variable epidermis[b] (applicable only for organic compounds)	B	unitless	naphthalene: 0.2 total PCBs:4.9 PCB TEQ: 5.6	EPA (2004b), Exhibit B-3

Note: These parameters are used to calculate the dermally absorbed dose based on exposure to surface water (see Section B.3.3.3.1 and Equations 3-4 through 3-7).

[a] Chemical-specific values for total PCBs and PCB TEQ were based on values for available individual chemicals. PCB 3 was used for total PCBs and 2,3,7,8-TCDD was used for PCB TEQ to calculate risks based on exposure to surface water.

[b] The stratum corneum and variable epidermis are the main two layers of skin. The stratum corneum is the main barrier preventing the absorption of chemicals.

EPA – US Environmental Protection Agency

PCB – polychlorinated biphenyl

TCDD – tetrachlorodibenzo-p-dioxin

TEQ – toxic equivalent

Table B.3-38. Summary of adult swimming scenario exposure parameters

Scenario Exposure Level	Incidental Water IR (ml/hr)	Event Duration (hours)	Exposure Frequency (days/yr)	Exposure Duration (years)[a]	Skin Surface Area Exposed (cm^2)	Location of Scenario-Specific Details	
						Incidental Ingestion	Dermal Absorption
Low	25	0.17	2	9	4,900	Table B.3-39	Table B.3-40
Medium	50	1	12	30	19,400		
High	70	2.6	24	70	21,800		

[a] Exposure duration for high-level exposure scenario was reduced from 75 years in the King County WQA (1999a) to 70 years, and exposure duration for medium-level exposure scenario was reduced from 33 years in the King County WQA (1999a) to 30 years to be consistent with other scenarios evaluated in this risk assessment.

IR – ingestion rate

WQA – water quality assessment

For consistency with other scenarios evaluated in this HHRA, several changes were made to the King County assessment parameterization. The King County assessment used a range of body weights ranging from 60 to 79 kg. A body weight of 71.8 kg (EPA

1997a) was used for low, medium, and high swimming exposure scenarios so that these could be combined with risk estimates for other scenarios that used this same body weight (e.g., habitat restoration worker and 7-day-per-year clamming). The assumed exposure duration for the high-level exposure scenario was reduced from 75 years in the King County assessment to 70 years, and the assumed exposure duration for the medium-level exposure scenario was reduced from 33 years in the King County assessment to 30 years to be consistent with other scenarios being evaluated in this risk assessment. The averaging times for cancer risk estimates were similarly adjusted. For non-cancer hazard estimates for all levels of exposure, the averaging time was equal to the exposure duration. Exposure parameters for incidental ingestion and dermal absorption for all three levels of exposure are provided in Tables B.3-39 and B.3-40, respectively.

Table B.3-39. Daily intake calculations – incidental ingestion of surface water during adult swimming scenario

Scenario timeframe: Current/future
Medium: Water
Exposure medium: Surface water
Exposure route: Ingestion (incidental)
Intake equation: Chronic daily intake (CDI) (mg/kg-day) = EPC × IR-w × t_{event} × EV × EF × ED × CF × 1/BW-a × 1/AT

Parameter Code	Parameter Definition	Unit	Value[a] Low	Value[a] Medium	Value[a] High	Rationale/ Reference
EPC	exposure point concentration in water	mg/cm^3	Table B.3-47			Section B.3.3.4
t_{event}	event duration	hrs/event	0.17	1	2.6	EPA (1988b); BPJ
EV	event frequency	events/day	1			BPJ
EF	exposure frequency	days/yr	2	12	24	EPA (1997a); BPJ
ED	exposure duration	years	9	30	70	EPA (1991a), consistency with other scenarios in EW HHRA
CF	conversion factor	L/mL	0.001			na
BW-a	body weight – adult	kg	71.8[b]			EPA (1997a)
IR-w	incidental ingestion rate	mL/hr	25	50	70	EPA (1991a), consistency with other scenarios in EW HHRA
AT-C	averaging time – cancer	days	25,550			EPA (2004b, 1989)
AT-N	averaging time – non-cancer	days	3,285	10,950	25,550	EPA (2004b, 1989)

Source: Standard Table 4 in EPA (2001c)

[a] A low, medium, and high exposure value was analyzed by King County (1999a).

[b] Mean body weight for male and female adults from Table 7-2 in EPA (1997a).

BPJ – best professional judgment

EPA – US Environmental Protection Agency

na – not applicable

Table B.3-40. Daily intake calculations – dermal exposure to surface water during adult swimming scenario

Scenario timeframe: Current/future
Medium: Water
Exposure medium: Surface water
Exposure route: Dermal
Intake equation: Chronic daily intake (CDI) (mg/kg-day) = $DA_{event} \times SA_w \times EV \times EF \times ED \times 1/BW\text{-}a \times 1/AT$

Parameter Code	Parameter Definition	Unit	Value[a]			Rationale/Reference
			Low	Medium	High	
DA_{event}	dermally absorbed dose per event[b]	mg/cm^2-event	Table B.3-47			Section B.3.3.3.1
t_{event}	event duration	hrs/event	0.17	1	2.6	EPA (1988b); BPJ
EV	event frequency	events/day	1			BPJ
EF	exposure frequency	days/yr	2	12	24	EPA (1997a); BPJ
ED	exposure duration	years	9	30	70	EPA (1991a), consistency with other scenarios in EW HHRA
BW-a	body weight – adult	kg	71.8[c]			EPA (1997a)
SA_w	skin surface area exposed	cm^2	4,900	19,400	21,800	EPA (1991a)
AT-C	averaging time – cancer	days	25,550			EPA (2004b, 1989)
AT-N	averaging time – non-cancer	days	3,285	10,950	25,550	EPA (2004b, 1989)

Source: Standard Table 4 in EPA (2001c)

[a] A low, medium, and high exposure value was analyzed by King County (1999a).

[b] The equations used to calculate the dermally absorbed dose are presented in Section B.3.3.3.1.

[c] Mean body weight for male and female adults from Table 7-2 in EPA (1997a).

BPJ – best professional judgment

EPA – US Environmental Protection Agency

na – not applicable

B.3.3.4 Exposure point concentrations

An EPC was calculated for each seafood consumption category, sediment exposure area, and the EW water exposure area. Figure B.3-3 shows the methods used to estimate EPCs based on the number of detected concentrations present in a given dataset. With the exception of the calculation of EPCs for the intertidal MIS sediment samples (described in Section B.3.3.4.2), the methods shown in Figure B.3-3 were applied for all other EPC calculations. Based on the COPC identification process, some chemicals were identified as COPCs even if they were never detected (i.e., if they have > 10% of reporting limits exceeding the RBC). Chemicals that were not detected in a particular media (water, sediment, or tissue) were evaluated in the uncertainty analysis for the appropriate pathways. However, if a chemical was detected and designated as a COPC for any seafood tissue type, EPCs were developed for the other tissue types so that market basket seafood exposure can be evaluated in the risk characterization section. Hence, EPCs were developed for some datasets for which there are no detected values.

No. of Detected Values		Method for Selecting EPC
0	→	Use one-half of the maximum reporting limit.
1 – 5	→	Select the higher of one-half the maximum reporting limit OR the maximum detected value.
6 or more	→	Use ProUCL 4.0, indicating detected and undetected values.

Figure B.3-3. Flow chart showing method for selecting EPC

The ProUCL software used for this analysis allows detected and undetected values to be indicated and creates interpolated values for non-detects based on the perceived distribution of the detected concentrations. Once any necessary interpolation is performed, the software conducts an analysis of the data to determine the most appropriate 95% UCL and makes a recommendation.

As stated previously, the rationale for selecting EPCs was based largely on the detection frequency for each chemical. The approach to calculate EPCs that is outlined above is intended to use all available data, be statistically defensible where possible, and adopt health-protective policies for deriving EPCs when statistical approaches for computing 95% UCLs are not available. This approach is also consistent with EPC calculation for the LDW HHRA (Windward 2007c). When fewer than six detected concentrations were available, the higher of either the maximum detected concentration or one-half the maximum RL was selected as the EPC. This approach was used because 95% UCLs calculated from datasets with very few detected concentrations are not expected to be reliable enough for deriving EPCs. Chemical contamination datasets are often positively skewed. For such positively skewed datasets, the true mean is greater than the 50th percentile and can be substantially greater when skewness is large. When the number of samples used to characterize an exposure area is very small (e.g., $n < 6$), there is a significant probability that the maximum result among those few samples will be less than the true mean. Even when using an approach that assigns the maximum sample result as an EPC, there is still a risk of underestimating exposures. This uncertainty is unavoidable when only a few samples are available to characterize an exposure area.

Certain classes of organic compounds are made up of individual compounds that have similar chemical structures as well as a common mechanism of toxicity. Exposure and toxicity are assessed for these classes on a group, rather than on an individual compound, basis. These compound groups include co-planar PCBs, chlorinated dioxins/furans and cPAHs. The methods for calculating totals (including PCB TEQ, dioxin/furan TEQ, and cPAH totals) on a sample-by-sample basis are briefly summarized here. The sum of the products of the concentration of each coplanar PCB and its TEF is called the PCB TEQ and is calculated on a per sample basis. Similarly, the sum of the products of each coplanar dioxin/furan and its TEF is the called the dioxin/furan TEQ and is also calculated on a per sample basis. The sum of the products of the concentration of each cPAH and its PEF is considered the cPAH TEQ and is calculated on a per sample basis. Once the TEQs for PCBs, dioxins/furans, and cPAHs are calculated on a per-sample bias, the methods for calculating the EPC for each of those is the same as that for other chemicals. Summary statistics, the distribution type, and the 95% UCL for chemical concentrations in tissue for all seafood consumption categories, sediment exposure areas, and water are presented in this HHRA. The methods for calculating the EPCs and the EPCs themselves for tissue, sediment, and water are described in detail in the following subsections.

B.3.3.4.1 Tissue

Based on the seafood consumption surveys summarized in Section B.3.3.1, 10 consumption categories based on seafood types were identified. Table B.3-41 lists the species for which tissue data were included to develop EPCs for each of the categories.

Given the high level of uncertainty in apportioning the consumption of multiple intertidal bivalve species at a specific site, it is important to consider how the concentrations of different contaminants vary across species present prior to selecting an apportionment method for the purpose of calculating a clam EPC. The decision to compute overall intertidal bivalve EPCs without considering species-specific EPC differences was made after it was determined that these differences had little impact on overall bivalve EPCs. In addition, unlike the other apportionment methods evaluated (i.e., a biomass-weighted approach), the use of one 95% UCL inclusive of all intertidal clam samples does not assume a specific distribution of different species will be collected repeatedly. Instead it is a 95% UCL for the mixture of intertidal clams actually collected in EW; the abundance of different clam species in the EW is not well understood but may differ from the assemblage of clams available in other areas of Puget Sound. The selection of this approach is specific to the EW and does not imply a precedent for selection of clam apportionment methods at other sites. Additional discussion of the uncertainty surrounding the use of a single 95% UCL that includes all clam species is provided in Section B.6.1.6.4.

Table B.3-41. Seafood consumption categories for developing EPCs

Seafood Category	EW Species and Tissue Types Included for Tissue Data
Benthic fish, fillet	English sole, skin-on and skinless fillet
Benthic fish, whole body	English sole, whole body and skin-on or skinless fillet and remainder[a]
Pelagic fish, rockfish	brown rockfish, whole body
Pelagic fish, perch[b]	shiner surfperch, whole body; striped perch, fillet
Crab, edible meat	Dungeness and red rock crab
Crab, whole body[c]	Dungeness and red rock crab
Clams[d]	all intertidal clams (butter clams, littleneck clams, cockles, and soft-shell clams)
Mussels	blue mussel
Geoduck clams, edible meat	geoducks
Geoduck clams, whole body[e]	geoduck, edible meat and gut ball

a The results for the fillet composite samples and the remainder composite samples were weighted based on the fraction of the whole-body mass represented by each sample in order to calculate whole-body results (Windward 2006b) (see Table B.2-3 and Attachment 1 for more details).

b Both shiner surfperch and striped perch are present in the EW. Consistent with the LDW, whole-body (shiner surfperch) and fillet data (striped perch) were treated together for the calculation of one EPC. Seafood consumption surveys indicate people eat both whole-body and filleted pelagic fish. However, because there were no fillet and whole-body data available for the same species (allowing for apportionment of fillet and whole-body consumption), these data were treated together as a single pelagic fish category.

c Data from hepatopancreas composite samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Whole-body (i.e., edible meat plus hepatopancreas) crab concentrations were calculated using the relative weights and concentrations of the edible meat and hepatopancreas.

d EPCs based on all clams collected from intertidal areas (regardless of species) were used for clam exposure estimates.

e Data from gut ball composite samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus gut ball. Whole-body (i.e., edible meat plus gut ball) geoduck concentrations were calculated using the relative weights and concentrations of the edible meat and gut ball.

API – Asian and Pacific Islander

EPC – exposure point concentration

EPCs were determined for each seafood category using the datasets described in Section B.2.1. EPCs for the entire EW were calculated for each seafood category in Table B.3-41 as is appropriate for the assumptions of the exposure scenarios and the size of the EW (i.e., exposure for subareas of the EW was not evaluated). For COPCs identified based on detected concentrations in tissue for all of the seafood consumption exposure scenarios (Table B.3-2), summary statistics, the distribution type, and the 95% UCL for chemical concentrations in tissue are presented in Table B.3-42 (EPCs for non-detected COPCs are presented in Section B.6.3.2). The tissue EPCs are summarized in Table B.3-43.

Table B.3-42. EPCs and summary statistics for detected COPCs in tissue

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum Detection (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used[a]	EPC (mg/kg ww)
Antimony						
Benthic fish, fillet	0/11	0.002	nd	0.004 U	one-half maximum RL	0.002
Benthic fish, whole body	4/11	0.004	0.005	0.008 U	maximum detect	0.005
Clams	8/10	0.02	0.06	0.008 UJ	95% KM (BCA) UCL	0.027
Crab, edible meat	1/9	0.002	0.004	0.008 U	maximum detect	0.004
Crab, whole body[b]	8/9	0.006	0.009	0.004 U	95% KM (BCA) UCL	0.0066
Geoduck clam, edible meat	0/6	0.004	nd	0.008 U	one-half maximum RL	0.004
Geoduck clam, whole body[b]	4/4	0.008	0.009	na	maximum detect	0.009
Mussels	9/17	0.007	0.010 J	0.02 U	95% KM (% Bootstrap) UCL	0.0065
Pelagic fish, perch	0/8	0.004	nd	0.008 U	one-half maximum RL	0.004
Pelagic fish, rockfish	0/13	0.002	nd	0.004 U	one-half maximum RL	0.002
Arsenic (inorganic)						
Benthic fish, fillet	0/11	0.005	nd	0.009 UJ	one-half maximum RL	0.0045
Benthic fish, whole body	11/11	0.032	0.059 J	na	95% Modified-t UCL	0.038
Clams	12/12	0.17	0.44	na	95% approximate gamma UCL	0.22
Crab, edible meat	9/9	0.032	0.043 J	na	95% Student's-t UCL	0.036
Crab, whole body[b]	9/9	0.042	0.057 J	na	95% Student's-t UCL	0.047
Geoduck clam, edible meat	6/6	0.029	0.063 J	na	95% Student's-t UCL	0.044
Geoduck clam, whole body[b]	4/4	0.036	0.049 J	na	maximum detect	0.049
Mussels	11/11	0.078	0.13 J	na	95% Student's-t UCL	0.096
Pelagic fish, perch	8/8	0.021	0.037 J	na	95% Student's-t UCL	0.027
Pelagic fish, rockfish	13/13	0.008	0.023 J	na	95% approximate gamma UCL	0.011
Cadmium						
Benthic fish, fillet	1/11	0.03	0.11	0.04 U	maximum detect	0.11
Benthic fish, whole body	1/11	0.02	0.04	0.04 U	maximum detect	0.04
Clams	10/10	0.08	0.11	na	95% Student's-t UCL	0.096

Table B.3-42. EPCs and summary statistics for detected COPCs in tissue (cont.)

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum Detection (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used[a]	EPC (mg/kg ww)
Crab, edible meat	9/9	0.7	0.98	na	95% Student's-t UCL	0.88
Crab, whole body [b]	9/9	2	3.1	na	95% Chebyshev (mean, Sd) UCL	3.6
Geoduck clam, edible meat	6/6	0.2	0.38	na	95% Student's-t UCL	0.30
Geoduck clam, whole body [b]	4/4	0.2	0.27	na	maximum detect	0.27
Mussels	17/17	0.45	0.66	na	95% Student's-t UCL	0.51
Pelagic fish, perch	0/8	0.03	nd	0.08 U	one-half maximum RL	0.04
Pelagic fish, rockfish	0/13	0.04	nd	0.08 U	one-half maximum RL	0.04
Chromium						
Benthic fish, fillet	5/11	0.07	0.1	0.1 U	maximum detect	0.1
Benthic fish, whole body	11/11	0.3	0.4	na	95% Student's-t UCL	0.36
Clams	10/10	0.6	1	na	95% Student's-t UCL	0.69
Crab, edible meat	7/9	0.09	0.1	0.1 UJ	maximum detect[c]	0.1
Crab, whole body [b]	8/9	0.09	0.1	0.05 UJ	maximum detect[c]	0.1
Geoduck clam, edible meat	3/6	0.2	0.5	0.1 U	maximum detect	0.5
Geoduck clam, whole body[b]	4/4	0.2	0.2	na	maximum detect	0.2
Mussels	17/17	0.2	0.93	na	95% Chebyshev (mean, Sd) UCL	0.41
Pelagic fish, perch	6/8	0.2	0.4	0.2 UJ	maximum detect	0.4
Pelagic fish, rockfish	13/13	0.4	0.6	na	95% Student's-t UCL	0.46
Cobalt						
Benthic fish, fillet	0/11	0.03	nd	0.06 U	one-half maximum RL	0.03
Benthic fish, whole body	0/11	0.03	nd	0.06 U	one-half maximum RL	0.03
Clams	10/10	0.17	0.36 J	na	95% modified-t UCL	0.21
Crab, edible meat	9/9	0.1	0.17	na	95% Student's-t UCL	0.13
Crab, whole body [b]	9/9	0.2	0.23	na	95% Student's-t UCL	0.21
Geoduck clam, edible meat	1/6	0.04	0.08	0.06 U	maximum detect	0.08
Geoduck clam, whole body [b]	4/4	0.07	0.08	na	maximum detect	0.08
Mussels	12/14	0.06	0.08	0.06 U	95% KM (BCA) UCL	0.072
Pelagic fish, perch	0/8	0.04	nd	0.1 U	one-half maximum RL	0.05

Table B.3-42. EPCs and summary statistics for detected COPCs in tissue (cont.)

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum Detection (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used[a]	EPC (mg/kg ww)
Pelagic fish, rockfish	0/13	0.05	nd	0.1 U	one-half maximum RL	0.05
Copper						
Benthic fish, fillet	11/11	0.48	1.5	na	95% modified-t UCL	0.68
Benthic fish, whole body	11/11	1.4	2.1	na	95% Student's-t UCL	1.7
Clams	10/10	5.8	9.7 J	na	95% Student's-t UCL	7.5
Crab, edible meat	9/9	15	16	na	95% approximate gamma UCL	16
Crab, whole body [b]	9/9	28	31	na	95% Student's-t UCL	29
Geoduck clam, edible meat	6/6	7.8	15	na	95% Student's-t UCL	11
Geoduck clam, whole body [b]	4/4	9.8	12	na	maximum detect	12
Mussels	17/17	1.8	2.6	na	95% Student's-t UCL	1.9
Pelagic fish, perch	8/8	1.7	3.2	na	95% approximate gamma UCL	2.3
Pelagic fish, rockfish	13/13	0.92	2.4	na	95% approximate gamma UCL	1.2
Lead						
Benthic fish, fillet	0/11	0.2	nd	0.4 U	one-half maximum RL	0.2
Benthic fish, whole body	0/11	0.2	nd	0.4 U	one-half maximum RL	0.2
Clams	7/10	0.6	1.2 J	0.4 UJ	95% KM (percentile bootstrap) UCL	0.79
Crab, edible meat	0/9	0.2	nd	0.4 U	one-half maximum RL	0.2
Crab, whole body [b]	0/9	0.1	nd	0.2 U	one-half maximum RL	0.1
Geoduck clam, edible meat	1/6	0.3	0.5	0.4 U	maximum detect	0.5
Geoduck clam, whole body[b]	4/4	0.4	0.4	na	maximum detect	0.4
Mussels	6/17	0.3	0.83	0.4 U	95% KM (t) UCL	0.54
Pelagic fish, perch	0/8	0.3	nd	0.8 U	one-half maximum RL	0.4
Pelagic fish, rockfish	0/13	0.4	nd	0.8 U	one-half maximum RL	0.4
Mercury						
Benthic fish, fillet	20/20	0.04	0.07	na	95% H-UCL	0.046
Benthic fish, whole body	13/13	0.03	0.042	na	95% Student's-t UCL	0.036
Clams	10/10	0.02	0.03	na	95% Student's-t UCL	0.021
Crab, edible meat	12/12	0.07	0.15	na	95% approximate gamma UCL	0.089

Table B.3-42. EPCs and summary statistics for detected COPCs in tissue (cont.)

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum Detection (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used[a]	EPC (mg/kg ww)
Crab, whole body [b]	9/9	0.05	0.12	na	95% approximate gamma UCL	0.069
Geoduck clam, edible meat	4/6	0.009	0.011	0.01 U	maximum detect	0.011
Geoduck clam, whole body [b]	4/4	0.01	0.01	na	maximum detect	0.010
Mussels	7/17	0.008	0.015	0.01 U	95% KM (percentile bootstrap) UCL	0.011
Pelagic fish, perch	13/17	0.03	0.07 J	0.02 UJ	95% KM (percentile bootstrap) UCL	0.043
Pelagic fish, rockfish	15/15	0.2	0.42	na	95% approximate gamma UCL	0.21
Molybdenum						
Benthic fish, fillet	11/11	0.2	0.3	na	95% modified-t UCL	0.24
Benthic fish, whole body	7/11	0.1	0.3	0.1 U	maximum detect[c]	0.3
Clams	10/10	0.5	0.6 J	na	95% Student's-t UCL	0.52
Crab, edible meat	9/9	0.4	0.4	na	95% Student's-t UCL	0.41
Crab, whole body [b]	9/9	0.4	0.4	na	95% Student's-t UCL	0.41
Geoduck clam, edible meat	6/6	1	1.4	na	95% Student's-t UCL	1.2
Geoduck clam, whole body [b]	4/4	1.3	1.5	na	maximum detect	1.5
Mussels	11/11	0.4	0.4	na	95% Student's-t UCL	0.39
Pelagic fish, perch	5/8	0.2	0.4	0.1 U	maximum detect	0.40
Pelagic fish, rockfish	6/13	0.2	0.4	0.2 U	95% KM (t) UCL	0.28
Selenium						
Benthic fish, fillet	11/11	0.55	0.67	na	95% Student's-t UCL	0.58
Benthic fish, whole body	11/11	0.6	0.68	na	95% Student's-t UCL	0.62
Clams	10/10	0.37	0.52	na	95% Student's-t UCL	0.41
Crab, edible meat	9/9	1	1.2	na	95% Student's-t UCL	1.1
Crab, whole body [b]	9/9	1	1.4	na	95% modified-t UCL	1.2
Geoduck clam, edible meat	6/6	0.5	0.6	na	95% Student's-t UCL	0.56
Geoduck clam, whole body [b]	4/4	0.59	0.62	na	maximum detect	0.62
Mussels	11/11	0.49	0.6	na	95% Student's-t UCL	0.54
Pelagic fish, perch	8/8	0.4	0.6 J	na	95% modified-t UCL	0.51
Pelagic fish, rockfish	13/13	0.66	0.85	na	95% Student's-t UCL	0.72

Table B.3-42. EPCs and summary statistics for detected COPCs in tissue (cont.)

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum Detection (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used[a]	EPC (mg/kg ww)
Vanadium						
Benthic fish, fillet	0/11	0.03	nd	0.06 U	one-half maximum RL	0.030
Benthic fish, whole body	11/11	0.35	0.49	na	95% Student's-t UCL	0.39
Clams	10/10	0.45	0.82 J	na	95% approximate gamma UCL	0.58
Crab, edible meat	3/9	0.04	0.08	0.06 U	maximum detect	0.08
Crab, whole body [b]	9/9	0.1	0.3	na	95% modified-t UCL	0.18
Geoduck clam, edible meat	4/6	0.1	0.34	0.06 U	maximum detect	0.34
Geoduck clam, whole body [b]	4/4	0.2	0.2	na	maximum detect	0.2
Mussels	11/11	0.32	0.42	na	95% Student's-t UCL	0.37
Pelagic fish, perch	4/8	0.1	0.28	0.1 U	maximum detect	0.28
Pelagic fish, rockfish	0/13	0.05	nd	0.1 U	one-half maximum RL	0.05
Zinc						
Benthic fish, fillet	11/11	9.7	13	na	95% approximate gamma UCL	10
Benthic fish, whole body	11/11	15	16	na	95% Student's-t UCL	15
Clams	10/10	18	21 J	na	95% Student's-t UCL	19
Crab, edible meat	9/9	52	60	na	95% Student's-t UCL	57
Crab, whole body [b]	9/9	49	59	na	95% Student's-t UCL	53
Geoduck clam, edible meat	6/6	10	15	na	95% Student's-t UCL	13
Geoduck clam, whole body [b]	4/4	13	16.1	na	maximum detect	16
Mussels	17/17	25.4	49.1	na	95% approximate gamma UCL	32
Pelagic fish, perch	8/8	24.4	26.8	na	95% Student's-t UCL	26
Pelagic fish, rockfish	13/13	16.8	21	na	95% Student's-t UCL	18
Dibutyltin						
Benthic fish, fillet	0/14	0.0047	nd	0.012 U	one-half maximum RL	0.006
Benthic fish, whole body	0/11	0.0053	nd	0.011 U	one-half maximum RL	0.0055
Clams	0/10	0.0053	nd	0.011 U	one-half maximum RL	0.0055
Crab, edible meat	0/9	0.0059	nd	0.012 U	one-half maximum RL	0.006
Crab, whole body [b]	1/9	0.0037	0.01	0.006 U	maximum detect	0.01

Table B.3-42. EPCs and summary statistics for detected COPCs in tissue (cont.)

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum Detection (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used[a]	EPC (mg/kg ww)
Geoduck clam, edible meat	0/6	0.0055	nd	0.011 U	one-half maximum RL	0.0055
Geoduck clam, whole body [b]	1/4	0.0046	0.01	0.0055 U	maximum detect	0.01
Mussels	3/14	0.0088	0.025	0.012 U	maximum detect	0.025
Pelagic fish, perch	0/8	0.0052	nd	0.011 U	one-half maximum RL	0.0055
Pelagic fish, rockfish	4/13	0.011	0.024	0.051 U	one-half maximum RL	0.026
Tributyltin						
Benthic fish, fillet	7/14	0.0057	0.014	0.0077 U	95% KM (percentile bootstrap) UCL	0.0096
Benthic fish, whole body	11/11	0.026	0.038	na	95% Student's-t UCL	0.030
Clams	10/10	0.047	0.14	na	95% approximate gamma UCL	0.072
Crab, edible meat	0/12	0.003	nd	0.0077 U	one-half maximum RL	0.0039
Crab, whole body [b]	1/9	0.0032	0.013	0.0039 U	maximum detect	0.013
Geoduck clam, edible meat	6/6	0.0076	0.0098	na	95% Student's-t UCL	0.0089
Geoduck clam, whole body [b]	4/4	0.0099	0.012	na	maximum detect	0.012
Mussels	16/17	0.033	0.0928	0.0077 U	97.5% KM (Chebyshev) UCL	0.082
Pelagic fish, perch	14/14	0.04	0.067	na	95% Student's-t UCL	0.052
Pelagic fish, rockfish	13/13	0.16	0.42	na	95% approximate gamma UCL	0.22
cPAH TEQ						
Benthic fish, fillet	3/11	0.00029	0.00042 J	0.00067 U	maximum detect	0.00042
Benthic fish, whole body	9/11	0.011	0.011	0.180 U	95% KM (Chebyshev) UCL	0.0068
Clams	11/11	0.016	0.063	na	95% approximate gamma UCL	0.027
Crab, edible meat	6/9	0.0006	0.0024 J	0.00045 U	95% KM (BCA) UCL	0.0011
Crab, whole body [b]	7/7	0.00096	0.0012 J	na	95% Student's-t UCL	0.0011
Geoduck clam, edible meat	6/6	0.0016	0.0028 J	na	95% Student's-t UCL	0.0022
Geoduck clam, whole body [b]	4/4	0.0031	0.0041 J	na	maximum detect	0.0041
Mussels	16/17	0.02	0.11	0.029 U	97.5% KM (Chebyshev) UCL	0.059
Pelagic fish, perch	6/8	0.0012	0.0022	0.0033 U	95% KM (t) UCL	0.0016
Pelagic fish, rockfish	0/12	0.00024	na	0.00058 U	one-half maximum RL	0.00029

Table B.3-42. EPCs and summary statistics for detected COPCs in tissue (cont.)

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum Detection (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used[a]	EPC (mg/kg ww)
1,4-Dichlorobenzene						
Benthic fish, fillet	0/11	0.17	nd	0.33 U	one-half maximum RL	0.17
Benthic fish, whole body	1/11	0.53	4.8	0.20 U	maximum detect	4.8
Clams	0/10	0.15	nd	0.30 U	one-half maximum RL	0.15
Crab, edible meat	0/9	0.11	nd	0.33 U	one-half maximum RL	0.17
Crab, whole body[b]	0/9	0.067	nd	0.17 U	one-half maximum RL	0.085
Geoduck clam, edible meat	0/6	0.034	nd	0.11 U	one-half maximum RL	0.055
Geoduck clam, whole body[b]	0/4	0.026	nd	0.062 U	one-half maximum RL	0.031
Mussels	0/17	0.068	nd	0.20 U	one-half maximum RL	0.1
Pelagic fish, perch	0/8	0.65	nd	1.3 U	one-half maximum RL	0.65
Pelagic fish, rockfish	0/13	0.17	nd	0.33 U	one-half maximum RL	0.17
Pentachlorophenol						
Benthic fish, fillet	0/3	0.002	nd	0.0040 U	one-half maximum RL	0.002
Benthic fish, whole body	0/3	0.0021	nd	0.0041 U	one-half maximum RL	0.0021
Clams	2/10	0.0031	0.0082 J	0.005 U	maximum detect	0.0082
Crab, edible meat	0/3	0.002	nd	0.0041 U	one-half maximum RL	0.0021
Crab, whole body[b]	0/3	0.001	nd	0.0021 U	one-half maximum RL	0.0011
Geoduck clam, edible meat	0/6	0.027	nd	0.30 U	one-half maximum RL	0.15
Geoduck clam, whole body[b]	0/4	0.036	nd	0.27 U	one-half maximum RL	0.14
Mussels	0/9	0.0097	nd	0.027 U	one-half maximum RL	0.014
Pelagic fish, perch	0/3	0.0049	nd	0.011 U	one-half maximum RL	0.0055
Pelagic fish, rockfish	0/13	0.0021	nd	0.0041 U	one-half maximum RL	0.0021
Total PCBs						
Benthic fish, fillet	20/20	1.7	5.7	na	95% approximate gamma UCL	2.4
Benthic fish, whole body	13/13	3.2	7.9 J	na	95% approximate gamma UCL	4.1
Clams	11/11	0.056	0.082	na	95% Student's-t UCL	0.069
Crab, edible meat	12/12	0.13	0.21 J	na	95% Student's-t UCL	0.16
Crab, whole body[b]	9/9	0.3	0.86 M	na	95% Modified-t UCL	0.45

Table B.3-42. EPCs and summary statistics for detected COPCs in tissue (cont.)

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum Detection (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used[a]	EPC (mg/kg ww)
Geoduck clam, edible meat	6/6	0.019	0.024 JN	na	95% Student's-t UCL	0.022
Geoduck clam, whole body[b]	4/4	0.028	0.034 JN	na	maximum detect	0.034
Mussels	14/17	0.026	0.044 J	0.013 U	95% KM (percentile bootstrap) UCL	0.031
Pelagic fish, perch	17/17	1	5.4	na	95% approximate gamma UCL	1.6
Pelagic fish, rockfish	15/15	2	6.2	na	95% H-UCL	4.0
PCB TEQ						
Benthic fish, fillet	3/3	1.3×10^{-5}	1.5×10^{-5}	na	maximum detect	1.5×10^{-5}
Benthic fish, whole body	3/3	3.5×10^{-5}	3.7×10^{-5} J	na	maximum detect	0.0000374
Clams	3/3	4.1×10^{-7}	7.3×10^{-7}	na	maximum detect	7.3×10^{-7}
Crab, edible meat	3/3	1.6×10^{-6}	1.7×10^{-6}	na	maximum detect	1.7×10^{-6}
Crab, whole body[b]	3/3	4.8×10^{-6}	5.6×10^{-6} M	na	maximum detect	5.6×10^{-6}
Geoduck clam, edible meat	3/3	1.4×10^{-7}	1.9×10^{-7}	na	maximum detect	1.9×10^{-7}
Geoduck clam, whole body[b]	1/1	2.3×10^{-7}	2.3×10^{-7}	na	maximum detect	2.3×10^{-7}
Mussels[d]	no data	no data	no data	no data	na	na
Pelagic fish, perch	3/3	1.3×10^{-5}	1.4×10^{-5}	na	maximum detect	1.4×10^{-5}
Pelagic fish, rockfish	6/6	2.5×10^{-5}	6.0×10^{-5} J	na	95% Student's-t UCL	4.0×10^{-5}
alpha-BHC						
Benthic fish, fillet	0/1	0.00044	nd	0.00088 U	one-half maximum RL	0.00044
Benthic fish, whole body	0/1	0.00046	nd	0.00092 U	one-half maximum RL	0.00046
Clams	0/6	0.00043	nd	0.00087 U	one-half maximum RL	0.00044
Crab, edible meat	0/1	0.00042	nd	0.00083 U	one-half maximum RL	0.00042
Crab, whole body[b]	0/1	0.00022	nd	0.00043 U	one-half maximum RL	0.00022
Geoduck clam, edible meat	1/1	0.0012	0.0012	na	maximum detect	0.0012
Geoduck clam, whole body[b]	1/1	0.0011	0.0011	na	maximum detect	0.0011
Mussels	0/1	0.00047	nd	0.00094 U	one-half maximum RL	0.00047
Pelagic fish, perch	0/1	0.00043	nd	0.00086 U	one-half maximum RL	0.00043
Pelagic fish, rockfish	2/9	0.00045	0.00058 J	0.00094 U	maximum detect	0.00058

Table B.3-42. EPCs and summary statistics for detected COPCs in tissue (cont.)

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum Detection (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used[a]	EPC (mg/kg ww)
beta-BHC						
Benthic fish, fillet	0/1	0.00044	nd	0.00088 U	one-half maximum RL	0.00044
Benthic fish, whole body	0/1	0.00046	nd	0.00092 U	one-half maximum RL	0.00046
Clams	0/6	0.00043	nd	0.00087 U	one-half maximum RL	0.00044
Crab, edible meat	0/1	0.00042	nd	0.00083 U	one-half maximum RL	0.00042
Crab, whole body [b]	0/1	0.00022	nd	0.00043 U	one-half maximum RL	0.00022
Geoduck clam, edible meat	1/1	0.0015	0.0015 J	na	maximum detect	0.0015
Geoduck clam, whole body [b]	1/1	0.0014	0.0014 J	na	maximum detect	0.0014
Mussels	0/1	0.00047	nd	0.00094 U	one-half maximum RL	0.00047
Pelagic fish, perch	0/1	0.00043	nd	0.00086 U	one-half maximum RL	0.00043
Pelagic fish, rockfish	0/9	0.00045	nd	0.00094 U	one-half maximum RL	0.00047
Total chlordane						
Benthic fish, fillet	1/1	0.0026	0.0026 J	na	maximum detect	0.0026
Benthic fish, whole body	1/1	0.0032	3.2	na	maximum detect	0.0032
Clams	2/6	0.0022	0.00068 J	0.0098 U	maximum detect	0.0049
Crab, edible meat	0/1	0.0011	nd	0.0021 U	one-half maximum RL	0.0011
Crab, whole body [b]	1/1	0.018	0.018 J	na	maximum detect	0.018
Geoduck clam, edible meat	0/1	0.0011	nd	0.0022 U	one-half maximum RL	0.0011
Geoduck clam, whole body [b]	0/1	0.00055	nd	0.0011 U	one-half maximum RL	0.00055
Mussels	1/1	0.00011	0.00011 J	na	maximum detect	0.00011
Pelagic fish, perch	1/1	0.003	0.003 J	na	maximum detect	0.003
Pelagic fish, rockfish	9/9	0.0062	0.000014	na	95% Student's-t UCL	0.0083
Total DDTs						
Benthic fish, fillet	1/1	0.013	0.013 J	na	maximum detect	0.013
Benthic fish, whole body	1/1	0.016	0.016 J	na	maximum detect	0.016
Clams	3/6	0.00075	0.00055 J	0.0022 U	one-half maximum RL	0.0011
Crab, edible meat	0/1	0.0011	nd	0.0021 U	one-half maximum RL	0.0011
Crab, whole body [b]	1/1	0.0046	0.0046 J	na	maximum detect	0.0046

Table B.3-42. EPCs and summary statistics for detected COPCs in tissue (cont.)

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum Detection (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used[a]	EPC (mg/kg ww)
Geoduck clam, edible meat	0/1	0.0011	nd	0.0022 U	one-half maximum RL	0.0011
Geoduck clam, whole body [b]	1/1	0.0008	0.0008 J	na	maximum detect	0.0008
Mussels	1/1	0.00017	0.00017 J	na	maximum detect	0.00017
Pelagic fish, perch	1/1	0.011	0.011 J	na	maximum detect	0.011
Pelagic fish, rockfish	9/9	0.023	0.054 J	na	95% Student's-t UCL	0.032
Dieldrin						
Benthic fish, fillet	1/1	0.00028	0.00028 J	na	maximum detect	0.00028
Benthic fish, whole body	1/1	0.00039	0.00039 J	na	maximum detect	0.00039
Clams	0/6	0.00043	nd	0.00087 U	one-half maximum RL	0.00044
Crab, edible meat	0/1	0.00042	nd	0.00083 U	one-half maximum RL	0.00042
Crab, whole body [b]	0/1	0.00022	nd	0.00043 U	one-half maximum RL	0.00022
Geoduck clam, edible meat	0/1	0.00044	nd	0.00088 U	one-half maximum RL	0.00044
Geoduck clam, whole body [b]	0/1	0.00022	nd	0.00044 U	one-half maximum RL	0.00022
Mussels	0/1	0.00047	nd	0.00094 U	one-half maximum RL	0.00047
Pelagic fish, perch	1/1	0.00076	0.00076 J	na	maximum detect	0.00076
Pelagic fish, rockfish	8/9	0.00045	0.00066 J	0.00094 U	95% KM (t) UCL	0.00054
Heptachlor						
Benthic fish, fillet	0/1	0.00022	nd	0.00044 U	one-half maximum RL	0.00022
Benthic fish, whole body	0/1	0.00023	nd	0.00046 U	one-half maximum RL	0.00023
Clams	0/6	0.00022	nd	0.00044 U	one-half maximum RL	0.00022
Crab, edible meat	0/1	0.00021	nd	0.00042 U	one-half maximum RL	0.00021
Crab, whole body [b]	0/1	0.00011	nd	0.00022 U	one-half maximum RL	0.00011
Geoduck clam, edible meat	0/1	0.00022	nd	0.00044 U	one-half maximum RL	0.00022
Geoduck clam, whole body [b]	0/1	0.00011	nd	0.00022 U	one-half maximum RL	0.00011
Mussels	1/1	0.0001	0.10 J	na	maximum detect	0.0001
Pelagic fish, perch	0/1	0.00022	nd	0.00043 U	one-half maximum RL	0.00022
Pelagic fish, rockfish	0/9	0.00024	nd	0.00047 U	one-half maximum RL	0.00024
Heptachlor epoxide						

Table B.3-42. EPCs and summary statistics for detected COPCs in tissue (cont.)

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum Detection (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used[a]	EPC (mg/kg ww)
Benthic fish, fillet	0/1	0.00022	nd	0.00044 U	one-half maximum RL	0.00022
Benthic fish, whole body	0/1	0.00023	nd	0.00046 U	one-half maximum RL	0.00023
Clams	0/6	0.00022	nd	0.00044 U	one-half maximum RL	0.00022
Crab, edible meat	0/1	0.00021	nd	0.00042 U	one-half maximum RL	0.00021
Crab, whole body [b]	1/1	0.00031	0.00031 J	na	maximum detect	0.00031
Geoduck clam, edible meat	0/1	0.00022	nd	0.00044 U	one-half maximum RL	0.00022
Geoduck clam, whole body [b]	0/1	0.00011	nd	0.00022 U	one-half maximum RL	0.00011
Mussels	0/1	0.00024	nd	0.00047 U	one-half maximum RL	0.00024
Pelagic fish, perch	0/1	0.00022	nd	0.00043 U	one-half maximum RL	0.00022
Pelagic fish, rockfish	1/9	0.00022	0.00014 J	0.00047 U	one-half maximum RL	0.00024
Mirex						
Benthic fish, fillet	0/1	0.00022	nd	0.00044 U	one-half maximum RL	0.00022
Benthic fish, whole body	0/1	0.00023	nd	0.00046 U	one-half maximum RL	0.00023
Clams	0/6	0.00022	nd	0.00044 U	one-half maximum RL	0.00022
Crab, edible meat	0/1	0.00021	nd	0.00042 U	one-half maximum RL	0.00021
Crab, whole body [b]	0/1	0.00011	nd	0.00022 U	one-half maximum RL	0.00011
Geoduck clam, edible meat	0/1	0.00022	nd	0.00044 U	one-half maximum RL	0.00022
Geoduck clam, whole body [b]	0/1	0.00011	nd	0.00022 U	one-half maximum RL	0.00011
Mussels	1/1	0.00011	0.00011 J	na	maximum detect	0.00011
Pelagic fish, perch	0/1	0.00022	nd	0.00043 U	one-half maximum RL	0.00022
Pelagic fish, rockfish	9/9	0.0003	0.00076	na	95% approximate gamma UCL	0.00044
Dioxin/Furan TEQ						
Benthic fish, fillet	3/3	7.5×10^{-7}	7.9×10^{-7} J	na	maximum detect	7.9×10^{-7}
Benthic fish, whole body	3/3	1.8×10^{-6}	1.9×10^{-6} J	na	maximum detect	1.9×10^{-6}
Clams	3/3	2.8×10^{-7}	3.8×10^{-7} J	na	maximum detect	3.8×10^{-7}
Crab, edible meat	3/3	4.7×10^{-7}	4.9×10^{-7} J	na	maximum detect	4.9×10^{-7}
Crab, whole body [b]	3/3	1.2×10^{-6}	1.3×10^{-6} J	na	maximum detect	1.3×10^{-6}
Geoduck clam, edible meat	3/3	2.3×10^{-7}	2.5×10^{-7} J	na	maximum detect	2.5×10^{-7}

Table B.3-42. EPCs and summary statistics for detected COPCs in tissue (cont.)

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum Detection (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used[a]	EPC (mg/kg ww)
Geoduck clam, whole body[b]	1/1	2.0×10^{-7}	2.0×10^{-7} J	na	maximum detect	2.0×10^{-7}
Mussels[d]	no data	no data	no data	no data	na	na
Pelagic fish, perch	3/3	1.2×10^{-6}	1.4×10^{-6} J	na	maximum detect	1.4×10^{-6}
Pelagic fish, rockfish	6/6	2.1×10^{-6}	3.0×10^{-6} J	na	95% Student's-t UCL	2.8×10^{-6}

[a] It should be noted that the although a 95% confidence was specified for ProUCL runs (i.e., a 95% UCL), in some cases the recommended UCL was a 97.5% or 99% UCL when the skewness of the dataset or presence of non-detected results required a higher percent UCL to ensure the specified 95% coverage.

[b] Whole-body crab and geoduck samples were calculated using relative weights and concentrations of the edible meat and hepatopancreas for crabs or edible meat and gut ball for geoduck. Details regarding this approach are provided in Table B.2-3 and in Attachment 1.

[c] For three consumption category-COPC combinations, the dataset used to calculate a 95% UCL had an insufficient number of distinct values (e.g., for chromium, all eight of the detected whole-body crab samples were equal to 0.1 mg/kg ww). In these three cases, the maximum detection was used as the EPC.

[d] No mussel data were available for this chemical. When the CDI and risk values were calculated, seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

BCA – bias-corrected accelerated bootstrap method
BHC – benzene hexachloride
CDI – chronic daily intake
COPC – chemical of potential concern
cPAH – carcinogenic polycyclic aromatic hydrocarbon
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
J – estimated concentration
KM – Kaplan-Meier
N – tentative identification
na – not applicable
nd – not detected
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RL – reporting limit
Sd – standard deviation
SVOC – semivolatile organic compound
TEQ – toxic equivalent
U – not detected at given concentration
UCL – upper confidence limit on the mean
ww – wet weight

Table B.3-43. Summary of EPCs for detected COPCs in tissue by seafood consumption category

COPC	EPC (mg/kg ww)									
	Benthic Fish, Fillet	Benthic Fish, Whole Body	Clams	Crab, Edible Meat	Crab, Whole Body[a]	Geoduck, Edible Meat	Geoduck, Whole Body[a]	Mussels	Pelagic Fish, Perch	Pelagic Fish, Rockfish
Metals										
Antimony	0.002	0.005	0.027	0.004	0.0066	0.004	0.009	0.0065	0.004	0.002
Arsenic (inorganic)	0.0045	0.038	0.22	0.036	0.047	0.044	0.049	0.096	0.027	0.011
Cadmium	0.11	0.04	0.096	0.88	3.6	0.30	0.27	0.51	0.04	0.04
Chromium	0.1	0.36	0.69	0.1	0.1	0.5	0.2	0.41	0.4	0.46
Cobalt	0.03	0.03	0.21	0.13	0.21	0.08	0.08	0.072	0.05	0.05
Copper	0.68	1.7	7.5	16	29	11	112	1.9	2.3	1.2
Lead	0.2	0.2	0.79	0.2	0.1	0.5	0.4	0.54	0.4	0.4
Mercury	0.046	0.036	0.021	0.090	0.069	0.011	0.01	0.011	0.043	0.21
Molybdenum	0.24	0.3	0.52	0.41	0.41	1.2	1.5	0.39	0.4	0.28
Selenium	0.58	0.62	0.41	1.1	1.2	0.56	0.62	0.54	0.51	0.72
Vanadium	0.03	0.39	0.58	0.08	0.18	0.34	0.2	0.37	0.28	0.05
Zinc	10	15	19	57	53	13	16	32	26	18
Organometals										
Dibutyltin	0.006	0.0055	0.0055	0.006	0.01	0.0055	0.01	0.025	0.0055	0.026
Tributyltin	0.0096	0.030	0.072	0.0039	0.013	0.0089	0.012	0.082	0.052	0.22
PAHs										
cPAH TEQ	0.00042	0.0069	0.027	0.0011	0.0011	0.0022	0.0041	0.059	0.0016	0.00029
SVOCs										
1,4-Dichlorobenzene	0.17	4.8	0.15	0.17	0.085	0.055	0.031	0.1	0.65	0.17
Pentachlorophenol	0.002	0.0021	0.0082	0.0021	0.0011	0.15	0.14	0.014	0.0055	0.0021
PCBs										
Total PCBs	2.4	4.1	0.069	0.16	0.45	0.022	0.034	0.031	1.6	4.0
PCB TEQ	1.5×10^{-5}	3.7×10^{-5}	7.3×10^{-7}	1.7×10^{-6}	5.6×10^{-6}	1.9×10^{-7}	2.3×10^{-7}	nd[b]	1.4×10^{-5}	4.0×10^{-5}

Table B.3-43. Summary of EPCs for detected COPCs in tissue by seafood consumption category (cont.)

COPC	EPC (mg/kg ww)									
	Benthic Fish, Fillet	Benthic Fish, Whole Body	Clams	Crab, Edible Meat	Crab, Whole Body[a]	Geoduck, Edible Meat	Geoduck, Whole Body[a]	Mussels	Pelagic Fish, Perch	Pelagic Fish, Rockfish
Pesticides										
alpha-BHC	0.00044	0.00046	0.00044	0.00042	0.00022	0.0012	0.0011	0.00047	0.00043	0.00058
beta-BHC	0.00044	0.00046	0.00044	0.00042	0.00022	0.0015	0.0014	0.00047	0.00043	0.00047
Total chlordane	0.0026	0.0032	0.0049	0.00105	0.018	0.0011	0.00055	0.00011	0.003	0.0083
Total DDTs	0.013	0.016	0.0011	0.00105	0.0046	0.0011	0.0008	0.00017	0.0107	0.032
Dieldrin	0.00028	0.00039	0.00044	0.00042	0.00022	0.00044	0.00022	0.00047	0.00076	0.00054
Heptachlor	0.00022	0.00023	0.00022	0.00021	0.00011	0.00022	0.00011	0.0001	0.00022	0.00024
Heptachlor epoxide	0.00022	0.00023	0.00022	0.00021	0.00031	0.00022	0.00011	0.00024	0.00022	0.00024
Mirex	0.00022	0.00023	0.00022	0.00021	0.00011	0.00022	0.00011	0.00011	0.00022	0.00044
Dioxin/Furan										
Dioxin/furan TEQ	7.9×10^{-7}	1.9×10^{-6}	3.8×10^{-7}	4.9×10^{-7}	1.3×10^{-6}	2.5×10^{-7}	2.0×10^{-7}	nd[b]	1.4×10^{-6}	2.8×10^{-6}

[a] Whole-body crab and geoduck samples were calculated using relative weights and concentrations of the edible meat and hepatopancreas for crabs or edible meat and gut ball for geoduck. Details regarding this approach are provided in Table B.2-3 and in Attachment 1.

[b] No mussel data were available for this chemical. When the CDI and risk values were calculated, seafood consumption that had been assigned to mussels was divided proportionally among the remaining shellfish consumption categories.

BHC – benzene hexachloride
CDI – chronic daily intake
COPC – chemical of potential concern
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
nd – no data
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TEQ – toxic equivalent
ww – wet weight

B.3.3.4.2 Sediment

This section describes the calculation of EPCs for the three sediment exposure areas for which EPCs were needed for the EW HHRA, which include the following:

- Site-wide intertidal exposure area (unrestricted access) for the habitat restoration worker, the tribal clamming RME scenario, and the tribal clamming 183-day-per-year scenario
- Public access intertidal area for the 7-day-per-year clamming scenario
- Site-wide sediment exposure area including subtidal and intertidal areas for the RME and CT netfishing scenarios

The following subsections present the methods for calculating these EPCs and present the final EPCs. Note that for areas where no sediment could be collected (because the substrate in those areas is gravel, cobble, or riprap), exposure to sediment cannot be assessed.

Site-Wide Intertidal Exposure Area

The EPCs for the tribal clamming and habitat restoration scenarios (for all COPCs) were determined using the 95% UCL for the three intertidal MIS samples that represent the intertidal study area as shown on Map B.3-1, each of which consists of 36 individual sediment grab samples randomly collected from the top 25 cm in intertidal areas throughout the EW. As discussed in the Section B.2.1.1, this depth was intended to reflect potential exposure during clamming (based on the burrowing depth of EW clams) and habitat restoration (e.g., reasonable average depth for digging or planting).

The approach used to calculate the MIS 95% UCL was developed in consultation with EPA (Kissinger 2010; Windward 2010h). MIS provides a more efficient way of estimating the mean compared with individual grab samples but does not provide information on the variability in the underlying distribution, or in particular, of extremes in the distribution.

The shape of the distribution cannot be properly evaluated with only three samples, so the central limit theorem is invoked and normality is assumed. Based on this assumption, the 95% UCL was derived with the standard equation for a normally distributed population using Equation 3-9.

$$95\%\ \mathrm{UCL} = \overline{X} + t_{\alpha=0.05(1),df=2}\mathrm{SD}(\overline{X})/\sqrt{n}$$ **Equation 3-9**

Where:

$\overline{X}$	=	mean of the three site-wide intertidal MIS samples
$t(\alpha=0.05,df=2)$	=	critical value for the theoretical distribution of standardized normal means where df is the degrees of freedom and alpha (α) is equal to the confidence
$SD(\overline{X})$	=	standard deviation of the mean of the three site-wide intertidal MIS samples (n = 3 site-wide intertidal samples)

Public Access Intertidal Exposure Area

The EPCs for the clamming 7-day-per-year scenario were derived using the single MIS sample for the intertidal areas that are publically accessible by walking, which was created from 32 individual grab samples (see Map B.2-2 and B.3-1). As with the site-wide intertidal EPC derivation, the approach used to calculate the MIS 95% UCL was developed in consultation with EPA (Kissinger 2010; Windward 2010h). The method for calculating the MIS 95% UCL for the public access intertidal area is similar to that presented above. However, because only a single MIS sample was available for the public access intertidal area, the variance (standard deviation and standard error) from the three site-wide intertidal MIS samples was used to derive the public access intertidal area 95% UCL. Equation 3-10 presents the method used to calculate the public access intertidal 95% UCL.

$$95\%\ \mathrm{UCL} = \overline{X} + t_{\alpha=0.05(1),df=2}\mathrm{SD}(\overline{X}) / \sqrt{n}$$ **Equation 3-10**

Where:

$\overline{X}$	=	value of the single public access MIS sample
$t(\alpha=0.05,df=2)$	=	critical value for the theoretical distribution of standardized normal means where df is the degrees of freedom for the estimate of variability, and alpha (α) is equal to the confidence level
$\mathrm{SD}(\overline{X})$	=	standard deviation of the mean of the three site-wide intertidal MIS samples, the best estimate for the standard deviation of the public-access MIS sample (n = 1 public access intertidal sample)

In Equation 3-10, the standard error of the mean of the single public access MIS sample was estimated using the standard deviation of the three site-wide MIS samples and the samples size of the public access intertidal samples (n = 1) (Kissinger 2010; Windward 2010h).

Site-Wide Sediment Exposure Area

The netfishing exposure scenarios include all relevant intertidal MIS and subtidal surface sediment samples (collected from the top 10 cm) in the EW from the dataset described in Section B.2.1. As described briefly in the Section B.2.1, subtidal and intertidal exposures were calculated separately and then combined to estimate study area–wide concentrations. For all chemicals except dioxin/furan TEQ and PCB TEQ, all appropriate subtidal samples (except the 13 composite grab samples) were used to develop a subtidal EPC (i.e., the 95% UCL of these samples), and the three intertidal-wide MIS samples were used to develop the EPC for the intertidal area. The approach used to calculate the intertidal MIS 95% UCL is the same as that described above. The subtidal and intertidal EPCs were then weighted based on the size of their relative areas to estimate a study area-wide EPC for the netfishing scenarios (2.7% for intertidal and 97.3% for subtidal).

A slightly different approach was used for dioxin/furan TEQ and PCB TEQ netfishing EPCs. For estimating TEQs for the netfishing scenarios, the EPC for the subtidal area was calculated using the 13 grab composite samples, and the EPC for the intertidal area was calculated using the 95% UCL for three intertidal-wide MIS samples, as described above. The EPC for the subtidal composites was the 95% UCL of the 13 composite samples. As was done for other chemicals, these two EPCs were then weighted based on the relative area of the intertidal and subtidal regions to develop study area-wide EPCs for dioxin/furan TEQ and PCB TEQ for the netfishing scenarios.

Sediment EPCs

For COPCs identified based on detected concentrations in sediment for all of the direct contact exposure scenarios (Table B.3-2), summary statistics, the distribution type, and EPC for chemical concentrations in sediment are presented in Table B.3-44 for scenarios with site-wide exposure (netfishing) and in Table B.3-45 for scenarios with only intertidal exposure (clamming and habitat restoration worker). The sediment EPCs are summarized in Table B.3-46.

Table B.3-44. EPCs and summary statistics for detected COPCs in sediment for exposure scenarios using both subtidal and intertidal sediment data

COPC	Netfishing Exposure Area	Detection Frequency	Mean Value (mg/kg dw)	Maximum Detection (mg/kg dw)	Maximum RL (mg/kg dw)	Standard Deviation (mg/kg dw)	Standard Error (mg/kg dw)	Statistic Used	EPC (mg/kg dw)
Antimony	intertidal	0/3	3	nd	6 U	0	0	site-wide intertidal MIS 95% UCL	3
	subtidal	3/181	5	44 J	38 UJ	na	na	maximum detect	44
	combined[a]	3/184	4.9	44 J	38 UJ	na	na	weighted 95% UCL	43
Arsenic	intertidal	3/3	10	13	na	2.8	1.6	site-wide intertidal MIS 95% UCL	15
	subtidal	157/227	10	241	63 U	na	na	95% KM (BCA) UCL	12
	combined[a]	160/230	10	241	63 U	na	na	weighted 95% UCL	12
Lead	intertidal	3/3	50	60	na	6	4	site-wide intertidal MIS 95% UCL	62
	subtidal	224/227	50	520	10 U	na	na	95% KM (Chebyshev) UCL	65
	combined[a]	227/230	50	520	10 U	na	na	weighted 95% UCL	65
Vanadium	intertidal	3/3	41	46 J	na	6.0	3.4	site-wide intertidal MIS 95% UCL	51
	subtidal	101/101	57	94	na	na	na	95% Student's-t UCL	59
	combined[a]	104/104	57	94	na	na	na	weighted 95% UCL	59
cPAHs TEQ	intertidal	3/3	1	1.9	na	0.76	0.44	site-wide intertidal MIS 95% UCL	2.3
	subtidal	229/237	0.5	16 J	0.048 U	na	na	95% KM (Chebyshev) UCL	0.84
	combined[a]	232/240	0.51	16 J	0.048 U	na	na	weighted 95% UCL	0.88
1,4-Dichloro-benzene	intertidal	1/3	0.006	0.012	0.006 U	0.0052	0.003	site-wide intertidal MIS 95% UCL	0.015
	subtidal	147/228	0.12	15	0.050 U	na	na	95% KM (Chebyshev) UCL	0.42
	combined[a]	148/231	0.12	15	0.050 U	na	na	weighted 95% UCL	0.41
PCB TEQ	intertidal	3/3	4.5×10^{-6}	6.3×10^{-6} J	na	1.6×10^{-6}	9.2×10^{-7}	site-wide intertidal MIS 95% UCL	7.2×10^{-6}
	subtidal	13/13	4.4×10^{-6}	9.5×10^{-6}	na	na	na	95% Student's-t UCL	5.6×10^{-6}
	combined[a]	16/16	4.4×10^{-6}	9.5×10^{-6}	na	na	na	weighted 95% UCL	5.6×10^{-6}
Total PCBs	intertidal	3/3	0.97	1.6	na	0.55	0.32	site-wide intertidal MIS 95% UCL	1.9
	subtidal	223/237	0.53	8.4	0.035 U	na	na	95% KM (Chebyshev) UCL	0.76
	combined[a]	226/240	0.54	8.4	0.035 U	na	na	weighted 95% UCL	0.79
Dioxin/furan	intertidal	3/3	1.2×10^{-5}	1.4×10^{-5} J	na	2.5×10^{-6}	1.5×10^{-6}	site-wide intertidal MIS 95% UCL	1.6×10^{-5}

Table B.3-44. EPCs and summary statistics for detected COPCs in sediment for exposure scenarios using both subtidal and intertidal sediment data

COPC	Netfishing Exposure Area	Detection Frequency	Mean Value (mg/kg dw)	Maximum Detection (mg/kg dw)	Maximum RL (mg/kg dw)	Standard Deviation (mg/kg dw)	Standard Error (mg/kg dw)	Statistic Used	EPC (mg/kg dw)
TEQ	subtidal	13/13	1.6×10^{-5}	3.1×10^{-5}	na	na	na	95% Student's-t UCL	1.9×10^{-5}
	combined[a]	16/16	1.6×10^{-5}	3.1×10^{-5}	na	na	na	weighted 95% UCL	1.9×10^{-5}

Note: No undetected chemicals were identified as COPCs for the netfishing exposure scenario.

[a] The combined netfishing mean value and EPC is the weighted average of the intertidal mean or EPC (2.7% of the exposure area) and subtidal mean or EPC (97.3% of the exposure area).

COPC – chemical of potential concern

dw – dry weight

EPC – exposure point concentration

MIS – multi-increment sampling

PAH – polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

U – not detected at given concentration

UCL – upper confidence limit

Table B.3-45. EPCs and summary statistics for detected COPCs in sediment for exposure scenarios using only intertidal sediment data

COPC	Intertidal Exposure Area	Detection Frequency	Concentration (mg/kg dw) Mean Value	Maximum Detection	Maximum RL	Standard Deviation	Standard Error	Statistic Used	EPC (mg/kg dw)
Arsenic	public access intertidal[a]	1/1	7.7	7.7	na	2.8	na	public access intertidal MIS 95% UCL	16
	site-wide intertidal[b]	3/3	10	13	na	2.8	1.6	site-wide intertidal MIS 95% UCL	15
Cobalt	public access intertidal[a]	1/1	5.0	5.0	na	0.55	na	public access intertidal MIS 95% UCL	6.6
	site-wide intertidal[b]	3/3	6.2	6.8	na	0.55	0.32	site-wide intertidal MIS 95% UCL	7.1
Lead	site-wide intertidal[b]	3/3	50	60	na	6	4	site-wide intertidal MIS 95% UCL	62
Vanadium	public access intertidal[a]	1/1	31 J	31 J	na	6.0	na	public access intertidal MIS 95% UCL	48
	site-wide intertidal[b]	3/3	41	46 J	na	6.0	3.4	site-wide intertidal MIS 95% UCL	51
cPAHs TEQ	public access intertidal[a]	1/1	0.39	0.39	na	0.76	na	public access intertidal MIS 95% UCL	2.6
	site-wide intertidal[b]	3/3	1	1.9	na	0.76	0.44	site-wide intertidal MIS 95% UCL	2.3
PCB TEQ	public access intertidal[a]	1/1	1.4×10^{-6}	1.4×10^{-6} J	na	1.6×10^{-6}	na	public access intertidal MIS 95% UCL	6.1×10^{-6}
	site-wide intertidal[b]	3/3	4.5×10^{-6}	6.3×10^{-6} J	na	1.6×10^{-6}	9.2×10^{-7}	site-wide intertidal MIS 95% UCL	7.2×10^{-6}
Total PCBs	public access intertidal[a]	1/1	0.37	0.37	na	0.55	na	public access intertidal MIS 95% UCL	2.0
	site-wide intertidal[b]	3/3	0.97	1.6	na	0.55	0.32	site-wide intertidal MIS 95% UCL	1.9
Dioxin/ furan TEQ	public access intertidal[a]	1/1	8.5×10^{-6}	8.5×10^{-6} J	na	2.5×10^{-6}	na	public access intertidal MIS 95% UCL	1.6×10^{-5}
	site-wide intertidal[b]	3/3	1.2×10^{-5}	1.4×10^{-5} J	na	2.5×10^{-6}	1.5×10^{-6}	site-wide intertidal MIS 95% UCL	1.6×10^{-5}

[a] The public access intertidal area was the exposure area used to evaluate the 7-days-per-year clamming scenario.

[b] The site-wide intertidal area was the exposure area used to evaluate the tribal clamming scenarios and the habitat restoration worker scenario.

COPC – chemical of potential concern

dw – dry weight

EPC – exposure point concentration

J – estimated concentration

MIS – multi-increment sampling

na – not applicable

PAH – polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

UCL – upper confidence limit

Table B.3-46. Summary of EPCs for detected COPCs in sediment by exposure scenario

COPC	EPC (mg/kg dw)				
	Netfishing RME[a]	Netfishing CT[a]	Habitat Restoration Worker	Tribal Clamming	Clamming – 7 Days per Year (Public Access Only)
Metals					
Antimony	43	4.9	not a COPC	not a COPC[b]	not a COPC[b]
Arsenic	12	10	15	15	16
Cobalt	not a COPC	not a COPC	not a COPC	7.1	6.6
Lead	65	50	not a COPC	62	not a COPC
Vanadium	59	57	51	51	48
PAHs					
cPAH TEQ	0.88	0.51	2.3	2.3	2.6
Other SVOCs					
1,4-Dichlorobenzene	0.41	0.12	not a COPC	not a COPC	not a COPC
PCBs					
Total PCBs	0.79	0.54	1.9	1.9	2.0
PCB TEQ	5.6×10^{-6} [c]	4.4×10^{-6} [c]	7.2×10^{-6} [c]	7.2×10^{-6}	6.1×10^{-6} [c]
Dioxins/Furans					
Dioxin/furan TEQ	1.9×10^{-5}	1.6×10^{-5}	not a COPC	1.6×10^{-5}	1.6×10^{-5}

a Netfishing EPCs are the combined (i.e., weighted) intertidal and subtidal values presented in Table B.3-45. Mean values were used to calculate the EPCs for the CT scenario.

b Chemical is a non-detected COPC for this exposure scenario. EPCs for non-detected COPCs are presented in Section B.6.3.2.

c For these scenarios, PCB TEQ did not screen in as a COPC but was retained as a COPC for consistency when total PCBs screened in as a COPC (see Table B.3-2).

COPC – chemical of potential concern
CT – central tendency
dw – dry weight
EPC – exposure point concentration
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent

B.3.3.4.3 Surface water

The available surface water dataset was described in Section B.2.1.3. Only water data collected within 1 m of the surface (Windward 2009f) was used for EPC calculation. Data from samples collected 1 m from the bottom were not included because people would not be expected to swim at those depths (these samples ranged in depth from 3.8 to 17 m below the water surface). In addition, the mixing of shallow and deep water is not expected to occur to a significant extent because the EW is known to be stratified, with higher salinity inflows from Elliott Bay near the bottom and freshwater outflows from the Duwamish River near the surface (Anchor and Windward 2008).

For COPCs identified based on detected concentrations in surface water for the swimming scenario (Table B.3-4), summary statistics, the distribution type, and the 95% UCL for chemical concentrations in water are presented in Table B.3-47. In addition, the dermally absorbed dose per event, which is used to calculate dermal exposure, as discussed in Section B.3.3.3, is presented in Table B.3-47.

B.3.3.5 Lead modeling

Existing evidence suggests that adverse health effects occur even at very low exposures to lead, so a reference dose, which is associated with no observable adverse effects, cannot be developed. Consequently, risk assessment methods other than those presented in other parts of Section B.3.3 are necessary for lead. The toxicokinetics of lead are well understood and indicate that lead is regulated based on blood lead concentration. Blood lead concentration can be correlated with both exposure and adverse health effects. For this HHRA, blood lead concentrations were estimated using the Integrated Exposure Uptake Biokinetic Model for Lead in Children (IEUBK) (EPA 1994) and the Adult Lead Model (ALM) (EPA 2003c). The parameterization of each model is discussed in the subsections below.

B.3.3.5.1 Children (IEUBK)

The IEUBK model (Version 1.1, Build 11 for Windows®) predicts blood-lead concentrations for children exposed to lead in their environment. The model requires input such as relevant absorption parameters and intake and exposure rates. The model then calculates and recalculates a complex set of equations to estimate the potential concentration of lead in the blood for a hypothetical population of children (aged 6 months to 7 years).

Default input parameters exist in the model for lead intake via air, drinking water, and diet. The IEUBK model allows for alternate dietary data to be used if data are available. If site-specific data are available, they are used to calculate the lead concentration for the alternate dietary source and the percentage of total dietary input that is represented by the alternate dietary source. The alternate dietary data are added to the other source data to derive a combined intake from all sources. For this HHRA, all default parameters recommended for use in the model by EPA were maintained except for alternate dietary source. The model pre-set value of 200 mg/kg dw was used for the soil concentration, which represents a "plausible value for urban soil lead concentration"(EPA 2002e). This value was not modified because no child-specific sediment exposure scenarios were evaluated for the EW.

Table B.3-47. EPCs and summary statistics for detected COPCs in surface water

COPC	Detection Frequency (ratio)	Mean Value (mg/L)	Maximum Detection (mg/L)	Maximum RL (mg/L)	Statistic Used	EPC (mg/L)	DA_{event} (mg/cm^2-event)
Arsenic[a]	28/28	0.0011	0.0016 J	na	95% Student's-t UCL	0.0012	3.1×10^{-9}
Chromium[a]	19/28	0.00091	0.0036 J	0.0024 U	95% KM (BCA) UCL	0.0011	2.9×10^{-9}
Vanadium[a]	27/28	0.0022	0.0093	0.000080 UJ	95% KM (Chebyshev) UCL	0.0039	1.0×10^{-8}
Naphthalene	8/28	0.00045	0.012	0.000042 U	95% KM (Chebyshev) UCL	0.0049	8.1×10^{-7}
Total PCBs	28/28	1.2×10^{-6}	5.8×10^{-6} J	na	95% Approximate Gamma UCL	1.5×10^{-6}	6.5×10^{-9}
PCB TEQ	28/28	5.8×10^{-10}	6.9×10^{-10} J	na	95% Student's-t UCL	6.1×10^{-10}	2.9×10^{-12}

[a] Exposure to surface water is based on total water concentrations, not dissolved concentrations.

COPC – chemical of potential concern

DA_{event} – dermally absorbed dose per event

EPC – exposure point concentration

J – estimated concentration

nd – not detected

PCB – polychlorinated biphenyl

RL – reporting limit

TEQ – toxic equivalent

U – not detected at given concentration

The default values for diet vary from 1.95 to 2.26 µg/day. These values are used to determine dietary lead exposure, unless data describing an alternate dietary source are entered. The alternate sources may include data for seafood from fishing, home-grown fruits and vegetables, and game animals from hunting. The model requires input on both the concentration of lead in the alternate dietary sources as well as the proportion of total dietary intake that these categories represent (the default concentration for all replacement foods = 0 mg/kg, default percentage of all food consumed = 0%). For the EW, only the seafood from fishing category was adjusted, because data for other food-borne sources of lead were not available. Table B.3-48 presents the alternate food source lead concentration for fish from the EW as well as the proportion of dietary intake represented by fish.

Table B.3-48. Input parameters for IEUBK lead model

Parameter	Value	Unit	Exposure Frequency
Alternate food source concentration[a]	0.21	µg lead/g	365 days per year
Alternate food source fraction[b]	12	%	na

a Alternate food source concentration was derived as a single value for all seafood categories by weighting the concentration in each seafood category by the amount of that category that is consumed. This calculation used the seafood tissue mean concentrations and the median child seafood consumption rates for each category. The alternate food source concentration was determined by summing the product of the mean EPC × median ingestion rate for each seafood category and then dividing that total by the sum of the median ingestion rates for each seafood category. Median values for ingestion rates were used per IEUBK model use guidelines (EPA 1994).

b 12 g/day (median amount of Puget Sound seafood consumed per day)/98.05 g/day (total meat consumed per day) (EPA 2006b).

dw – dry weight

EPC – exposure point concentration

IEUBK – Integrated Exposure Uptake Biokinetic Model for Lead in Children

na – not applicable

ww – wet weight

Alternate dietary data from the child tribal scenario based on Tulalip data for the consumption of fish and shellfish was included in the model. The IEUBK model applies average or CT estimates for all terms (EPA 1994). For seafood consumption rates, the median child seafood consumption rate was identified based on 40% of the median adult tribal seafood consumption rate based on Tulalip data of 29.9 g/day (EPA 2006b). Furthermore, the percentage of the alternate food source (fish) of its food group (all meat) was set at 12%. In order to calculate the average food lead concentration in the variety of seafood consumed by tribal children, the median ingestion rate was multiplied by the mean lead concentration for each seafood category. Consistent with the LDW HHRA (Windward 2007c), anadromous fish were included in the seafood consumption rate for children in the IEUBK model. Lead concentration in anadromous fish was estimated based on data collected by the Puget Sound Ambient Monitoring Program (West et al. 2001). The sum of the results of this calculation was then divided by the total ingestion rate to get the average lead concentration for EW fish. Table B.3-49 presents the details of this calculation.

Table B.3-49. Median ingestion rate and mean exposure point concentration for tribal children based on Tulalip ingestion rates by seafood category

Seafood Consumption Category	Median Ingestion Rate (g/day)	Mean Lead Concentration (mg/kg ww)	Ingestion Rate × Mean Lead Concentration (µg/day)
Anadromous fish[a]	6.0	0.04[b]	0.24
Benthic fish, fillet	0.48	0.2	0.10
Benthic fish, whole body	0	0.2	0
Clams	2.4	0.6	1.4
Crab, edible meat	1.6	0.2	0.32
Crab, whole body	0.5	0.1	0.05
Geoduck clam, edible meat	0.4	0.3	0.12
Geoduck clam, whole body	0.04	0.4	0.02
Mussels	0.04	0.3	0.01
Pelagic fish, perch	0.44	0.3	0.13
Pelagic fish, rockfish	0.08	0.4	0.03
Total	12	na	2.46

Note: The alternate food source concentration of lead, equal to 0.21 µg/g, was calculated by dividing the sum of the product of the median ingestion rate and the mean lead concentration for each seafood consumption category (2.46 µg/day) by the total median ingestion rate (12 g/day).

[a] As directed by EPA and consistent with the LDW HHRA (Windward 2007c), anadromous fish were included in the seafood consumption rate for children in the IEUBK model. This model is intended to quantify the cumulative exposure to lead for children, regardless of source. There are dietary sources other than seafood that may contain lead, but there are no site-specific data to quantify the exposure, so the default food lead concentration was used as a surrogate for all other food-borne sources of lead exposure.

[b] The mean lead concentration was set equal to the maximum detected concentration from the PSAMP database (n = 36) (West et al. 2001). All but one result were non-detects at 0.02 or 0.03 mg/kg ww.

EPA – US Environmental Protection Agency

HHRA – human health risk assessment

IEUBK – Integrated Exposure Uptake Biokinetic Model for Lead in Children

LDW – Lower Duwamish Waterway

na – not applicable

PSAMP – Puget Sound Ambient Monitoring Program

ww – wet weight

B.3.3.5.2 Adults (ALM)

The ALM is based on protecting the developing fetus of a pregnant woman, the most sensitive subpopulation affected by adult lead exposure. Accordingly, EPA has used this model to estimate soil lead cleanup levels for sites at which the likely exposed population would be older children or adults (i.e., the model is considered protective of these subpopulations). Although the model was developed to assess soil exposures, it was applied in the EW, consistent with its application to the LDW (Windward 2007c), to evaluate exposure to lead in both sediment and in seafood. Adjustments were made to the model to account for seafood intake (EPA 2007c). Specifically, Kissinger (2002) provided a revised algorithm that incorporates an exposure term for seafood consumption. This approach provided a way to evaluate cumulative exposure to lead in the EW from both direct sediment contact and seafood ingestion while still using the ALM spreadsheets developed by EPA.

The ALM is first used to estimate an average blood lead concentration in adults based on additional exposure (i.e., exposure above a baseline level) to lead in sediment, seafood, and air (Equation 3-11). This average blood lead concentration in adults is then used to estimate a fetal blood lead concentration, as is described later in this section. It should be noted that the contribution of lead from air at the EW site was considered negligible because blood lead concentrations are much less sensitive to passive re-entrainment of lead from soil in air.

$$PbB_{adult,central} = \frac{PbB_0 + BKSF \times FI \times ((PB_s \times IR_s \times AF_s \times EF_s) + (PB_f \times IR_f \times AF_f \times EF_f))}{AT}$$

Equation 3-11

Where $PbB_{adult,central}$ is the geometric mean blood lead concentration (µg/dL) in exposed adults. The definition and parameterization of the other variables in the equation above are provided in Table B.3-50.

Table B.3-50. Input parameters for the ALM

Parameter	Description	Value	Unit
General Exposure Parameters			
PbB_0	adult baseline (geometric mean) blood lead concentration	1.53[a]	µg/dL
BKSF	biokinetic slope factor	0.4 (EPA default)	µg/dL per µg/day
FI	fractional intake	1[b]	unitless
AT	averaging time	365	days
Sediment Exposure Scenario Parameters			
IR_s	sediment ingestion rate –netfishing	50 (EPA default)[c]	mg/day
IR_s	sediment ingestion rate – clamming	100 (EPA default)[c]	mg/day
Pb_s	mean lead concentration in sediment – netfishing	50	mg/kg dw
Pb_s	mean lead concentration in sediment – tribal clamming RME	50	mg/kg dw
EF_s	exposure frequency for netfishing	119	days/yr
EF_s	exposure frequency for tribal clamming RME	120	days/yr
AF_s	gastrointestinal absorbance fraction for lead in sediment	0.12 (EPA default for soil)[f]	unitless
Seafood Consumption Scenario Parameters			
IR_f	median seafood ingestion rate	15[d]	g/day
Pb_f	mean lead concentration in seafood	0.37[e]	mg/kg ww
EF_f	exposure frequency for seafood consumption	365	days/yr
AF_f	gastrointestinal absorbance fraction for lead in tissue	0.12[g]	unitless

a The average baseline blood lead concentration of women in the US was used (EPA 2002a).

b As was done for all other COPCs for the seafood consumption and direct sediment exposure scenarios (Section B.3.3.1 and B.3.3.2), a fractional intake of 1 was used for the lead model.

Table B.3-50. Input parameters for the ALM (cont.)

[c] Although EPA has not developed default exposure assumptions for sediment, a conservative assumption that assumed sediment consumption would be equivalent to 100% of the assumed soil and dust intake on each day an individual visits the EW was applied.

[d] The median Puget Sound seafood consumption rate is equal to the rate used for the adult tribal CT scenario based on Tulalip data. These ingestion rates, as well as the approach used to develop them, are discussed in Section B.3.3.1. Anadromous fish consumption was not specifically addressed in the tissue lead calculations because it is considered to be part of baseline dietary exposure, which is included in the baseline blood lead concentration.

[e] Lead concentration in seafood equals the sum of (mean lead concentration × ingestion rate) for each seafood category/total ingestion rate. Mean lead concentrations for each consumption category are presented in Table B.3-42.

[f] Gastrointestinal absorption fraction for lead in sediment (EPA 2003c).

[g] Gastrointestinal absorption fraction for lead in tissue (EPA 2007c).

ALM – Adult Lead Model
CT – central tendency
dw – dry weight
EPA – US Environmental Protection Agency
LDW – Lower Duwamish Waterway
ww – wet weight

As recommended in the ALM guidelines (EPA 2003c), mean sediment and tissue values were used to calculate risks from lead exposure and are presented in Table B.3-50. In addition, the ingestion rate for the adult tribal CT scenario based on Tulalip data (Hiltner 2007) was used in the lead model because EPA guidance calls for use of median ingestion rates in the ALM (see Section B.3.3.1.1 for additional discussion of these ingestion rates).

As indicated previously in this section, the second step in the ALM is to calculate the 95th percentile fetal blood lead concentration (Equation 3-12):

$$PbB_{fetal95} = PbB_{adult,central} \times GSD_{i,adult}^{1.645} \times R_{fetal/maternal}$$ **Equation 3-12**

Where:

$PbB_{fetal95}$	=	95th percentile fetal blood lead concentration (µg/dL)
$PbB_{adult,central}$	=	central estimate of maternal adult blood lead concentration from Equation 3-11
$GSD_{i,adult}$	=	geometric standard deviation of the blood lead distribution
1.645	=	95th percentile value for the Student's t distribution
$R_{fetal/maternal}$	=	proportionality constant between fetal and maternal blood lead concentration

The geometric standard deviation (GSD) used in Equation 3-12 is an estimation of variation in blood lead concentrations around the geometric mean. It is used to estimate upper percentile blood lead concentrations for an individual and provide a health-protective estimate of the probability of an individual exceeding a given blood lead concentration (target risk goal). In accordance with EPA (2002a), a GSD of 2.29 was applied to this model. Fetal blood lead concentrations were predicted based on the EPA assumption that fetal blood lead concentrations at birth are 90% of the maternal blood lead concentration. A 10-µg/dL blood lead concentration for a fetus is associated with a 11.1-µg/dL blood lead concentration for the mother according to EPA (2003c). The probability of exceeding the 10-µg/dL blood lead threshold for an individual was calculated using the following mathematical function in Microsoft® Excel®:

$$P_{exceedance} = 1 - \text{Normdist}(\text{Ln}(Pb_{target}/Pb_{central} \times R_{fetal/maternal}))/\text{Ln}(GSD)) \qquad \textbf{Equation 3-13}$$

Where:

Pb_{target}	=	child threshold blood lead concentration (in this application, 10 µg/dL)
$Pb_{central}$	=	child central tendency blood lead estimate
$R_{fetal/maternal}$	=	proportionality constant between fetal and maternal blood lead concentration
GSD	=	geometric standard deviation of the blood lead distribution

The results of the ALM are presented in Section B.5.4.2.

B.3.4 CHRONIC DAILY INTAKE RATES

CDI rates represent the estimated daily chemical dose for an individual averaged over the exposure duration for each scenario. Separate CDIs are calculated for chemicals with carcinogenic and non-carcinogenic effects because the averaging times over which the doses are calculated are different.

Tables 1 through 7 in Attachment 3 present the results of the CDI calculations performed using Equations 3-1 through 3-8 and the exposure parameters presented in Tables B.3-5 through B.3-40. The CDI results were used in the risk characterization and uncertainty analysis (Sections B.5 and B.6, respectively).

B.4 Toxicity Assessment

The toxicity assessment is an evaluation of each chemical's potential to cause health effects based on available toxicological information. The methodology used for this assessment is consistent with EPA guidance (2003b) and, when possible, with the LDW HHRA (Windward 2007c).

B.4.1 Hierarchy of Toxicity References and Selection of Toxicity Values

Quantitative estimates of toxicity potential have been developed by EPA and other agencies. EPA (2003b) has developed a hierarchical order of toxicity values for use in HHRAs, which was applied for the development of toxicity values for COPCs for this risk assessment:

- **Tier 1 –** EPA's Integrated Risk Information System (IRIS) database
- **Tier 2 –** EPA's Provisional Peer-Reviewed Toxicity Values (PPRTVs), Office of Research and Development/National Center for Environmental Assessment.
- **Tier 3 –** Other toxicity values. Tier 3 includes additional EPA and non-EPA sources of toxicity information. Priority is given to those sources of information that are the most current and have been peer reviewed and for which the bases are transparent and publicly available. Sources include toxicity information from EPA regional offices, EPA Health Effects Assessment Summary Tables (HEAST), the California Environmental Protection Agency (Cal EPA), and the Agency for Toxic Substance and Disease Registry (ATSDR) minimal risk levels. The source of these values was EPA's RSL tables(EPA 2010a).

Chemicals may be quantitatively evaluated on the basis of their non-carcinogenic and/or carcinogenic potential. The toxicity values used for evaluating exposure to chemicals with non-carcinogenic and carcinogenic effects are called the RfD and SF, respectively.

The RfD is an estimate, with an uncertainty spanning perhaps an order of magnitude or greater, of the daily exposure of the human population, including sensitive sub-populations, that is likely to be without an appreciable risk of deleterious effects. In developing toxicity values for non-cancer effects, EPA reviews available data to identify the most sensitive endpoint (i.e., the effects that occur at the lowest concentration) for the most sensitive population or test organism. These available data include effects on children and other sensitive subpopulations. Chemicals may have additional adverse effects that occur at higher exposure levels.

The SF represents a plausible upper-bound estimate of the probability of a carcinogenic response per unit intake of a chemical over a lifetime. EPA has recently updated their carcinogen risk assessment guidance to emphasize consideration of the mode of action (e.g., mutagenesis) in the development of SFs (EPA 2005c). In general, the SF is based on a dose-response curve using available carcinogenic data for a given chemical. For most carcinogens, mathematical models are used to extrapolate from high experimental doses to the low doses expected for human contact in the environment. The selection of the mathematical model for dose extrapolation (e.g., linear or non-linear) should be informed by the mode of action of the chemical (EPA 2005c).

The toxicity values used in this HHRA are summarized in Tables B.4-1 (non-cancer) and B.4-2 (cancer). The toxicological endpoints that were used to establish the RfDs are presented in Table B.4-3. Many chemicals may have adverse effects that are not included in Table B.4-3 because these effects occur at doses that are higher than the doses that cause the effects upon which the RfDs are based. For some chemicals for which the RfD was from a source other than IRIS, the toxicological endpoint(s) were identified using ATSDR or the Risk Assessment Information System (RAIS). For example, although not identified in IRIS as a critical effect for the development of the RfD for PCBs (or PCB Aroclors), nervous system effects, particularly neurodevelopmental effects, are well-documented across a range of PCB exposure levels (ATSDR 2000; Longnecker et al. 2003). Therefore, PCBs were included in the evaluation of non-cancer hazards associated with the neurological endpoint in the risk characterization. Similarly, although no critical effect toxicological endpoint for the RfD was identified from IRIS or ATSDR for chromium VI, gastrointestinal effects are known to result from exposure to chromium VI (ATSDR 2008), and thus this endpoint was added to Table 4-3. A discussion of the uncertainties associated with the RfDs used for risk characterization is provided in Section B.6.2.1.

The pharmacokinetics, acute toxicity, chronic toxicity, and potential carcinogenicity of each COPC are discussed in further detail in Attachment 4. The discussion of toxic effects in Attachment 4 includes many different exposure routes, some of which are not relevant to environmental exposure within the EW, such as occupational inhalation exposure. The additional exposure routes are included in Attachment 4 only for completeness. The information on pharmacokinetics, acute toxicity, chronic toxicity, and the potential carcinogenicity of each COPC was obtained primarily from the following sources:

- EPA's IRIS database
- EPA's PPRTV database
- EPA's 1997 values contained in the HEAST
- Toxicological profiles presented in EPA's *Guidance for Assessing Chemical Contaminant Data for Use in Fish Advisories* (EPA 2000d)
- EPA's Office of Ground Water and Drinking Water
- The ATSDR ToxFAQs™
- Hazardous Substance Data Bank
- RAIS

Table B.4-1. Non-cancer toxicity data (oral) for chemicals of potential concern

Chemical[a]	Oral RfD (mg/kg day)	Endpoint (Critical Effect from RfD Source)	Uncertainty Factor	RfD Source	Source Date[b]	Notes
Detected COPCs						
Metals						
Antimony	0.0004	endocrine and hematologic systems (adverse effects on longevity, blood glucose, and cholesterol)	1,000	IRIS	6/29/2010	
Arsenic	0.0003	cardiovascular and integumentary systems (hyperpigmentation, keratosis, and possible vascular complications)	3	IRIS	6/29/2010	surrogate = inorganic arsenic
Cadmium (in food)	0.001	kidney (significant proteinuria)	10	IRIS	6/29/2010	
Chromium	0.003	digestive system (irritation of and ulcers in the stomach and small intestine, anemia, male reproductive damage [ATSDR])	300	IRIS	6/29/2010	surrogate = hexavalent chromium
Cobalt	0.0003	endocrine system (thyroid – decreased iodine uptake)	3,000	PPRTV	8/15/2011	
Copper	0.04[c]	digestive system (irritation)	na	HEAST	5/2010	
Lead	na[d]	nervous system (neurotoxicant)	na	na	na	
Mercury	0.0001	development and nervous system (developmental neuropsychological impairment)	10	IRIS	6/29/2010	surrogate = methylmercury
Molybdenum	0.005	kidney (increased uric acid levels)	30	IRIS	6/29/2010	
Selenium	0.005	hematologic, nervous, and integumentary systems (clinical selenosis)	3	IRIS	6/29/2010	selenium and compounds
Vanadium	0.009	integumentary system (decreased hair cystine)	100	IRIS	6/29/2010	vanadium pentaoxide
Zinc	0.3	hematologic system (decreases in erythrocyte copper, zinc-superoxide dismutase activity in healthy adults)	3	IRIS	6/29/2010	zinc and compounds

Table B.4-1. Non-cancer toxicity data (oral) for chemicals of potential concern (cont.)

Chemical[a]	Oral RfD (mg/kg day)	Endpoint (Critical Effect from RfD Source)	Uncertainty Factor	RfD Source	Source Date[b]	Notes
Organometals						
Dibutyltin as ion	0.0003	immune system (immunotoxicity and reduced body weight)	1,000	PPRTV	8/15/2011	dibutyltin compounds
Tributyltin as ion	0.00015	immune system (immunosuppression)	100	IRIS	6/30/2010	surrogate = by conversion from tributyltin oxide (multiply IRIS oral RfD by 0.49)
Other SVOCs						
1,4-Dichlorobenzene	0.07	liver (adverse effects on liver, kidney, and blood, nervous system during development, skin blotches and anemia from regular exposure over long periods)	na	ATSDR	5/2010	
Pentachlorophenol	0.03	kidney and liver (adverse effects on liver and kidney pathology)	100	IRIS	6/29/2010	
PAHs						
Naphthalene	0.02	body weight (decreased mean terminal body weight in males)	3,000	IRIS	6/29/2010	
PCBs						
Total PCBs[e] (total includes Aroclors 1016, 1221, 1232, 1242, 1248, 1254, and 1260)	0.00002	immune and integumentary systems, eyes (ocular exudate, inflamed and prominent Meibomian glands, distorted nail growth, decreased antibody response)	300	IRIS	6/29/2010	surrogate = Aroclor 1254, the lowest and most protective RfD available for PCBs in IRIS. Note that this RfD was also applied for calculation of nervous system effects (see Table 4-3).
	0.00007	development (reduced birth weights)	100	IRIS	8/16/2011	surrogate = Aroclor 1016
PCB TEQ	na	An RfD for 2,3,7,8-TCDD (the chemical upon with the toxicity of PCB TEQ is based) was made available by EPA on IRIS in February 2012 as the EW HHRA was nearing completion. Thus, non-cancer hazards were not incorporated into this document, but will be presented as part of the SRI.				
Pesticides						
alpha-BHC	0.008	liver	na	ATSDR	5/2010	
Dieldrin	0.00005	liver (liver lesions)	100	IRIS	6/29/2010	
Heptachlor	0.0005	liver (liver weight increases in males)	300	IRIS	8/27/2010	

Table B.4-1. Non-cancer toxicity data (oral) for chemicals of potential concern (cont.)

Chemical[a]	Oral RfD (mg/kg day)	Endpoint (Critical Effect from RfD Source)	Uncertainty Factor	RfD Source	Source Date[b]	Notes
Heptachlor epoxide	0.000013	liver and body weight (increased liver-to-body weight ratio in both males and females)	1,000	IRIS	8/27/2010	
Mirex	0.0002	liver (liver cytomegaly, fatty metamorphosis, angiectasis; thyroid cystic follicles)	300	IRIS	8/27/2010	
Total DDTs	0.0005	liver (liver lesions)	100	IRIS	6/29/2010	surrogate = 4,4′-DDT; total includes DDDs, DDEs, and DDTs
Total chlordane	0.0005	liver (hepatic necrosis)	300	IRIS	6/29/2010	surrogate = chlordane (technical); total includes alpha-chlordane, gamma-chlordane, oxychlordane, cis-nonachlor, and trans-nonachlor
Dioxins/Furans						
Dioxin/furan TEQ	na	An RfD for 2,3,7,8-TCDD (the chemical upon with the toxicity of Dioxin/furan TEQ is based) was made available by EPA on IRIS in February 2012 as the EW HHRA was nearing completion. Thus, non-cancer hazards were not incorporated into this document, but will be presented as part of the SRI.				
Non-Detected COPCs						
Phthalates						
BEHP	0.020	liver (liver weight increases)	1,000	IRIS	9/28/2010	
Butyl benzyl phthalate	0.200	liver (significantly increased liver-to-body weight and liver-to-brain weight ratios)	1,000	IRIS	9/28/2010	
Other SVOCs						
1,2,4-Trichlorobenzene	0.01	endocrine system (adrenal weight increases; vacuolization of zona fasciculata in the cortex)	1,000	IRIS	9/27/2010	
1,2-Diphenylhydrazine	na	na	na	na	na	
2,4,6-Trichlorophenol	0.001	development (decreased litter size)	3,000	PPRTV	8/15/2011	
2,4-Dichlorophenol	0.003	immune system (decreased delayed hypersensitivity response)	100	IRIS	9/27/2010	

Table B.4-1. Non-cancer toxicity data (oral) for chemicals of potential concern (cont.)

Chemical[a]	Oral RfD (mg/kg day)	Endpoint (Critical Effect from RfD Source)	Uncertainty Factor	RfD Source	Source Date[b]	Notes
2,4-Dinitrophenol	0.002	eyes (cataract formation)	1,000	IRIS	9/27/2010	
2,4-Dinitrotoluene	0.002	liver, nervous system, hematologic system (neurotoxicity, Heinz bodies and biliary tract hyperplasia)	100	IRIS	9/28/2010	
2,6-Dinitrotoluene	0.001	liver, nervous system, hematologic system (neurological, hematological, and liver histopathology)	3,000	PPRTV	8/15/2011	
2-Nitroaniline	0.01	na	10,000	PPRTV appendix	8/15/2011	screening value
3,3'-Dichlorobenzidine	na	na	na	na	na	
4,6-Dinitro-o-cresol	0.00008	na	10,000	PPRTV appendix	8/15/2011	screening value
4-Chloroaniline	0.004	hematologic system (non-neoplastic lesions of splenic capsule)	3,000	IRIS	9/28/2010	
4-Nitroaniline	0.004	hematologic system (increases in methemoglobin and hemosiderosis)	100	PPRTV	8/15/2011	
Aniline	0.007	hematologic system (spleen/blood effects)	1,000	PPRTV	8/15/2011	
Bis(2-chloroethoxy)methane	0.003	liver (liver lesions)	3,000	PPRTV	8/15/2011	
Bis(2-chloroethyl)ether	na	na	na	na	na	
Hexachlorobenzene	0.0008	liver (liver effects)	100	IRIS	9/28/2010	
Hexachlorobutadiene	0.001	kidney (tubule regeneration)	100	PPRTV	8/15/2011	
Hexachlorocyclopentadiene	0.006	digestive system (chronic irritation such as forestomach lesions)	1,000	IRIS	9/28/2010	
Hexachloroethane	0.001	kidney (atrophy and degeneration of the renal tubules)	1,000	IRIS	9/28/2010	
Nitrobenzene	0.002	hematologic system (increased methemoglobin levels)	1,000	IRIS	9/28/2010	
n-Nitroso-di-n-propylamine	na	na	na	na	na	

Table B.4-1. Non-cancer toxicity data (oral) for chemicals of potential concern (cont.)

Chemical[a]	Oral RfD (mg/kg day)	Endpoint (Critical Effect from RfD Source)	Uncertainty Factor	RfD Source	Source Date[b]	Notes
n-Nitrosodimethylamine	0.000008	development (weanling sex ration and perinatal mortality)	3,000	PPRTV	8/15/2011	
n-Nitrosodiphenylamine	na	na	na	na	na	
PAHs						
Benzo(a)pyrene	na	na	na	na	na	
Pesticides						
Aldrin	0.00003	liver (liver toxicity)	1,000	IRIS	9/28/2010	
Toxaphene	na	na	na	na	na	

a Chemicals for which no RfDs were available were excluded from this table. These chemicals include beta-BHC, cPAH TEQ, dioxin/furan TEQ, and PCB TEQ.

b For IRIS and PPRTV, the source date represents the date that the database was searched; for ATSDR and HEAST, the source dates represent the dates that the RSL tables (EPA 2010a) (the sources of these values) were updated.

c The source of the RfD for copper is HEAST, which per EPA has not been updated since 1997. The HEAST document provides a drinking water criteria value for copper (EPA 1997b), which was converted into a provisional RfD by EPA for use in the RSL tables (EPA 2010a). Although uncertain, this provisional RfD was considered acceptable for use in the HHRA based on its inclusion in EPA's RSL tables.

d No RfD is available for lead because existing toxicity information for lead indicates adverse effects even at very low concentrations (RfDs are established as the concentration below which studies have found there to be no adverse effects). The method used to evaluate risks associated with exposure to lead is discussed in detail in Section B.3.3.5.

e Two RfDs are listed for total PCBs. HQs based on both of these RfDs are presented in Section B.5 to allow for an evaluation of the effects of exposure to PCBs on different endpoints.

ATSDR – Agency for Toxic Substance and Disease Registry
BHC – benzene hexachloride
BEHP – bis(2-ethylhexyl) phthalate
cPAH – carcinogenic polycyclic aromatic hydrocarbon
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EPA – US Environmental Protection Agency
HEAST – Health Effects Assessment Summary Tables
HHRA – human health risk assessment
IRIS – Integrated Risk Information System
na – not available
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PPRTV – Provisional Peer-Reviewed Toxicity Values
RfD – reference dose
RSL – regional screening level
SVOC – semivolatile organic compound
TEQ – toxic equivalent
TOC – toxic equivalency

Table B.4-2. Cancer toxicity data (oral/dermal) for chemicals of potential concern

Chemical[a]	Oral Cancer Slope Factor (mg/kg-day)$^{-1}$	Cancer Description Guideline[b]	Source	Source Date[c]	Notes
Detected COPCs					
Metals					
Arsenic	1.5	A	IRIS	6/29/2010	surrogate = inorganic arsenic
Cobalt	na	B1 (RAIS)	na	6/29/2010	
Copper	na	D	IRIS	6/29/2010	
Mercury	na	C	IRIS	6/29/2010	surrogate = methylmercury
Selenium	na	D	IRIS	6/29/2010	selenium and compounds
Zinc	na	D	IRIS	6/29/2010	zinc and compounds
Other SVOCs					
1,4-Dichlorobenzene	0.0054	C (RAIS)	Cal EPA	5/2010	
Pentachlorophenol	0.12	B2	IRIS	6/29/2010	
PAHs					
Naphthalene	na	C	IRIS	6/29/2010	
cPAH TEQ	7.3	B2	IRIS	6/29/2010	slope factor based on benzo(a)pyrene
PCBs					
Total PCBs	2	B2	IRIS	6/29/2010	upper-bound slope factor used for this risk estimate; total includes Aroclors 1016, 1221, 1232, 1242, 1248, 1254, and 1260
PCB TEQ	150,000[d]	B2 (RAIS)	HEAST	4/2006	slope factor based on 2,3,7,8-TCDD; consistent with the slope factor used in the LDW HHRA
Pesticides					
alpha-BHC	6.3	B2	IRIS	8/27/2010	
beta-BHC	1.8	C	IRIS	8/27/2010	
Dieldrin	16	B2	IRIS	6/29/2010	
Heptachlor	4.5	B2	IRIS	8/27/2010	
Heptachlor epoxide	9.1	B2	IRIS	8/27/2010	
Mirex	18	B2 (RAIS)	Cal EPA	5/2010	

Table B.4-2. Cancer toxicity data (oral/dermal) for chemicals of potential concern (cont.)

Chemical[a]	Oral Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Cancer Description Guideline[b]	Source	Source Date[c]	Notes
Total chlordane	0.35	B2	IRIS	6/29/2010	surrogate = chlordane (technical); total includes alpha-chlordane, gamma-chlordane, oxychlordane, cis-nonachlor, and trans-nonachlor
Total DDTs	0.34	B2	IRIS	6/29/2010	surrogate = 4,4′-DDT; total includes DDDs, DDEs, and DDTs
Dioxins/Furans					
Dioxin/furan TEQ	150,000[d]	B2 (RAIS)	HEAST	4/2006	slope factor based on 2,3,7,8-TCDD; consistent with the slope factor used in the LDW HHRA
Non-Detected COPCs					
Phthalates					
BEHP	0.01	B2	IRIS	9/28/2010	
Butyl benzyl phthalate	na	C	IRIS	9/28/2010	
Other SVOCs					
1,2,4-Trichlorobenzene	na	D	IRIS	9/27/2010	
1,2-Diphenylhydrazine	0.8	B2	IRIS	9/27/2010	
2,4,6-Trichlorophenol	0.01	B2	IRIS	9/27/2010	
2,4-Dichlorophenol	na	na	na	na	
2,4-Dinitrophenol	na	na	na	na	
2,4-Dinitrotoluene	0.31	na	Cal EPA	5/2010	
2,6-Dinitrotoluene	0.68	B2	IRIS	9/28/2010	surrogate = 2,4-/2,6-dinitrotoluene mixture
2-Nitroaniline	na	D (RAIS)	na	na	
3,3'-Dichlorobenzidine	0.45	B2	IRIS	9/28/2010	
4,6-Dinitro-o-cresol	na	D (RAIS)	na	9/28/2010	
4-Chloroaniline	na	C (RAIS)	na	9/28/2010	
4-Nitroaniline	0.02	C (RAIS)	PPRTV	8/2011	
Aniline	0.006	B2	IRIS	9/28/2010	
Bis(2-chloroethoxy)methane	na	D	IRIS	9/28/2010	
Bis(2-chloroethyl)ether	1.1	B2	IRIS	9/28/2010	
Hexachlorobenzene	1.6	B2	IRIS	9/28/2010	

Table B.4-2. Cancer toxicity data (oral/dermal) for chemicals of potential concern (cont.)

Chemical[a]	Oral Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Cancer Description Guideline[b]	Source	Source Date[c]	Notes
Hexachlorobutadiene	0.08	C	IRIS	9/28/2010	
Hexachloroethane	0.01	C	IRIS	9/28/2010	
Nitrobenzene	na	likely human carcinogen	IRIS	9/28/2010	
n-Nitroso-di-n-propylamine	7.0	B2	IRIS	8/15/2011	
n-Nitrosodimethylamine	51	B2	IRIS	9/28/2010	
n-Nitrosodiphenylamine	0.005	B2	IRIS	9/28/2010	
PAHs					
Benzo(a)pyrene	7.3	B2	IRIS	9/28/2010	
Pesticides					
Aldrin	17	B2	IRIS	9/28/2010	
Toxaphene	1.1	B2	IRIS	9/28/2010	

[a] Chemicals included in this table are either Class A, B, or C chemicals with regard to their cancer-causing potential, with available cancer slope factors, or are Class D chemicals (see Footnote b). Cadmium and chromium, although known carcinogens, have been excluded from this table because they are carcinogens only via the inhalation pathway, which is not a pathway of concern for this HHRA.

[b] Classifications are as follows: A = known human carcinogen; B1 = probable human carcinogen (based on limited evidence of carcinogenicity in humans); B2 = probable human carcinogen (sufficient evidence in animals and inadequate or no evidence in humans); C = possible human carcinogen (limited evidence from animal studies and inadequate or no data in humans); D = not classifiable as to human carcinogenicity.

[c] For IRIS and PPRTV, the source date represents the date that the database was searched; for Cal EPA and HEAST, the source date represents the dates that the RSL tables (the sources of these values) was updated.

[d] Although HEAST has not been updated recently, the HEAST slope factor for 2,3,7.8-TCDD was used per EPA comments for consistency with the LDW HHRA (Windward 2007c). However, it should be noted that there is uncertainty associated with this value. Several alternate value slope factors are available for dioxins (e.g., EPA's past dioxin slope factor of 156,000 $[mg/kg\text{-}day]^{-1}$ or Cal EPA's slope factor of 130,000 $[mg/kg\text{-}day]^{-1}$), and EPA is currently in the process of completing a reassessment of the toxicity of dioxins (EPA 2012).

BHC – benzene hexachloride
BEHP – bis(2-ethylhexyl) phthalate
Cal EPA – California Environmental Protection Agency
cPAH – carcinogenic PAH
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EPA – Environmental Protection Agency
HEAST – Health Effects Assessment Summary Tables
HHRA – human health risk assessment
IRIS – Integrated Risk Information System
LDW – Lower Duwamish Waterway
na – not applicable
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PPRTV – Provisional Peer-Reviewed Toxicity Values
RAIS – Risk Assessment Information System
SVOC – semivolatile organic compound
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent

Table B.4-3. Toxicological endpoints for detected COPCs with non-carcinogenic effects

Chemical	Endpoint											
	Kidney	Liver	Develop-ment	Cardio-vascular System	Endocrine System	Hematologic System	Immune System	Nervous System	Integumentary System (Skin, Hair, Nails, and Teeth)	Eyes	Body Weight	Digestive System
Metals												
Antimony					X	X						
Arsenic (as inorganic arsenic)				X					X			
Cadmium	X											
Chromium (as hexavalent chromium)[a]												X
Cobalt					X							
Copper[b]												X
Lead[c]								X				
Mercury (as methylmercury)			X					X				
Molybdenum	X											
Selenium						X		X	X			
Vanadium[d]									X			
Zinc						X						
Organometals												
Dibutyltin as ion[e]							X					
Tributyltin as ion							X					
Other SVOCs												
1,4-Dichlorobenzene[f]		X										
Pentachlorophenol	X	X										
PAHs												
Naphthalene											X	
PCBs												
Total PCBs (based on Aroclor 1254)[g]			X[h]				X[h]	X[h]	X[h]	X[h]		

Table B.4-3. Toxicological endpoints for detected COPCs with non-carcinogenic effects (cont.)

Chemical	Endpoint											
	Kidney	Liver	Develop-ment	Cardio-vascular System	Endocrine System	Hematologic System	Immune System	Nervous System	Integumentary System (Skin, Hair, Nails, and Teeth)	Eyes	Body Weight	Digestive System
Pesticides												
alpha-BHC		X										
Total chlordane		X										
Total DDTs		X										
Dieldrin		X										
Heptachlor		X										
Heptachlor epoxide		X									X	
Mirex		X										

Note: Each of these chemicals will likely have additional toxic effects (endpoints) at exposures above the RfD. The toxicological profile section of the HHRA (Attachment 4) may be consulted by readers desiring more information on toxic effect endpoints. ATSDR toxicological profiles are an excellent resource for this information.

[a] The supporting study used to establish the IRIS RfD observed no toxic effects. The endpoint for chromium was identified using ATSDR (2008).

[b] The RfD for this chemical is from a source other than IRIS. The endpoint for copper was identified using ATSDR (2004).

[c] No RfD was available for this chemical. Lead was not evaluated using the RfD approach, but through the modeling of blood levels in children and fetuses (see Section B.3.3.5). However, lead is known to be a neurotoxicant, and thus this endpoint is shown in this table.

[d] The RfD for this chemical is from a source other than IRIS. The endpoint for vanadium was identified using ATSDR (1995).

[e] The RfD for this chemical is from a source other than IRIS. The endpoint for dibutyltin was identified using RAIS (2009).

[f] The RfD for this chemical is from a source other than IRIS. The endpoint for this chemical was identified using ATSDR (2006).

[g] The effects of PCBs on skin (chloracne) are well-documented, but these are associated with acute exposures at levels much higher than the RfD (ATSDR 2000).

[h] As shown in Table 4-1, two RfDs are presented for total PCBs: the first based on effects to the immune system, integumentary system, and eyes; and the second based on developmental effects (reduced birth weight associated with Aroclor 1016). Separate HQs were calculated in Section B.5 based on these two RfDs. In addition, nervous system effects for PCBs were not identified in IRIS for derivation of the RfDs; but such effects, particularly neurodevelopmental effects, are well-documented (ATSDR 2000). The lower and more health protective RfD (associated with immune, integumentary systems, and eyes for Aroclor 1254) was applied for calculation of nervous system effects.

ATSDR – Agency for Toxic Substance and Disease Registry
COPC – chemical of potential concern
HHRA – human health risk assessment
HI – hazard index
HQ – hazard quotient
IRIS – Integrated Risk Information System
PCB – polychlorinated hydrocarbon
RAIS – Risk Assessment Information System
RfD – reference dose
X – Indicates that the IRIS RfD for a particular chemical was calculated for the identified endpoint.

B.4.2 Toxicity Assessment for PCB TEQ, Dioxin/Furan TEQ, and cPAH TEQ

As previously noted in Sections B.2.2.4 and B.3.3.4, certain classes of chemicals are composed of individual compounds that have similar chemical structures as well as a common mechanism of toxicity. Toxicity assessment for these classes is done on a group, rather than on an individual compound basis. These compound groups include co-planar PCB congeners, dioxins/furans, and cPAHs. The specific chemicals in each group were presented in Table B.2-5.

The toxicity of co-planar PCBs and dioxins/furans was assessed using a toxic equivalency approach. Each congener was assigned a TEF that describes the toxicity of that congener relative to that of the reference compound 2,3,7,8-TCDD. A congener that is assumed to be equal in toxicity to 2,3,7,8-TCDD would have a TEF of 1.0. A congener that is assumed to be half as toxic as 2,3,7,8-TCDD would have a TEF of 0.5.

There have been several efforts to develop TCDD TEFs for dioxins/furans and co-planar PCBs that have 2,3,7,8-TCDD-like toxicity (EPA 2000b). The most recent effort occurred at an expert meeting organized by the World Health Organization (WHO) in 2005 (Van den Berg et al. 2006). The WHO effort examined a number of lines of evidence to develop a consensus-based list of TEFs. Table B.2-5 provides the 2005 WHO TEFs for co-planar PCBs and dioxins/furans.

As discussed in Section B.2.2.4, the sum of the products of the concentration of each co-planar PCB and its TEF is referred to as the PCB TEQ and is calculated on a per-sample basis. Similarly, the sum of the products of each co-planar dioxin/furan and its TEF is referred to as the dioxin/furan TEQ and is also calculated on a per-sample basis. The excess cancer risk posed by PCB TEQ and dioxin/furan TEQ is then determined by multiplying the TEQ CDI by the SF for the reference compound 2,3,7,8-TCDD (see Table B.4-2). In this document, PCB TEQ and dioxin/furan TEQ exposure and risk estimates are presented both separately and together as a total TEQ excess cancer risk.

The toxicity of multiple cPAHs may be evaluated using the relative potency approach. This approach involves a comparison of the cancer-causing ability of a particular cPAH with a reference compound, benzo[a]pyrene, by means of a PEF. A cPAH with a PEF of 1.0 is assumed to be as effective as benzo[a]pyrene in inducing cancer. A cPAH with a PEF of 0.5 would be assumed to be half as effective as benzo[a]pyrene in inducing cancer.

PEFs for individual cPAHs have been developed by Cal EPA (1994) based on various toxic endpoints. EPA has also developed relative potency factors (RPFs), which are similar to PEFs, for these cPAHs (EPA 1993). The California EPA cPAH PEFs were used in the EW HHRA because they had better documentation than did the EPA RPF values.

As discussed in Section B.2.2.4, the sum of the products of the concentration of each cPAH and its PEF is considered the cPAH TEQ and is calculated on a per-sample basis.

The excess cancer risk posed by the cPAH TEQ is then computed by multiplying the cPAH TEQ (i.e., total benzo[a]pyrene equivalents) CDI by the benzo[a]pyrene SF (Table B.4-2).

There are additional considerations for calculating risks for children associated with cPAH TEQ, which has a mutagenic mode of action. EPA guidance provides adjustments for chemicals that have a mutagenic mode of action (EPA 2005d). Of the COPCs in this HHRA, only cPAHs fall into this category. As stated in the guidance, the toxicity values (i.e., SFs) of chemicals with mutagenic modes of action should be adjusted upward by a factor of 10 for children aged 0 to 2 and by a factor of 3 for children aged 3 to 16. This approach, which is discussed further in Section B.5.1.1, is based on the assumption that carcinogens with a mutagenic mode of action may have more deleterious effects when exposure occurs during early life stages. In this HHRA, all children's scenarios consisted of exposures from ages 0 to 6 years, for which SF adjustments for cPAHs were incorporated. Adjustments to the SF for older children are discussed in Section 6.3.3.

B.5 Risk Characterization

This section presents risk estimates for all exposure scenarios presented in Section B.3 based on the toxicity information presented in Section B.4. The equations used to calculate the risk estimates are presented, followed by the calculation results. These estimates are useful for characterizing risks to people who could be exposed to chemicals present in EW seafood, sediment, and surface water and for identifying chemicals of concern (COCs), which are defined here as chemicals with an estimated excess cancer risk greater than 1×10^{-6} or a non-cancer HQ greater than 1 for any RME scenario. In addition, the risks estimates presented in this section are used to identify those chemicals that represent the greatest contribution to total risk.

Cancer risks and non-cancer hazards were calculated separately in a manner consistent with EPA guidance (EPA 1989). In addition, EPA's guidance for children's carcinogenic risk assessment (EPA 2005d) was used.[35] Excess cancer risks and HQs are presented according to the format recommended by EPA (2001c) for chemicals detected in EW seafood, sediment, or surface water. This section also presents incremental risk estimates (i.e., the difference between the risk estimates calculated for the EW and those calculated based on background or upstream concentrations) to allow an evaluation of the contribution of background concentrations to risks at the site.

B.5.1 Calculation of Risk Estimates

Carcinogenic risks and non-carcinogenic health effects were evaluated separately because of fundamental differences in their critical toxicity values. Equations for each type of effect are presented in separate subsections that follow.

B.5.1.1 Carcinogenic risks

For relatively low probabilities (i.e., below 1 in 100), carcinogenic risks are calculated by multiplying the estimated exposure level (the CDI) by the cancer SF for each chemical.[36]

$$\text{Risk} = \text{CDI} \times \text{SF} \qquad \textbf{Equation 5-1}$$

Where:

Risk	=	estimated chemical-specific individual lifetime excess cancer risk (unitless)[37]
CDI	=	chemical-specific chronic daily intake (mg/kg-day)
SF	=	route- and chemical-specific carcinogenic SF $(\text{mg/kg-day})^{-1}$

[35] This approach is consistent with that used in the LDW HHRA.

[36] In cases where excess cancer risk estimates exceed 1 in 100, the exponential version of the risk equation will be used, per EPA guidance (1989).

[37] Excess cancer risk refers to risks associated with site-specific exposure, above and beyond risks associated with exposure from all other causes, including exposure to carcinogenic sources outside of the site.

Excess cancer risks were summed across chemicals for each exposure scenario. Consistent with the LDW HHRA (Windward 2007c), two risk estimate totals are presented in tables in Section 5.3, one consisting of excess cancer risks from all detected non-PCB chemicals together with risk from total PCBs (excluding the PCB TEQ excess cancer risk) and the other consisting of excess cancer risks from all detected non-PCB chemicals together with risk from PCB TEQs (excluding the total PCB excess cancer risk). These two methods were used because two types of PCB data are available for the site (i.e., total PCBs as Aroclors and PCB TEQ). It is not appropriate to sum the excess cancer risks from both types of PCB data because some level of double-counting of PCB risks will occur. This is because the PCB congeners with dioxin-like properties that are used to calculate the PCB TEQ are also present in the commercial Aroclor mixtures used to derive the total PCB sum. However, it is possible that bioaccumulated PCB mixtures may have altered congener compositions, which will result in greater toxicity than the toxicity predicted for commercial Aroclor mixtures or co-planar PCB TEQ alone. The true risk may be between that determined for total PCBs or PCB TEQ and the sum of these risks. The uncertainty associated with PCB risk characterization, as well as alternative methods for calculating total risk (e.g., including both total PCBs [as Aroclors] and PCB TEQ estimates) are discussed in the uncertainty analysis (Section B.6.3.1). PCB TEQ and dioxin/furan TEQ are also subtotaled in the second approach to estimate the total TEQ risk.

EPA guidance (2005d) provides direction for the calculation of carcinogenic risks to children from chemicals with mutagenic modes of action. For cPAHs, which have been identified as having a mutagenic mode of action (the only COPC in this category for this HHRA), the SF was adjusted upwards in the risk calculation (as shown in Equation 5-2) to account for potential greater susceptibility of children from 0 to 6 years of age compared with that of older children and adults.

$$\text{Risk} = \left(\text{CDI}_{\text{age}0-2} \times (\text{SF} \times 10)\right) + \left(\text{CDI}_{\text{age}3-6} \times (\text{SF} \times 3)\right) \qquad \textbf{Equation 5-2}$$

Where:

Risk	=	chemical-specific lifetime excess cancer risk (unitless)
CDI	=	chemical-specific chronic daily intake rate based on the exposure duration for each age group (i.e., 2 years for ages 0 to 2 and 4 years for ages 3 to 6)
SF	=	route- and chemical-specific cancer slope factor $(\text{mg/kg-day})^{-1}$

When risks to children (aged 0 to 6 years) were evaluated, this SF adjustment was made to ensure that risk estimates would be sufficiently protective of early life stage exposure, per EPA guidance (2005d). The implementation of this approach resulted in an approximately five-fold increase in the cPAH TEQ cancer risk estimate for children and was based on the assumption that the toxicity of carcinogens with a mutagenic mode of action could be greater for younger children than for older children or adults.

Excess cancer risks are probabilities expressed in scientific notation. For example, 1×10^{-5} is equivalent to 0.00001 or 1 in 100,000. Excess cancer risks are presented with only one significant figure to acknowledge the uncertainty in the cancer SFs, per EPA guidance (1989), and in the exposure assumptions underlying the calculations.

B.5.1.2 Non-carcinogenic health effects

The potential for non-carcinogenic health effects is represented by the ratio of a chemical's CDI and the route-specific RfD and is expressed as an HQ:

$$HQ = \frac{CDI}{RfD} \quad \textbf{Equation 5-3}$$

Where:

HQ = estimated chemical-specific hazard quotient (unitless)
CDI = chemical-specific chronic daily intake (mg/kg-day)
RfD = route- and chemical-specific reference dose (mg/kg-day)

The HQ is recommended by EPA as a way to quantify the potential for non-carcinogenic health effects (EPA 1989). HQs are not risk probabilities; the likelihood of an adverse effect does not usually increase linearly with the calculated value. An HQ greater than 1 indicates potential adverse health effects from chemical exposure, although the same HQ may not equate to the same magnitude of adverse health effects for all chemicals. HQ interpretation considers the shape and slope of the dose-response curve in the area of observation, the magnitude of uncertainty and modifying factors to the RfD, and the confidence assigned to the RfD by EPA.

Individual COPCs with similar toxicological effects may be summed to yield an effect-specific hazard index (HI) (EPA 1989). The effect-specific HI is an expression of the additivity of non-carcinogenic health effects. An effect-specific HI can be calculated by summing HQs for chemicals with similar toxicological effects (e.g., immunotoxicity). If the sum of all HQs for a given scenario evaluated in the EW HHRA was less than 1, no effect-specific HIs were calculated because they would also not exceed 1. Effect-specific HIs were calculated for scenarios when the sum of all HQs was greater than 1. This is consistent with the approach used in the LDW HHRA (Windward 2007c).

B.5.1.3 Risk thresholds

CERCLA risk thresholds are discussed in the National Contingency Plan (NCP) (40 CFR 300), which states that "for known or suspected carcinogens, acceptable exposure levels are generally concentration levels that represent an excess upper bound lifetime cancer risk to an individual of between 1×10^{-4} (1 in 10,000) and 1×10^{-6} (1 in 1,000,000)." This range is referred to as the acceptable risk range in EPA guidance (1991b). In addition, non-carcinogenic hazards that do not exceed 1 are considered to be acceptable by EPA (1991b).

In this HHRA, chemicals were identified as COCs if they had an excess cancer risk estimate greater than 1×10^{-6} or an HQ greater than 1 for any RME scenario. As a health-protective approach (i.e., to account for the presence of multiple chemicals and exposure pathways), the lower end of EPA's acceptable cancer risk range (1×10^{-6}) was selected for identifying COCs.

B.5.2 Risk Characterization Format

Excess cancer risks and HQs are presented according to the format recommended in EPA (2001c). The primary purpose of the HHRA is to characterize risks to people who may be exposed to chemicals present in the EW. This HHRA will also support risk management decisions and the evaluation of remedial options related to the EW. Risk estimates provide information to the public about what their health risks may be from engaging in different activities associated with the EW (e.g., consumption of EW seafood, netfishing, habitat restoration, clamming, and swimming). Therefore, risks have been characterized and quantified for chemicals detected in EW seafood, sediment, or surface water.

Some chemicals that were never detected but have RLs that exceed RSLs have been identified as COPCs for each exposure pathway and medium. Consistent with the LDW HHRA (Windward 2007c), hypothetical risk estimates for these undetected COPCs are quantified and discussed in the uncertainty analysis (Section B.6.3.2). Risks estimates attributable to these undetected chemicals have very high uncertainty. In cases where the RLs of non-detected chemicals resulted in high risk estimates, historical site uses were considered to determine whether there was reason to believe the chemical was actually present. In certain cases, additional analyses were run to achieve lower detection limits, thus reducing the estimates of potential risk associated with these chemicals (e.g., for chlorinated pesticides).

Excess cancer risks are summed for all COPCs associated with each exposure scenario.[38] Exposure scenarios in which the same receptor is exposed via multiple pathways simultaneously were addressed by summing the RME estimates for those pathways. This approach was applied to all direct sediment and water exposure scenarios that involved both dermal absorption and incidental sediment ingestion. In addition, excess cancer risk estimates were summed across some potentially related scenarios (e.g., netfishing and seafood consumption). For some combinations of scenarios, the highest RME pathway risk estimate may be several orders of magnitude higher than the other scenarios. The resulting risk estimate for the combination of multiple scenarios may then differ only slightly or not at all from the risk estimate for the RME scenario alone.

In this section, CDIs are presented with two significant figures; excess cancer risks and HQs are presented with only one significant figure. The former makes the calculations

[38] COPCs that were never detected in a given media were not included in this sum but instead were evaluated in the uncertainty analysis (Section B.6).

behind the risk estimates easier to follow, and the latter reflects the accuracy of the cancer SFs and reference doses, per the EPA Integrated Risk Information System (IRIS) database. Sums of excess cancer risk estimates are reported with one significant figure as well. For example, the sum of the excess risk estimates of 2×10^{-4} and 3×10^{-5} would be reported as 2×10^{-4}, not 2.3×10^{-4}. HIs (sums of HQs) are presented with one significant figure if they are less than 1, or to the nearest integer if they are greater than 1. This is to allow the reader to follow summations. For example, HQs of 4 and 10 would be summed to an HI of 14 and not rounded to 10. However, HQs of 0.01 and 0.001 would be summed to an HI of 0.01, not 0.011.

B.5.3 RISK CHARACTERIZATION RESULTS

This section presents the risk characterization results for each exposure scenario type: seafood consumption (Section B.5.3.1), direct sediment exposure (Section B.5.3.2), and surface water exposure (Section B.5.3.5). Excess cancer risks and HQs for the various exposure scenarios are presented in tables, as appropriate, in the following subsections.

Detection frequency was quite variable across chemicals and media. The risk estimates for each seafood consumption scenario that consisted of a market basket approach were based on the consumption of 7 to 10 seafood categories (a range of categories was used because not all scenarios included the consumption of whole-body benthic fish[39] or geoduck, and mussel samples were not analyzed for all COPCs). For the seafood consumption scenarios, some chemicals were not detected in many of the seafood categories (e.g., crab whole body, mussels) included in the risk calculations. Per the methodology described in Section B.3.3.4, in these instances one-half of the maximum RL was used as the EPC. This resulted in contributions to the risk estimate that sometimes exceeded 50% for a given chemical in seafood categories based entirely on the laboratory RL. The risk estimates for these chemicals are footnoted. The uncertainties related to the use of RL values for non-detected results are discussed in greater detail in the uncertainty analysis (Section B.6).

The adult one-meal-per-month risk estimates may be used to inform the public about the risks that might occur if people were to consume specific seafood type(s) from the EW at a particular frequency. It should be noted that one-meal-per-month risk estimates are not meant to describe the actual behavior of any group that may consume seafood from the EW (e.g., recreational anglers). These risk estimates can be adjusted to account for specific patterns of higher or lower consumption and the consumption of multiple seafood categories. For example, if someone eats two meals per month of EW crab and one meal per month of EW pelagic fish, he or she could multiply the one-meal-per-month crab risk estimate by two and add the sum to the one-meal-per-month pelagic fish risk estimate to approximate the risk associated with his or her own EW seafood consumption. However, it is important to take into account that the one-meal-per-month scenario risks are based on the assumption that EW seafood is consumed at this

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[39] As outlined in Section B.3, the benthic fish consumption category was composed of English sole.

rate for 30 years (see Table B.3-22). For exposure durations less than 30 years, risks might be overestimated. Similarly, if the exposure duration is more than 30 years, risks might be underestimated. As with other seafood consumption risk estimates presented in this section, for the one-meal-per-month scenario it was assumed that the concentrations of chemicals in tissue (i.e., the EPCs summarized in Table B.3-43) do not change over the exposure duration of 30 years.

B.5.3.1 Seafood consumption

B.5.3.1.1 Excess cancer risk estimates

Total upper bound excess cancer estimates for seafood consumption significantly exceeded 1×10^{-6} for each of the scenarios evaluated (Tables B.5-1 through B.5-8), regardless of the PCB summation approach (i.e., the inclusion of total PCBs or PCB TEQ in the sum). The highest total excess cancer risk estimates were for the adult tribal scenario based on Suquamish data (1×10^{-2} [Table B.5-5]), followed by the adult tribal RME scenario based on Tulalip data (1×10^{-3} [Table B.5-1]), the adult API RME scenario (less than or equal to 6×10^{-4} [Table B.5-6]), and the child tribal RME scenario based on Tulalip data (less than or equal to 4×10^{-4} [Table B.5-3]). The lowest risk estimate was for the adult API CT scenario (1×10^{-5} [Table B.5-7]).

Additionally, this section presents the excess cancer risks for the adult one-meal-per-month scenarios (Table B.5-8). Risks associated with these scenarios ranged from 2×10^{-5} to 4×10^{-4} depending on the type of seafood consumed. However, it should be noted that the risks for this scenario are presented for informational purposes only (as discussed in Section B.5.3), and are not used by EPA for risk management decisions.

Table B.5-1. Excess cancer risk estimates for the adult tribal RME seafood consumption scenario based on Tulalip data

Scenario timeframe: Current/future **Medium:** Sediment **Exposure medium:** Fish and shellfish tissue **Receptor population:** Tribal fish and shellfish consumers Receptor age: Adult

Chemical	EPC (mg/kg ww)[a]	Cancer CDI (mg/kg-day)	Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk
Arsenic[b]	Table B.3-42	1.3×10^{-4}	1.5	2×10^{-4}
cPAH TEQ	Table B.3-42	1.4×10^{-5}	7.3	1×10^{-4}
1,4-Dichlorobenzene[c]	Table B.3-42	2.1×10^{-4}	0.0054	1×10^{-6}
Pentachlorophenol[c]	Table B.3-42	1.9×10^{-5}	0.12	2×10^{-6}
Total PCBs	Table B.3-42	5.4×10^{-4}	2	1×10^{-3}
PCB TEQ[d]	Table B.3-42	4.6×10^{-9}	150,000	7×10^{-4}
Total DDTs	Table B.3-42	4.0×10^{-6}	0.34	1×10^{-6}
alpha-BHC[c]	Table B.3-42	5.6×10^{-7}	6.3	4×10^{-6}
beta-BHC[c]	Table B.3-42	5.9×10^{-7}	1.8	1×10^{-6}
Dieldrin[c]	Table B.3-42	5.1×10^{-7}	16	8×10^{-6}
Total chlordane	Table B.3-42	5.2×10^{-6}	0.35	2×10^{-6}
Heptachlor[c]	Table B.3-42	2.5×10^{-7}	4.5	1×10^{-6}
Heptachlor epoxide[c]	Table B.3-42	2.7×10^{-7}	9.1	2×10^{-6}
Mirex[c]	Table B.3-42	2.5×10^{-7}	18	4×10^{-6}
Dioxin/furan TEQ[d]	Table B.3-42	7.3×10^{-10}	150,000	1×10^{-4}
Total TEQ excess cancer risk for dioxins/furans and coplanar PCBs				$\mathbf{8 \times 10^{-4}}$
Total excess cancer risk (excluding PCB TEQ)				$\mathbf{1 \times 10^{-3}}$
Total excess cancer risk (excluding total PCBs)				$\mathbf{1 \times 10^{-3}}$

a An EPC for each seafood category was calculated in the exposure section.

b Arsenic EPCs and risk estimates are based on inorganic arsenic.

c Greater than 50% of the risk associated with this chemical is derived from seafood categories with non-detected concentrations.

d No mussel data were available for this chemical. When the CDI and risk values were calculated, the portion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

BHC – benzene hexachloride
CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent
ww – wet weight

Table B.5-2. Excess cancer risk estimates for the adult tribal CT seafood consumption scenario based on Tulalip data

Scenario timeframe: Current/future **Medium:** Sediment **Exposure medium:** Fish and shellfish tissue **Receptor population:** Tribal fish and shellfish consumers Receptor age: Adult

Chemical	EPC (mg/kg ww)[a]	Cancer CDI (mg/kg-day)	Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk
Arsenic[b]	Table B.3-42	6.7×10^{-6}	1.5	1×10^{-5}
cPAH TEQ	Table B.3-42	5.5×10^{-7}	7.3	4×10^{-6}
1,4-Dichlorobenzene[c]	Table B.3-42	1.3×10^{-5}	0.0054	7×10^{-8}
Pentachlorophenol[c]	Table B.3-42	3.5×10^{-7}	0.12	4×10^{-8}
Total PCBs	Table B.3-42	2.5×10^{-5}	2	5×10^{-5}
PCB TEQ[d]	Table B.3-42	2.7×10^{-10}	150,000	4×10^{-5}
Total DDTs	Table B.3-42	2.5×10^{-7}	0.34	9×10^{-8}
alpha-BHC[c]	Table B.3-42	3.7×10^{-8}	6.3	2×10^{-7}
beta-BHC[c]	Table B.3-42	3.8×10^{-8}	1.8	7×10^{-8}
Dieldrin[c]	Table B.3-42	3.3×10^{-8}	16	5×10^{-7}
Total chlordane	Table B.3-42	2.6×10^{-7}	0.35	9×10^{-8}
Heptachlor[c]	Table B.3-42	1.6×10^{-8}	4.5	7×10^{-8}
Heptachlor epoxide[c]	Table B.3-42	1.8×10^{-8}	9.1	2×10^{-7}
Mirex[c]	Table B.3-42	1.6×10^{-8}	18	3×10^{-7}
Dioxin/furan TEQ[d]	Table B.3-42	4.2×10^{-11}	150,000	6×10^{-6}
Total TEQ excess cancer risk for dioxins/furans and coplanar PCBs				$\mathbf{5 \times 10^{-5}}$
Total excess cancer risk (excluding PCB TEQ)				$\mathbf{7 \times 10^{-5}}$
Total excess cancer risk (excluding total PCBs)				$\mathbf{6 \times 10^{-5}}$

a An EPC for each seafood category was calculated in the exposure section.

b Arsenic EPCs and risk estimates are based on inorganic arsenic.

c Greater than 50% of the risk associated with this chemical is derived from seafood categories with non-detected concentrations.

d No mussel data were available for this chemical. When the CDI and risk values were calculated, the portion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

BHC – benzene hexachloride
CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
CT – central tendency
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
ww – wet weight

Table B.5-3. Excess cancer risk estimates for the child tribal RME seafood consumption scenario based on Tulalip data

Scenario timeframe: Current/future **Medium:** Sediment **Exposure medium:** Fish and shellfish tissue **Receptor population:** Tribal fish and shellfish consumers Receptor age: Child

Chemical	EPC (mg/kg ww)[a]	Cancer CDI (mg/kg-day)	Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk
Arsenic[b]	Table B.3-42	2.4×10^{-5}	1.5	4×10^{-5}
cPAH TEQ[c]	Table B.3-42	2.6×10^{-6}	7.3	1×10^{-4}
1,4-Dichlorobenzene[d]	Table B.3-42	3.9×10^{-5}	0.0054	2×10^{-7}
Pentachlorophenol[d]	Table B.3-42	3.5×10^{-6}	0.12	4×10^{-7}
Total PCBs	Table B.3-42	9.9×10^{-5}	2	2×10^{-4}
PCB TEQ[e]	Table B.3-42	8.4×10^{-10}	150,000	1×10^{-4}
Total DDTs	Table B.3-42	7.3×10^{-7}	0.34	2×10^{-7}
alpha-BHC[d]	Table B.3-42	1.0×10^{-7}	6.3	7×10^{-7}
beta-BHC[d]	Table B.3-42	1.1×10^{-7}	1.8	2×10^{-7}
Dieldrin[d]	Table B.3-42	9.4×10^{-8}	16	1×10^{-6}
Total chlordane	Table B.3-42	9.6×10^{-7}	0.35	3×10^{-7}
Heptachlor[d]	Table B.3-42	4.5×10^{-8}	4.5	2×10^{-7}
Heptachlor epoxide[d]	Table B.3-42	4.9×10^{-8}	9.1	4×10^{-7}
Mirex[d]	Table B.3-42	4.6×10^{-8}	18	8×10^{-7}
Dioxin/furan TEQ[e]	Table B.3-42	1.3×10^{-10}	150,000	2×10^{-5}
Total TEQ excess cancer risk for dioxins/furans and coplanar PCBs				**1×10^{-4}**
Total excess cancer risk (excluding PCB TEQ)				**4×10^{-4}**
Total excess cancer risk (excluding total PCBs)				**3×10^{-4}**

a An EPC for each seafood category was calculated in the exposure section.

b Arsenic EPCs and risk estimates are based on inorganic arsenic.

c Because of the potential for the increased susceptibility of children to carcinogens with mutagenic activity, as described in EPA guidance (2005d), the risk estimate for cPAHs is based on dose adjustments across the 0-to-6-year age range of children. See Section B.5.1.1 for more information.

d Greater than 50% of the risk associated with this chemical is derived from seafood categories with non-detected concentrations.

e No mussel data were available for this chemical. When the CDI and risk values were calculated, the portion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

BHC – benzene hexachloride
CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent
ww – wet weight

Table B.5-4. Excess cancer risk estimates for the child tribal CT seafood consumption scenario based on Tulalip data

Scenario timeframe: Current/future **Medium:** Sediment **Exposure medium:** Fish and shellfish tissue **Receptor population:** Tribal fish and shellfish consumers Receptor age: Child

Chemical	EPC (mg/kg ww)a	Cancer CDI (mg/kg-day)	Cancer Slope Factor (mg/kg-day)-1	Excess Cancer Risk
Arsenic[b]	Table B.3-42	2.9×10^{-6}	1.5	4×10^{-6}
cPAH TEQ[c]	Table B.3-42	2.4×10^{-7}	7.3	9×10^{-6}
1,4-Dichlorobenzene[d]	Table B.3-42	5.5×10^{-6}	0.0054	3×10^{-8}
Pentachlorophenol[d]	Table B.3-42	1.5×10^{-7}	0.12	2×10^{-8}
Total PCBs	Table B.3-42	1.1×10^{-5}	2	2×10^{-5}
PCB TEQ[e]	Table B.3-42	1.2×10^{-10}	150,000	2×10^{-5}
Total DDTs	Table B.3-42	1.1×10^{-7}	0.34	4×10^{-8}
alpha-BHC[d]	Table B.3-42	1.6×10^{-8}	6.3	1×10^{-7}
beta-BHC[d]	Table B.3-42	1.6×10^{-8}	1.8	3×10^{-8}
Dieldrin[d]	Table B.3-42	1.4×10^{-8}	16	2×10^{-7}
Total chlordane	Table B.3-42	1.1×10^{-7}	0.35	4×10^{-8}
Heptachlor[d]	Table B.3-42	7.0×10^{-9}	4.5	3×10^{-8}
Heptachlor epoxide[d]	Table B.3-42	7.6×10^{-9}	9.1	7×10^{-8}
Mirex[d]	Table B.3-42	7.0×10^{-9}	18	1×10^{-7}
Dioxin/furan TEQ[e]	Table B.3-42	1.8×10^{-11}	150,000	3×10^{-6}
Total TEQ excess cancer risk for dioxins/furans and coplanar PCBs				**2×10^{-5}**
Total excess cancer risk (excluding PCB TEQ)				**4×10^{-5}**
Total excess cancer risk (excluding total PCBs)				**4×10^{-5}**

a An EPC for each seafood category was calculated in the exposure section.

b Arsenic EPCs and risk estimates are based on inorganic arsenic.

c Because of the potential for the increased susceptibility of children to carcinogens with mutagenic activity, as described in EPA guidance (2005d), the risk estimate for cPAHs is based on dose adjustments across the 0-to-6-year age range of children. See Section B.5.1.1 for more information.

d Greater than 50% of the risk associated with this chemical is derived from seafood categories with non-detected concentrations.

e No mussel data were available for this chemical. When the CDI and risk values were calculated, the portion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

BHC – benzene hexachloride
CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
CT – central tendency
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
ww – wet weight

Table B.5-5. Excess cancer risk estimates for the adult tribal seafood consumption scenario based on Suquamish data

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Receptor population: Tribal fish and shellfish consumers
Receptor age: Adult

Chemical	EPC (mg/kg ww)[a]	Cancer CDI (mg/kg-day)	Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk
Arsenic[b]	Table B.3-42	1.2×10^{-3}	1.5	2×10^{-3}
cPAH TEQ	Table B.3-42	1.4×10^{-4}	7.3	1×10^{-3}
1,4-Dichlorobenzene[c]	Table B.3-42	1.3×10^{-3}	0.0054	7×10^{-6}
Pentachlorophenol[c]	Table B.3-42	1.4×10^{-4}	0.12	2×10^{-5}
Total PCBs	Table B.3-42	4.3×10^{-3}	2	9×10^{-3}
PCB TEQ[d]	Table B.3-42	4.0×10^{-8}	150,000	6×10^{-3}
Total DDTs	Table B.3-42	3.5×10^{-5}	0.34	1×10^{-5}
alpha-BHC[c]	Table B.3-42	3.8×10^{-6}	6.3	2×10^{-5}
beta-BHC[c]	Table B.3-42	3.9×10^{-6}	1.8	7×10^{-6}
Dieldrin[c]	Table B.3-42	3.2×10^{-6}	16	5×10^{-5}
Total chlordane	Table B.3-42	3.5×10^{-5}	0.35	1×10^{-5}
Heptachlor[c]	Table B.3-42	1.6×10^{-6}	4.5	7×10^{-6}
Heptachlor epoxide[c]	Table B.3-42	1.6×10^{-6}	9.1	1×10^{-5}
Mirex[c]	Table B.3-42	1.7×10^{-6}	18	3×10^{-5}
Dioxin/furan TEQ[d]	Table B.3-42	4.8×10^{-9}	150,000	7×10^{-4}
Total TEQ excess cancer risk for dioxins/furans and coplanar PCBs				$\mathbf{7 \times 10^{-3}}$
Total excess cancer risk (excluding PCB TEQ)				$\mathbf{1 \times 10^{-2}}$
Total excess cancer risk (excluding total PCBs)				$\mathbf{1 \times 10^{-2}}$

a An EPC for each seafood category was calculated in the exposure section.
b Arsenic EPCs and risk estimates are based on inorganic arsenic.
c Greater than 50% of the risk associated with this chemical is derived from seafood categories with non-detected concentrations.
d No mussel data were available for this chemical. When the CDI and risk values were calculated, the portion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

BHC – benzene hexachloride
CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
ww – wet weight

Table B.5-6. Excess cancer risk estimates for the adult API RME seafood consumption scenario

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Receptor population: Asian and Pacific Islander fish and shellfish consumers
Receptor age: Adult

Chemical	EPC (mg/kg ww)[a]	Cancer CDI (mg/kg-day)	Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk
Arsenic[b]	Table B.3-42	5.0×10^{-5}	1.5	8×10^{-5}
cPAH TEQ	Table B.3-42	7.3×10^{-6}	7.3	5×10^{-5}
1,4-Dichlorobenzene[c]	Table B.3-42	6.5×10^{-5}	0.0054	4×10^{-7}
Pentachlorophenol	Table B.3-42	2.3×10^{-6}	0.12	3×10^{-7}
Total PCBs	Table B.3-42	2.0×10^{-4}	2	4×10^{-4}
PCB TEQ[d]	Table B.3-42	2.1×10^{-9}	150,000	3×10^{-4}
Total DDTs	Table B.3-42	1.6×10^{-6}	0.34	6×10^{-7}
alpha-BHC[c]	Table B.3-42	1.5×10^{-7}	6.3	9×10^{-7}
beta-BHC[c]	Table B.3-42	1.5×10^{-7}	1.8	3×10^{-7}
Dieldrin[c]	Table B.3-42	1.5×10^{-7}	16	2×10^{-6}
Total chlordane	Table B.3-42	1.9×10^{-6}	0.35	7×10^{-7}
Heptachlor[c]	Table B.3-42	7.0×10^{-8}	4.5	3×10^{-7}
Heptachlor epoxide[c]	Table B.3-42	8.1×10^{-8}	9.1	7×10^{-7}
Mirex[c]	Table B.3-42	7.6×10^{-8}	18	1×10^{-6}
Dioxin/furan TEQ[d]	Table B.3-42	2.6×10^{-10}	150,000	4×10^{-5}
Total TEQ excess cancer risk for dioxins/furans and coplanar PCBs				$\mathbf{3 \times 10^{-4}}$
Total excess cancer risk (excluding PCB TEQ)				$\mathbf{6 \times 10^{-4}}$
Total excess cancer risk (excluding total PCBs)				$\mathbf{5 \times 10^{-4}}$

a An EPC for each seafood category was calculated in the exposure section.
b Arsenic EPCs and risk estimates are based on inorganic arsenic.
c Greater than 50% of the risk associated with this chemical is derived from seafood categories with non-detected concentrations.
d No mussel data were available for this chemical. When the CDI and risk values were calculated, the portion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

API – Asian and Pacific Islander
BHC – benzene hexachloride
CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent
ww – wet weight

Table B.5-7. Excess cancer risk estimates for the adult API CT seafood consumption scenario

Scenario timeframe: Current/future **Medium:** Sediment **Exposure medium:** Fish and shellfish tissue **Receptor population:** Asian and Pacific Islander fish and shellfish consumers Receptor age: Adult

Chemical	EPC (mg/kg ww)[a]	Cancer CDI (mg/kg-day)	Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk
Arsenic[b]	Table B.3-42	1.2×10^{-6}	1.5	2×10^{-6}
cPAH TEQ	Table B.3-42	1.2×10^{-7}	7.3	9×10^{-7}
1,4-Dichlorobenzene[c]	Table B.3-42	1.5×10^{-6}	0.0054	8×10^{-9}
Pentachlorophenol	Table B.3-42	3.6×10^{-8}	0.12	4×10^{-9}
Total PCBs	Table B.3-42	3.7×10^{-6}	2	7×10^{-6}
PCB TEQ[d]	Table B.3-42	5.0×10^{-11}	150,000	8×10^{-6}
Total DDTs	Table B.3-42	4.0×10^{-8}	0.34	1×10^{-8}
alpha-BHC[c]	Table B.3-42	4.5×10^{-9}	6.3	3×10^{-8}
beta-BHC[c]	Table B.3-42	4.5×10^{-9}	1.8	8×10^{-9}
Dieldrin[c]	Table B.3-42	4.5×10^{-9}	16	7×10^{-8}
Total chlordane	Table B.3-42	4.1×10^{-8}	0.35	1×10^{-8}
Heptachlor[c]	Table B.3-42	2.2×10^{-9}	4.5	1×10^{-8}
Heptachlor epoxide[c]	Table B.3-42	2.5×10^{-9}	9.1	2×10^{-8}
Mirex[c]	Table B.3-42	2.2×10^{-9}	18	4×10^{-8}
Dioxin/furan TEQ[d]	Table B.3-42	6.6×10^{-12}	150,000	1×10^{-6}
Total TEQ excess cancer risk for dioxins/furans and coplanar PCBs				$\mathbf{9 \times 10^{-6}}$
Total excess cancer risk (excluding PCB TEQ)				$\mathbf{1 \times 10^{-5}}$
Total excess cancer risk (excluding total PCBs)				$\mathbf{1 \times 10^{-5}}$

a An EPC for each seafood category was calculated in the exposure section.

b Arsenic EPCs and risk estimates are based on inorganic arsenic.

c Greater than 50% of the risk associated with this chemical is derived from seafood categories with non-detected concentrations.

d No mussel data were available for this chemical. When the CDI and risk values were calculated, the portion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

API – Asian and Pacific Islander
BHC – benzene hexachloride
CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
CT – central tendency
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
ww – wet weight

Table B.5-8. Excess cancer risk estimates associated with the consumption of one-meal-per-month of seafood by adults

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Receptor population: Adult one-meal-per-month fish and shellfish consumers
Receptor age: Adult

Chemical	Seafood Category	EPC (mg/kg ww)	Cancer CDI (mg/kg-day)	Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk
Arsenic[a]	benthic fish[b]	0.0045	2.0×10^{-7}	1.5	3×10^{-7}
	clam	0.22	9.8×10^{-6}	1.5	1×10^{-5}
	crab edible meat	0.036	1.6×10^{-6}	1.5	2×10^{-6}
	pelagic fish, perch	0.027	1.2×10^{-6}	1.5	2×10^{-6}
	pelagic fish, rockfish	0.011	4.9×10^{-7}	1.5	7×10^{-7}
cPAH TEQ	benthic fish	0.00042	1.9×10^{-8}	7.3	1×10^{-7}
	clam	0.027	1.2×10^{-6}	7.3	9×10^{-6}
	crab edible meat	0.0011	4.9×10^{-8}	7.3	4×10^{-7}
	pelagic fish, perch	0.0016	7.2×10^{-8}	7.3	5×10^{-7}
	pelagic fish, rockfish[b]	0.00029	1.3×10^{-8}	7.3	9×10^{-8}
1,4-Dichlorobenzene	benthic fish[b]	0.17	7.6×10^{-6}	0.0054	4×10^{-8}
	clam[b]	0.15	6.7×10^{-6}	0.0054	4×10^{-8}
	crab edible meat[b]	0.17	7.6×10^{-6}	0.0054	4×10^{-8}
	pelagic fish, perch[b]	0.65	2.9×10^{-5}	0.0054	2×10^{-7}
	pelagic fish, rockfish[b]	0.17	7.6×10^{-6}	0.0054	4×10^{-8}
Pentachlorophenol	benthic fish[b]	0.002	9.0×10^{-8}	0.12	1×10^{-8}
	clam	0.0082	3.7×10^{-7}	0.12	4×10^{-8}
	crab edible meat[b]	0.0021	9.4×10^{-8}	0.12	1×10^{-8}
	pelagic fish, perch[b]	0.0055	2.5×10^{-7}	0.12	3×10^{-8}
	pelagic fish, rockfish[b]	0.0021	9.4×10^{-8}	0.12	1×10^{-8}
Total PCBs	benthic fish	2.4	1.1×10^{-4}	2	2×10^{-4}
	clam	0.069	3.1×10^{-6}	2	6×10^{-6}
	crab edible meat	0.16	7.2×10^{-6}	2	1×10^{-5}
	pelagic fish, perch	1.6	7.2×10^{-5}	2	1×10^{-4}
	pelagic fish, rockfish	4	1.8×10^{-4}	2	4×10^{-4}
PCB TEQ	benthic fish	0.000015	6.7×10^{-10}	150,000	1×10^{-4}
	clam	0.00000073	3.3×10^{-11}	150,000	5×10^{-6}
	crab edible meat	0.0000017	7.6×10^{-11}	150,000	1×10^{-5}
	pelagic fish, perch	0.000014	6.3×10^{-10}	150,000	9×10^{-5}
	pelagic fish, rockfish	0.00004	1.8×10^{-9}	150,000	3×10^{-4}
Total DDTs	benthic fish	0.013	5.8×10^{-7}	0.34	2×10^{-7}
	clam	0.0011	4.9×10^{-8}	0.34	2×10^{-8}

Table B.5-8. Excess cancer risk estimates associated with the consumption of one-meal-per-month of seafood by adults (cont.)

Chemical	Seafood Category	EPC (mg/kg ww)	Cancer CDI (mg/kg-day)	Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk
	crab edible meat[b]	0.0011	4.9×10^{-8}	0.34	2×10^{-8}
	pelagic fish, perch	0.011	4.9×10^{-7}	0.34	2×10^{-7}
	pelagic fish, rockfish	0.032	1.4×10^{-6}	0.34	5×10^{-7}
alpha-BHC	benthic fish[b]	0.00044	2.0×10^{-8}	6.3	1×10^{-7}
	clam[b]	0.00044	1.9×10^{-8}	6.3	1×10^{-7}
	crab edible meat[b]	0.00042	1.9×10^{-8}	6.3	1×10^{-7}
	pelagic fish, perch[b]	0.00043	1.9×10^{-8}	6.3	1×10^{-7}
	pelagic fish, rockfish	0.00058	2.6×10^{-8}	6.3	2×10^{-7}
beta-BHC	benthic fish[b]	0.00044	2.0×10^{-8}	1.8	4×10^{-8}
	clam[b]	0.00044	1.9×10^{-8}	1.8	4×10^{-8}
	crab edible meat[b]	0.00042	1.9×10^{-8}	1.8	3×10^{-8}
	pelagic fish, perch[b]	0.00043	1.9×10^{-8}	1.8	3×10^{-8}
	pelagic fish, rockfish[b]	0.00047	2.1×10^{-8}	1.8	4×10^{-8}
Dieldrin	benthic fish	0.0003	1.3×10^{-8}	16	2×10^{-7}
	clam[b]	0.00044	2.0×10^{-8}	16	3×10^{-7}
	crab edible meat[b]	0.00042	1.9×10^{-8}	16	3×10^{-7}
	pelagic fish, perch	0.00076	3.4×10^{-8}	16	5×10^{-7}
	pelagic fish, rockfish	0.00054	2.4×10^{-8}	16	4×10^{-7}
Total chlordane	benthic fish	0.0026	1.2×10^{-7}	0.35	4×10^{-8}
	clam	0.0049	2.2×10^{-7}	0.35	8×10^{-8}
	crab edible meat[b]	0.0011	4.9×10^{-8}	0.35	2×10^{-8}
	pelagic fish, perch	0.003	1.3×10^{-7}	0.35	5×10^{-8}
	pelagic fish, rockfish	0.0083	3.7×10^{-7}	0.35	1×10^{-7}
Heptachlor	benthic fish[b]	0.00022	9.8×10^{-9}	4.5	4×10^{-8}
	clam[b]	0.00022	9.8×10^{-9}	4.5	4×10^{-8}
	crab edible meat[b]	0.00021	9.4×10^{-9}	4.5	4×10^{-8}
	pelagic fish, perch[b]	0.00022	9.6×10^{-9}	4.5	4×10^{-8}
	pelagic fish, rockfish[b]	0.00024	1.1×10^{-8}	4.5	5×10^{-8}
Heptachlor epoxide	benthic fish[b]	0.00022	9.8×10^{-9}	9.1	9×10^{-8}
	clam[b]	0.00022	9.8×10^{-9}	9.1	9×10^{-8}
	crab edible meat[b]	0.00021	9.4×10^{-9}	9.1	9×10^{-8}
	pelagic fish, perch[b]	0.00022	9.6×10^{-9}	9.1	9×10^{-8}
	pelagic fish, rockfish	0.00024	1.1×10^{-8}	9.1	1×10^{-7}
Mirex	benthic fish[b]	0.00022	9.8×10^{-9}	18	2×10^{-7}
	clam[b]	0.00022	9.8×10^{-9}	18	2×10^{-7}
	crab edible meat[b]	0.00021	9.4×10^{-9}	18	2×10^{-7}
	pelagic fish, perch[b]	0.00022	9.6×10^{-9}	18	2×10^{-7}
	pelagic fish, rockfish	0.00044	2.0×10^{-8}	18	4×10^{-7}

Table B.5-8. Excess cancer risk estimates associated with the consumption of one-meal-per-month of seafood by adults (cont.)

Chemical	Seafood Category	EPC (mg/kg ww)	Cancer CDI (mg/kg-day)	Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk
Dioxin/furan TEQ	benthic fish	0.00000079	3.5×10^{-11}	150,000	5×10^{-6}
	clam	0.00000038	1.7×10^{-11}	150,000	3×10^{-6}
	crab edible meat	0.00000049	2.2×10^{-11}	150,000	3×10^{-6}
	pelagic fish, perch	0.0000014	6.3×10^{-11}	150,000	9×10^{-6}
	pelagic fish, rockfish	0.0000028	1.3×10^{-10}	150,000	2×10^{-5}
Total TEQ excess cancer risk for dioxins/furans and coplanar PCBs			benthic fish		**1×10^{-4}**
			clam		**8×10^{-6}**
			crab edible meat		**1×10^{-5}**
			pelagic fish, perch		**1×10^{-4}**
			pelagic fish, rockfish		**3×10^{-4}**
Total excess cancer risk (excluding PCB TEQ)			benthic fish		**2×10^{-4}**
			clam		**3×10^{-5}**
			crab edible meat		**2×10^{-5}**
			pelagic fish, perch		**1×10^{-4}**
			pelagic fish, rockfish		**4×10^{-4}**
Total excess cancer risk (excluding total PCBs)			benthic fish		**1×10^{-4}**
			clam		**3×10^{-5}**
			crab edible meat		**2×10^{-5}**
			pelagic fish, perch		**1×10^{-4}**
			pelagic fish, rockfish		**3×10^{-4}**

Note: The adult one-meal-per-month scenarios are presented for informational purposes only, and are not used by EPA for risk management decisions.

[a] Arsenic EPCs and risk estimates are based on inorganic arsenic.

[b] No detected values in this seafood category. CDI and risk estimate are based on one-half the maximum reporting limit.

BHC – benzene hexachloride
CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
DDT – dichlorodiphenyltrichloroethane
EPA – US Environmental Protection Agency
EPC – exposure point concentration
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
ww – wet weight

Excess cancer risks were greater than 1 × 10^{-6} for 5 to 11 COPCs, depending on the RME seafood consumption scenario. Specifically, the COPCs with excess cancer risks greater than 1 × 10^{-6} for each RME scenario were as follows:

- **Adult tribal RME scenario based on Tulalip data (Table B.5-1) –** arsenic, cPAH TEQ, pentachlorophenol, total PCBs, PCB TEQ, alpha-benzene hexachloride (BHC), dieldrin, total chlordane, heptachlor epoxide, mirex, and dioxin/furan TEQ
- **Child tribal RME scenario based on Tulalip data (Table B.5-3)** – arsenic, cPAH TEQ, total PCBs, PCB TEQ, and dioxin/furan TEQ
- **Adult API RME scenario (Table B.5-6) –** arsenic, cPAH TEQ, total PCBs, PCB TEQ, dieldrin, and dioxin/furan TEQ
- Based on the fact that their excess cancer risks were greater than 1 × 10^{-6}, these 11 COPCs were identified as COCs for the RME seafood consumption scenarios.

In addition, excess cancer risks were greater than 1 × 10^{-6} for the COPCs listed for each of the following non-RME scenarios:

- Adult and child tribal CT scenario based on Tulalip data (Tables B.5-2 and B.5-4) – arsenic, cPAH TEQ, total PCBs, PCB TEQ, and dioxin/furan TEQ
- **Adult tribal scenario based on Suquamish data (Table B.5-5) –** arsenic, cPAH TEQ, 1,4-dichlorobenzene, pentachlorophenol, total PCBs, PCB TEQ, total DDTs, alpha-BHC, beta-BHC, dieldrin, total chlordane, heptachlor, heptachlor epoxide, mirex, and dioxin/furan TEQ
- Adult API CT scenario (Table B.5-7) – arsenic, total PCBs, and PCB TEQ

Total excess cancer risks were equal to 1 × 10^{-3} for the adult tribal RME scenario based on Tulalip data, 4 × 10^{-4} for the child tribal RME scenario based on Tulalip data, and 6 × 10^{-4} for the adult API RME scenario. Of the non-RME scenarios, risks were highest for the adult tribal scenario based on Suquamish data (1 × 10^{-2}). Total excess cancer risk estimates for the CT scenarios (adult tribal scenario based on Tulalip data, child tribal scenario based on Tulalip data, and adult API scenario [Tables B.5-2, B.5-4, and B.5-7, respectively]) were one or more orders of magnitude lower than those for the adult tribal RME scenario based on Tulalip data (Table B.5-1).

For the adult one-meal-per-month scenarios (Table B.5-8),[40] total excess cancer risks estimates were highest for pelagic fish (rockfish) consumption and lowest for crab consumption and ranged from 2 × 10^{-5} to 3 × 10^{-4}. For some or all of the consumption categories evaluated, excess cancer risks were greater than 1 × 10^{-6} for arsenic, cPAH TEQ, total PCBs, PCB TEQ, and dioxin/furan TEQ.

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[40] The adult one-meal-per-month scenarios are presented for informational purposes only, and are not used by EPA for risk management decisions.

For most scenarios, estimates of total excess cancer risk were relatively similar regardless of the PCB summation approach. For all scenarios, the total PCB excess cancer risk estimate was equal to or greater than the PCB TEQ excess cancer risk, but differences were not more than two-fold for all scenarios except the benthic fish adult one-meal-per-month scenario.

Because PCB TEQ and dioxin/furan TEQ were not analyzed in mussels (as described in Section B.3.3.4.1), for the calculation of PCB TEQ and dioxin/furan TEQ risks for consumption scenarios that included a market basket of seafood categories, the consumption of mussels was apportioned to other seafood categories. Because mussel consumption made up only a small percentage of total seafood consumption in these scenarios, this reapportioning had little impact on total risk estimates.

As indicated in the footnotes to Tables B.5-1 to B.5-8, for all scenarios that included a market basket of seafood categories, the excess cancer risk for some COPCs was primarily attributable to non-detected concentrations (for chemical seafood categories in which there were no detected concentrations, one-half the maximum RL was used as the EPC). These COPCs were not major contributors to the total risk estimate; the footnotes to Tables B.5-1 to B.5-8 indicate which COPCs have a majority (over 50%) of risk associated with non-detects. The uncertainty associated with risk estimates for infrequently detected chemicals is discussed in the uncertainty analysis (Section B.6).

B.5.3.1.2 Non-cancer hazard estimates

This section presents non-cancer HQs for the seafood consumption scenarios (Tables B.5-9 to B.5-16). All RME seafood consumption scenarios had at least one chemical with an HQ greater than 1. Non-cancer HQs were greater than 1 for total PCBs for the adult tribal RME scenario based on Tulalip data (Table B.5-9), for cadmium and total PCBs for the child tribal RME scenario based on Tulalip data (Table B.5-11), and for total PCBs for the adult API RME scenario (Table B.5-14). Thus, these two chemicals (cadmium and total PCBs) were identified as COCs based on non-cancer HQs. In addition, HQs were greater than 1 for one chemical (total PCBs) for the adult and child tribal CT scenarios based on Tulalip data (Tables B.5-10 and B.5-12) and for six chemicals (arsenic, cadmium, cobalt, mercury, TBT, and total PCBs) for the adult tribal scenario based on Suquamish data (Table B.5-13). No COPCs had HQs greater than 1 for the adult API CT scenario (Table B.5-15).

In addition to the calculation of non-cancer HQs for all individual COPCs, HIs (sums of HQs for chemicals with similar effects endpoints) were also calculated and are presented in this section. Effect-specific HIs were calculated for hematological, immunological, kidney, liver, neurological, endocrine, integumentary, digestive system, and developmental endpoints, as described in Section B.4 and B.5.1.2. The chemicals associated with each endpoint are identified in the footnotes of Tables B.5-9 to B.5-16.

The immunological, integumentary, and neurological HIs exceeded 1 for all scenarios except the adult API CT scenario; and the developmental HI exceeded 1 for all scenarios except the adult tribal CT scenario based on Tulalip data and the adult API CT scenario, primarily because of the contribution of PCBs. All four of these HIs were greater than 20 for the three RME scenarios: the adult tribal RME scenario based on Tulalip data (equal to 27 or 28), the child tribal RME scenario based on Tulalip data (equal to 59), and the adult API RME scenario (equal to 24 or 25). For the adult tribal scenario based on Suquamish data, these four HIs were also well above 20, equal to 217 to 219. In addition, the HIs for the hematological, kidney, endocrine, and digestive system endpoints were greater than 1 for the adult tribal scenario based on Suquamish data (Table B.5-13), as was the HI for the kidney endpoint for the child tribal RME scenario based on Tulalip data (Table B.5-11). For all scenarios, the majority of the total immunological, integumentary, neurological, and developmental HIs (greater than or equal to 97%) was attributable to total PCBs.

For the adult one-meal-per-month scenarios, the benthic fish, perch, and rockfish scenarios had HQs greater than 1 for one COPC: total PCBs (Table B.5-16). No COPCs had HQs greater than 1 for the adult one-meal-per-month clam or crab scenarios (Table B.5-16). Immunological, integumentary, neurological, and developmental HIs for the adult one-meal-per-month scenarios ranged from less than 1 to 21, depending on the seafood category and the RfD on which the HQ was based (Table B.5-16). No other HIs were greater than 1 for any of the adult one-meal-per-month scenarios.

Table B.5-9. Non-cancer hazard estimates for the adult tribal RME seafood consumption scenario based on Tulalip data

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Receptor population: Tribal fish and shellfish consumers
Receptor age: Adult

Chemical	EPC (mg/kg ww)[a]	Non-Cancer CDI (mg/kg-day)	Reference Dose (mg/kg-day)	Hazard Quotient
Antimony	Table B.3-42	1.6×10^{-5}	0.0004	0.04
Arsenic[b]	Table B.3-42	1.3×10^{-4}	0.0003	0.4
Cadmium	Table B.3-42	7.4×10^{-4}	0.001	0.7
Chromium	Table B.3-42	4.7×10^{-4}	0.003	0.2
Cobalt	Table B.3-42	1.8×10^{-4}	0.0003	0.6
Copper	Table B.3-42	1.3×10^{-2}	0.04	0.3
Mercury	Table B.3-42	5.7×10^{-5}	0.0001	0.6
Molybdenum	Table B.3-42	6.0×10^{-4}	0.005	0.1
Selenium	Table B.3-42	8.3×10^{-4}	0.005	0.2
Vanadium	Table B.3-42	3.8×10^{-4}	0.009	0.04
Zinc	Table B.3-42	3.8×10^{-2}	0.3	0.1
Dibutyltin as ion[c]	Table B.3-42	7.7×10^{-6}	0.0003	0.03
Tributyltin as ion	Table B.3-42	4.7×10^{-5}	0.00015	0.3
1,4-Dichlorobenzene[c]	Table B.3-42	2.1×10^{-4}	0.07	0.003
Pentachlorophenol[c]	Table B.3-42	1.9×10^{-5}	0.03	0.0006
Total PCBs	Table B.3-42	5.4×10^{-4}	0.00002	27[d]
			0.00007	8[e]
Total DDTs	Table B.3-42	4.0×10^{-6}	0.0005	0.008
alpha-BHC[c]	Table B.3-42	5.6×10^{-7}	0.008	0.00007
Dieldrin[c]	Table B.3-42	5.1×10^{-7}	0.00005	0.01
Total chlordane	Table B.3-42	5.2×10^{-6}	0.0005	0.01
Heptachlor[c]	Table B.3-42	2.5×10^{-7}	0.0005	0.0005
Heptachlor epoxide[c]	Table B.3-42	2.7×10^{-7}	0.000013	0.02
Mirex[c]	Table B.3-42	2.5×10^{-7}	0.0002	0.001
Hazard indices by effect:				
Hazard index for hematological endpoint[f]				**0.3**
Hazard index for immunological endpoint[f]				**27**
Hazard index for kidney endpoint[h]				**0.8**
Hazard index for liver endpoint[i]				**0.06**
Hazard index for neurological endpoint[j]				**28**
Hazard index for endocrine endpoint[k]				**0.6**
Hazard index for integumentary endpoint[l]				**28**
Hazard index for digestive system endpoint[m]				**0.5**
Hazard index for developmental endpoint[n]				**9**

Table B.5-9. Non-cancer hazard estimates for the adult tribal RME seafood consumption scenario based on Tulalip data (cont.)

[a] An EPC for each seafood category was calculated in the exposure section.
[b] Arsenic EPCs and risk estimates are based on inorganic arsenic.
[c] Greater than 50% of the HQ associated with this chemical is derived from seafood categories with non-detected concentrations.
[d] HQ used for the calculation of the immunological, integumentary, and neurological endpoint hazard indices (Table B.4-1).
[e] HQ used for the calculation of the developmental endpoint hazard index (Table B.4-1).
[f] Hematological endpoint includes the following chemicals: antimony, selenium, and zinc.
[g] Immunological endpoint includes the following chemicals: dibutyltin, total PCBs, and TBT.
[h] Kidney endpoint includes the following chemicals: cadmium, molybdenum, and pentachlorophenol.
[i] Liver endpoint includes the following chemicals: 1,4-dichlorobenzene, alpha-BHC, total chlordane, total DDTs, dieldrin, heptachlor, heptachlor epoxide, mirex, and pentachlorophenol.
[j] Neurological endpoint includes the following chemicals: mercury, total PCBs, and selenium. Neurological effects associated with exposure to lead are discussed in Section B.5.4.
[k] Endocrine endpoint includes the following chemicals: antimony and cobalt.
[l] Integumentary endpoint includes the following chemicals: arsenic, total PCBs, selenium, and vanadium.
[m] Digestive system endpoint includes the following chemicals: chromium and copper.
[n] Developmental endpoint includes the following chemicals: mercury and total PCBs.

BHC – benzene hexachloride
CDI – chronic daily intake
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
HQ – hazard quotient
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent
ww – wet weight

Table B.5-10. Non-cancer hazard estimates for the adult tribal CT seafood consumption scenario based on Tulalip data

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Receptor population: Tribal fish and shellfish consumers
Receptor age: Adult

Chemical	EPC (mg/kg ww)[a]	Non-Cancer CDI (mg/kg-day)	Reference Dose (mg/kg-day)	Hazard Quotient
Antimony	Table B.3-42	1.8×10^{-6}	0.0004	0.005
Arsenic[b]	Table B.3-42	1.6×10^{-5}	0.0003	0.05
Cadmium	Table B.3-42	7.6×10^{-5}	0.001	0.08
Chromium	Table B.3-42	5.7×10^{-5}	0.003	0.02
Cobalt	Table B.3-42	2.2×10^{-5}	0.0003	0.07
Copper	Table B.3-42	1.7×10^{-3}	0.04	0.04
Mercury	Table B.3-42	7.3×10^{-6}	0.0001	0.07
Molybdenum	Table B.3-42	8.3×10^{-5}	0.005	0.02
Selenium	Table B.3-42	1.1×10^{-4}	0.005	0.02
Vanadium	Table B.3-42	4.0×10^{-5}	0.009	0.004
Zinc	Table B.3-42	5.3×10^{-3}	0.3	0.02
Dibutyltin as ion[c]	Table B.3-42	9.9×10^{-7}	0.0003	0.003
Tributyltin as ion	Table B.3-42	4.8×10^{-6}	0.00015	0.03
1,4-Dichlorobenzene[c]	Table B.3-42	3.0×10^{-5}	0.07	0.0004
Pentachlorophenol[c]	Table B.3-42	8.3×10^{-7}	0.03	0.00003
Total PCBs	Table B.3-42	5.9×10^{-5}	0.00002	3[d]
			0.00007	0.8[e]
Total DDTs	Table B.3-42	5.9×10^{-7}	0.0005	0.001
alpha-BHC[c]	Table B.3-42	8.6×10^{-8}	0.008	0.00001
Dieldrin[c]	Table B.3-42	7.7×10^{-8}	0.00005	0.002
Total chlordane	Table B.3-42	6.1×10^{-7}	0.0005	0.001
Heptachlor[c]	Table B.3-42	3.8×10^{-8}	0.0005	0.00008
Heptachlor epoxide[c]	Table B.3-42	4.1×10^{-8}	0.000013	0.003
Mirex[c]	Table B.3-42	3.8×10^{-8}	0.0002	0.0002
Hazard indices by effect:				
Hazard index for hematological endpoint[f]				**0.05**
Hazard index for immunological endpoint[g]				**3**
Hazard index for kidney endpoint[h]				**0.1**
Hazard index for liver endpoint[i]				**0.008**
Hazard index for neurological endpoint[j]				**3**
Hazard index for endocrine endpoint[k]				**0.08**
Hazard index for integumentary endpoint[l]				**3**
Hazard index for digestive system endpoint[m]				**0.06**
Hazard index for developmental endpoint[n]				**0.9**

Table B.5-10. Non-cancer hazard estimates for the adult tribal CT seafood consumption scenario based on Tulalip data (cont.)

[a] An EPC for each seafood category was calculated in the exposure section.

[b] Arsenic EPCs and risk estimates are based on inorganic arsenic.

[c] Greater than 50% of the HQ associated with this chemical is derived from seafood categories with non-detected concentrations.

[d] HQ used for the calculation of the immunological, integumentary, and neurological endpoint hazard indices (Table B.4-1).

[e] HQ used for the calculation of the developmental endpoint hazard index (Table B.4-1).

[f] Hematological endpoint includes the following chemicals: antimony, selenium, and zinc.

[g] Immunological endpoint includes the following chemicals: dibutyltin, total PCBs, and TBT.

[h] Kidney endpoint includes the following chemicals: cadmium, molybdenum, and pentachlorophenol.

[i] Liver endpoint includes the following chemicals: 1,4-dichlorobenzene, alpha-BHC, total chlordane, total DDTs, dieldrin, heptachlor, heptachlor epoxide, mirex, and pentachlorophenol.

[j] Neurological endpoint includes the following chemicals: mercury, total PCBs, and selenium. Neurological effects associated with exposure to lead are discussed in Section B.5.4.

[k] Endocrine endpoint includes the following chemicals: antimony and cobalt.

[l] Integumentary endpoint includes the following chemicals: arsenic, total PCBs, selenium, and vanadium.

[m] Digestive system endpoint includes the following chemicals: chromium and copper.

[n] Developmental endpoint includes the following chemicals: mercury and total PCBs.

BHC – benzene hexachloride
CDI – chronic daily intake
CT – central tendency
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
HQ – hazard quotient
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
ww – wet weight

Table B.5-11. Non-cancer hazard estimates for the child tribal RME seafood consumption scenario based on Tulalip data

Scenario timeframe: Current/future **Medium:** Sediment **Exposure medium:** Fish and shellfish tissue **Receptor population:** Tribal fish and shellfish consumers Receptor age: Child

Chemical	EPC (mg/kg ww)[a]	Non-Cancer CDI (mg/kg-day)	Reference Dose (mg/kg-day)	Hazard Quotient
Antimony	Table B.3-42	3.4×10^{-5}	0.0004	0.09
Arsenic[b]	Table B.3-42	2.8×10^{-4}	0.0003	0.9
Cadmium	Table B.3-42	1.6×10^{-3}	0.001	2
Chromium	Table B.3-42	1.0×10^{-3}	0.003	0.3
Cobalt	Table B.3-42	3.9×10^{-4}	0.0003	1
Copper	Table B.3-42	2.8×10^{-2}	0.04	0.7
Mercury	Table B.3-42	1.2×10^{-4}	0.0001	1
Molybdenum	Table B.3-42	1.3×10^{-3}	0.005	0.3
Selenium	Table B.3-42	1.8×10^{-3}	0.005	0.4
Vanadium	Table B.3-42	8.2×10^{-4}	0.009	0.09
Zinc	Table B.3-42	8.1×10^{-2}	0.3	0.3
Dibutyltin as ion[c]	Table B.3-42	1.7×10^{-5}	0.0003	0.06
Tributyltin as ion	Table B.3-42	1.0×10^{-4}	0.00015	0.7
1,4-Dichlorobenzene[c]	Table B.3-42	4.6×10^{-4}	0.07	0.007
Pentachlorophenol[c]	Table B.3-42	4.1×10^{-5}	0.03	0.001
Total PCBs	Table B.3-42	1.2×10^{-3}	0.00002	58[d]
			0.00007	17[e]
Total DDTs	Table B.3-42	8.5×10^{-6}	0.0005	0.02
alpha-BHC[c]	Table B.3-42	1.2×10^{-6}	0.008	0.0002
Dieldrin[c]	Table B.3-42	1.1×10^{-6}	0.00005	0.02
Total chlordane	Table B.3-42	1.1×10^{-5}	0.0005	0.02
Heptachlor[c]	Table B.3-42	5.3×10^{-7}	0.0005	0.001
Heptachlor epoxide[c]	Table B.3-42	5.7×10^{-7}	0.000013	0.04
Mirex[c]	Table B.3-42	5.3×10^{-7}	0.0002	0.003
Hazard indices by effect:				
Hazard index for hematological endpoint[f]				**0.8**
Hazard index for immunological endpoint[g]				**59**
Hazard index for kidney endpoint[h]				**2**
Hazard index for liver endpoint[i]				**0.1**
Hazard index for neurological endpoint[j]				**59**
Hazard index for endocrine endpoint[k]				**1**
Hazard index for integumentary endpoint[l]				**59**
Hazard index for digestive system endpoint[m]				**1**
Hazard index for developmental endpoint[n]				**18**

Table B.5-11. Non-cancer hazard estimates for the child tribal RME seafood consumption scenario based on Tulalip data (cont.)

[a] An EPC for each seafood category was calculated in the exposure section.

[b] Arsenic EPCs and risk estimates are based on inorganic arsenic.

[c] Greater than 50% of the HQ associated with this chemical is derived from seafood categories with non-detected concentrations.

[d] HQ used for the calculation of the immunological, integumentary, and neurological endpoint hazard indices (Table B.4-1).

[e] HQ used for the calculation of the developmental endpoint hazard index (Table B.4-1).

[f] Hematological endpoint includes the following chemicals: antimony, selenium, and zinc.

[g] Immunological endpoint includes the following chemicals: dibutyltin, total PCBs, and TBT.

[h] Kidney endpoint includes the following chemicals: cadmium, molybdenum, and pentachlorophenol.

[i] Liver endpoint includes the following chemicals: 1,4-dichlorobenzene, alpha-BHC, total chlordane, total DDTs, dieldrin, heptachlor, heptachlor epoxide, mirex, and pentachlorophenol.

[j] Neurological endpoint includes the following chemicals: mercury, total PCBs and selenium. Neurological effects associated with exposure to lead are discussed in Section B.5.4.

[k] Endocrine endpoint includes the following chemicals: antimony, and cobalt.

[l] Integumentary endpoint includes the following chemicals: arsenic, total PCBs, selenium, and vanadium.

[m] Digestive system endpoint includes the following chemicals: chromium and copper.

[n] Developmental endpoint includes the following chemicals: mercury and total PCBs.

BHC – benzene hexachloride
CDI – chronic daily intake
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
HQ – hazard quotient
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent
ww – wet weight

Table B.5-12. Non-cancer hazard estimates for the child tribal CT seafood consumption scenario based on Tulalip data

Scenario timeframe: Current/future **Medium:** Sediment **Exposure medium:** Fish and shellfish tissue **Receptor population:** Tribal fish and shellfish consumers Receptor age: Child

Chemical	EPC (mg/kg ww)[a]	Non-Cancer CDI (mg/kg-day)	Reference Dose (mg/kg-day)	Hazard Quotient
Antimony	Table B.3-42	3.9×10^{-6}	0.0004	0.01
Arsenic[b]	Table B.3-42	3.3×10^{-5}	0.0003	0.1
Cadmium	Table B.3-42	1.6×10^{-4}	0.001	0.2
Chromium	Table B.3-42	1.2×10^{-4}	0.003	0.04
Cobalt	Table B.3-42	4.8×10^{-5}	0.0003	0.2
Copper	Table B.3-42	3.7×10^{-3}	0.04	0.09
Mercury	Table B.3-42	1.6×10^{-5}	0.0001	0.2
Molybdenum	Table B.3-42	1.8×10^{-4}	0.005	0.04
Selenium	Table B.3-42	2.5×10^{-4}	0.005	0.05
Vanadium	Table B.3-42	8.7×10^{-5}	0.009	0.01
Zinc	Table B.3-42	1.1×10^{-2}	0.3	0.04
Dibutyltin as ion[c]	Table B.3-42	2.1×10^{-6}	0.0003	0.007
Tributyltin as ion	Table B.3-42	1.0×10^{-5}	0.00015	0.07
1,4-Dichlorobenzene[c]	Table B.3-42	6.4×10^{-5}	0.07	0.0009
Pentachlorophenol[c]	Table B.3-42	1.8×10^{-6}	0.03	0.00006
Total PCBs	Table B.3-42	1.3×10^{-4}	0.00002	6[d]
			0.00007	2[e]
Total DDTs	Table B.3-42	1.3×10^{-6}	0.0005	0.003
alpha-BHC[c]	Table B.3-42	1.8×10^{-7}	0.008	0.00002
Dieldrin[c]	Table B.3-42	1.7×10^{-7}	0.00005	0.003
Total chlordane	Table B.3-42	1.3×10^{-6}	0.0005	0.003
Heptachlor[c]	Table B.3-42	8.1×10^{-8}	0.0005	0.0002
Heptachlor epoxide[c]	Table B.3-42	8.8×10^{-8}	0.000013	0.007
Mirex[c]	Table B.3-42	8.2×10^{-8}	0.0002	0.0004
Hazard indices by effect:				
Hazard index for hematological endpoint[f]				**0.1**
Hazard index for immunological endpoint[g]				**6**
Hazard index for kidney endpoint[h]				**0.2**
Hazard index for liver endpoint[i]				**0.02**
Hazard index for neurological endpoint[j]				**6**
Hazard index for endocrine endpoint[k]				**0.2**
Hazard index for integumentary endpoint[l]				**6**
Hazard index for digestive system endpoint[m]				**0.1**
Hazard index for developmental endpoint[n]				**2**

Table B.5-12. Non-cancer hazard estimates for the child tribal CT seafood consumption scenario based on Tulalip data (cont.)

[a] An EPC for each seafood category was calculated in the exposure section.

[b] Arsenic EPCs and risk estimates are based on inorganic arsenic.

[c] Greater than 50% of the HQ associated with this chemical is derived from seafood categories with non-detected concentrations.

[d] HQ used for the calculation of the immunological, integumentary, and neurological endpoint hazard indices (Table B.4-1).

[e] HQ used for the calculation of the developmental endpoint hazard index (Table B.4-1).

[f] Hematological endpoint includes the following chemicals: antimony, selenium, and zinc.

[g] Immunological endpoint includes the following chemicals: dibutyltin, total PCBs, and TBT.

[h] Kidney endpoint includes the following chemicals: cadmium, molybdenum, and pentachlorophenol.

[i] Liver endpoint includes the following chemicals: 1,4-dichlorobenzene, alpha-BHC, total chlordane, total DDTs, dieldrin, heptachlor, heptachlor epoxide, mirex, and pentachlorophenol.

[j] Neurological endpoint includes the following chemicals: mercury, total PCBs, and selenium. Neurological effects associated with exposure to lead are discussed in Section B.5.4.

[k] Endocrine endpoint includes the following chemicals: antimony and cobalt.

[l] Integumentary endpoint includes the following chemicals: arsenic, total PCBs, selenium, and vanadium.

[m] Digestive system endpoint includes the following chemicals: chromium and copper.

[n] Developmental endpoint includes the following chemicals: mercury and total PCBs.

BHC – benzene hexachloride
CDI – chronic daily intake
CT – central tendency
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
HQ – hazard quotient
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
ww – wet weight

Table B.5-13. Non-cancer hazard estimates for the adult tribal seafood consumption scenario based on Suquamish data

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Receptor population: Tribal fish and shellfish consumers
Receptor age: Adult

Chemical	EPC (mg/kg ww)[a]	Non-Cancer CDI (mg/kg-day)	Reference Dose (mg/kg-day)	Hazard Quotient
Antimony	Table B.3-42	1.4×10^{-4}	0.0004	0.4
Arsenic[b]	Table B.3-42	1.2×10^{-3}	0.0003	4
Cadmium	Table B.3-42	1.7×10^{-3}	0.001	2
Chromium	Table B.3-42	4.2×10^{-3}	0.003	1
Cobalt	Table B.3-42	1.2×10^{-3}	0.0003	4
Copper	Table B.3-42	5.8×10^{-2}	0.04	1
Mercury	Table B.3-42	3.3×10^{-4}	0.0001	3
Molybdenum	Table B.3-42	3.9×10^{-3}	0.005	0.8
Selenium	Table B.3-42	3.9×10^{-3}	0.005	0.8
Vanadium	Table B.3-42	3.2×10^{-3}	0.009	0.4
Zinc	Table B.3-42	1.6×10^{-1}	0.3	0.5
Dibutyltin as ion[c]	Table B.3-42	5.8×10^{-5}	0.0003	0.2
Tributyltin as ion	Table B.3-42	5.3×10^{-4}	0.00015	4
1,4-Dichlorobenzene[c]	Table B.3-42	1.3×10^{-3}	0.07	0.02
Pentachlorophenol[c]	Table B.3-42	1.4×10^{-4}	0.03	0.005
Total PCBs	Table B.3-42	4.3×10^{-3}	0.00002	214[d]
			0.00007	61[e]
Total DDTs	Table B.3-42	3.5×10^{-5}	0.0005	0.07
alpha-BHC[c]	Table B.3-42	3.8×10^{-6}	0.008	0.0005
Dieldrin[c]	Table B.3-42	3.2×10^{-6}	0.00005	0.06
Total chlordane	Table B.3-42	3.5×10^{-5}	0.0005	0.07
Heptachlor[c]	Table B.3-42	1.6×10^{-6}	0.0005	0.003
Heptachlor epoxide[c]	Table B.3-42	1.6×10^{-6}	0.000013	0.1
Mirex[c]	Table B.3-42	1.7×10^{-6}	0.0002	0.01
Hazard indices by effect:				
Hazard index for hematological endpoint[f]				**2**
Hazard index for immunological endpoint[g]				**218**
Hazard index for kidney endpoint[h]				**3**
Hazard index for liver endpoint[i]				**0.3**
Hazard index for neurological endpoint[j]				**218**
Hazard index for endocrine endpoint[k]				**4**
Hazard index for integumentary endpoint[l]				**219**
Hazard index for digestive system endpoint[m]				**2**
Hazard index for developmental endpoint[n]				**64**

Table B.5-13. Non-cancer hazard estimates for the adult tribal seafood consumption scenario based on Suquamish data (cont.)

[a] An EPC for each seafood category was calculated in the exposure section.

[b] Arsenic EPCs and risk estimates are based on inorganic arsenic.

[c] Greater than 50% of the HQ associated with this chemical is derived from seafood categories with non-detected concentrations.

[d] HQ used for the calculation of the immunological, integumentary, and neurological endpoint hazard indices (Table B.4-1).

[e] HQ used for the calculation of the developmental endpoint hazard index (Table B.4-1).

[f] Hematological endpoint includes the following chemicals: antimony, selenium, and zinc.

[g] Immunological endpoint includes the following chemicals: dibutyltin, total PCBs, and TBT.

[h] Kidney endpoint includes the following chemicals: cadmium, molybdenum, and pentachlorophenol.

[i] Liver endpoint includes the following chemicals: 1,4-dichlorobenzene, alpha-BHC, total chlordane, total DDTs, dieldrin, heptachlor, heptachlor epoxide, mirex, and pentachlorophenol.

[j] Neurological endpoint includes the following chemicals: mercury, total PCBs, and selenium. Neurological effects associated with exposure to lead are discussed in Section B.5.4.

[k] Endocrine endpoint includes the following chemicals: antimony and cobalt.

[l] Integumentary endpoint includes the following chemicals: arsenic, total PCBs, selenium, and vanadium.

[m] Digestive system endpoint includes the following chemicals: chromium and copper.

[n] Developmental endpoint includes the following chemicals: mercury and total PCBs.

BHC – benzene hexachloride
CDI – chronic daily intake
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
HQ – hazard quotient
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
ww – wet weight

Table B.5-14. Non-cancer hazard estimates for the adult API RME seafood consumption scenario

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Receptor population: Asian and Pacific Islander fish and shellfish consumers
Receptor age: Adult

Chemical	EPC (mg/kg ww)[a]	Non-Cancer CDI (mg/kg-day)	Reference Dose (mg/kg-day)	Hazard Quotient
Antimony	Table B.3-42	1.4×10^{-5}	0.0004	0.04
Arsenic[b]	Table B.3-42	1.2×10^{-4}	0.0003	0.4
Cadmium	Table B.3-42	4.5×10^{-4}	0.001	0.4
Chromium	Table B.3-42	4.1×10^{-4}	0.003	0.1
Cobalt	Table B.3-42	1.4×10^{-4}	0.0003	0.5
Copper	Table B.3-42	7.4×10^{-3}	0.04	0.2
Mercury	Table B.3-42	4.1×10^{-5}	0.0001	0.4
Molybdenum	Table B.3-42	3.7×10^{-4}	0.005	0.07
Selenium	Table B.3-42	5.0×10^{-4}	0.005	0.1
Vanadium	Table B.3-42	3.3×10^{-4}	0.009	0.04
Zinc	Table B.3-42	2.2×10^{-2}	0.3	0.07
Dibutyltin as ion	Table B.3-42	7.8×10^{-6}	0.0003	0.03
Tributyltin as ion	Table B.3-42	5.7×10^{-5}	0.00015	0.4
1,4-Dichlorobenzene[c]	Table B.3-42	1.5×10^{-4}	0.07	0.002
Pentachlorophenol	Table B.3-42	5.4×10^{-6}	0.03	0.0002
Total PCBs	Table B.3-42	4.8×10^{-4}	0.00002	24[c]
			0.00007	7[d]
Total DDTs	Table B.3-42	3.8×10^{-6}	0.0005	0.008
alpha-BHC[c]	Table B.3-42	3.5×10^{-7}	0.008	0.00004
Dieldrin[c]	Table B.3-42	3.5×10^{-7}	0.00005	0.007
Total chlordane	Table B.3-42	4.5×10^{-6}	0.0005	0.009
Heptachlor[c]	Table B.3-42	1.6×10^{-7}	0.0005	0.0003
Heptachlor epoxide[c]	Table B.3-42	1.9×10^{-7}	0.000013	0.01
Mirex[c]	Table B.3-42	1.8×10^{-7}	0.0002	0.0009
Hazard indices by effect:				
Hazard index for hematological endpoint[f]				**0.2**
Hazard index for immunological endpoint[g]				**24**
Hazard index for kidney endpoint[h]				**0.5**
Hazard index for liver endpoint[i]				**0.04**
Hazard index for neurological endpoint[j]				**25**
Hazard index for endocrine endpoint[k]				**0.5**
Hazard index for integumentary endpoint[l]				**25**
Hazard index for digestive system endpoint[m]				**0.3**
Hazard index for developmental endpoint[n]				**7**

Table B.5-14. Non-cancer hazard estimates for the adult API RME seafood consumption scenario (cont.)

[a] An EPC for each seafood category was calculated in the exposure section.

[b] Arsenic EPCs and risk estimates are based on inorganic arsenic.

[c] Greater than 50% of the HQ associated with this chemical is derived from seafood categories with non-detected concentrations.

[d] HQ used for the calculation of the immunological, integumentary, and neurological endpoint hazard indices (Table B.4-1).

[e] HQ used for the calculation of the developmental endpoint hazard index (Table B.4-1).

[f] Hematological endpoint includes the following chemicals: antimony, selenium, and zinc.

[g] Immunological endpoint includes the following chemicals: dibutyltin, total PCBs, and TBT.

[h] Kidney endpoint includes the following chemicals: cadmium, molybdenum, and pentachlorophenol.

[i] Liver endpoint includes the following chemicals: 1,4-dichlorobenzene, alpha-BHC, total chlordane, total DDTs, dieldrin, heptachlor, heptachlor epoxide, mirex, and pentachlorophenol.

[j] Neurological endpoint includes the following chemicals: mercury, total PCBs, and selenium. Neurological effects associated with exposure to lead are discussed in Section B.5.4.

[k] Endocrine endpoint includes the following chemicals: antimony and cobalt.

[l] Integumentary endpoint includes the following chemicals: arsenic, total PCBs, selenium, and vanadium.

[m] Digestive system endpoint includes the following chemicals: chromium and copper.

[n] Developmental endpoint includes the following chemicals: mercury and total PCBs.

API – Asian and Pacific Islander
BHC – benzene hexachloride
CDI – chronic daily intake
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
HQ – hazard quotient
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent
ww – wet weight

Table B.5-15. Non-cancer hazard estimates for the adult API CT seafood consumption scenario

Scenario timeframe: Current/future **Medium:** Sediment **Exposure medium:** Fish and shellfish tissue **Receptor population:** Asian and Pacific Islander fish and shellfish consumers Receptor age: Adult

Chemical	EPC (mg/kg ww)[a]	Non-Cancer CDI (mg/kg-day)	Reference Dose (mg/kg-day)	Hazard Quotient
Antimony	Table B.3-42	1.1×10^{-6}	0.0004	0.003
Arsenic[b]	Table B.3-42	9.5×10^{-6}	0.0003	0.03
Cadmium	Table B.3-42	3.0×10^{-5}	0.001	0.03
Chromium	Table B.3-42	3.5×10^{-5}	0.003	0.01
Cobalt	Table B.3-42	1.2×10^{-5}	0.0003	0.04
Copper	Table B.3-42	6.7×10^{-4}	0.04	0.02
Mercury	Table B.3-42	3.7×10^{-6}	0.0001	0.04
Molybdenum	Table B.3-42	3.6×10^{-5}	0.005	0.007
Selenium	Table B.3-42	4.6×10^{-5}	0.005	0.009
Vanadium	Table B.3-42	2.6×10^{-5}	0.009	0.003
Zinc	Table B.3-42	2.1×10^{-3}	0.3	0.007
Dibutyltin as ion	Table B.3-42	5.1×10^{-7}	0.0003	0.002
Tributyltin as ion	Table B.3-42	3.8×10^{-6}	0.00015	0.03
1,4-Dichlorobenzene[c]	Table B.3-42	1.2×10^{-5}	0.07	0.0002
Pentachlorophenol	Table B.3-42	2.8×10^{-7}	0.03	0.000009
Total PCBs	Table B.3-42	2.9×10^{-5}	0.00002	1[c]
			0.00007	0.4[d]
Total DDTs	Table B.3-42	3.1×10^{-7}	0.0005	0.0006
alpha-BHC[c]	Table B.3-42	3.5×10^{-8}	0.008	0.000004
Dieldrin[c]	Table B.3-42	3.5×10^{-8}	0.00005	0.0007
Total chlordane	Table B.3-42	3.2×10^{-7}	0.0005	0.0006
Heptachlor[c]	Table B.3-42	1.7×10^{-8}	0.0005	0.00003
Heptachlor epoxide[c]	Table B.3-42	1.9×10^{-8}	0.000013	0.001
Mirex[c]	Table B.3-42	1.7×10^{-8}	0.0002	0.00009
Hazard indices by effect:				
Hazard index for hematological endpoint[f]				**0.02**
Hazard index for immunological endpoint[g]				**1**
Hazard index for kidney endpoint[h]				**0.04**
Hazard index for liver endpoint[i]				**0.003**
Hazard index for neurological endpoint[j]				**1**
Hazard index for endocrine endpoint[k]				**0.04**
Hazard index for integumentary endpoint[l]				**1**
Hazard index for digestive system endpoint[m]				**0.03**
Hazard index for developmental endpoint[n]				**0.4**

Table B.5-15. Non-cancer hazard estimates for the adult API CT seafood consumption scenario (cont.)

[a] An EPC for each seafood category was calculated in the exposure section.
[b] Arsenic EPCs and risk estimates are based on inorganic arsenic.
[c] Greater than 50% of the HQ associated with this chemical is derived from seafood categories with non-detected concentrations.
[d] HQ used for the calculation of the immunological, integumentary, and neurological endpoint hazard indices (Table B.4-1).
[e] HQ used for the calculation of the developmental endpoint hazard index (Table B.4-1).
[f] Hematological endpoint includes the following chemicals: antimony, selenium, and zinc.
[g] Immunological endpoint includes the following chemicals: dibutyltin, total PCBs, and TBT.
[h] Kidney endpoint includes the following chemicals: cadmium, molybdenum, and pentachlorophenol.
[i] Liver endpoint includes the following chemicals: 1,4-dichlorobenzene, alpha-BHC, total chlordane, total DDTs, dieldrin, heptachlor, heptachlor epoxide, mirex, and pentachlorophenol.
[j] Neurological endpoint includes the following chemicals: mercury, total PCBs, and selenium. Neurological effects associated with exposure to lead are discussed in Section B.5.4.
[k] Endocrine endpoint includes the following chemicals: antimony and cobalt.
[l] Integumentary endpoint includes the following chemicals: arsenic, total PCBs, selenium, and vanadium.
[m] Digestive system endpoint includes the following chemicals: chromium and copper.
[n] Developmental endpoint includes the following chemicals: mercury and total PCBs.

API – Asian and Pacific Islander
BHC – benzene hexachloride
CDI – chronic daily intake
CT – central tendency
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
EPC – exposure point concentration
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
ww – wet weight

Table B.5-16. Non-cancer hazard estimates associated with the consumption of one-meal-per-month of seafood by adults

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Receptor population: Adults consuming one meal per month of fish or shellfish
Receptor age: Adult

Chemical	Seafood Category	EPC (mg/kg ww)	Non-Cancer CDI (mg/kg-day)	Reference Dose (mg/kg-day)	Hazard Quotient
Antimony	benthic fish[b]	0.002	2.1×10^{-7}	0.0004	0.0005
	clam	0.027	2.8×10^{-6}	0.0004	0.007
	crab edible meat	0.004	4.2×10^{-7}	0.0004	0.001
	pelagic fish, perch[b]	0.004	4.2×10^{-7}	0.0004	0.001
	pelagic fish, rockfish[b]	0.002	2.1×10^{-7}	0.0004	0.0005
Arsenic[a]	benthic fish[b]	0.0045	4.7×10^{-7}	0.0003	0.002
	clam	0.22	2.3×10^{-5}	0.0003	0.08
	crab edible meat	0.036	3.8×10^{-6}	0.0003	0.01
	pelagic fish, perch	0.027	2.8×10^{-6}	0.0003	0.009
	pelagic fish, rockfish	0.011	1.1×10^{-6}	0.0003	0.004
Cadmium	benthic fish	0.11	1.1×10^{-5}	0.001	0.01
	clam	0.096	1.0×10^{-5}	0.001	0.01
	crab edible meat	0.88	9.2×10^{-5}	0.001	0.09
	pelagic fish, perch[b]	0.04	4.2×10^{-6}	0.001	0.004
	pelagic fish, rockfish[b]	0.04	4.2×10^{-6}	0.001	0.004
Chromium	benthic fish	0.1	1.0×10^{-5}	0.003	0.003
	clam	0.69	7.2×10^{-5}	0.003	0.02
	crab edible meat	0.1	1.0×10^{-5}	0.003	0.003
	pelagic fish, perch	0.4	4.2×10^{-5}	0.003	0.01
	pelagic fish, rockfish	0.46	4.8×10^{-5}	0.003	0.02
Cobalt	benthic fish[b]	0.03	3.1×10^{-6}	0.0003	0.01
	clam	0.21	2.2×10^{-5}	0.0003	0.07
	crab edible meat	0.13	1.4×10^{-5}	0.0003	0.05
	pelagic fish, perch[b]	0.05	5.2×10^{-6}	0.0003	0.02
	pelagic fish, rockfish[b]	0.05	5.2×10^{-6}	0.0003	0.02
Copper	benthic fish	0.68	7.1×10^{-5}	0.04	0.002
	clam	7.5	7.8×10^{-4}	0.04	0.02
	crab edible meat	16	1.7×10^{-3}	0.04	0.04
	pelagic fish, perch	2.3	2.4×10^{-4}	0.04	0.006
	pelagic fish, rockfish	1.2	1.3×10^{-4}	0.04	0.003
Mercury	benthic fish	0.046	4.8×10^{-6}	0.0001	0.05
	clam	0.021	2.2×10^{-6}	0.0001	0.02

Table B.5-16. Non-cancer hazard estimates associated with the consumption of one-meal-per-month of seafood by adults (cont.)

Chemical	Seafood Category	EPC (mg/kg ww)	Non-Cancer CDI (mg/kg-day)	Reference Dose (mg/kg-day)	Hazard Quotient
	crab edible meat	0.089	9.3×10^{-6}	0.0001	0.09
	pelagic fish, perch	0.043	4.5×10^{-6}	0.0001	0.04
	pelagic fish, rockfish	0.21	2.2×10^{-5}	0.0001	0.2
Molybdenum	benthic fish	0.24	2.5×10^{-5}	0.005	0.005
	clam	0.52	5.4×10^{-5}	0.005	0.01
	crab edible meat	0.41	4.3×10^{-5}	0.005	0.009
	pelagic fish, perch	0.4	4.2×10^{-5}	0.005	0.008
	pelagic fish, rockfish	0.28	2.9×10^{-5}	0.005	0.006
Selenium	benthic fish	0.58	6.1×10^{-5}	0.005	0.01
	clam	0.41	4.3×10^{-5}	0.005	0.009
	crab edible meat	1.1	1.1×10^{-4}	0.005	0.02
	pelagic fish, perch	0.51	5.3×10^{-5}	0.005	0.01
	pelagic fish, rockfish	0.72	7.5×10^{-5}	0.005	0.02
Vanadium	benthic fish[b]	0.03	3.1×10^{-6}	0.009	0.0003
	clam	0.58	6.1×10^{-5}	0.009	0.007
	crab edible meat	0.08	8.4×10^{-6}	0.009	0.0009
	pelagic fish, perch	0.28	2.9×10^{-5}	0.009	0.003
	pelagic fish, rockfish[b]	0.05	5.2×10^{-6}	0.009	0.0006
Zinc	benthic fish	10	1.0×10^{-3}	0.3	0.003
	clam	19	2.0×10^{-3}	0.3	0.007
	crab edible meat	57	6.0×10^{-3}	0.3	0.02
	pelagic fish, perch	26	2.7×10^{-3}	0.3	0.009
	pelagic fish, rockfish	18	1.9×10^{-3}	0.3	0.006
Dibutyltin as ion	benthic fish[b]	0.006	6.3×10^{-7}	0.0003	0.002
	clam[b]	0.0055	5.7×10^{-7}	0.0003	0.002
	crab edible meat[b]	0.006	6.3×10^{-7}	0.0003	0.002
	pelagic fish, perch[b]	0.0055	5.7×10^{-7}	0.0003	0.002
	pelagic fish, rockfish	0.026	2.7×10^{-6}	0.0003	0.009
Tributyltin as ion	benthic fish	0.0096	1.0×10^{-6}	0.000147	0.007
	clam	0.072	7.5×10^{-6}	0.000147	0.05
	crab edible meat[b]	0.0039	4.1×10^{-7}	0.000147	0.003
	pelagic fish, perch	0.052	5.4×10^{-6}	0.000147	0.04
	pelagic fish, rockfish	0.22	2.3×10^{-5}	0.000147	0.2
1,4-Dichlorobenzene	benthic fish[b]	0.17	1.8×10^{-5}	0.07	0.0003
	clam[b]	0.15	1.6×10^{-5}	0.07	0.0002
	crab edible meat[b]	0.17	1.8×10^{-5}	0.07	0.0003
	pelagic fish, perch[b]	0.65	6.8×10^{-5}	0.07	0.001
	pelagic fish, rockfish[b]	0.17	1.8×10^{-5}	0.07	0.0003
Pentachlorophenol	benthic fish[b]	0.002	2.1×10^{-7}	0.03	0.000007

Table B.5-16. Non-cancer hazard estimates associated with the consumption of one-meal-per-month of seafood by adults (cont.)

Chemical	Seafood Category	EPC (mg/kg ww)	Non-Cancer CDI (mg/kg-day)	Reference Dose (mg/kg-day)	Hazard Quotient
	clam	0.0082	8.6×10^{-7}	0.03	0.00003
	crab edible meat[b]	0.0021	2.2×10^{-7}	0.03	0.000007
	pelagic fish, perch[b]	0.0055	5.7×10^{-7}	0.03	0.00002
	pelagic fish, rockfish[b]	0.0021	2.2×10^{-7}	0.03	0.000007
Total PCBs	benthic fish	2.4	2.5×10^{-4}	0.00002	13[c]
				0.00007	4[d]
	clam	0.069	7.2×10^{-6}	0.00002	0.4[c]
				0.00007	0.1[d]
	crab edible meat	0.16	1.7×10^{-5}	0.00002	0.8[c]
				0.00007	0.2[d]
	pelagic fish, perch	1.6	1.7×10^{-4}	0.00002	8[c]
				0.00007	2[d]
	pelagic fish, rockfish	4	4.2×10^{-4}	0.00002	21[c]
				0.00007	6[d]
Total DDTs	benthic fish	0.013	1.4×10^{-6}	0.0005	0.003
	clam	0.0011	1.1×10^{-7}	0.0005	0.0002
	crab edible meat[b]	0.0011	1.1×10^{-7}	0.0005	0.0002
	pelagic fish, perch	0.011	1.1×10^{-6}	0.0005	0.002
	pelagic fish, rockfish	0.032	3.3×10^{-6}	0.0005	0.007
alpha-BHC	benthic fish[b]	0.00044	4.6×10^{-8}	0.008	0.000006
	clam[b]	0.00044	4.5×10^{-8}	0.008	0.000006
	crab edible meat[b]	0.00042	4.3×10^{-8}	0.008	0.000005
	pelagic fish, perch[b]	0.00043	4.5×10^{-8}	0.008	0.000006
	pelagic fish, rockfish	0.00058	6.1×10^{-8}	0.008	0.000008
Dieldrin	benthic fish	0.0003	3.1×10^{-8}	0.00005	0.0006
	clam[b]	0.00044	4.6×10^{-8}	0.00005	0.0009
	crab edible meat[b]	0.00042	4.4×10^{-8}	0.00005	0.0009
	pelagic fish, perch	0.00076	7.9×10^{-8}	0.00005	0.002
	pelagic fish, rockfish	0.00054	5.6×10^{-8}	0.00005	0.001
Total chlordane	benthic fish	0.0026	2.7×10^{-7}	0.0005	0.0005
	clam	0.0049	5.1×10^{-7}	0.0005	0.001
	crab edible meat[b]	0.0011	1.1×10^{-7}	0.0005	0.0002
	pelagic fish, perch	0.003	3.1×10^{-7}	0.0005	0.0006
	pelagic fish, rockfish	0.0083	8.7×10^{-7}	0.0005	0.002
Heptachlor	benthic fish[b]	0.00022	2.3×10^{-8}	0.0005	0.00005
	clam[b]	0.00022	2.3×10^{-8}	0.0005	0.00005
	crab edible meat[b]	0.00021	2.2×10^{-8}	0.0005	0.00004
	pelagic fish, perch[b]	0.00022	2.2×10^{-8}	0.0005	0.00004
	pelagic fish, rockfish[b]	0.00024	2.5×10^{-8}	0.0005	0.00005

Table B.5-16. Non-cancer hazard estimates associated with the consumption of one-meal-per-month of seafood by adults (cont.)

Chemical	Seafood Category	EPC (mg/kg ww)	Non-Cancer CDI (mg/kg-day)	Reference Dose (mg/kg-day)	Hazard Quotient
Heptachlor epoxide	benthic fish[b]	0.00022	2.3×10^{-8}	0.000013	0.002
	clam[b]	0.00022	2.3×10^{-8}	0.000013	0.002
	crab edible meat[b]	0.00021	2.2×10^{-8}	0.000013	0.002
	pelagic fish, perch[b]	0.00022	2.2×10^{-8}	0.000013	0.002
	pelagic fish, rockfish	0.00024	2.5×10^{-8}	0.000013	0.002
Mirex	benthic fish[b]	0.00022	2.3×10^{-8}	0.0002	0.0001
	clam[b]	0.00022	2.3×10^{-8}	0.0002	0.0001
	crab edible meat[b]	0.00021	2.2×10^{-8}	0.0002	0.0001
	pelagic fish, perch[b]	0.00022	2.2×10^{-8}	0.0002	0.0001
	pelagic fish, rockfish	0.00044	4.6×10^{-8}	0.0002	0.0002
Hazard indices by effect:					
Hazard index for hematological endpoint[e]			**benthic fish**		**0.01**
			clam		**0.02**
			crab edible meat		**0.04**
			pelagic fish, perch		**0.02**
			pelagic fish, rockfish		**0.03**
Hazard index for immunological endpoint[f]			**benthic fish**		**13**
			clam		**0.5**
			crab edible meat		**0.8**
			pelagic fish, perch		**8**
			pelagic fish, rockfish		**21**
Hazard index for kidney endpoint[g]			**benthic fish**		**0.02**
			clam		**0.02**
			crab edible meat		**0.1**
			pelagic fish, perch		**0.01**
			pelagic fish, rockfish		**0.01**
Hazard index for liver endpoint[h]			**benthic fish**		**0.007**
			clam		**0.006**
			crab edible meat		**0.004**
			pelagic fish, perch		**0.008**
			pelagic fish, rockfish		**0.01**
Hazard index for neurological endpoint[i]			**benthic fish**		**13**
			clam		**0.4**
			crab edible meat		**0.9**
			pelagic fish, perch		**8**
			pelagic fish, rockfish		**21**
Hazard index for endocrine endpoint[j]			**benthic fish**		**0.01**
			clam		**0.08**
			crab edible meat		**0.05**
			pelagic fish, perch		**0.02**
			pelagic fish, rockfish		**0.02**
Hazard index for integumentary endpoint[k]			**benthic fish**		**13**

Table B.5-16. Non-cancer hazard estimates associated with the consumption of one-meal-per-month of seafood by adults (cont.)

Chemical	Seafood Category	EPC (mg/kg ww)	Non-Cancer CDI (mg/kg-day)	Reference Dose (mg/kg-day)	Hazard Quotient
	clam				0.5
	crab edible meat				0.8
	pelagic fish, perch				8
	pelagic fish, rockfish				21
Hazard index for digestive system endpoint[l]	benthic fish				0.005
	clam				0.04
	crab edible meat				0.04
	pelagic fish, perch				0.02
	pelagic fish, rockfish				0.02
Hazard index for developmental endpoint[m]	benthic fish				4
	clam				0.1
	crab edible meat				0.3
	pelagic fish, perch				2
	pelagic fish, rockfish				6

Note: The adult one-meal-per-month scenarios are presented for informational purposes only, and are not used by EPA for risk management decisions.

[a] Arsenic EPCs and risk estimates are based on inorganic arsenic.

[b] No detected values in this seafood category. CDI and risk estimate are based on one-half the maximum reporting limit.

[c] HQ used for the calculation of the immunological, integumentary, and neurological endpoint hazard indices (Table B.4-1).

[d] HQ used for the calculation of the developmental endpoint hazard index (Table B.4-1).

[c] Hematological endpoint includes the following chemicals: antimony, selenium, and zinc.

[d] Immunological endpoint includes the following chemicals: dibutyltin, total PCBs, and TBT.

[e] Kidney endpoint includes the following chemicals: cadmium, molybdenum, and pentachlorophenol.

[f] Liver endpoint includes the following chemicals: 1,4-dichlorobenzene, alpha-BHC, total chlordane, total DDTs, dieldrin, heptachlor, heptachlor epoxide, mirex, and pentachlorophenol.

[g] Neurological endpoint includes the following chemicals: mercury, total PCBs, and selenium. Neurological effects associated with exposure to lead are discussed in Section B.5.4.

[h] Endocrine endpoint includes the following chemicals: antimony and cobalt.

[i] Integumentary endpoint includes the following chemicals: arsenic, total PCBs, selenium, and vanadium.

[j] Digestive system endpoint includes the following chemicals: chromium and copper.

[k] Developmental endpoint includes the following chemicals: mercury and total PCBs.

BHC – benzene hexachloride
CDI – chronic daily intake
DDT – dichlorodiphenyltrichloroethane
EPA – US Environmental Protection Agency
EPC – exposure point concentration
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
ww – wet weight

B.5.3.1.3 Percent contribution of COPCs to the total seafood consumption risk

In addition to an evaluation of the excess cancer risks and non-cancer HQs, the percent contribution of individual COPCs was also examined. Table B.5-17 presents the relative contribution of different chemicals to excess cancer risk estimates for all seafood consumption scenarios. The majority of the total excess cancer risk for the two adult RME seafood consumption scenarios was attributable to total PCBs; the majority of the total excess cancer risk for the child RME seafood consumption scenarios was attributable to cPAH TEQ. This type of assessment was not performed for non-cancer hazards because HQs are not directly additive across endpoints, and therefore the contribution of different chemicals cannot be characterized as fractions of the overall hazard. The overwhelming majority of the non-cancer hazards associated with seafood consumption were contributed by total PCBs (greater than 80% of the total immunological, neurological, and integumentary HIs).

Table B.5-17. Contributions to risks by chemical for adult and child seafood consumption scenarios

Chemical	Percentage of Contribution to Total Excess Cancer Risk[a]											
								Adult One Meal per Month[b]				
	Adult Tribal RME (Tulalip Data)	Adult Tribal CT (Tulalip Data)	Child Tribal RME (Tulalip Data)	Child Tribal CT (Tulalip Data)	Adult Tribal (Suquamish Data)	Adult API RME	Adult API CT	Benthic Fish	Clam	Crab	Pelagic Fish, Rockfish	Pelagic Fish, Perch
Total Excess Cancer Risk (excluding PCB TEQ)[c]												
Arsenic	14%	14%	11%	11%	16%	14%	18%	0.1%	35%	12%	0.2%	2%
cPAH TEQ	7%	6%	27%	25%	8%	9%	8%	0.05%	31%	2%	0.02%	0.4%
Total PCBs	70%	70%	55%	55%	70%	69%	63%	97%	21%	62%	95%	89%
Dioxin/furan TEQ	7%	8%	5%	8%	5%	7%	9%	2%	10%	18%	5%	8%
Other COPCs[d]	2%	2%	1%	2%	1%	1%	2%	0.5%	3%	5%	0.4%	1%
Total Excess Cancer Risk (excluding total PCBs)[c]												
Arsenic	18%	16%	15%	11%	20%	17%	17%	0.3%	36%	12%	0.2%	2%
cPAH TEQ	9%	6%	38%	25%	10%	10%	7%	0.09%	32%	2%	0.03%	0.5%
PCB TEQ	62%	65%	38%	55%	61%	63%	66%	94%	18%	62%	93%	87%
Dioxin/furan TEQ	9%	10%	8%	8%	7%	8%	8%	5%	11%	18%	6%	9%
Other COPCs[d]	2%	3%	2%	2%	2%	2%	2%	0.9%	3%	5%	0.6%	1%

a The sum of all percentages may not equal exactly 100% because of the presentation of significant figures.

b The adult one-meal-per-month scenarios are presented for informational purposes only, and are not used by EPA for risk management decisions.

c As was done in the excess cancer risk tables for the seafood consumption scenarios (Tables B.5-1 through B.5-8), total excess cancer risks are presented both excluding PCB TEQ and excluding total PCBs to avoid double-counting the risk associated with PCBs. Percent contributions were calculated for both sums in this table.

d Includes all other COPCs detected in tissue.

API – Asian and Pacific Islander
COPC – chemical of potential concern
CT – central tendency
cPAH – carcinogenic polycyclic aromatic hydrocarbon
EPA – US Environmental Protection Agency
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent

For each scenario, the magnitude of the excess cancer risk, along with the average percent contribution of total PCBs, arsenic, cPAH TEQ, dioxin/furan TEQ, and other COPCs, is shown in Figure B.5-1. Four chemicals (total PCBs, arsenic, cPAH TEQ, dioxin/furan TEQ) were selected for further examination in this HHRA because they had excess cancer risks greater than 1×10^{-6} and each represented more than 5% of the total excess cancer risk for at least one scenario.

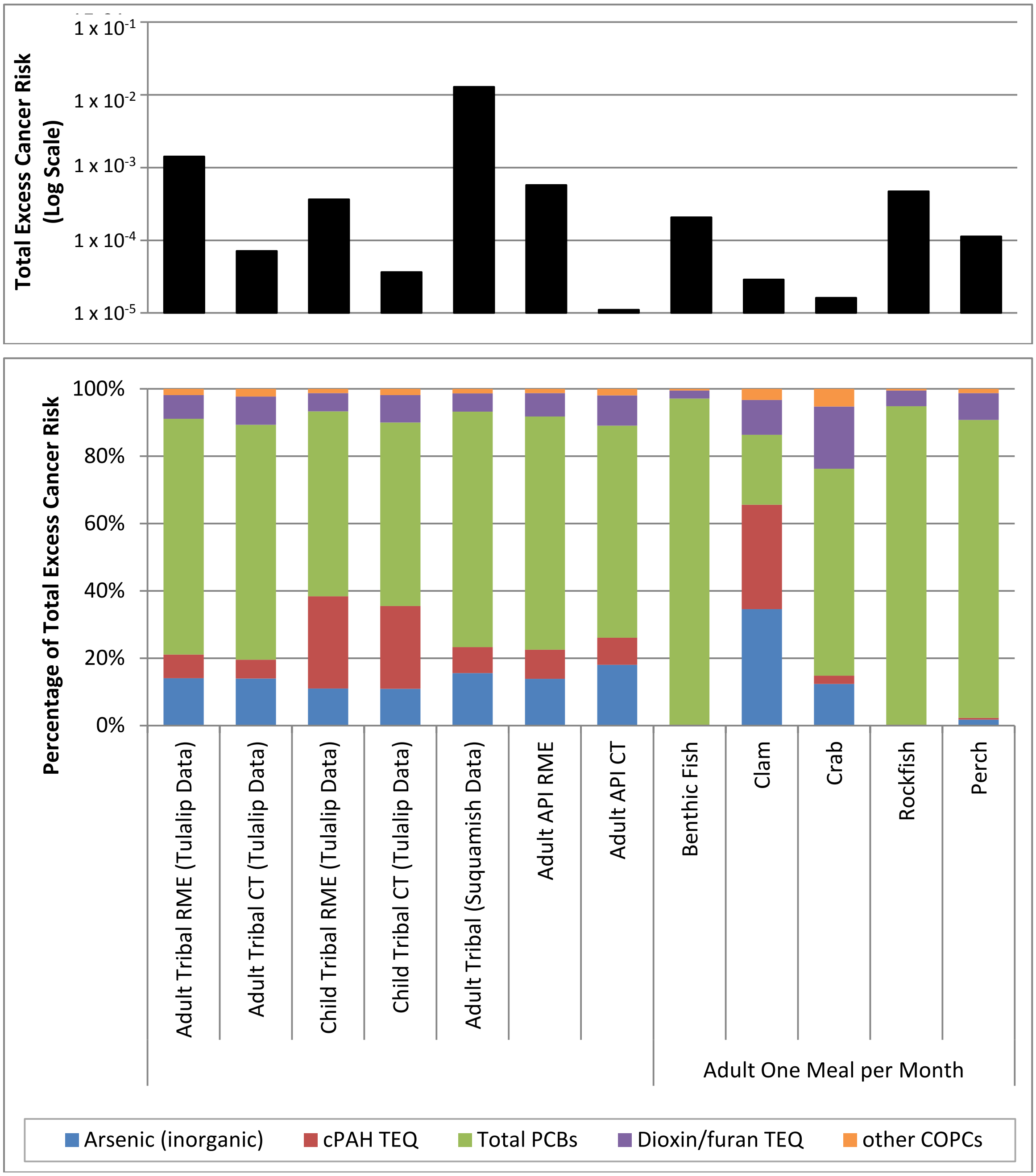


Figure B.5-1. Excess cancer risks by chemical for seafood consumption scenario

Overall, arsenic, cPAH TEQ, and PCBs (as total PCBs or PCB TEQ) were the greatest contributors to excess cancer risk estimates. The percent contribution to the overall risk from cPAH TEQ for the child scenarios was higher than those for the adult scenarios because EPA risk assessment procedures consider children to be more sensitive than adults to chemicals with mutagenic modes of action, such as cPAHs (see Section B.5.1.1). Dioxin/furan TEQ contributed a lower percentage to the overall estimated cancer risk, between 5 and 10% for the tribal and API scenarios. The percent contribution of dioxin/furan TEQ to the total risk estimate was more variable for the adult one-meal-per-month scenarios, highlighting the variability in chemical concentrations in the different consumption categories. PCBs were the dominant contributor for benthic fish, crab, rockfish, and perch; arsenic, cPAH TEQ, and PCBs were the dominant contributors for clams. For all scenarios, chemicals in the "other chemicals" group included two SVOCs and eight pesticides. The "other chemicals" group contributed 5% or less of the overall cancer risk estimate for all seafood consumption scenarios.

B.5.3.1.4 Risk estimates by seafood category for chemicals that contribute the greatest percentage to seafood consumption risk estimates

The previous sections summarized excess cancer risks, non-cancer hazards, and the percent contribution of COPCs to the total risk. For those COPCs that contribute the greatest percentages of the total risk, this section discusses the contribution to risk (for each COPC) by the different seafood categories (e.g. clam, crab edible meat, etc.). As discussed in Section B.5.3.1.3, arsenic, cPAH TEQ, total PCBs, PCB TEQ, and dioxin/furan TEQ were determined to be the dominant contributors to excess cancer risk estimates, with excess cancer risks for each of these chemicals being greater than 1×10^{-6} for nine or more seafood consumption scenarios. In addition, each of these chemicals also contributed 5% or more of the total excess cancer risks for eight or more seafood consumption scenarios. Total PCBs and cadmium had HQs greater than 1 for one or more of the seafood consumption RME scenarios, with total PCBs contributing the largest proportion to non-cancer HI estimates. Arsenic, cPAH TEQ, total PCBs, PCB TEQ, and dioxin/furan TEQ were among the most frequently detected chemicals in seafood, and their concentrations varied greatly across the ten seafood categories. For example, total PCB EPCs ranged from 0.022 mg/kg ww for geoduck edible meat to 4.1 mg/kg ww for whole-body benthic fish. Similarly, consumption rates for the 10 different seafood categories also varied across scenarios because of differences in the overall quantity consumed and the relative distribution of the consumption rates (Section B.3.3.1). Together, these variations in chemical concentrations and consumption rates resulted in ranges of risk estimates among the various seafood consumption scenarios. Although body weight and exposure duration assumptions also differed across scenarios and contributed to the differences in risk estimates, these exposure parameters had a smaller influence on risk estimates than did the seafood consumption rates.

The proportional contributions of each seafood category to risk estimates are presented in Tables B.5-18 to B.5-22 and discussed below. The adult and child tribal RME scenarios based on Tulalip data are presented together in Table B.5-18 because the apportionment of the market basket was done in the same way, meaning that the percentage of total risk associated with each consumption category was the same. Likewise, the adult and child tribal CT scenarios based on Tulalip data are presented together in Table B.5-19.

Table B.5-18. Comparison of excess cancer risks and non-cancer hazards by consumption category for select chemicals in the adult tribal RME and child tribal RME seafood consumption scenarios based on Tulalip data

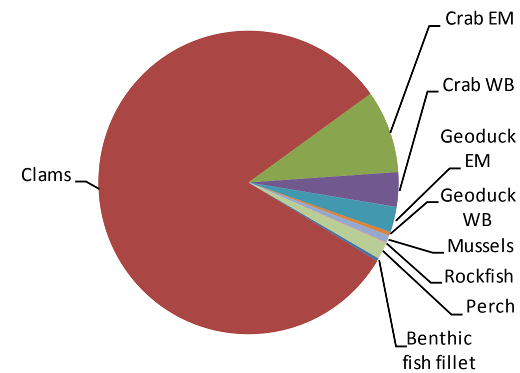
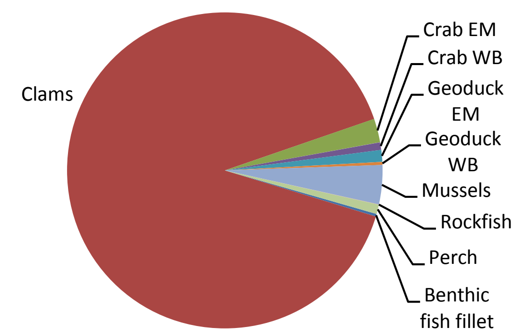
Consumption Category	EPC (mg/kg ww)	Ingestion Rate (g/day)		Percentage of Chemical's Risk	Excess Cancer Risk		Non-Cancer Hazard Quotient		Adult or Child Tribal RME (Tulalip Data) Seafood Consumption Risk Expressed as Pie Chart[a]
		Adult	Child		Adult	Child	Adult	Child	
Arsenic[b]									
Benthic fish fillet	0.0045	7.5	3	0.3%	6×10^{-7}	1×10^{-7}	0.001	0.003	
Clams	0.22	39.3	15.7	81.4%	2×10^{-4}	3×10^{-5}	0.3	0.8	
Crab EM	0.036	26.1	10.4	8.9%	2×10^{-5}	3×10^{-6}	0.04	0.08	
Crab WB	0.047	8.3	3.3	3.7%	7×10^{-6}	1×10^{-6}	0.02	0.03	
Geoduck EM	0.044	6.5	2.6	2.7%	5×10^{-6}	1×10^{-6}	0.01	0.03	
Geoduck WB	0.049	0.9	0.4	0.4%	8×10^{-7}	2×10^{-7}	0.002	0.004	
Mussels	0.096	0.8	0.3	0.7%	1×10^{-6}	2×10^{-7}	0.003	0.006	
Pelagic fish, rockfish	0.011	1	0.4	0.1%	2×10^{-7}	4×10^{-8}	0.0004	0.001	
Pelagic fish, perch	0.027	7.1	2.8	1.8%	4×10^{-6}	6×10^{-7}	0.01	0.02	
				Total risk from arsenic	**2×10^{-4}**	**4×10^{-5}**	**0.4**	**0.9**	
cPAH TEQ									
Benthic fish fillet	0.00042	7.5	3	0.3%	3×10^{-7}	3×10^{-7}	na	na	
Clams	0.027	39.3	15.7	90.0%	9×10^{-5}	9×10^{-5}	na	na	
Crab EM	0.0011	26.1	10.4	2.4%	3×10^{-6}	3×10^{-6}	na	na	
Crab WB	0.0011	8.3	3.3	0.8%	8×10^{-7}	8×10^{-7}	na	na	
Geoduck EM	0.0022	6.5	2.6	1.2%	1×10^{-6}	1×10^{-6}	na	na	
Geoduck WB	0.0041	0.9	0.4	0.3%	3×10^{-7}	4×10^{-7}	na	na	
Mussels	0.059	0.8	0.3	4.0%	4×10^{-6}	4×10^{-6}	na	na	
Pelagic fish, rockfish	0.00029	1	0.4	0.02%	3×10^{-8}	3×10^{-8}	na	na	
Pelagic fish, perch	0.0016	7.1	2.8	0.9%	1×10^{-6}	1×10^{-6}	na	na	
				Total risk from cPAH TEQ	**1×10^{-4}**	**1×10^{-4}**	**na**	**na**	

Table B.5-18. Comparison of excess cancer risks and non-cancer hazards by consumption category for select chemicals in the adult tribal RME and child tribal RME seafood consumption scenarios based on Tulalip data (cont.)

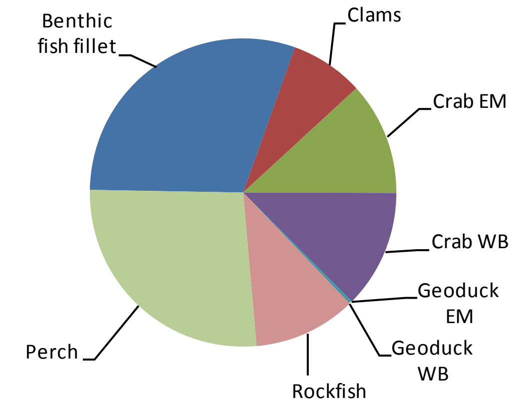
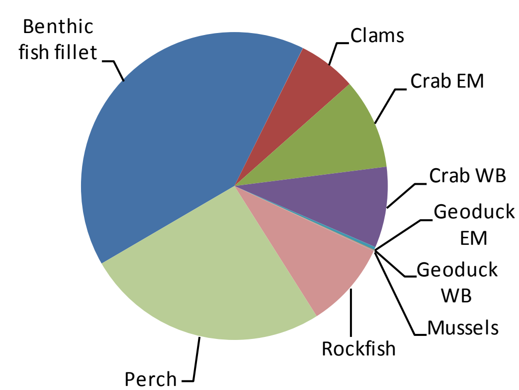
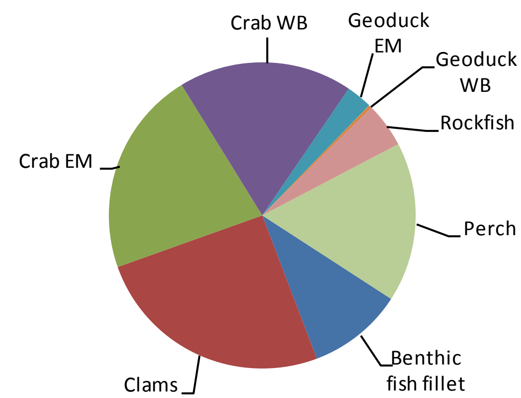
Consumption Category	EPC (mg/kg ww)	Ingestion Rate (g/day)		Percentage of Chemical's Risk	Excess Cancer Risk		Non-Cancer Hazard Quotient		Adult or Child Tribal RME (Tulalip Data) Seafood Consumption Risk Expressed as Pie Chart[a]
		Adult	Child		Adult	Child	Adult	Child	
PCB TEQ[c]									
Benthic fish fillet	1.5×10^{-5}	7.6	3	30.2%	2×10^{-4}	4×10^{-5}	na	na	
Clams	7.3×10^{-7}	39.6	15.8	7.7%	5×10^{-5}	1×10^{-5}	na	na	
Crab EM	1.7×10^{-6}	26.3	10.5	11.9%	8×10^{-5}	2×10^{-5}	na	na	
Crab WB	5.6×10^{-6}	8.4	3.4	12.5%	9×10^{-5}	2×10^{-5}	na	na	
Geoduck EM	1.9×10^{-7}	6.5	2.6	0.3%	2×10^{-6}	4×10^{-7}	na	na	
Geoduck WB	2.3×10^{-7}	0.9	0.4	0.1%	4×10^{-7}	8×10^{-8}	na	na	
Pelagic fish, rockfish	4.0×10^{-5}	1	0.4	10.6%	7×10^{-5}	1×10^{-5}	na	na	
Pelagic fish, perch	1.4×10^{-5}	7.2	2.8	26.7%	2×10^{-4}	3×10^{-5}	na	na	
Total risk from PCB TEQ					**7×10^{-4}**	**1×10^{-4}**	**na**	**na**	
Total PCBs									
Benthic fish fillet	2.4	7.5	3	40.7%	4×10^{-4}	8×10^{-5}	11	24	
Clams	0.069	39.3	15.7	6.1%	7×10^{-5}	1×10^{-5}	2	4	
Crab EM	0.16	26.1	10.4	9.5%	1×10^{-4}	2×10^{-5}	3	5	
Crab WB	0.45	8.3	3.3	8.5%	9×10^{-5}	2×10^{-5}	2	5	
Geoduck EM	0.022	6.5	2.6	0.3%	3×10^{-6}	6×10^{-7}	0.09	0.2	
Geoduck WB	0.034	0.9	0.4	0.1%	7×10^{-7}	2×10^{-7}	0.02	0.04	
Mussels	0.031	0.8	0.3	0.1%	6×10^{-7}	1×10^{-7}	0.02	0.03	
Pelagic fish, rockfish	4.0	1	0.4	9.1%	1×10^{-4}	2×10^{-5}	2	5	
Pelagic fish, perch	1.6	7.1	2.8	25.7%	3×10^{-4}	5×10^{-5}	7	15	
Total risk from total PCBs					**1×10^{-3}**	**2×10^{-4}**	**27**	**58**	

Table B.5-18. Comparison of excess cancer risks and non-cancer hazards by consumption category for select chemicals in the adult tribal RME and child tribal RME seafood consumption scenarios based on Tulalip data (cont.)

Consumption Category	EPC (mg/kg ww)	Ingestion Rate (g/day)		Percentage of Chemical's Risk	Excess Cancer Risk		Non-Cancer Hazard Quotient	
		Adult	Child		Adult	Child	Adult	Child
Dioxin/Furan TEQ[c]								
Benthic fish fillet	7.9×10^{-7}	7.6	3	10.1%	1×10^{-5}	2×10^{-6}	na	na
Clams	3.8×10^{-7}	39.6	15.8	25.3%	3×10^{-5}	5×10^{-6}	na	na
Crab EM	4.9×10^{-7}	26.3	10.5	21.6%	2×10^{-5}	4×10^{-6}	na	na
Crab WB	1.3×10^{-6}	8.4	3.4	18.3%	2×10^{-5}	4×10^{-6}	na	na
Geoduck EM	2.5×10^{-7}	6.5	2.6	2.7%	3×10^{-6}	5×10^{-7}	na	na
Geoduck WB	2.0×10^{-7}	0.9	0.4	0.3%	3×10^{-7}	7×10^{-8}	na	na
Pelagic fish, rockfish	2.8×10^{-6}	1	0.4	4.7%	5×10^{-6}	9×10^{-7}	na	na
Pelagic fish, perch	1.4×10^{-6}	7.2	2.8	16.9%	2×10^{-5}	3×10^{-6}	na	na
Total risk from dioxin/furan TEQ					**1×10^{-4}**	**2×10^{-5}**	**na**	**na**

Adult or Child Tribal RME (Tulalip Data) Seafood Consumption Risk Expressed as Pie Chart[a]



[a] Pie charts represent both cancer and non-cancer risks. Risk percentages are based on EPCs and ingestion rates, meaning that the percentage of risk from each consumption category is the same for cancer and non-cancer risks.

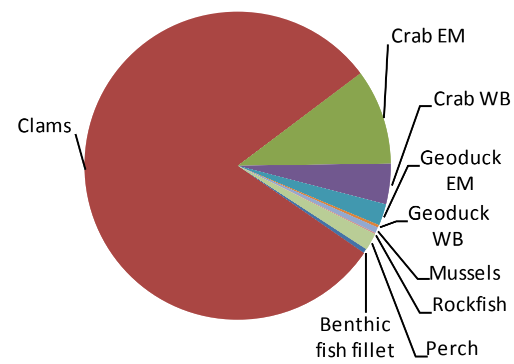
[b] Arsenic EPCs and risk estimates are based on inorganic arsenic.

[c] No mussel data were available for this chemical. When the CDI and risk values were calculated, the proportion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
EM – edible meat
EPC – exposure point concentration
na – not available
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent
WB – whole body
ww – wet weight

Table B.5-19. Comparison of excess cancer and non-cancer risks by consumption category for select chemicals in the adult tribal CT and child tribal CT seafood consumption scenarios based on Tulalip data

Consumption Category	EPC (mg/kg ww)[a]	Ingestion Rate (g/day)		Percentage of Chemical's Risk	Excess Cancer Risk		Non-Cancer Hazard Quotient		Adult or Child Tribal CT (Tulalip Data) Seafood Consumption Risk Expressed as Pie Chart[b]
		Adult	Child		Adult	Child	Adult	Child	
Arsenic[c]									
Benthic fish fillet	0.0050	1.2	0.48	0.5%	5×10^{-8}	2×10^{-8}	0.0002	0.0005	
Clams	0.17	6	2.4	80.1%	8×10^{-6}	3×10^{-6}	0.04	0.09	
Crab EM	0.032	4	1.6	10.1%	1×10^{-6}	4×10^{-7}	0.005	0.01	
Crab WB	0.042	1.3	0.5	4.3%	4×10^{-7}	2×10^{-7}	0.002	0.005	
Geoduck EM	0.029	1	0.4	2.3%	2×10^{-7}	1×10^{-7}	0.001	0.003	
Geoduck WB	0.036	0.1	0.04	0.3%	3×10^{-8}	1×10^{-8}	0.0001	0.0003	
Mussels	0.078	0.1	0.04	0.6%	6×10^{-8}	3×10^{-8}	0.0003	0.0007	
Pelagic fish, rockfish	0.0080	0.2	0.08	0.1%	1×10^{-8}	5×10^{-9}	0.0001	0.0001	
Pelagic fish, perch	0.021	1.1	0.44	1.8%	2×10^{-7}	8×10^{-8}	0.001	0.002	
Total risk from arsenic					**1×10^{-5}**	**4×10^{-6}**	**0.05**	**0.1**	
cPAH TEQ									
Benthic fish fillet	0.00029	1.2	0.48	0.3%	1×10^{-8}	3×10^{-8}	na	na	
Clams	0.016	6	2.4	91.2%	4×10^{-6}	8×10^{-6}	na	na	
Crab EM	0.00060	4	1.6	2.3%	9×10^{-8}	2×10^{-7}	na	na	
Crab WB	0.00096	1.3	0.5	1.2%	5×10^{-8}	1×10^{-7}	na	na	
Geoduck EM	0.0016	1	0.4	1.5%	6×10^{-8}	1×10^{-7}	na	na	
Geoduck WB	0.0031	0.1	0.04	0.3%	1×10^{-8}	3×10^{-8}	na	na	
Mussels	0.020	0.1	0.04	1.9%	8×10^{-8}	2×10^{-7}	na	na	
Pelagic fish, rockfish	0.00024	0.2	0.08	1.05%	2×10^{-9}	4×10^{-9}	na	na	
Pelagic fish, perch	0.0012	1.1	0.44	1.3%	5×10^{-8}	1×10^{-7}	na	na	
Total risk from cPAH TEQ					**4×10^{-6}**	**9×10^{-6}**	**na**	**na**	



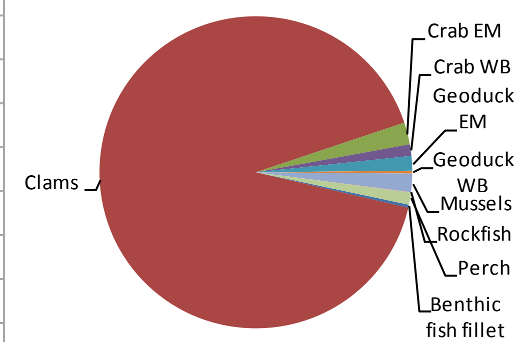


Table B.5-19. Comparison of excess cancer and non-cancer risks by consumption category for select chemicals in the adult tribal CT and child tribal CT seafood consumption scenarios based on Tulalip data (cont.)

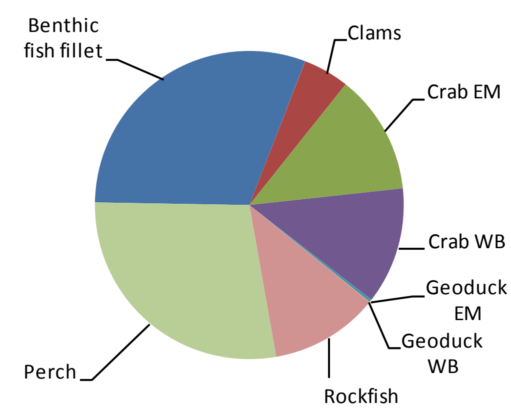
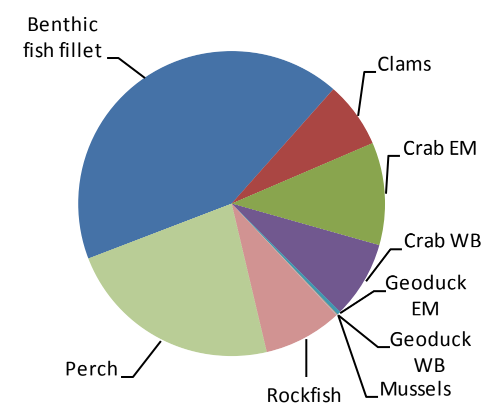
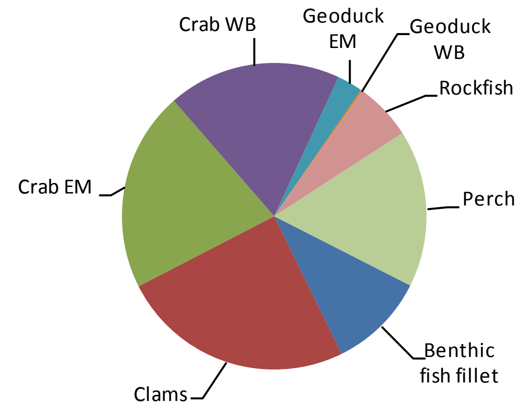
Consumption Category	EPC (mg/kg ww)[a]	Ingestion Rate (g/day)		Percentage of Chemical's Risk	Excess Cancer Risk		Non-Cancer Hazard Quotient		Adult or Child Tribal CT (Tulalip Data) Seafood Consumption Risk Expressed as Pie Chart[b]
		Adult	Child		Adult	Child	Adult	Child	
PCB TEQ[d]									
Benthic fish fillet	1.3×10^{-5}	1.2	0.49	30.6%	1×10^{-5}	5×10^{-6}	na	na	
Clams	4.1×10^{-7}	6.1	2.4	4.9%	2×10^{-6}	8×10^{-7}	na	na	
Crab EM	1.6×10^{-6}	4	1.6	12.5%	5×10^{-6}	2×10^{-6}	na	na	
Crab WB	4.8×10^{-6}	1.3	0.5	12.2%	5×10^{-6}	2×10^{-6}	na	na	
Geoduck EM	1.4×10^{-7}	1	0.4	0.3%	1×10^{-7}	5×10^{-8}	na	na	
Geoduck WB	2.3×10^{-7}	0.1	0.04	0.05%	2×10^{-8}	8×10^{-9}	na	na	
Pelagic fish, rockfish	2.9×10^{-5}	0.2	0.08	11.4%	5×10^{-6}	2×10^{-6}	na	na	
Pelagic fish, perch	1.3×10^{-5}	1.1	0.44	28.0%	1×10^{-5}	5×10^{-6}	na	na	
				Total risk from PCB TEQ	**4×10^{-5}**	**2×10^{-5}**	**na**	**na**	
Total PCBs									
Benthic fish fillet	1.7	1.2	0.48	42.4%	2×10^{-5}	9×10^{-6}	1	3	
Clams	0.056	6	2.4	7.0%	4×10^{-6}	2×10^{-6}	0.2	0.4	
Crab EM	0.13	4	1.6	10.8%	5×10^{-6}	2×10^{-6}	0.3	0.6	
Crab WB	0.30	1.3	0.5	8.1%	4×10^{-6}	2×10^{-6}	0.2	0.5	
Geoduck EM	0.019	1	0.4	0.4%	2×10^{-7}	9×10^{-8}	0.01	0.02	
Geoduck WB	0.028	0.1	0.04	0.06%	2×10^{-8}	1×10^{-8}	0.001	0.003	
Mussels	0.026	0.1	0.04	0.05%	2×10^{-8}	1×10^{-8}	0.001	0.003	
Pelagic fish, rockfish	2.0	0.2	0.08	8.3%	4×10^{-6}	2×10^{-6}	0.3	0.7	
Pelagic fish, perch	1.0	1.1	0.44	22.9%	1×10^{-5}	5×10^{-6}	1	2	
				Total risk from total PCBs	**5×10^{-5}**	**2×10^{-5}**	**3**	**6**	

Table B.5-19. Comparison of excess cancer and non-cancer risks by consumption category for select chemicals in the adult tribal CT and child tribal CT seafood consumption scenarios based on Tulalip data (cont.)

Consumption Category	EPC (mg/kg ww)[a]	Ingestion Rate (g/day)		Percentage of Chemical's Risk	Excess Cancer Risk		Non-Cancer Hazard Quotient		Adult or Child Tribal CT (Tulalip Data) Seafood Consumption Risk Expressed as Pie Chart[b]
		Adult	Child		Adult	Child	Adult	Child	
Dioxin/Furan TEQ[d]									
Benthic fish fillet	7.5×10^{-7}	1.2	0.49	11.2%	7×10^{-7}	3×10^{-7}	na	na	
Clams	2.8×10^{-7}	6.1	2.4	21.2%	1×10^{-6}	6×10^{-7}	na	na	
Crab EM	4.7×10^{-7}	4	1.6	23.5%	1×10^{-6}	6×10^{-7}	na	na	
Crab WB	1.2×10^{-6}	1.3	0.5	19.4%	1×10^{-6}	5×10^{-7}	na	na	
Geoduck EM	2.3×10^{-7}	1	0.4	2.9%	2×10^{-7}	8×10^{-8}	na	na	
Geoduck WB	2.0×10^{-7}	0.1	0.04	0.2%	2×10^{-8}	7×10^{-9}	na	na	
Pelagic fish, rockfish	2.1×10^{-6}	0.2	0.08	5.2%	3×10^{-7}	1×10^{-7}	na	na	
Pelagic fish, perch	1.2×10^{-6}	1.1	0.44	16.4%	1×10^{-6}	4×10^{-7}	na	na	
Total risk from dioxin/furan TEQ					**6×10^{-6}**	**3×10^{-6}**	**na**	**na**	

[a] EPCs used for the CT scenarios are mean values.

[b] Pie charts represent both cancer and non-cancer risks. Risk percentages are based on EPCs and ingestion rates, meaning that the percentage of risk from each consumption category is the same for cancer and non-cancer risks.

[c] Arsenic EPCs and risk estimates are based on inorganic arsenic.

[d] No mussel data were available for this chemical. When the CDI and risk values were calculated, the proportion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
CT – central tendency
EM – edible meat
EPC – exposure point concentration
na – not available
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
WB – whole body
ww – wet weight

Table B.5-20. Comparison of excess cancer and non-cancer risks by consumption category for select chemicals in the adult tribal seafood consumption scenario based on Suquamish data

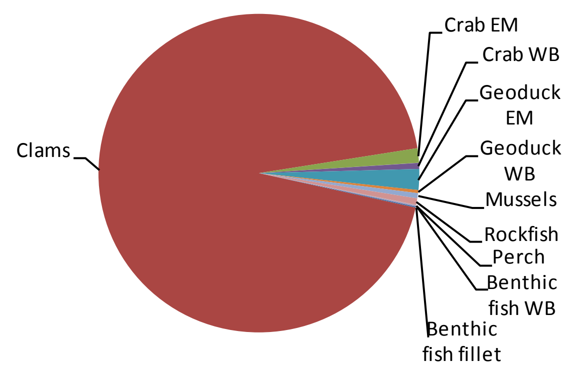
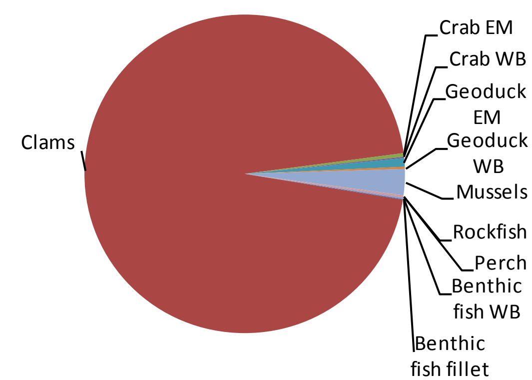
Consumption Category	EPC (mg/kg ww)	Ingestion Rate (g/day)	Percentage of Chemical's Risk	Excess Cancer Risk	Non-Cancer Hazard Quotient	Adult Tribal (Suquamish data) Seafood Consumption Risk Expressed as Pie Chart[a]
Arsenic[b]						
Benthic fish fillet	0.0045	25.9	0.1%	2×10^{-6}	0.005	
Benthic fish WB	0.22	3.2	0.1%	2×10^{-6}	0.01	
Clams	0.036	393.7	94.0%	2×10^{-3}	4	
Crab EM	0.047	37.8	1.5%	3×10^{-5}	0.06	
Crab WB	0.044	12	0.6%	1×10^{-5}	0.02	
Geoduck EM	0.049	43.8	2.1%	4×10^{-5}	0.08	
Geoduck WB	0.096	6	0.3%	6×10^{-6}	0.01	
Mussels	0.011	5	0.5%	9×10^{-6}	0.02	
Pelagic fish, rockfish	0.027	55.4	0.7%	1×10^{-5}	0.03	
Pelagic fish, perch	0.0045	0.6	0.02%	3×10^{-7}	0.001	
			Total risk from arsenic	**2×10^{-3}**	**4**	
cPAH TEQ						
Benthic fish fillet	0.00042	25.9	0.1%	1×10^{-6}	na	
Benthic fish WB	0.0068	3.2	0.2%	2×10^{-6}	na	
Clams	0.027	393.7	95.3%	1×10^{-3}	na	
Crab EM	0.0011	37.8	0.4%	4×10^{-6}	na	
Crab WB	0.0011	12	0.1%	1×10^{-6}	na	
Geoduck EM	0.0022	43.8	0.9%	9×10^{-6}	na	
Geoduck WB	0.0041	6	0.2%	2×10^{-6}	na	
Mussels	0.059	5	2.6%	3×10^{-5}	na	
Pelagic fish, rockfish	0.00029	55.4	0.1%	1×10^{-6}	na	
Pelagic fish, perch	0.0016	0.6	0.009%	9×10^{-8}	na	
			Total risk from cPAH TEQ	**1×10^{-3}**	**na**	

Table B.5-20. Comparison of excess cancer and non-cancer risks by consumption category for select chemicals in the adult tribal seafood consumption scenario based on Suquamish data (cont.)

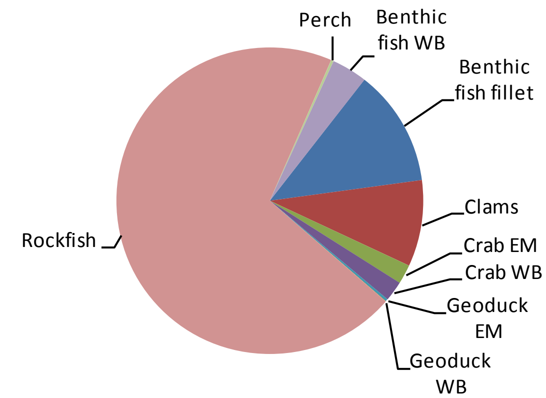
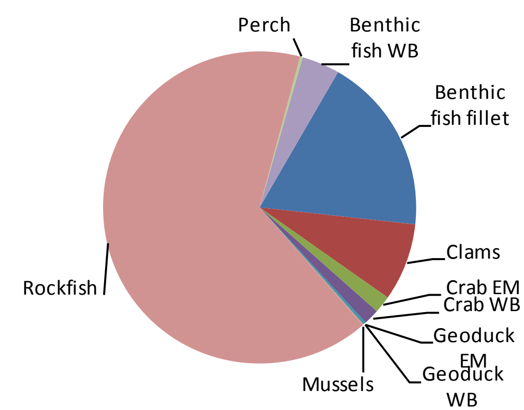
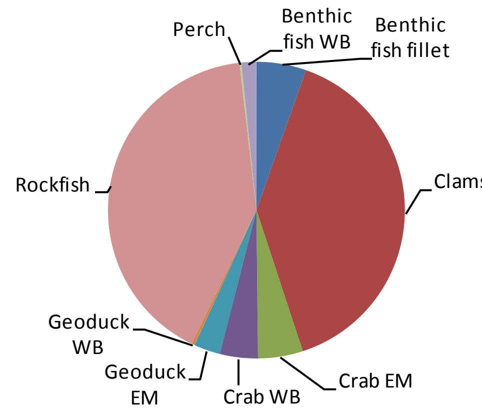
Consumption Category	EPC (mg/kg ww)	Ingestion Rate (g/day)	Percentage of Chemical's Risk	Excess Cancer Risk	Non-Cancer Hazard Quotient	Adult Tribal (Suquamish data) Seafood Consumption Risk Expressed as Pie Chart[a]
PCB TEQ[c]						
Benthic fish fillet	1.5×10^{-5}	26.1	12.3%	7×10^{-4}	na	
Benthic fish WB	3.7×10^{-5}	3.2	3.7%	2×10^{-4}	na	
Clams	7.3×10^{-7}	397.1	9.1%	6×10^{-4}	na	
Crab EM	1.7×10^{-6}	38.1	2.0%	1×10^{-4}	na	
Crab WB	5.6×10^{-6}	12.1	2.1%	1×10^{-4}	na	
Geoduck EM	1.9×10^{-7}	44.2	0.3%	2×10^{-5}	na	
Geoduck WB	2.3×10^{-7}	6.1	0.04%	3×10^{-6}	na	
Pelagic fish, rockfish	4.0×10^{-5}	55.9	70.2%	4×10^{-3}	na	
Pelagic fish, perch	1.4×10^{-5}	0.6	0.3%	2×10^{-5}	na	
			Total risk from PCB TEQ	**6×10^{-3}**	**na**	
Total PCBs						
Benthic fish fillet	2.4	25.9	18.4%	2×10^{-3}	39	
Benthic fish WB	4.1	3.2	3.9%	3×10^{-4}	8	
Clams	0.069	393.7	8.0%	7×10^{-4}	17	
Crab EM	0.16	37.8	1.8%	2×10^{-4}	4	
Crab WB	0.45	12	1.6%	1×10^{-4}	3	
Geoduck EM	0.022	43.8	0.3%	2×10^{-5}	0.6	
Geoduck WB	0.034	6	0.06%	5×10^{-6}	0.1	
Mussels	0.031	5	0.05%	4×10^{-6}	0.1	
Pelagic fish, rockfish	4.0	55.4	65.6%	6×10^{-3}	140	
Pelagic fish, perch	1.6	0.6	0.3%	2×10^{-5}	0.6	
			Total risk from total PCBs	**9×10^{-3}**	**214**	

Table B.5-20. Comparison of excess cancer and non-cancer risks by consumption category for select chemicals in the adult tribal seafood consumption scenario based on Suquamish data (cont.)

Consumption Category	EPC (mg/kg ww)	Ingestion Rate (g/day)	Percentage of Chemical's Risk	Excess Cancer Risk	Non-Cancer Hazard Quotient	Adult Tribal (Suquamish data) Seafood Consumption Risk Expressed as Pie Chart[a]
Dioxin/Furan TEQ[c]						
Benthic fish fillet	7.9×10^{-7}	26.1	5.4%	4×10^{-5}	na	
Benthic fish WB	3.8×10^{-7}	3.2	1.6%	1×10^{-5}	na	
Clams	4.9×10^{-7}	397.1	39.5%	3×10^{-4}	na	
Crab EM	1.3×10^{-6}	38.1	4.9%	4×10^{-5}	na	
Crab WB	2.5×10^{-7}	12.1	4.1%	3×10^{-5}	na	
Geoduck EM	2.0×10^{-7}	44.2	2.9%	2×10^{-5}	na	
Geoduck WB	2.8×10^{-6}	6.1	0.3%	2×10^{-6}	na	
Pelagic fish, rockfish	2.8×10^{-6}	55.9	41.0%	3×10^{-4}	na	
Pelagic fish, perch	1.4×10^{-6}	0.6	0.2%	2×10^{-6}	na	
			Total risk from dioxin/furan TEQ	**7×10^{-4}**	**na**	

[a] Pie charts represent both cancer and non-cancer risks. Risk percentages are based on EPCs and ingestion rates, meaning that the percentage of risk from each consumption category is the same for cancer and non-cancer risks.

[b] Arsenic EPCs and risk estimates are based on inorganic arsenic.

[c] No mussel data were available for this chemical. When the CDI and risk values were calculated, the proportion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
EM – edible meat
EPC – exposure point concentration
na – not available
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
WB – whole body
ww – wet weight

Table B.5-21. Comparison of excess cancer and non-cancer risks by consumption category for select chemicals in the adult API RME seafood consumption scenario

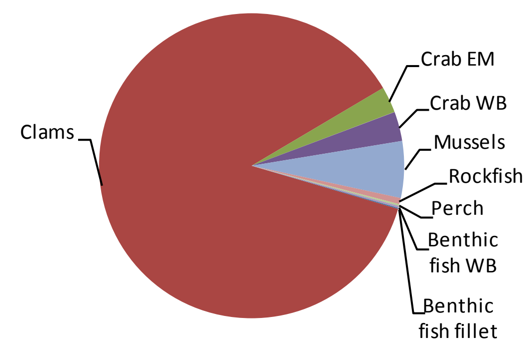
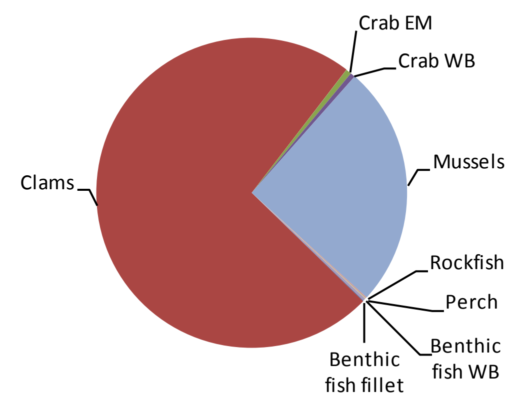
Consumption Category	EPC (mg/kg ww)	Ingestion Rate (g/day)	Percentage of Chemical's Risk	Excess Cancer Risk	Non-Cancer Hazard Quotient	Adult API RME Seafood Consumption Risk Expressed as Pie Chart[a]
Arsenic[b]						
Benthic fish fillet	0.0045	2	0.1%	9×10^{-8}	0.0005	
Benthic fish WB	0.038	0.4	0.2%	2×10^{-7}	0.001	
Clams	0.22	29.1	86.9%	7×10^{-5}	0.3	
Crab EM	0.036	5.7	2.8%	2×10^{-6}	0.01	
Crab WB	0.047	4.9	3.1%	2×10^{-6}	0.01	
Mussels	0.096	4.6	6.0%	5×10^{-6}	0.02	
Pelagic fish, rockfish	0.011	4.4	0.7%	5×10^{-7}	0.003	
Pelagic fish, perch	0.027	0.5	0.2%	1×10^{-7}	0.001	
			Total risk from arsenic	**8×10^{-5}**	**0.4**	
cPAH TEQ						
Benthic fish fillet	0.00042	2	0.08%	4×10^{-8}	na	
Benthic fish WB	0.0068	0.4	0.3%	1×10^{-7}	na	
Clams	0.027	29.1	73.1%	4×10^{-5}	na	
Crab EM	0.0011	5.7	0.6%	3×10^{-7}	na	
Crab WB	0.0011	4.9	0.5%	3×10^{-7}	na	
Mussels	0.059	4.6	25.3%	1×10^{-5}	na	
Pelagic fish, rockfish	0.00029	4.4	0.1%	6×10^{-8}	na	
Pelagic fish, perch	0.0016	0.5	0.07%	4×10^{-8}	na	
			Total risk from cPAH TEQ	**5×10^{-5}**	**na**	

Table B.5-21. Comparison of excess cancer and non-cancer risks by consumption category for select chemicals in the adult API RME seafood consumption scenario (cont.)

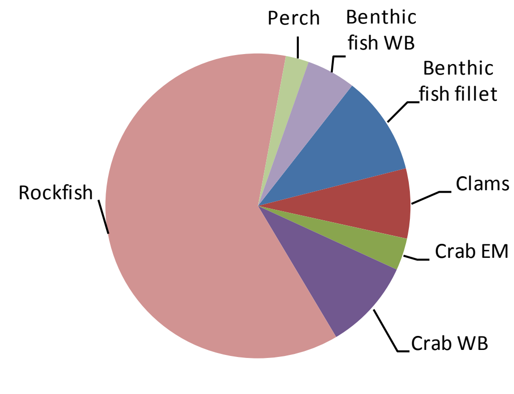
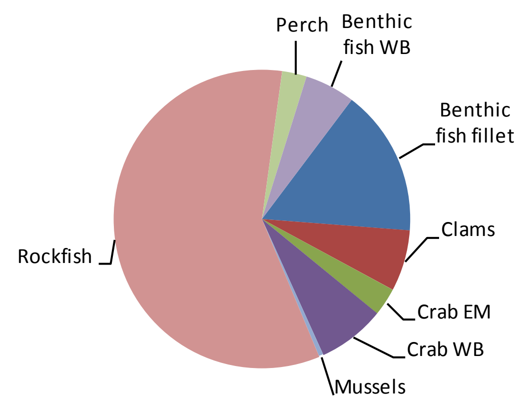
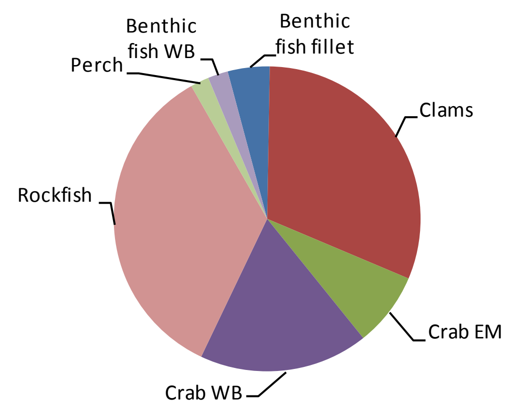
Consumption Category	EPC (mg/kg ww)	Ingestion Rate (g/day)	Percentage of Chemical's Risk	Excess Cancer Risk	Non-Cancer Hazard Quotient	Adult API RME Seafood Consumption Risk Expressed as Pie Chart[a]
PCB TEQ[c]						
Benthic fish fillet	1.5×10^{-5}	2.2	10.6%	3×10^{-5}	na	
Benthic fish WB	3.7×10^{-5}	0.4	4.8%	2×10^{-5}	na	
Clams	7.3×10^{-7}	32	7.5%	2×10^{-5}	na	
Crab EM	1.7×10^{-6}	6.3	3.4%	1×10^{-5}	na	
Crab WB	5.6×10^{-6}	5.4	9.7%	3×10^{-5}	na	
Pelagic fish, rockfish	4.0×10^{-5}	4.5	61.7%	2×10^{-4}	na	
Pelagic fish, perch	1.4×10^{-5}	0.5	2.3%	7×10^{-6}	na	
			Total risk from PCB TEQ	**3×10^{-4}**	**na**	
Total PCBs						
Benthic fish fillet	2.4	2	15.9%	7×10^{-5}	4	
Benthic fish WB	4.1	0.4	5.5%	2×10^{-5}	1	
Clams	0.069	29.1	6.7%	3×10^{-5}	2	
Crab EM	0.16	5.7	3.0%	1×10^{-5}	0.7	
Crab WB	0.45	4.9	7.3%	3×10^{-5}	2	
Mussels	0.031	4.6	0.5%	2×10^{-6}	0.1	
Pelagic fish, rockfish	4.0	4.4	58.5%	2×10^{-4}	14	
Pelagic fish, perch	1.6	0.5	2.7%	1×10^{-5}	0.6	
			Total risk from total PCBs	4×10^{-4}	24	

Table B.5-21. Comparison of excess cancer and non-cancer risks by consumption category for select chemicals in the adult API RME seafood consumption scenario (cont.)

Consumption Category	EPC (mg/kg ww)	Ingestion Rate (g/day)	Percentage of Chemical's Risk	Excess Cancer Risk	Non-Cancer Hazard Quotient	Adult API RME Seafood Consumption Risk Expressed as Pie Chart[a]
Dioxin/Furan TEQ[c]						
Benthic fish fillet	7.9×10^{-7}	2.2	4.5%	2×10^{-6}	na	
Benthic fish WB	3.8×10^{-7}	0.4	2.0%	8×10^{-7}	na	
Clams	4.9×10^{-7}	32	31.3%	1×10^{-5}	na	
Crab EM	1.3×10^{-6}	6.3	7.9%	3×10^{-6}	na	
Crab WB	2.5×10^{-7}	5.4	18.0%	7×10^{-6}	na	
Pelagic fish, rockfish	2.8×10^{-6}	4.5	34.5%	1×10^{-5}	na	
Pelagic fish, perch	1.4×10^{-6}	0.5	1.8%	7×10^{-7}	na	
Total risk from dioxin/furan TEQ				**4×10^{-5}**	**na**	



[a] Pie charts represent both cancer and non-cancer risks. Risk percentages are based on EPCs and ingestion rates, meaning that the percentage of risk from each consumption category is the same for cancer and non-cancer risks.

[b] Arsenic EPCs and risk estimates are based on inorganic arsenic.

[c] No mussel data were available for this chemical. When the CDI and risk values were calculated, the proportion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

API – Asian and Pacific Islander
CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
EM – edible meat
EPC – exposure point concentration
na – not available
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent
WB – whole body
ww – wet weight

Table B.5-22. Comparison of excess cancer and non-cancer risks by consumption category for select chemicals in the adult API CT seafood consumption scenario

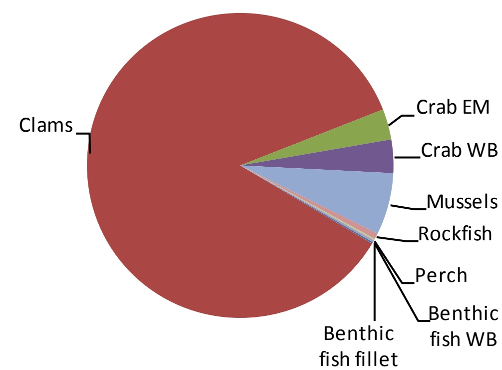
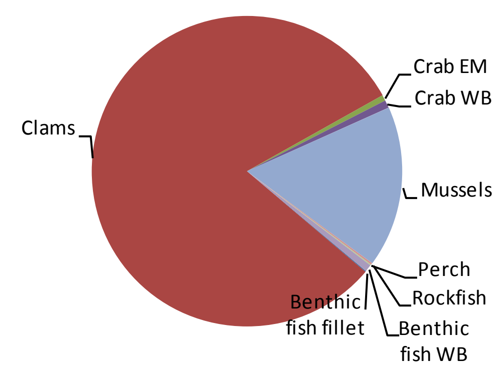
Consumption Category	EPC (mg/kg ww)[a]	Ingestion Rate (g/day)	Percentage of Chemical's Risk	Excess Cancer Risk	Non-Cancer Hazard Quotient	Adult API CT Seafood Consumption Risk Expressed as Pie Chart[b]
Arsenic[c]						
Benthic fish fillet	0.0050	0.2	0.2%	3×10^{-9}	0.00005	
Benthic fish WB	0.032	0.04	0.2%	4×10^{-9}	0.0001	
Clams	0.17	3	85.6%	2×10^{-6}	0.03	
Crab EM	0.032	0.6	3.2%	6×10^{-8}	0.001	
Crab WB	0.042	0.5	3.5%	6×10^{-8}	0.001	
Mussels	0.078	0.5	6.5%	1×10^{-7}	0.002	
Pelagic fish, rockfish	0.0080	0.45	0.6%	1×10^{-8}	0.0002	
Pelagic fish, perch	0.021	0.05	0.2%	3×10^{-9}	0.0001	
			Total risk from arsenic	**2×10^{-6}**	**0.03**	
cPAH TEQ						
Benthic fish fillet	0.00029	0.2	0.1%	9×10^{-10}	na	
Benthic fish WB	0.011	0.04	0.7%	7×10^{-9}	na	
Clams	0.016	3	80.7%	7×10^{-7}	na	
Crab EM	0.00060	0.6	0.6%	5×10^{-9}	na	
Crab WB	0.00096	0.5	0.8%	7×10^{-9}	na	
Mussels	0.020	0.5	16.8%	1×10^{-7}	na	
Pelagic fish, rockfish	0.00024	0.45	0.2%	2×10^{-9}	na	
Pelagic fish, perch	0.0012	0.05	0.1%	9×10^{-10}	na	
			Total risk from cPAH TEQ	**9×10^{-7}**	**na**	

Table B.5-22. Comparison of excess cancer and non-cancer risks by consumption category for select chemicals in the adult API CT seafood consumption scenario (cont.)

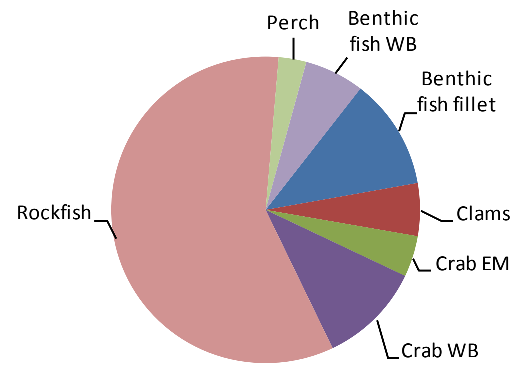
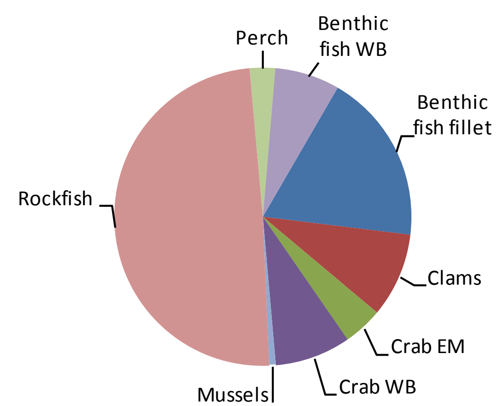
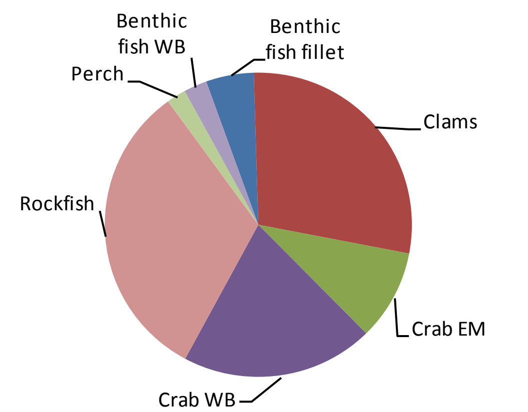
Consumption Category	EPC (mg/kg ww)[a]	Ingestion Rate (g/day)	Percentage of Chemical's Risk	Excess Cancer Risk	Non-Cancer Hazard Quotient	Adult API CT Seafood Consumption Risk Expressed as Pie Chart[b]
PCB TEQ[d]						
Benthic fish fillet	1.3×10^{-5}	0.2	10.6%	8×10^{-7}	na	
Benthic fish WB	3.5×10^{-5}	0.04	5.7%	4×10^{-7}	na	
Clams	4.1×10^{-7}	3.3	5.5%	4×10^{-7}	na	
Crab EM	1.6×10^{-6}	0.7	4.5%	3×10^{-7}	na	
Crab WB	4.8×10^{-6}	0.6	11.7%	9×10^{-7}	na	
Pelagic fish, rockfish	2.9×10^{-5}	0.5	58.9%	4×10^{-6}	na	
Pelagic fish, perch	1.3×10^{-5}	0.06	3.2%	2×10^{-7}	na	
			Total risk from PCB TEQ	**8×10^{-6}**	**na**	
Total PCBs						
Benthic fish fillet	1.7	0.2	18.6%	1×10^{-6}	0.3	
Benthic fish WB	3.2	0.04	7.0%	5×10^{-7}	0.1	
Clams	0.056	3	9.2%	7×10^{-7}	0.1	
Crab EM	0.13	0.6	4.3%	3×10^{-7}	0.06	
Crab WB	0.30	0.5	8.2%	6×10^{-7}	0.1	
Mussels	0.026	0.5	0.7%	5×10^{-8}	0.01	
Pelagic fish, rockfish	2.0	0.45	49.3%	4×10^{-6}	0.7	
Pelagic fish, perch	1.0	0.05	2.7%	2×10^{-7}	0.04	
			Total risk from total PCBs	**7×10^{-6}**	**1**	

Table B.5-22. Comparison of excess cancer and non-cancer risks by consumption category for select chemicals in the adult API CT seafood consumption scenario (cont.)

Consumption Category	EPC (mg/kg ww)[a]	Ingestion Rate (g/day)	Percentage of Chemical's Risk	Excess Cancer Risk	Non-Cancer Hazard Quotient	Adult API CT Seafood Consumption Risk Expressed as Pie Chart[b]
Dioxin/Furan TEQ[d]						
Benthic fish fillet	7.5×10^{-7}	0.2	4.5%	5×10^{-8}	na	
Benthic fish WB	1.8×10^{-7}	0.04	2.2%	2×10^{-8}	na	
Clams	2.8×10^{-7}	3.3	27.9%	3×10^{-7}	na	
Crab EM	4.7×10^{-7}	0.7	9.9%	1×10^{-7}	na	
Crab WB	1.2×10^{-7}	0.6	21.7%	2×10^{-7}	na	
Pelagic fish, rockfish	2.1×10^{-6}	0.5	31.7%	3×10^{-7}	na	
Pelagic fish, perch	1.2×10^{-6}	0.06	2.2%	2×10^{-8}	na	
			Total risk from dioxin/furan TEQ	**1×10^{-6}**	**na**	

[a] EPCs used for the CT scenarios were mean values.

[b] Pie charts represent both cancer and non-cancer risks. Risk percentages are based on EPCs and ingestion rates, meaning that the percentage of risk from each consumption category is the same for cancer and non-cancer risks.

[c] Arsenic EPCs and risk estimates are based on inorganic arsenic.

[d] No mussel data were available for this chemical. When the CDI and risk values were calculated, the proportion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
CT – central tendency
EM – edible meat
EPC – exposure point concentration
na – not available
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
WB – whole body
ww – wet weight

For the adult and child tribal RME and CT consumption scenarios based on Tulalip data, the majority (approximately 80 to 90%) of risk estimated for arsenic and cPAH TEQ was attributable to clam consumption (Tables B.5-18 and B.5-19). Clams have higher arsenic and cPAH concentrations than do other seafood categories, and consumption rates for clams were higher than those for other seafood categories. For PCB TEQ for these scenarios, pelagic fish (perch) and benthic fish fillet each contributed approximately 30% of the PCB TEQ risk estimates, even though these consumption categories were a relatively small portion of consumption by mass (Tables B.5-18 and B.5-19). Crabs (including both whole body and edible meat) also contributed significantly to the estimated PCB TEQ excess cancer risk. For total PCBs in the adult tribal scenarios based on Tulalip data, the same categories (i.e., pelagic fish [perch], benthic fish fillet, and crab) were key contributors, with benthic fish fillet contributing the most, followed by pelagic fish (perch) and crabs (both whole body and edible meat). For dioxin/furan TEQ, the main contributors to risk were crab (edible meat and whole body) and clams.

The relative contribution of the different seafood categories to risks associated with arsenic and cPAH TEQ was similar for the adult API scenarios (Tables B.5-21, and B.5-22) and the tribal scenarios based on Tulalip data, with the exception that mussels were a more significant contributor to the cPAH TEQ risks for the adult API scenarios. The relative contribution of consumption categories for PCBs and dioxin/furan TEQ was also different for the adult API scenarios (Tables B.5-21, and B.5-22). These differences were primarily the result of differences in ingestion rates. The API scenario assumed more consumption of whole-body fish than the tribal scenarios and assumed no consumption of geoduck.[41] The majority of PCB risks were the result of rockfish consumption, with lesser contributions from benthic fish fillet, crab, and clam consumption. Dioxin/furan risks were largely the result of the consumption of rockfish, clams, and crabs.

There were significant differences between the adult tribal scenario based on Suquamish data (Table B.5-20) and other scenarios. Over 94% of the arsenic and cPAH TEQ excess cancer risk for the adult tribal scenario based on Suquamish data was attributable to clams, which constituted the majority of their seafood diet (393.7 of 583.5 g total daily non-anadromous seafood consumption). The concentration of inorganic arsenic in clams was over two times higher than that in other seafood categories. Rockfish was the main contributor to the overall PCB TEQ or total PCB risks (65% to 70% of the total), with lesser contributions from benthic fish fillets followed by clams. Although rockfish represented a much smaller portion of the diet in the adult tribal

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[41] Although geoduck consumption was reported in EPA's seafood consumption survey of APIs in King County (EPA 1999a), non-tribal groups, including API populations, do not have commercial harvesting rights to geoducks in the EW. Because of this and the fact that special equipment (including scuba gear) and training in its use are required to harvest geoducks, no EW geoduck consumption was assumed for the API scenario.

scenario based on Suquamish data (55.4 g/day, as compared with 393.7 g/day for clams), they were a major contributor to PCB and dioxin/furan risks because of the high concentrations of these chemicals in rockfish compared with clams.

As described above, the percent contribution of the various seafood consumption categories differed by scenario. The general patterns are described below by chemical.

- **Arsenic –** The main contributor for all scenarios was clams, contributing 80 to 94% of the inorganic arsenic risk. Section B.6.2.6 provides a brief discussion of the percentage of total arsenic that is inorganic.
- **cPAH TEQ –** The main contributor for all scenarios was clams, contributing 73 to 95% of the cPAH TEQ risk.
- **PCBs (both total PCBs and PCB TEQ) –** The main contributors differed greatly by scenario, with benthic fish fillet contributing 11 to 42%, perch contributing less than 1 to 28%, and rockfish contributing 8 to 70%.
- **Dioxin/furan TEQ –** The main contributors differed by scenario, with clams contributing 21 to 40%, crab edible meat contributing 5 to 24%, whole-body crab contributing 4 to 22%, and rockfish contributing 5 to 41%.

Whole-body benthic fish and mussels were the only consumption categories not identified as being major contributors to the risk for any of these chemicals, primarily because of the low proportion of consumption rates for these categories.

B.5.3.1.5 Influence of chemicals with low detection frequencies on seafood risk estimates

Risks were estimated for chemicals detected in at least one tissue sample; however, many chemicals were detected in only one or a few of the seafood categories included in each of the risk estimates. As described in Section B.3.3.4, if a chemical was not detected in a seafood category, one-half the highest RL was selected as the EPC.[42] In cases where the RLs for undetected analytes were high, the related risk estimates could be based predominantly on one-half the highest RL for a particular consumption category. Of the 28 COPCs detected in tissue, 18 were not detected in at least one of the 10 seafood consumption categories.

To investigate whether RLs for COPCs with low detection frequencies might have influenced risk estimates for individual COPCs, the percentage of each chemical's risk that was attributed to consumption categories with no detected concentrations (i.e., the EPC was based on one-half the highest RL) was evaluated for several seafood consumption scenarios. For the adult and child tribal scenarios based on Tulalip data

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[42] The highest RL from the available samples in the dataset was selected as the EPC, as discussed in Section B.3.3.4. This should not be confused with the selection of the lowest RL as the best result for a given sample. As discussed in Section B.2.2.1, when multiple results were available because of the analysis of a laboratory duplicate or replicate, the lowest RL was selected as the best result if all of the results for a given sample were non-detects.

(CT and RME), greater than 50% of the risk for 9 of the 28 detected COPCs was derived from consumption categories in which the COPC was not detected, although only 5 of these COPCs had excess cancer risks greater than 1×10^{-6} (see Table B.5-23). For the adult tribal scenario based on Suquamish data, greater than 50% of the risk for 9 of the 28 detected COPCs was derived from consumption categories in which the COPC was not detected; 8 of these COPCs had excess cancer risks greater than 1×10^{-6} (see Table B.5-23). For the adult API scenarios (CT and RME), greater than 50% of the risk for 7 of the 28 detected COPCs was derived from consumption categories in which the COPC was not detected, although only 1 of these COPCs (dieldrin) had an excess cancer risk greater than 1×10^{-6} (see Table B.5-23). In addition, it should be noted that the COPCs identified in Table B.5-23 were not major contributors to the total risk, with each contributing less than 1% to the total excess cancer risk, and these COPCs had HQs that were well below 1 (all were less than or equal to 0.2).

Table B.5-23. Summary of chemicals with greater than 50% of risk derived from seafood categories with no detected values

Chemical	Seafood Consumption Scenarios with Greater than 50% of Risk from Consumption Categories with No Detected Values[a]							Seafood Consumption Categories with No Detected Values									
	Adult Tribal RME (Tulalip Data)	Adult Tribal CT (Tulalip Data)	Child Tribal RME (Tulalip Data)	Child Tribal CT (Tulalip Data)	API RME	API CT	Adult Tribal (Suquamish Data)	Benthic Fish, Fillet	Benthic Fish, WB	Clams	Crab, EM	Crab, WB	Geoduck, EM	Geoduck, WB	Mussels	Pelagic Fish. Perch	Pelagic Fish, Rockfish
Organometals																	
Dibutyltin	*	*	*	*	na	na	*	X	X	X	X		X			X	
SVOCs																	
1,4-Dichlorobenzene	*	*	*	*	*	*	7×10^{-6}	X		X	X	X	X	X	X	X	X
Pentachlorophenol	2×10^{-6}	*	*	*	na	na	2×10^{-5}	X	X		X	X	X	X	X	X	X
Pesticides																	
alpha-BHC	4×10^{-6}	*	*	*	*	*	2×10^{-5}	X	X	X	X	X			X	X	
beta-BHC	*	*	*	*	*	*	7×10^{-6}	X	X	X	X	X			X	X	X
Dieldrin	8×10^{-6}	*	*	*	2×10^{-6}	*	5×10^{-5}			X	X	X	X	X	X		
Heptachlor	*	*	*	*	*	*	7×10^{-6}	X	X	X	X	X	X	X		X	X
Heptachlor epoxide	2×10^{-6}	*	*	*	*	*	1×10^{-5}	X	X	X	X		X	X	X	X	
Mirex	4×10^{-6}	*	*	*	*	*	3×10^{-5}	X	X	X	X	X	X	X		X	

[a] Chemicals with greater than 50% of risk from consumption categories with no detected values that also have excess cancer risks greater than 1×10^{-6} are noted in the table by showing the excess cancer risk estimate; those with excess cancer risks less than or equal to 1×10^{-6} or with HQs less than or equal to 1 are noted in the table with an asterisk (*). None of the chemicals in this table have non-cancer HQs greater than 1.

API – Asian and Pacific Islander
BHC – benzene hexachloride
CT – central tendency
EM – edible meat
EPC – exposure point concentration
HQ – hazard quotient
na – not applicable
RME – reasonable maximum exposure
SVOC – semivolatile organic compound
WB – whole body

The chemicals identified in Table B.5-23 were also identified with footnotes in Tables B.5-1 to B.5-16. Table B.5-23 also presents the seafood categories for which there were no detected concentrations. The chemicals identified in Table B.5-23 are primarily organochlorine pesticides.

RLs also influenced EPCs when there was a mixture of detected and non-detected concentrations within a seafood category. For over 20 chemicals, the detection frequency was less than 50% in at least one seafood category. However, there were relatively few cases where the selected EPC was greater than the maximum detected concentration (only eight chemical-consumption category pairs). In most cases, the selected EPC was only slightly greater than the maximum detected concentration; thus, these EPCs are not expected to significantly impact risk estimates.

B.5.3.2 Direct sediment exposure

This section presents excess cancer risks and non-cancer HQs for the netfishing, habitat restoration worker, and clamming exposure scenarios, along with a summary of the percent contribution of COPCs to the total excess cancer risk for each of these scenarios.

B.5.3.2.1 Netfishing

Netfishing risks were estimated for the entire EW (intertidal and subtidal) for both the RME and CT scenarios. Overall, the risk estimates associated with netfishing were much lower than those for seafood consumption. For the netfishing RME scenario, the total excess cancer risk for both the incidental sediment ingestion and dermal absorption exposure routes was 7×10^{-6}, regardless of the summation approach (Table B.5-24). Arsenic and cPAH TEQ had excess cancer risks greater than 1×10^{-6} and were thus identified as COCs for this scenario.

Table B.5-24. Excess cancer risk estimates for the netfishing RME scenario based on exposure by incidental sediment ingestion and dermal absorption

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Receptor population: Commercial fisherman
Receptor age: Adult

Chemical	EPC (mg/kg dw)[a]	Cancer CDI (mg/kg-day)		Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk		
		Incidental Ingestion	Dermal Absorption		Incidental Ingestion	Dermal Absorption	Total
Arsenic	12	1.5×10^{-6}	6.5×10^{-7}	1.5	2×10^{-6}	1×10^{-6}	3×10^{-6}
cPAH TEQ	0.88	1.1×10^{-7}	2.1×10^{-7}	7.3	8×10^{-7}	2×10^{-6}	3×10^{-6}
1,4-Dichlorobenzene	0.41	5.1×10^{-8}	7.4×10^{-8}	0.0054	3×10^{-10}	4×10^{-10}	7×10^{-10}
Total PCBs	0.79	9.9×10^{-8}	2.0×10^{-7}	2	2×10^{-7}	4×10^{-7}	6×10^{-7}
PCB TEQ	5.6×10^{-6}	7.0×10^{-13}	1.4×10^{-12}	150,000	1×10^{-7}	2×10^{-7}	3×10^{-7}
Dioxin/furan TEQ	1.9×10^{-5}	2.4×10^{-12}	1.0×10^{-12}	150,000	4×10^{-7}	2×10^{-7}	6×10^{-7}
Total TEQ excess cancer risk for dioxins/furans and coplanar PCBs							**9×10^{-7}**
Total excess cancer risk (excluding PCB TEQ)							**7×10^{-6}**
Total excess cancer risk (excluding total PCBs)							**7×10^{-6}**

[a] All EPCs are medium-specific, rather than route-specific (i.e., the same EPC was used for the dermal absorption and incidental sediment ingestion exposure routes).

CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
dw –dry weight
EPC – exposure point concentration
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent

The total excess cancer risk estimate for the netfishing CT scenario was 1×10^{-6}, regardless of the summation approach (Table B.5-25), and therefore no COPCs exceeded the excess cancer risk threshold for this CT scenario.

Table B.5-25. Excess cancer risk estimates for the netfishing CT scenario based on exposure by incidental sediment ingestion and dermal absorption

Scenario timeframe: Current/future **Medium:** Sediment Exposure medium: Sediment Receptor population: Commercial fisherman Receptor age: Adult

Chemical	EPC (mg/kg dw)[a]	Cancer CDI (mg/kg-day)		Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk		
		Incidental Ingestion	Dermal Absorption		Incidental Ingestion	Dermal Absorption	Total
Arsenic	10	4.4×10^{-7}	1.9×10^{-8}	1.5	7×10^{-7}	3×10^{-8}	7×10^{-7}
cPAH TEQ	0.51	2.2×10^{-8}	4.2×10^{-9}	7.3	2×10^{-7}	3×10^{-8}	2×10^{-7}
1,4-Dichlorobenzene	0.12	5.2×10^{-9}	7.6×10^{-10}	0.0054	3×10^{-11}	4×10^{-12}	3×10^{-11}
Total PCBs	0.54	2.4×10^{-8}	4.8×10^{-9}	2.0	5×10^{-8}	1×10^{-8}	6×10^{-8}
PCB TEQ	4.4×10^{-6}	1.9×10^{-13}	3.9×10^{-14}	150,000	3×10^{-8}	6×10^{-9}	4×10^{-8}
Dioxin/furan TEQ	1.6×10^{-5}	7.0×10^{-13}	3.0×10^{-14}	150,000	1×10^{-7}	5×10^{-9}	1×10^{-7}
Total TEQ excess cancer risk for dioxins/furans and coplanar PCBs							$\mathbf{1 \times 10^{-7}}$
Total excess cancer risk (excluding PCB TEQ)							$\mathbf{1 \times 10^{-6}}$
Total excess cancer risk (excluding total PCBs)							$\mathbf{1 \times 10^{-6}}$

[a] All EPCs are medium-specific, rather than route-specific (i.e., the same EPC was used for the dermal absorption and incidental sediment ingestion exposure routes).

CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
CT – central tendency
dw –dry weight
EPC – exposure point concentration
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

Non-cancer hazards for the netfishing scenario were found to be below levels of concern, with the sum of HQs across all COPCs for both RME and CT scenarios significantly less than 1 (Tables B.5-26 and B.5-27). Effect-specific HIs were not calculated because the sum of HQs across all effects was less than 1 (see Section B.5.1.2). No non-cancer COCs were identified for the netfishing RME scenario.

Table B.5-26. Non-cancer hazard estimates for the netfishing RME scenario based on exposure by incidental sediment ingestion and dermal absorption

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Receptor population: Commercial fisherman
Receptor age: Adult

Chemical	EPC (mg/kg dw)[a]	Non-Cancer CDI (mg/kg-day)		Reference Dose (mg/kg-day)	Hazard Quotient		
		Incidental Ingestion	Dermal Absorption		Incidental Ingestion	Dermal Absorption	Total
Antimony	43	8.6×10^{-6}	na[b]	0.0004	0.02	na[b]	0.02
Arsenic	12	2.4×10^{-6}	1.0×10^{-6}	0.0003	0.008	0.003	0.01
Vanadium	59	1.2×10^{-5}	na[b]	0.009	0.001	na[b]	0.001
1,4-Dichlorobenzene	0.41	8.2×10^{-8}	1.2×10^{-7}	0.07	1×10^{-6}	2×10^{-6}	3×10^{-6}
Total PCBs	0.79	1.6×10^{-7}	3.2×10^{-7}	0.00002	0.008	0.02	0.03
Sum of HQs							**0.06**

a All EPCs are medium-specific, rather than route-specific (i.e., the same EPC was used for the dermal absorption and incidental sediment ingestion exposure routes).

b No absorption factor is available for this chemical. Dermal exposure for this chemical is discussed in the uncertainty analysis (Section B.6).

CDI – chronic daily intake
dw –dry weight
EPC – exposure point concentration
HQ – hazard quotient
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure

Table B.5-27. Non-cancer hazard estimates for the netfishing CT scenario based on exposure by incidental sediment ingestion and dermal absorption

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Receptor population: Commercial fisherman
Receptor age: Adult

Chemical	EPC (mg/kg dw)[a]	Non-Cancer CDI (mg/kg-day)		Reference Dose (mg/kg-day)	Hazard Quotient		
		Incidental Ingestion	Dermal Absorption		Incidental Ingestion	Dermal Absorption	Total
Antimony	4.9	5.2×10^{-7}	na[b]	0.0004	0.001	na[b]	0.001
Arsenic	10	1.1×10^{-6}	4.6×10^{-8}	0.0003	0.004	0.0002	0.004
Vanadium	57	6.0×10^{-6}	na[b]	0.009	0.0007	na[b]	0.0007
1,4-Dichlorobenzene	0.12	1.3×10^{-8}	1.8×10^{-9}	0.07	2×10^{-7}	3×10^{-8}	2×10^{-7}
Total PCBs	0.54	5.7×10^{-8}	1.1×10^{-8}	0.00002	0.003	0.0006	0.003
Sum of HQs							**0.009**

[a] All EPCs are medium-specific, rather than route-specific (i.e., the same EPC was used for the dermal absorption and incidental sediment ingestion exposure routes).

[b] No absorption factor is available for this chemical. Dermal exposure for this chemical is discussed in the uncertainty analysis (Section B.6).

CDI – chronic daily intake
CT – central tendency
dw –dry weight
EPC – exposure point concentration
HQ – hazard quotient
PCB – polychlorinated biphenyl

B.5.3.2.2 Habitat restoration worker

Habitat restoration worker risks were estimated for the intertidal portion of the EW, as shown in Map B.3-1. Overall, the total excess cancer risk for the combined exposure routes (dermal absorption and incidental sediment ingestion) was equal to 1×10^{-6}, regardless of the summation approach (Table B.5-28). Therefore, none of the COPCs had excess cancer risks greater than EPA's risk threshold of 1×10^{-6}.

Table B.5-28. Excess cancer risk estimates for the habitat restoration worker scenario based on exposure by incidental sediment ingestion and dermal absorption

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Receptor population: Restoration worker
Receptor age: Adult

Chemical	EPC (mg/kg dw)[a]	Cancer CDI (mg/kg-day)		Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk		
		Incidental Ingestion	Dermal Absorption		Incidental Ingestion	Dermal Absorption	Total
Arsenic	15	2.5×10^{-7}	8.9×10^{-8}	1.5	4×10^{-7}	1×10^{-7}	5×10^{-7}
cPAH TEQ	2.3	3.8×10^{-8}	5.9×10^{-8}	7.3	3×10^{-7}	4×10^{-7}	7×10^{-7}
Total PCBs	1.9	3.1×10^{-8}	5.3×10^{-8}	2	6×10^{-8}	1×10^{-7}	2×10^{-7}
PCB TEQ	7.2×10^{-6}	1.2×10^{-13}	2.0×10^{-13}	150,000	2×10^{-8}	3×10^{-8}	5×10^{-8}
Total excess cancer risk (excluding PCB TEQ)							**1×10^{-6}**
Total excess cancer risk (excluding total PCBs)							**1×10^{-6}**

[a] All EPCs are medium-specific, rather than route-specific (i.e., the same EPC was used for the dermal absorption and incidental sediment ingestion exposure routes).

CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
dw –dry weight
EPC – exposure point concentration
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

Non-cancer HQs for individual chemicals for the habitat restoration worker scenario were less than 1 (Table B.5-29). Effect-specific HIs were not calculated because the sum of HQs across all effects was less than 1 (see Section B.5.1.2).

Table B.5-29. Non-cancer hazard estimates for the habitat restoration worker scenario based on exposure by incidental sediment ingestion and dermal absorption

Scenario timeframe: Current/future **Medium:** Sediment Exposure medium: Sediment Receptor population: Restoration worker Receptor age: Adult

Chemical	EPC (mg/kg dw)[a]	Non-Cancer CDI (mg/kg-day)		Reference Dose (mg/kg-day)	Hazard Quotient		
		Incidental Ingestion	Dermal Absorption		Incidental Ingestion	Dermal Absorption	Total
Arsenic	15	8.6×10^{-7}	3.1×10^{-7}	0.0003	0.003	0.001	0.004
Vanadium	51	2.9×10^{-6}	na[b]	0.009	0.0003	na[b]	0.0003
Total PCBs	1.9	1.1×10^{-7}	1.8×10^{-7}	0.00002	0.005	0.009	0.01
Sum of HQs							**0.01**

[a] All EPCs are medium-specific, rather than route-specific (i.e., the same EPC was used for the dermal absorption and incidental sediment ingestion exposure routes).

[b] No absorption factor is available for this chemical. Dermal exposure for this chemical is discussed in the uncertainty analysis (Section B.6).

CDI – chronic daily intake
dw –dry weight
EPC – exposure point concentration
HQ – hazard quotient
na – not available
PCB – polychlorinated biphenyl

B.5.3.2.3 Clamming

Risks were estimated for three clamming scenarios, including a tribal clamming RME scenario (120 days per year), a tribal clamming 183-days-per-year scenario, and a 7-day-per-year clamming scenario, as described in Section B.3.3.2. The clamming scenarios presented in this section consist only of exposures to sediment via incidental ingestion and dermal contact. The consumption of clams is evaluated in the seafood consumption scenarios. The effect of summing risks from clam collection with clam consumption is evaluated in Section B.5.6.3.

For the tribal clamming RME scenario, the total excess cancer risk for the combined incidental sediment ingestion and dermal absorption exposure routes was 3×10^{-5}, regardless of the summation approach (Table B.5-30). Arsenic, cPAH TEQ, and total PCBs had risks greater than 1×10^{-6}, and were thus identified as COCs for this scenario. In addition, although PCB TEQ and dioxin/furan TEQ individually did not exceed this threshold for the tribal clamming RME scenario; when these two TEQ risks were combined, the excess cancer risk was equal to 2×10^{-6}.Thus both of PCB and dioxin/furan TEQ were identified as COCs for this scenario.

Table B.5-30. Excess cancer risk estimates for the tribal clamming RME scenario (120 days per year) based on incidental sediment ingestion and dermal absorption

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Receptor population: Tribal/subsistence clam collectors
Receptor age: Adult

Chemical	EPC (mg/kg dw)[a]	Cancer CDI (mg/kg-day)		Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk		
		Incidental Ingestion	Dermal Absorption		Incidental Ingestion	Dermal Absorption	Total
Arsenic	15	5.5×10^{-6}	2.0×10^{-6}	1.5	8×10^{-6}	3×10^{-6}	1×10^{-5}
cPAH TEQ	2.3	8.5×10^{-7}	1.3×10^{-6}	7.3	6×10^{-6}	1×10^{-5}	2×10^{-5}
Total PCBs	1.9	7.0×10^{-7}	1.2×10^{-6}	2	1×10^{-6}	2×10^{-6}	3×10^{-6}
PCB TEQ	7.2×10^{-6}	2.6×10^{-12}	4.5×10^{-12}	150,000	4×10^{-7}	7×10^{-7}	1×10^{-6}
Dioxin/furan TEQ	1.6×10^{-5}	5.9×10^{-12}	2.1×10^{-12}	150,000	9×10^{-7}	3×10^{-7}	1×10^{-6}
Total TEQ excess cancer risk for dioxins/furans and coplanar PCBs							**2×10^{-6}**
Total excess cancer risk (excluding PCB TEQ)							**3×10^{-5}**
Total excess cancer risk (excluding total PCBs)							**3×10^{-5}**

[a] All EPCs are medium-specific, rather than route-specific (i.e., the same EPC was used for the dermal absorption and incidental sediment ingestion exposure routes).

CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
dw –dry weight
EPC – exposure point concentration
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent

For the tribal clamming 183-day-per-year scenario, the total excess cancer risk for the incidental ingestion and dermal absorption pathways was 5×10^{-5} or 6×10^{-5}, depending on the summation approach (Table B.5-31). Risks for all five COPCs were greater than 1×10^{-6}, with the highest risks being associated with arsenic and cPAH TEQ. The total excess cancer risk estimate for the 7-day-per-year clamming scenario was 1×10^{-6}, regardless of the summation approach, and thus no COPCs exceeded the risk threshold (Table B.5-32).

Table B.5-31. Excess cancer risk estimates for the tribal clamming 183-day-per-year scenario based on incidental sediment ingestion and dermal absorption

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Receptor population: Tribal/subsistence clam collectors
Receptor age: Adult

Chemical	EPC (mg/kg dw)[a]	Cancer CDI (mg/kg-day)		Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk		
		Incidental Ingestion	Dermal Absorption		Incidental Ingestion	Dermal Absorption	Total
Arsenic	15	9.2×10^{-6}	3.3×10^{-6}	1.5	1×10^{-5}	5×10^{-6}	2×10^{-5}
cPAH TEQ	2.3	1.4×10^{-6}	2.2×10^{-6}	7.3	1×10^{-5}	2×10^{-5}	3×10^{-5}
Total PCBs	1.9	1.2×10^{-6}	2.0×10^{-6}	2	2×10^{-6}	4×10^{-6}	6×10^{-6}
PCB TEQ	7.2×10^{-6}	4.4×10^{-12}	7.5×10^{-12}	150,000	7×10^{-7}	1×10^{-6}	2×10^{-6}
Dioxin/furan TEQ	1.6×10^{-5}	9.8×10^{-12}	3.6×10^{-12}	150,000	1×10^{-6}	5×10^{-7}	2×10^{-6}
Total TEQ excess cancer risk for dioxins/furans and coplanar PCBs							**4×10^{-6}**
Total excess cancer risk (excluding PCB TEQ)							**6×10^{-5}**
Total excess cancer risk (excluding total PCBs)							**5×10^{-5}**

[a] All EPCs are medium-specific, rather than route-specific (i.e., the same EPC was used for the dermal absorption and incidental sediment ingestion exposure routes).

CDI – chronic daily intake

cPAH – carcinogenic polycyclic aromatic hydrocarbon

dw –dry weight

EPC – exposure point concentration

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

Table B.5-32. Excess cancer risk estimates for the 7-day-per-year clamming scenario based on incidental sediment ingestion and dermal absorption

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Receptor population: Occasional clam collectors
Receptor age: Adult

Chemical	EPC (mg/kg dw)[a]	Cancer CDI (mg/kg-day)		Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk		
		Incidental Ingestion	Dermal Absorption		Incidental Ingestion	Dermal Absorption	Total
Arsenic	16	1.8×10^{-7}	6.6×10^{-8}	1.5	3×10^{-7}	1×10^{-7}	4×10^{-7}
cPAH TEQ	2.6	3.0×10^{-8}	4.7×10^{-8}	7.3	2×10^{-7}	3×10^{-7}	5×10^{-7}
Total PCBs	2.0	2.3×10^{-8}	3.9×10^{-8}	2	5×10^{-8}	8×10^{-8}	1×10^{-7}
PCB TEQ	6.1×10^{-6}	7.0×10^{-14}	1.2×10^{-13}	150,000	1×10^{-8}	2×10^{-8}	3×10^{-8}
Dioxin/furan TEQ	1.6×10^{-5}	1.8×10^{-13}	6.6×10^{-14}	150,000	3×10^{-8}	1×10^{-8}	4×10^{-8}
Total TEQ excess cancer risk for dioxins/furans and coplanar PCBs							$\mathbf{7 \times 10^{-8}}$
Total excess cancer risk (excluding PCB TEQ)							$\mathbf{1 \times 10^{-6}}$
Total excess cancer risk (excluding total PCBs)							$\mathbf{1 \times 10^{-6}}$

[a] All EPCs are medium-specific, rather than route-specific (i.e., the same EPC was used for the dermal absorption and incidental sediment ingestion exposure routes).

CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
dw –dry weight
EPC – exposure point concentration
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

For all of the clamming scenarios, HQs for all chemicals were less than 1 (Tables B.5-33 to B.5-35). Effect-specific HIs were not calculated because the sum of HQs across all endpoints did not exceed 1. No non-cancer COCs were identified for clamming.

Table B.5-33. Non-cancer hazard estimates for the tribal clamming RME scenario (120 days per year) based on incidental sediment ingestion and dermal absorption

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Receptor population: Tribal/subsistence clam collectors
Receptor age: Adult

Chemical	EPC (mg/kg dw)[a]	Non-Cancer CDI (mg/kg-day)		Reference Dose (mg/kg-day)	Hazard Quotient		
		Incidental Ingestion	Dermal Absorption		Incidental Ingestion	Dermal Absorption	Total
Arsenic	15	6.0×10^{-6}	2.2×10^{-6}	0.0003	0.02	0.007	0.03
Cobalt	7.1	2.9×10^{-6}	na[b]	0.0003	0.01	na[b]	0.01
Vanadium	51	2.0×10^{-5}	na[b]	0.009	0.002	na[b]	0.002
Total PCBs	1.9	7.6×10^{-7}	1.3×10^{-6}	0.00002	0.04	0.06	0.1
Sum of HQs							**0.1**

a All EPCs are medium-specific, rather than route-specific (i.e., the same EPC was used for the dermal absorption and incidental sediment ingestion exposure routes).

b No absorption factor is available for this chemical. Dermal exposure for this chemical is discussed in the uncertainty analysis (Section B.6).

CDI – chronic daily intake

dw –dry weight

EPC – exposure point concentration

HQ – hazard quotient

PCB – polychlorinated biphenyl

RME – reasonable maximum exposure

Table B.5-34. Non-cancer hazard estimates for the tribal clamming 183-day-per-year scenario based on incidental sediment ingestion and dermal absorption

Scenario timeframe: Current/future **Medium:** Sediment Exposure medium: Sediment **Receptor population:** Tribal/subsistence clam collectors Receptor age: Adult

Chemical	EPC (mg/kg dw)[a]	Non-Cancer CDI (mg/kg-day)		Reference Dose (mg/kg-day)	Hazard Quotient		
		Incidental Ingestion	Dermal Absorption		Incidental Ingestion	Dermal Absorption	Total
Arsenic	15	9.2×10^{-6}	3.3×10^{-6}	0.0003	0.03	0.01	0.04
Cobalt	7.1	4.4×10^{-6}	na[b]	0.0003	0.01	na[b]	0.01
Vanadium	51	3.1×10^{-5}	na[b]	0.009	0.003	na[b]	0.003
Total PCBs	1.9	1.2×10^{-6}	2.0×10^{-6}	0.00002	0.06	0.1	0.2
Sum of HQs							**0.3**

a All EPCs are medium-specific, rather than route-specific (i.e., the same EPC was used for the dermal absorption and incidental sediment ingestion exposure routes).

b No absorption factor is available for this chemical. Dermal exposure for this chemical is discussed in the uncertainty analysis (Section B.6).

CDI – chronic daily intake

dw –dry weight

EPC – exposure point concentration

HQ – hazard quotient

PCB – polychlorinated biphenyl

Table B.5-35. Non-cancer hazard estimates for the 7-day-per-year clamming scenario based on incidental sediment ingestion and dermal absorption

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Sediment
Receptor population: Occasional clam collectors
Receptor age: Adult

Chemical	EPC (mg/kg dw)[a]	Non-Cancer CDI (mg/kg-day)		Reference Dose (mg/kg-day)	Hazard Quotient		
		Incidental Ingestion	Dermal Absorption		Incidental Ingestion	Dermal Absorption	Total
Arsenic	16	4.3×10^{-7}	1.5×10^{-7}	0.0003	0.001	0.0005	0.002
Cobalt	6.6	1.8×10^{-7}	na[b]	0.0003	0.0006	na[b]	0.0006
Vanadium	48	1.3×10^{-6}	na[b]	0.009	0.0001	na[b]	0.0001
Total PCBs	2.0	5.3×10^{-8}	9.0×10^{-8}	0.00002	0.003	0.005	0.008
Sum of HQs							**0.01**

[a] All EPCs are medium-specific, rather than route-specific (i.e., the same EPC was used for the dermal absorption and incidental sediment ingestion exposure routes).

[b] No absorption factor is available for this chemical. Dermal exposure for this chemical is discussed in the uncertainty analysis (Section B.6).

CDI – chronic daily intake
dw –dry weight
EPC – exposure point concentration
HQ – hazard quotient
PCB – polychlorinated biphenyl

B.5.3.2.4 Percent contribution of COPCs to the direct sediment exposure total risk

As was done for the seafood consumption scenarios, an evaluation of risk contribution was performed for the direct sediment exposure scenarios (Table B.5-36 and Figure B.5-2). Figure B.5-2 shows both the risk magnitude and the percent risk by chemical for each exposure scenario. Arsenic, cPAH TEQ, dioxins/furans, and PCBs were evaluated because these chemicals had excess cancer risks greater than 1×10^{-6} and each contributed greater than 5% of the total excess cancer risk for at least one scenario. Note that only arsenic, cPAH TEQ, and total PCBs had excess cancer risks greater than 1×10^{-6} for at least one of the RME scenarios. In addition, these chemicals were detected in greater than 10% of EW sediment samples.

Table B.5-36. Contribution to risks by chemical for direct sediment exposure scenarios with total excess cancer risks greater than the 1×10^{-6} threshold

Chemical	Percentage of Contribution to Total Excess Cancer Risk		
	Netfishing RME	Tribal Clamming RME	Tribal Clamming – 183 Days per Year
Total Excess Cancer Risk (excluding PCB TEQ)			
Arsenic	42%	29%	34%
cPAH TEQ	42%	59%	52%
Total PCBs	8%[a]	9%	10%
Dioxin/furan TEQ	8%[a]	3%[a]	3%
Other COPCs[b]	0.01%[a]	na	na
Total Excess Cancer Risk (excluding total PCBs)			
Arsenic	43%	31%	37%
cPAH TEQ	43%	63%	56%
PCB TEQ	4%[a]	3%	4%
Dioxin/furan TEQ	9%[a]	3%[a]	4%
Other COPCs[b]	0.01%[a]	na	na

[a] Excess cancer risks for these chemical-scenario combinations did not exceed the risk threshold of 1×10^{-6}.
[b] Includes all other COPCs detected in sediment.
COPC – chemical of potential concern
CT – central tendency
cPAH – carcinogenic polycyclic aromatic hydrocarbon
na – not applicable
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent

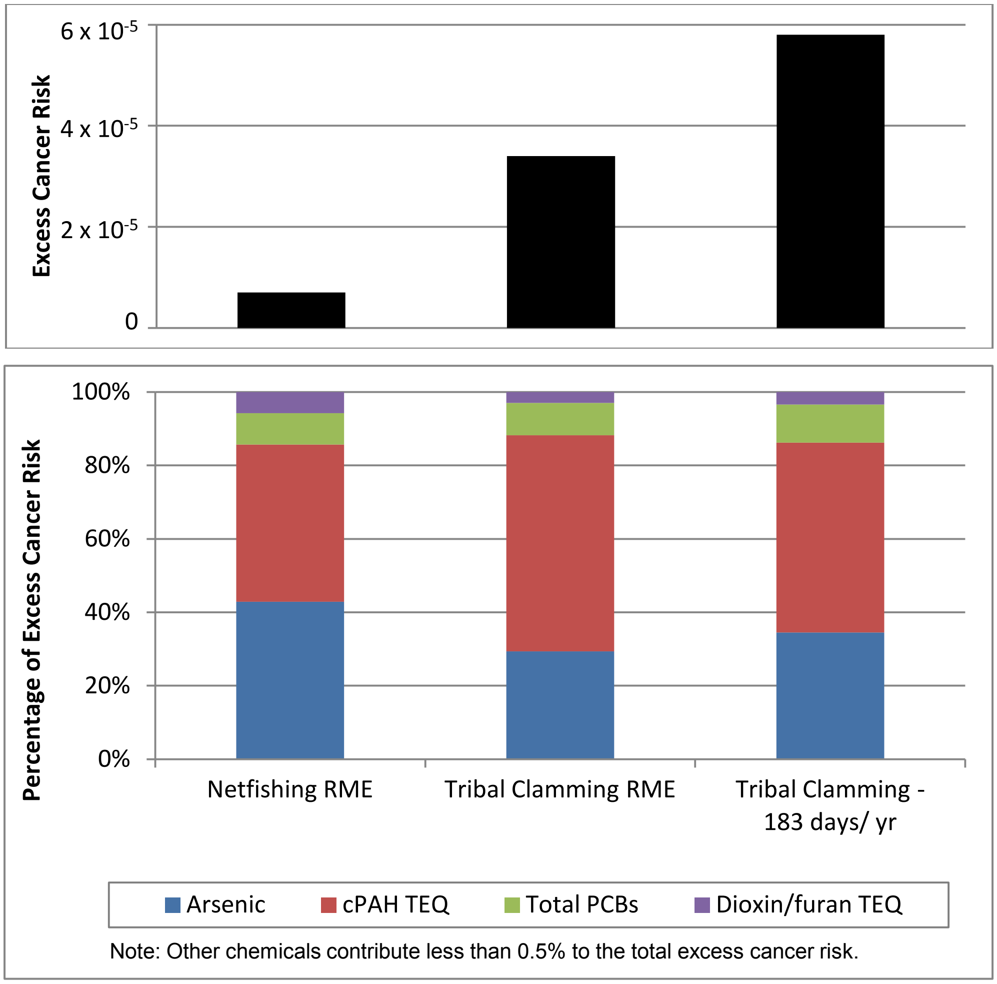


Figure B.5-2. Excess cancer risks by chemical for direct sediment exposure scenarios with total excess cancer risks greater than the 1×10^{-6} threshold

The percent contribution by chemical was fairly consistent across the six sediment exposure scenarios. Arsenic and cPAH TEQ were the main contributors to the total excess cancer risk for the sediment exposure scenarios, contributing 29 to 43% for arsenic and 42 to 63% for cPAH TEQ for the three scenarios with total excess cancer risks greater than the 1×10^{-6} threshold (Figure B.5-2). PCBs and dioxins/furans contributed to a lesser extent.

B.5.3.3 Surface water exposure

Risks to individuals based on exposure to surface water while swimming were estimated for the EW, as described in Section B.3.3.3. The swimming scenario presented in this section consists of exposure to surface water via incidental ingestion and dermal absorption for three levels of exposure (high, medium, and low). These three levels of exposure were evaluated to provide a range of possible risk estimates and for consistency with the King County WQA for the Duwamish River and Elliott Bay (King County 1999a). However, it should be noted that these levels of exposure are likely a significant overestimate of swimming exposure levels for the EW, given that these levels of exposure were developed for areas (e.g., Elliott Bay) that include a larger number of recreational access points than does the EW and do not include surface water within navigation channels that are used by large ships and tug boats.

For the low level of swimming exposure (a 10-minute swim, 2 days per year for 9 years), the total excess cancer risk was 2×10^{-8}. No COPCs exceeded the cancer risk threshold for this exposure level. For the medium level of swimming exposure (a 1-hour swim, 12 days per year for 30 years) and high level of swimming exposure (a 2.6-hour swim, 24 days per year for 70 years), the total excess cancer risk estimates were equal to 2×10^{-6} and 9×10^{-6}, respectively, for the summation approach that excluded total PCBs (Table B.5-37). For these totals, PCB TEQ contributed over 95% of the excess cancer risk and was the only COPC with an excess cancer risk greater than the 1×10^{-6} risk threshold. Although the PCB TEQ excess cancer risk was greater than 1×10^{-6} for the medium and high levels of exposure, these swimming risk estimates are considered to be highly uncertain based on two factors:

Current and anticipated future site use – Given the limited number of public recreational access points along the EW and the waterway's use as an active shipping channel, the EW is not known to be used for swimming at the present time and is not an expected to be used for swimming in the future. Thus, risks calculated for the swimming scenario, particularly for the medium level of exposure (which assumes 1 hour of swimming, 12 times per year over a period of 30 years) and the high level of exposure (which assumes 2.6 hours of swimming, 24 times per year over a period of 70 years), likely represents an overestimate of risk based on surface water exposure.

Application of dioxin-like TEQ approach for dermal exposure – Considerable uncertainty is associated with the application of the dioxin-like TEQ approach for dermal exposure (discussed in Section B.6.2.3), which contributed the nearly all (over 99%) of the PCB TEQ swimming risk (as compared with the incidental ingestion of water). The dioxin-like TEQ approach was developed for the consideration of the risk associated with the consumption of tissue (Van den Berg et al. 2006), and its applicability to dermal absorption exposure is uncertain because bioavailability for non-dietary exposures is not well characterized.

Based on these uncertainties and because no RME level of exposure was defined for the EW, no COCs were identified for the swimming scenario. The calculation of risks associated with swimming was intended to provide a general idea of the risks associated with surface water exposure, not to quantify a level of EW exposure that has been observed or is expected to occur in the future.

Table B.5-37. Excess cancer risk estimates for the swimming scenario based on exposure by incidental surface water ingestion and dermal absorption

Scenario timeframe: Current/future Medium: Water Exposure medium: Surface water Receptor population: Swimmers Receptor age: Adult

Chemical	EPC (mg/L)	Cancer CDI (mg/kg-day)		Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk		
		Incidental Ingestion	Dermal Absorption		Incidental Ingestion	Dermal Absorption	Total
High Level of Exposure (2.6 hours/event, 24 days/year, 70 years)							
Arsenic[a]	0.0012	2.1×10^{-7}	6.1×10^{-8}	1.5	3×10^{-7}	9×10^{-8}	4×10^{-7}
Total PCBs	1.5×10^{-6}	2.5×10^{-10}	1.3×10^{-7}	2	5×10^{-10}	3×10^{-7}	3×10^{-7}
PCB TEQ	6.1×10^{-10}	1.0×10^{-13}	5.7×10^{-11}	150,000	2×10^{-8}	9×10^{-6}	9×10^{-6}
Total excess cancer risk (excluding PCB TEQ)							**7×10^{-7}**
Total excess cancer risk (excluding total PCBs)							**9×10^{-6}**
Medium Level of Exposure (1 hour/event, 12 days/year, 30 years)							
Arsenic[a]	0.0012	1.2×10^{-8}	1.2×10^{-8}	1.5	2×10^{-8}	2×10^{-8}	4×10^{-8}
Total PCBs	1.5×10^{-6}	1.5×10^{-11}	2.5×10^{-8}	2	3×10^{-11}	5×10^{-8}	5×10^{-8}
PCB TEQ	6.1×10^{-10}	6.0×10^{-15}	1.1×10^{-11}	150,000	9×10^{-10}	2×10^{-6}	2×10^{-6}
Total excess cancer risk (excluding PCB TEQ)							**9×10^{-8}**
Total excess cancer risk (excluding total PCBs)							**2×10^{-6}**
Low Level of Exposure (0.17 hour/event, 2 days/year, 9 years)							
Arsenic[a]	0.0012	4.9×10^{-11}	1.5×10^{-10}	1.5	7×10^{-11}	2×10^{-10}	3×10^{-10}
Total PCBs	1.5×10^{-6}	6.3×10^{-14}	3.1×10^{-10}	2	1×10^{-13}	6×10^{-10}	6×10^{-10}
PCB TEQ	6.1×10^{-10}	2.5×10^{-17}	1.4×10^{-13}	150,000	4×10^{-12}	2×10^{-8}	2×10^{-8}
Total excess cancer risk (excluding PCB TEQ)							**9×10^{-10}**
Total excess cancer risk (excluding total PCBs)							**2×10^{-8}**

[a] Exposure to surface water is based on total water concentrations, not dissolved concentrations.

CDI – chronic daily intake

EPC – exposure point concentration

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

Non-cancer HQs for the swimming scenario were less than 1 for all three levels of exposure, as was the sum of HQs across COPCs (Table B.5-38). Effect-specific HIs were not calculated because the sum of HQs across all effects was less than 1 (see Section B.5.1.2).

Table B.5-38. Non-cancer hazard estimates for the swimming scenario based on exposure by incidental surface water ingestion and dermal absorption

Scenario timeframe: Current/future
Medium: Water
Exposure medium: Surface water
Receptor population: Swimmers
Receptor age: Adult

Chemical	EPC (mg/L)	Non-Cancer CDI (mg/kg-day)		Reference Dose (mg/kg-day)	Hazard Quotient		
		Incidental Ingestion	Dermal Absorption		Incidental Ingestion	Dermal Absorption	Total
High Level of Exposure (2.6 hours/event, 24 days/year, 70 years)							
Arsenic[a]	0.0012	2.0×10^{-7}	6.1×10^{-8}	0.0003	0.0007	0.0002	0.0009
Chromium[a]	0.0011	1.9×10^{-7}	5.8×10^{-8}	0.003	0.00006	0.0008	0.0009
Vanadium[a]	0.0039	6.6×10^{-7}	2.0×10^{-7}	0.009	0.00007	0.0009	0.001
Naphthalene	0.0049	8.2×10^{-7}	1.6×10^{-5}	0.02	0.00004	0.0008	0.0008
Total PCBs	1.5×10^{-6}	2.5×10^{-10}	1.3×10^{-7}	0.00002	0.00001	0.006	0.006
Sum of HQs							0.01
Medium Level of Exposure (1 hour/event, 12 days/year, 30 years)							
Arsenic[a]	0.0012	2.7×10^{-8}	2.7×10^{-8}	0.0003	0.00009	0.00009	0.0002
Chromium[a]	0.0011	2.6×10^{-8}	2.6×10^{-8}	0.003	0.000009	0.0003	0.0003
Vanadium[a]	0.0039	9.0×10^{-8}	9.1×10^{-8}	0.009	0.00001	0.0004	0.0004
Naphthalene	0.0049	1.1×10^{-7}	7.2×10^{-6}	0.02	0.000006	0.0004	0.0004
Total PCBs	1.5×10^{-6}	$3.4 \times 10\text{-}^{11}$	5.8×10^{-8}	0.00002	0.000002	0.003	0.003
Sum of HQs							0.004
Low Level of Exposure (0.17 hour/event, 2 days/year, 9 years)							
Arsenic[a]	0.0012	3.8×10^{-10}	1.1×10^{-9}	0.0003	0.000001	0.000004	0.000005
Chromium[a]	0.0011	3.6×10^{-10}	1.1×10^{-9}	0.003	0.0000001	0.00001	0.00001
Vanadium[a]	0.0039	1.3×10^{-9}	3.8×10^{-9}	0.009	0.0000001	0.00002	0.00002
Naphthalene	0.0049	1.6×10^{-9}	3.0×10^{-7}	0.02	0.00000008	0.00002	0.00002
Total PCBs	1.5×10^{-6}	4.9×10^{-13}	2.4×10^{-9}	0.00002	0.00000002	0.0001	0.0001
Sum of HQs							0.0002

[a] Exposure to surface water is based on total water concentrations, not dissolved concentrations.

CDI – chronic daily intake
EPC – exposure point concentration
HQ – hazard quotient
PCB – polychlorinated biphenyl

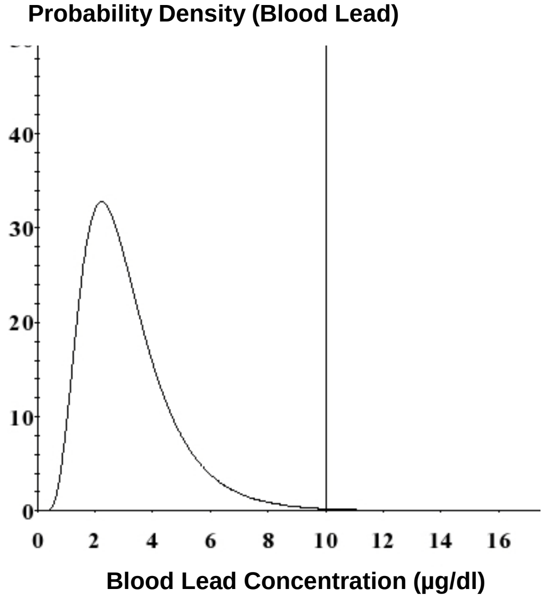
B.5.4 LEAD

As described in Section B.3.3.5, risks from exposure to lead were not quantified following the methods used for other COPCs. Because the toxicokinetics (absorption, distribution, metabolism, and excretion) of lead are well understood, health risks from lead exposure were evaluated based on blood lead concentration, which can be modeled. The results of blood lead modeling for children (IEUBK) and adults (ALM) are presented in the following subsections.

B.5.4.1 Children

The IEUBK lead model (Version 1.1, Build 11 for Windows®) was run using default parameters, except for the inclusion of site-specific tissue concentration data, as described earlier in Table B.3-48 and B.3-49. Model output was provided in the form of a probability density curve that described the probability of blood lead concentrations occurring in a hypothetical population of children. The Centers for Disease Control and Prevention (CDC) has established 10 µg/dl as a level-of-concern threshold for children blood lead concentrations; appropriate medical follow-up is warranted for concentrations above this level.

A probability density curve designates the percentage of children that are predicted to have blood lead concentrations that may exceed the threshold. A probability density curve was generated for the EW using site-specific dietary data for the child tribal RME seafood consumption scenario based on Tulalip data (Figure B.5-3) (see Section B.3.3.5.1 for further explanation of the approach to calculating site-specific dietary values). The IEUBK model was run using alternative dietary data for seafood consumption (see Table B.3-48 and B.3-49). Based on this level of exposure, less than 1% of the modeled child population would have blood lead concentrations that exceeded the CDC level of concern (which would be shown in Figure B.5-3 as the area under the curve to the right of the vertical line, which represents 10 µg/dl). EPA's risk reduction goal for contaminated sites is that no more than 5% of the population of children exposed to lead will have blood lead concentrations greater than 10 µg/dl. Based on the results of the IEUBK model for the EW, lead is not considered to be a COC in the EW.



Cutoff = 10.000 µg/dl
Geo mean = 2.880
GSD = 1.600
% Above = 0.405
% Below = 99.595

Figure B.5-3. Probability density curve for predicted blood lead concentrations using input data from children's seafood consumption

B.5.4.2 Adults

The ALM was run to estimate risks from lead exposure to the most sensitive population, which is a developing fetus. Lead risks were assessed by estimating the probability of exceeding the threshold blood lead concentration of 10 µg/dl in the fetus through the evaluation of the exposure of a pregnant mother. Results for the netfishing RME scenario and tribal clamming RME scenario are presented in Table B.5-39. As described in Section B.3.3.5.2, the model was run in two modes (with and without seafood consumption) so that the incremental effects of seafood consumption could be evaluated. The risks from lead exposure in the other sediment exposure scenarios are not presented because the lead concentrations in sediment for these scenarios were lower than those for the scenarios used in the modeling.

Table B.5-39. Risk estimates for predicted fetal and adult lead concentrations using the adult lead model

Model Output Categories	Unit	Concentration or Percentage by Scenario	
		Tribal Clamming RME	Netfishing RME
Incidental Ingestion of Sediment Alone			
Estimated adult blood lead concentrations, CTE[a]	µg/dl	1.6	1.6
Estimated fetal blood lead concentrations, 95th percentile[b]	µg/dl	5.7	5.5
Probability of fetal blood lead concentrations exceeding 10 µg/dl (lognormal)[c]	%	0.98	0.91
Incidental Ingestion of Sediment and Adult Tulalip RME seafood consumption			
Estimated adult blood lead concentrations, CTE[d]	µg/dl	1.9	1.8
Estimated fetal blood lead concentrations, 95th percentile[b]	µg/dl	6.6	6.5
Probability of fetal blood lead concentrations exceeding 10 µg/dl (lognormal)[c]	%	1.6	1.5

a CTE of adult blood lead concentration is only for sediment intake.
b Estimate of 95th percentile fetal blood lead concentration.
c Probability of exceeding EPA's threshold for fetal exposure, a blood lead level of 10 µg/dl (EPA 2003c).
d CTE of adult blood lead concentration for sediment ingestion and seafood consumption.
CTE – central tendency estimate
RME – reasonable maximum exposure

The 95th percentiles of predicted blood lead concentrations for the developing fetus ranged from 5.5 to 5.7 µg/dl for the scenarios that assumed no EW seafood consumption (Table B.5-39). The probability of exceeding the 10 µg/dl blood lead threshold was less than 1% for both the tribal clamming RME and netfishing RME scenarios. The 95th percentiles of blood lead concentrations and probabilities of exceeding the 10 µg/dl threshold were slightly higher when the seafood consumption scenario was included. However, the probability for exceeding the 10 µg/dl blood lead threshold remained less than 2%. These results are consistent with the results from the IEUBK model and indicate that lead in the EW is not considered to be a COC for human health.

B.5.4.3 Neurological effects associated with exposure to lead

As noted in Section B.4, lead is a known neurotoxicant, and thus lead was identified in Table B.4-3 as having neurological effects. However, because the models used to evaluate exposure to lead for children and adults do not result in the calculation of an HQ, it is not possible to evaluate the cumulative effects on the nervous system from lead and the other COPCs with neurological effects (i.e., mercury, total PCBs, and selenium). Rather, potential cumulative effects must be discussed qualitatively.

Nervous system HIs ranged from 25 to 59 for the three RME seafood consumption scenarios,[43] over 95% of which was based on the HQ for total PCBs. The lead model results for adults and children (presented in Sections B.5.4.1 and B.5.4.2, respectively) indicate that there is less than a 2% chance that blood lead levels would exceed the threshold. This is well below EPA's target that the probability of exceedance be no more than 5% for lead. Compared with the HQs for total PCBs that are well above EPA's threshold of an HQ of 1, the low exceedance probabilities for lead indicate that the absence of lead from the neurological HI sums in Tables B.5-9 through B.5-16 does not represent a significant underestimate of non-cancer neurological effects. However, because no threshold for lead toxicity has been identified, exposure to lead at any concentration may cause a small degree of neurological dysfunction.

B.5.5 Background Concentrations for Determining Incremental Risk

CERCLA includes provisions for evaluating chemical concentrations in background areas and distinguishes between natural background and anthropogenic background. Anthropogenic background is defined as the combination of both natural and anthropogenic substances present in the environment as a result of human activities not specifically related to the CERCLA release in question. The chemicals for which an evaluation of background is appropriate are determined on a site-specific basis based on the most significant contributors to the total risk estimates and a preliminary evaluation of the magnitude of differences between site and background concentrations.

Although CERCLA allows for the consideration of background concentrations in the risk assessment, the evaluation of background in this HHRA is not intended or designed for the purpose of selecting cleanup levels under CERCLA. The background evaluations in this HHRA are intended only to provide additional information relevant to the exposure and risk estimates, as was done in the LDW HHRA (Windward 2007c). Background data evaluation in support of CERCLA determinations of cleanup levels will be provided in the RI and/or FS reports.

Consistent with CERCLA guidance and the LDW HHRA (Windward 2007c), this section provides an incremental risk evaluation for arsenic. According to EPA (2002c) guidance, the risks from chemicals with suspected background sources are characterized initially without consideration of those background sources, as was done in Section B.5.3. Next, a separate risk estimate may be made for assumed exposures to these chemicals from background sources. Finally, background risks are compared with the EW risks presented in Section B.5.3. The difference between these two estimates, if any, is called the incremental risk and is assumed to be the result of sources related to the site and therefore referred to as the site-related risk.

[43] The sum of HQs (across all endpoints) were less than 1 for all the direct sediment exposure and swimming scenarios, and thus no endpoint specific HIs were calculated for these scenarios. Adverse neurological effects are not expected from exposure to EW sediment or surface water.

The four chemicals with the highest contributions to total excess cancer risk estimates for multiple exposure pathways and scenarios are arsenic, cPAH TEQ, PCBs, and dioxins/furans. These chemicals are known to be present in sediment from background areas of Puget Sound (Windward 2010g) and in upstream sediment from the Green/Duwamish River.[44] The relationship between EW and background (or upstream) sediment and tissue concentrations are described below by chemical.

- **Arsenic –** Tissue and sediment concentrations over most of the EW are relatively similar to anthropogenic background tissue and upstream sediment concentrations, with only a small number of EW tissue and sediment samples exhibiting substantially higher concentrations, and thus incremental risk estimates are presented for both sediment and tissue.
- **PCBs and cPAHs –** Tissue and sediment concentrations over broad areas of the EW are well above those found in background areas, so no incremental risk estimates are presented in this section (Windward 2010g; Malcolm Pirnie 2005; PSAT 2007; WSDOH 2009).[45]
- **Dioxins/furans –** An initial evaluation of tissue and sediment dioxin/furan concentrations in the EW indicate that there are similarities between EW concentrations and anthropogenic background concentrations. However, sediment dioxin/furan concentrations in the EW are above those in sediment upstream of the LDW Superfund site and sediment in Puget Sound that has been sampled for natural background considerations (EPA 2008). For tissue, it was difficult to determine an appropriate background dataset with sufficient samples to conduct an incremental risk analysis. Thus, no evaluation of background concentrations of dioxins/furans is presented in this HHRA. Instead, background for dioxins and furans will be discussed in the RI and/or FS.

The incremental risk analyses presented in this section were completed using available background tissue data or upstream sediment data. Sampling focused on collecting data on inorganic arsenic in tissue from background areas was conducted in 2004 and 2005 as part of the LDW HHRA (Windward 2007c) and the LDW Phase 2 RI work plan (Windward 2004). A number of studies have involved the collection of sediment arsenic data upstream of the LDW Superfund site, as discussed below. The background/ upstream datasets for arsenic are discussed briefly in the Sections B.5.5.1.1 (arsenic sediment analysis) and B.5.5.1.2 (arsenic tissue analysis) and in more detail in Attachment 5.

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[44] The Green/Duwamish River that is referenced here is upstream of the LDW Superfund site.

[45] EW EPCs for sediment (PCBs and cPAH TEQ) and for tissue (PCBs) were compared with values presented in the background section of the LDW RI (Windward 2010g). EW EPCs for cPAH TEQ in tissue were compared with several sources, including background values presented in the RI for the marine environment near the former Rayonier Mill site in Port Angeles, Washington (Malcolm Pirnie 2005), the health consultation for the Wyckoff/Eagle Harbor Superfund site (WSDOH 2009), and the 2007 Puget Sound update report (PSAT 2007).

B.5.5.1 Arsenic incremental risk analysis

Excess cancer risks for arsenic exceeded 1×10^{-6} for most of the direct sediment exposure scenarios and exceeded 1×10^{-5} for most of the seafood consumption scenarios. These risk estimates do not take into account the percentage of the total arsenic risk in the EW that may be attributable to background sources, such as arsenic that occurs naturally in the Puget Sound Basin or arsenic from anthropogenic (i.e., man-made) sources outside the EW. All discussions of background levels for arsenic presented here should be considered preliminary, and these levels will be re-evaluated in the RI and FS.

Arsenic occurs naturally in all sediment and soil worldwide (e.g., in Puget Sound sediment as part of native rock and natural geological features, such as volcanoes) (NOAA and Ecology 1999, 2000, 2002). In addition, historical anthropogenic sources within the region have contributed additional arsenic in some areas of the EW. For example, a recently completed soil survey for arsenic and lead in south King County suggested that the former Asarco smelter, which was located in Ruston, Washington, is likely one of the sources responsible for elevated arsenic and lead concentrations in soil throughout the Duwamish River watershed (Pacific Groundwater Group and TeraStat 2005).

B.5.5.1.1 Sediment

The incremental risk assessment of arsenic in sediment was conducted using all available data from the Green/Duwamish River upstream of the LDW and the EW because it is affected by similar natural (i.e., soils of volcanic origin) and anthropogenic sources outside the LDW and EW (i.e., the Asarco smelter plume). This approach is consistent with the LDW HHRA (Windward 2007c). A total of 24 samples from five sampling events from the LDW RI (Windward 2010g), along with 72 samples from a more recent Ecology upstream sampling event (Ecology and Environment 2009) were available for use in this assessment. A summary of these data is presented in Table B.5-40, and station locations are shown on Map B.5-1 (Windward 2005c). The full dataset is presented in Attachment 5.

Table B.5-40. Duwamish River surface sediment arsenic data collected upstream of the LDW and EW

Sampling Event	Sampling Year	No. of Samples	Range of Arsenic Concentrations (mg/kg dw)
Norfolk cleanup – 1	1994	2	11 – 22
Boeing site characterization	1997	3	4.5 – 7.2
EPA site investigation	1998	5	4 – 5.1
LDW RI – Round 2 surface sediment	2005	6	3.2 – 7.3

Table B.5-40. Duwamish River surface sediment arsenic data collected upstream of the LDW and EW (cont.)

Sampling Event	Sampling Year	No. of Samples	Range of Arsenic Concentrations (mg/kg dw)
LDW RI – surface sediment background	2005	8	4.6 – 10.9
Ecology upstream	2008	72	3.7 – 16
Overall Summary Statistics			
No. of samples			**96**
Range of concentrations			**3.2 – 22**
Mean			**6.8**
95% UCL (calculated using ProUCL)			**7.3**

dw – dry weight
EW – East Waterway
LDW – Lower Duwamish Waterway
RI – remedial investigation
UCL – upper confidence limit on the mean

The ProUCL-recommended upstream 95% UCL (7.3 mg/kg dw) was compared with arsenic EPCs for the direct sediment exposure scenarios with excess cancer risks greater than 1×10^{-6}, which ranged from 12 to 15 mg/kg dw (Table B.5-41). As noted previously, this background value should be considered preliminary. Additional data may be evaluated in the RI and FS to determine the appropriate background concentration for the EW.

Table B.5-41. Sediment arsenic EPCs for upstream and EW exposure areas

Exposure Scenario[a]	EW EPC (mg/kg dw)	Upstream 95% UCL (mg/kg dw)	Difference Between the EPC and the Upstream 95% UCL (mg/kg dw)
Netfishing RME	12	7.3	4.7
Tribal clamming RME (120 days per year)	15	7.3	7.7
Tribal clamming – 183 days per year	15	7.3	7.7

[a] Only those scenarios with excess cancer risks greater than 1×10^{-6} are presented.
CT – central tendency
dw – dry weight
EPC – exposure point concentration
EW – East Waterway
RME – reasonable maximum exposure
UCL – upper confidence limit on the mean

Excess cancer risks associated with upstream concentrations of arsenic were calculated for each of these EW direct sediment exposure scenarios with excess cancer risks greater than 1×10^{-6} by replacing each EW EPC with the upstream 95% UCL. Incremental cancer risks were then estimated for each scenario by subtracting the upstream-related risks from the scenario-related risks. The incremental risks for the direct sediment exposure scenarios with EW excess cancer risks greater than 1×10^{-6} are presented in Table B.5-42. Incremental risks for the netfishing and tribal clamming RME scenarios were 1×10^{-6} and 5×10^{-6}, respectively (approximately 33 to 50% of the EW excess cancer risk for arsenic). The incremental risk for the tribal clamming 183 day-per-year-scenario was 1×10^{-5} (approximately 50% of the EW excess cancer risk for arsenic).

Table B.5-42. Incremental excess cancer risks from exposure to arsenic in sediment

Exposure Scenario	EW Excess Cancer Risk Estimate	Upstream Excess Cancer Risk Estimate	Incremental Excess Cancer Risk Estimate
Netfishing RME	3×10^{-6}	2×10^{-6}	1×10^{-6}
Tribal clamming RME (120 days per year)	1×10^{-5}	5×10^{-6}	5×10^{-6}
Tribal clamming – 183 days per year	2×10^{-5}	9×10^{-6}	1×10^{-5}

[a] Only those scenarios with excess cancer risks greater than 1×10^{-6} are presented.
EW – East Waterway
RME – reasonable maximum exposure

B.5.5.1.2 Tissue

Background inorganic arsenic tissue data were collected as part of the LDW RI sampling program and was used in this HHRA as part of this preliminary background evaluation. The samples collected as part of the LDW background tissue evaluation are appropriate for use in the EW HHRA because the locations sampled are not influenced by the EW and the same types of fish, crab, and clams collected for these background tissue samples were collected in the EW.

The arsenic background dataset includes data from three field studies, which include a 2004 clam sampling event (Windward 2005a), a 2005 clam sampling event (Windward 2006a), and a 2004 fish and crab sampling event (Windward 2005b). The sampling locations are shown in Map B.5-2, and additional details regarding these sampling events are provided in Attachment 5. Data are available from two types of background locations: one that represents areas with only naturally occurring arsenic and another that represents areas that were affected by emissions from the Asarco smelter, which was located near Tacoma, Washington. Two types of locations were chosen for evaluation because arsenic concentrations may differ between the two environments, and EPA (2002d) acknowledges that both natural and anthropogenic sources may be relevant as background for risk characterization. As with the EW samples, only the inorganic arsenic tissue data from these background samples were used in this HHRA because EPA's cancer SF for estimating carcinogenic risks is specific to inorganic arsenic. In addition, it should be noted that no background mussel samples were available for use in this assessment.[46] In addition, no geoduck or rockfish background inorganic arsenic data were available because the background sampling events were conducted for the LDW HHRA, and these species were not evaluated as part of that assessment.

[46] Mussels represent a relatively small fraction of the total seafood consumption rate used for the seafood consumption scenarios (less than 10% of the total ingestion rate).

The inorganic arsenic EPCs for the EW and background tissue samples were calculated using ProUCL and are presented in Table B.5-43, along with sample counts and mean values. Additional statistics are presented in Attachment 5. In both the EW and background samples, inorganic arsenic concentrations in clams were the highest relative to other tissue types. However, inorganic arsenic concentrations in EW tissue samples are not elevated as compared with background concentrations (either for Asarco-influenced or non-Asarco-influenced background) for any tissue type.

Table B.5-43. Inorganic arsenic EPCs for tissue samples collected from the EW and background areas

Sampling Area	Species and/or Tissue Type	Number of Samples	Inorganic Arsenic Concentration (mg/kg ww)	
			Mean	EPC
Clams				
EW	four clam species[a]	12	0.17	0.22
Both Asarco-influenced areas (Vashon Island and Seahurst Park)	*Mya arenaria* and other species[b]	12	0.31	0.46
Both non-Asarco background areas (Dungeness NWR and Bainbridge Island)	*Mya arenaria* and other species[b]	12	0.12	0.28
Perch (shiner surfperch, pile perch, striped perch)				
EW	whole body plus fillet	8	0.021	0.027
Asarco-influenced background (East Passage)	whole body plus fillet	3	0.008	0.01 (max)
Background (Blake Island)	whole body plus fillet	6	0.02	0.03
Crabs (Dungeness crab, slender crab)				
EW	edible meat	9	0.032	0.036
	calculated whole body[c]	9	0.042	0.047
Asarco-influenced background (East Passage)	edible meat	6	0.018	0.03
	calculated whole body[c]	6	0.037	0.05
Background (Blake Island)	edible meat	6	0.023	0.03
	calculated whole body[c]	6	0.11	0.1
Benthic Fish (English sole)				
EW	fillet	11	0.0050	0.0045
	whole body	11	0.032	0.038
Asarco-influenced background (East Passage)	fillet	6	0.0019	0.004 (max)
	whole body	6	0.011	0.015
Background (Blake Island)	fillet	6	0.0026	0.004 (max)
	whole body	6	0.018	0.025

Note: Additional details regarding the background dataset are presented in Attachment 5. Sources of the background data include a 2004 clam sampling event (Windward 2005a), a 2005 clam sampling event (Windward 2006a), and a 2004 fish and crab sampling event (Windward 2005b).

a EW samples were available for the following species: butter clam (*Saxidomus giganteus*), Eastern soft-shell (*Mya arenaria*), cockle (*Clinocardium nuttali*), and native littleneck (*Protothaca staminea*).

b For this assessment, composite clam samples both of Eastern soft-shell clams (Mya arenaria) and of other clam species (including butter clam, cockle, Macoma species [*Macoma nasuta and Macoma secta]*, gaper clam [*Tresus capax*], and native littleneck) were used.

Table B.5-43. Inorganic arsenic EPCs for tissue samples collected from the EW and background areas (cont.)

[c] Data from hepatopancreas composite samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Whole-body (i.e., edible meat plus hepatopancreas) crab concentrations were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weight of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data).

EPC – exposure point concentration

EW – East Waterway

NWR – national wildlife refuge

ww – wet weight

For the seafood consumption exposure scenarios, incremental excess cancer risk estimates associated with arsenic were calculated by subtracting the background risk estimates from the EW risk estimates (Table B.5-44). Background inorganic arsenic tissue concentrations were not available for all consumption categories used to assess risks in the HHRA, including geoduck (edible meat and whole body), rockfish, and mussels. For geoduck and rockfish, substitute species were identified based on similarities in inorganic arsenic concentrations in the EW, as described below:

- **Geoduck –** In the EW, inorganic arsenic concentrations in geoduck and crab were similar, both for edible meat and whole-body samples. For geoduck and crab, EPCs for edible meat were equal to 0.044 and 0.036 mg/kg ww, respectively; and EPCs for whole-body samples were equal to 0.049 and 0.047 mg/kg ww, respectively. Thus, background inorganic arsenic concentrations in crab were used as a substitute for geoduck.
- **Rockfish –** In the EW, inorganic arsenic concentrations in perch and rockfish were relatively similar (EPCs were equal to 0.027 and 0.011 mg/kg ww, respectively). Although a larger difference exists between the perch and rockfish concentrations (as compared with the crab and geoduck concentrations), the similarities in the species (both are pelagic fish) are also important indicators of the suitability of using perch concentrations as a substitute for rockfish concentrations. Thus, background inorganic arsenic concentrations in perch were used as a substitute for rockfish.

It should be noted that geoduck and rockfish together accounted for less than 5% of the arsenic excess cancer risk for the RME scenarios in the EW (see Tables B.5-18 and B.5-21), and thus the absence of data for these consumption categories should not impact conclusions regarding background arsenic tissue concentrations.

To account for the lack of mussel data, the portion of the diet represented by mussels was distributed proportionally to the other consumption categories. This was the same as the approach taken for several COPCs (e.g., PCB TEQ) for which mussel data were unavailable.

Table B.5-44. Incremental cancer risk estimates associated with inorganic arsenic for the seafood consumption exposure scenarios

Exposure Scenario	Dietary Composition	EW Excess Cancer Risk Estimate	Background Type	Background Excess Cancer Risk Estimate	Incremental Excess Cancer Risk Estimate[a]
Adult tribal RME (Tulalip data)[b]	mixed diet	2×10^{-4}	Asarco-influenced	4×10^{-4}	0
			non-Asarco	2×10^{-4}	0
Adult tribal CT (Tulalip data)[b]	mixed diet	1×10^{-5}	Asarco-influenced	4×10^{-5}	0
			non-Asarco	2×10^{-5}	0
Child tribal RME (Tulalip data)[b]	mixed diet	4×10^{-5}	Asarco-influenced	7×10^{-5}	0
			non-Asarco	4×10^{-5}	0
Child tribal CT (Tulalip data)[b]	mixed diet	4×10^{-6}	Asarco-influenced	7×10^{-6}	0
			non-Asarco	3×10^{-6}	1×10^{-6}
Adult tribal (Suquamish data)[b]	mixed diet	2×10^{-3}	Asarco-influenced	4×10^{-3}	0
			non-Asarco	2×10^{-3}	0
Adult API RME[b]	mixed diet	8×10^{-5}	Asarco-influenced	2×10^{-4}	0
			non-Asarco	1×10^{-4}	0
Adult API CT[b]	mixed diet	2×10^{-6}	Asarco-influenced	3×10^{-6}	0
			non-Asarco	1×10^{-6}	1×10^{-6}
Adult one meal per month[c]	clam	1×10^{-5}	Asarco-influenced	3×10^{-5}	0
			non-Asarco	2×10^{-5}	0
	pelagic fish, perch	2×10^{-6}	Asarco-influenced	7×10^{-7}	1×10^{-6}
			non-Asarco	2×10^{-6}	0
	pelagic fish, rockfish	7×10^{-7}	Asarco-influenced	nd	nd
			non-Asarco	nd	nd
	crab	2×10^{-6}	Asarco-influenced	2×10^{-6}	0
			non-Asarco	2×10^{-6}	0
	benthic fish	3×10^{-7}	Asarco-influenced	3×10^{-7}	0
			non-Asarco	3×10^{-7}	0

a Incremental risk estimates were equal to zero when EW concentrations were equal to or less than background concentrations.

b No mussel data were available. When the CDI and risk values were calculated, the portion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories. In addition, surrogate species were used for consumption categories for which no background data were available (crab data were used to represent geoduck, and perch data were used to represent rockfish).

c The adult one-meal-per-month scenarios are presented for informational purposes only, and are not used by EPA for risk management decisions.

API – Asian and Pacific Islander
CDI – a chronic daily intake
CT – central tendency
EPA – US Environmental Protection Agency
EW – East Waterway
nd – no data
RME – reasonable maximum exposure

Incremental risks were equal to zero for most of the seafood consumption scenarios, regardless of the type of background data used for the comparison (i.e., Asarco-influenced or non-Asarco-influenced). The exceptions were for the child tribal CT scenario based on Tulalip data and the adult API CT scenario using the non-Asarco influenced background, as well as for the adult one-meal-per-month perch consumption scenario using the Asarco-influenced background concentrations. For these three cases, the incremental risk was equal to 1×10^{-6}. These results highlight the fact that inorganic arsenic concentrations in EW tissue samples are similar to those in background tissue samples.

B.5.5.2 Summary of incremental risk analysis for arsenic

Sections B.5.5.1.1 and B.5.5.1.2 presented an assessment of incremental risks (i.e., the difference between risks estimates for the EW and those calculated for background areas) for arsenic in sediment and tissue, respectively. As noted above, the discussions of background levels for arsenic presented here should be considered preliminary, and these levels will be re-evaluated in the RI and FS. Table B.5-45 provides a brief summary of the background data and results of this assessment. .

Table B.5-45. Summary of incremental risk analysis for arsenic

Scenario Type	Description of Background Data	Comparison of EW and Background	Incremental Excess Cancer Risks[a]
Direct sediment contact	samples collected from the Duwamish River (upstream of the EW and LDW)	Exposure concentrations in the EW (10 to 16 mg/kg dw) were higher than the upstream concentration (7.3 mg/kg ww).	1×10^{-6} to 1×10^{-5}
Seafood consumption	samples collected from Puget Sound to represent naturally occurring regional concentrations and Asarco-influenced concentrations	Inorganic arsenic concentrations in EW tissue samples were similar to those in background tissue samples.	0 for all except three scenarios, which were equal to 1×10^{-6} [b]

a The incremental excess cancer risk is equal to the EW risk minus the background risk, and was calculated only for scenarios with EW excess cancer risks greater than 1×10^{-6}.

b The child tribal CT scenario based on Tulalip data and the adult API CT scenario had incremental excess cancer risks greater than 0 (equal to 1×10^{-6}) using the non-Asarco influenced background (but had incremental risks of 0 using the Asarco-influenced background). The adult one-meal-per-month scenario based on the consumption of pelagic fish using the Asarco-influenced background concentration had an incremental excess cancer risk greater than 0 (equal to 1×10^{-6}) (but had incremental risks of 0 using non-Asarco-influenced background).

EPC – exposure point concentration

EW – East Waterway

LDW – Lower Duwamish Waterway

UCL – upper confidence limit on the mean

B.5.6 Risk Characterization Summary

This section summarizes the risk estimates for each scenario and discusses cumulative risks across multiple scenarios. Risks have been evaluated for a number of different types of exposure scenarios in order to describe different intensities of site use or seafood consumption. RME scenarios represent the highest exposures that are reasonably expected to occur at a site and are generally used by EPA to evaluate remedial actions at a site (EPA 1989). RME scenarios by definition likely overestimate exposure for many individuals. CT scenarios are intended to reflect average exposures. CT exposures and risks are not favored in decision-making because they will underestimate exposure and risk for a substantial number of individuals (EPA 1989). CT exposures and risks are useful in characterizing the exposure and risk range (National Research Council 1994). Another method of examining risk and exposure is to quantify risks based on a unit of exposure that a member of the public can then use to assess risks associated with their individual behavior. This last method was used to characterize seafood consumption risk on an individual basis, with the unit of exposure being one meal per month, and was also used to characterize risk associated with the collection of clams, with the unit of exposure equal to 7-days-per-year.

As discussed throughout Section B.5, chemicals with excess cancer risks greater than 1×10^{-6} or an HQ greater than 1 for any RME scenario were identified as COCs. Table B.5-46 presents a summary of the COCs identified by scenario.

Table B.5-46. Summary of COCs

COPC Selected as COCs for One or More RME Scenarios	Seafood Consumption Scenarios			Direct Sediment Exposure Scenarios	
	Adult Tribal RME (Tulalip Data)	Child Tribal RME (Tulalip Data)	Adult API RME	Netfishing RME	Tribal Clamming RME
Arsenic	X	X	X	X	X
Cadmium		X			
cPAH TEQ	X	X	X	X	X
Pentachlorophenol	X				
Total PCBs	X	X	X		X
PCB TEQ	X	X	X		
alpha-BHC	X				
Dieldrin	X		X		
Total chlordane	X				
Heptachlor epoxide	X				
Mirex	X				
Dioxin/furan TEQ	X	X	X		
Total TEQ[a]					X

[a] Total TEQ is equal to the sum of PCB TEQ and dioxin/furan TEQ. When excess cancer risks for either PCB TEQ or dioxin/furan TEQ were not independently greater than 1×10^{-6}, the sum of these two chemicals (total TEQ) was identified as a COC if it was greater than this threshold.

API – Asian and Pacific Islander
BHC – benzene hexachloride
COC – chemical of concern
COPC – chemical of potential concern
cPAH – carcinogenic polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent

The following subsections present a summary of the seafood consumption scenarios (Section B.5.6.1), a summary of the direct sediment contact and swimming scenarios (Section B.5.6.2), cumulative risk estimates across multiple exposure scenarios (Section B.5.6.3), and graphics that illustrate the relationship between consumption rates and risk estimates for the seafood consumption scenarios (Section B.5.6.4). It should be noted that the relationships between tissue and sediment concentrations of COCs have not yet been evaluated, but will be explored as part of the SRI.

B.5.6.1 Summary of seafood consumption scenarios

The excess cancer risk and non-cancer hazard estimates for the seafood consumption scenarios are summarized in Tables B.5-47 and B.5-48, respectively. For the purpose of brevity, chemical-specific risk and HQ estimates are provided only for chemicals that had a cancer risk estimate greater than 1×10^{-6} or an HQ greater than 1 for any scenario. However, for a chemical to be considered a COC, the excess cancer risk must be greater than 1×10^{-6} and/or have an HQ greater than 1 for one or more of the RME scenarios. As shown in Table B.5-46, those chemicals designated as COCs for the seafood consumption scenarios include arsenic, cadmium, cPAH TEQ, pentachlorophenol, total PCBs, PCB TEQ, alpha-BHC, dieldrin, total chlordane, heptachlor epoxide, mirex, and dioxin/furan TEQ.

All seafood consumption scenarios evaluated were associated with total excess cancer risk estimates that exceeded 1×10^{-6} (Table B.5-47). Total excess cancer risks for the three RME scenarios were greater 1×10^{-4} (the upper end of EPA's acceptable risk range), and each of the three RME scenarios had between 5 and 11 of the 15 individual COPCs identified as COCs (i.e., had risks greater than 1×10^{-6}). The non-RME scenarios also had COPCs with risks greater than 1×10^{-6}. Of the non-RME scenarios, risks were highest for the adult tribal scenario based on Suquamish data (1×10^{-2}). Total excess cancer risk estimates for the CT scenarios (i.e., adult tribal scenario based on Tulalip data, child tribal scenario based on Tulalip data, and adult API scenario) were one or more orders of magnitude lower than those for the adult tribal RME scenario based on Tulalip data.

In addition to having excess cancer risks above thresholds, all three of the RME scenarios had HQs for at least one chemical that were greater than 1: the PCB HQ ranged from 7 to 58 for all three RME scenarios, and the cadmium HQ was equal to 2 for the child tribal RME scenario based on Tulalip data (Table B.5-48). The total PCB HQ was also above 1 for most of the non-RME scenarios (except for the adult API CT scenario [neither PCB HQ was greater than 1] and for the adult tribal CT scenario based on Tulalip data [1 of the PCB HQs was less than 1]). In addition, arsenic, cadmium, cobalt, mercury, and TBT had HQs above 1 for the adult tribal scenario based on Suquamish data. Thus, with the exception of the API CT scenario listed above, all of the market basket seafood consumption scenarios had non-cancer HIs above 1.

Total excess cancer risks for the adult one-meal-per-month seafood consumption scenarios ranged from 2×10^{-5} to 4×10^{-4} (with the highest risks based on the consumption of benthic or pelagic fish). HQs for the adult one-meal-per-month scenarios were only greater than 1 for total PCBs (for the scenarios based on the consumption of benthic fish and pelagic fish [both rockfish and perch]). HQs and excess cancer risks were highest for the scenario based on the consumption of rockfish.

Table B.5-47. Summary of estimated excess cancer risks for the seafood consumption scenarios

Chemical	Estimated Excess Cancer Risk											
	Adult Tribal RME (Tulalip Data)	Adult Tribal CT (Tulalip Data)	Child Tribal RME (Tulalip Data)	Child Tribal CT (Tulalip Data)	Adult Tribal (Suquamish Data)	Adult API RME	Adult API CT	Adult One Meal per Month				
								Benthic Fish	Clam	Crab	Pelagic Fish, Rockfish	Pelagic Fish, Perch
Arsenic[b]	2×10^{-4}	1×10^{-5}	4×10^{-5}	4×10^{-6}	2×10^{-3}	8×10^{-5}	2×10^{-6}	3×10^{-7}[c]	1×10^{-5}	2×10^{-6}	7×10^{-7}	2×10^{-6}
cPAH TEQ	1×10^{-4}	4×10^{-6}	1×10^{-4}	9×10^{-6}	1×10^{-3}	5×10^{-5}	9×10^{-7}	1×10^{-7}	9×10^{-6}	4×10^{-7}	9×10^{-8}	5×10^{-7}[c]
1,4-Dichlorobenzene	1×10^{-6}[d]	7×10^{-8}[d]	2×10^{-7}[d]	3×10^{-8}[d]	7×10^{-6}[d]	4×10^{-7}[d]	8×10^{-9}[d]	4×10^{-8}[c]	4×10^{-8}[c]	4×10^{-8}[c]	4×10^{-8}[c]	2×10^{-7}[c]
Pentachlorophenol	2×10^{-6}[d]	4×10^{-8}[d]	4×10^{-7}[d]	2×10^{-8}[d]	2×10^{-5}[d]	3×10^{-7}	4×10^{-9}	1×10^{-8}[c]	4×10^{-8}	1×10^{-8}[c]	1×10^{-8}[c]	3×10^{-8}[c]
Total PCBs	1×10^{-3}	5×10^{-5}	2×10^{-4}	2×10^{-5}	9×10^{-3}	4×10^{-4}	7×10^{-6}	2×10^{-4}	6×10^{-6}	1×10^{-5}	4×10^{-4}	1×10^{-4}
PCB TEQ[e]	7×10^{-4}	4×10^{-5}	1×10^{-4}	2×10^{-5}	6×10^{-3}	3×10^{-4}	8×10^{-6}	1×10^{-4}	5×10^{-6}	1×10^{-5}	3×10^{-4}	9×10^{-5}
Total DDTs	1×10^{-6}	9×10^{-8}	2×10^{-7}	4×10^{-8}	1×10^{-5}	6×10^{-7}	1×10^{-8}	2×10^{-7}	2×10^{-8}	2×10^{-8}[c]	5×10^{-7}	2×10^{-7}
alpha-BHC	4×10^{-6}[d]	2×10^{-7}[d]	7×10^{-7}[d]	1×10^{-7}[d]	2×10^{-5}[d]	9×10^{-7}[d]	3×10^{-8}[d]	1×10^{-7}[c]	1×10^{-7}[c]	1×10^{-7}[c]	2×10^{-7}	1×10^{-7}[c]
beta-BHC	1×10^{-6}[d]	7×10^{-8}[d]	2×10^{-7}[d]	3×10^{-8}[d]	7×10^{-6}[d]	3×10^{-7}[d]	8×10^{-9}[d]	4×10^{-8}[c]	4×10^{-8}[c]	3×10^{-8}[c]	4×10^{-8}[c]	3×10^{-8}[c]
Dieldrin	8×10^{-6}[d]	5×10^{-7}[d]	1×10^{-6}[d]	2×10^{-7}[d]	5×10^{-5}[d]	2×10^{-6}[d]	7×10^{-8}[d]	2×10^{-7}	3×10^{-7}[c]	3×10^{-7}[c]	4×10^{-7}	5×10^{-7}
Total chlordane	2×10^{-6}	9×10^{-8}	3×10^{-7}	4×10^{-8}	1×10^{-5}	7×10^{-7}	1×10^{-8}	4×10^{-8}	8×10^{-8}	2×10^{-8}[c]	1×10^{-7}	5×10^{-8}
Heptachlor	1×10^{-6}[d]	7×10^{-8}[d]	2×10^{-7}[d]	3×10^{-8}[d]	7×10^{-6}[d]	3×10^{-7}[d]	1×10^{-8}[d]	4×10^{-8}[c]	4×10^{-8}[c]	4×10^{-8}[c]	5×10^{-8}[c]	4×10^{-8}[c]
Heptachlor epoxide	2×10^{-6}[d]	2×10^{-7}[d]	4×10^{-7}[d]	7×10^{-8}[d]	1×10^{-5}[d]	7×10^{-7}[d]	2×10^{-8}[d]	9×10^{-8}[c]	9×10^{-8}[c]	9×10^{-8}[c]	1×10^{-7}	9×10^{-8}[c]
Mirex	4×10^{-6}[d]	3×10^{-7}[d]	8×10^{-7}[d]	1×10^{-7}[d]	3×10^{-5}[d]	1×10^{-6}[d]	4×10^{-8}[d]	2×10^{-7}[c]	2×10^{-7}[c]	2×10^{-7}[c]	4×10^{-7}	2×10^{-7}[c]
Dioxin/furan TEQ[e]	1×10^{-4}	6×10^{-6}	2×10^{-5}	3×10^{-6}	7×10^{-4}	4×10^{-5}	1×10^{-6}	5×10^{-6}	3×10^{-6}	3×10^{-6}	2×10^{-5}	9×10^{-6}
Total TEQ excess cancer risk for dioxins/furans and coplanar PCBs	**8×10^{-4}**	**5×10^{-5}**	**1×10^{-4}**	**2×10^{-5}**	**7×10^{-3}**	**3×10^{-4}**	**9×10^{-6}**	**1×10^{-4}**	**8×10^{-6}**	**1×10^{-5}**	**3×10^{-4}**	**1×10^{-4}**
Total excess cancer risk (excluding PCB TEQ)[f]	**1×10^{-3}**	**7×10^{-5}**	**4×10^{-4}**	**4×10^{-5}**	**1×10^{-2}**	**6×10^{-4}**	**1×10^{-5}**	**2×10^{-4}**	**3×10^{-5}**	**2×10^{-5}**	**4×10^{-4}**	**1×10^{-4}**

Table B.5-47. Summary of estimated excess cancer risks for the seafood consumption scenarios (cont.)

Chemical	Estimated Excess Cancer Risk											
	Adult Tribal RME (Tulalip Data)	Adult Tribal CT (Tulalip Data)	Child Tribal RME (Tulalip Data)	Child Tribal CT (Tulalip Data)	Adult Tribal (Suquamish Data)	Adult API RME	Adult API CT	Adult One Meal per Month				
								Benthic Fish	Clam	Crab	Pelagic Fish, Rockfish	Pelagic Fish, Perch
Total excess cancer risk (excluding total PCBs)[f]	1×10^{-3}	6×10^{-5}	3×10^{-4}	4×10^{-5}	1×10^{-2}	5×10^{-4}	1×10^{-5}	1×10^{-4}	3×10^{-5}	2×10^{-5}	3×10^{-4}	1×10^{-4}

[a] The adult one-meal-per-month scenarios are presented for informational purposes only, and are not used by EPA for risk management decisions.

[b] Arsenic EPCs and risk estimates are based on inorganic arsenic.

[c] There were no detected values of this chemical for this seafood category. Risk estimate was based on one-half the maximum RL.

[d] Greater than 50% of the risk associated with this chemical was derived from seafood categories with no detected values.

[e] No mussel data were available for this chemical. When the CDI and risk values were calculated, the portion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

[f] Total risk values include the risks associated with all COPCs.

API – Asian and Pacific Islander
BHC – benzene hexachloride
CDI – chronic daily intake
COPC – chemical of potential concern
cPAH – carcinogenic polycyclic aromatic hydrocarbon
CT – central tendency
DDT – dichlorodiphenyltrichloroethane
EPA – US Environmental Protection Agency
PCB – polychlorinated biphenyl
RL – reporting limit
RME – reasonable maximum exposure
TEQ – toxic equivalent

Table B.5-48. Summary of estimated non-cancer hazards for the seafood consumption scenarios

Chemical	Estimated Non-Cancer Hazard											
	Adult Tribal RME (Tulalip Data)	Adult Tribal CT (Tulalip Data)	Child Tribal RME (Tulalip Data)	Child Tribal CT (Tulalip Data)	Adult Tribal (Suquamish Data)	Adult API RME	Adult API CT	Adult One Meal per Month[a]				
								Benthic Fish	Clam	Crab	Pelagic Fish, Rockfish	Pelagic Fish, Perch
Arsenic[b]	0.4	0.05	0.9	0.1	4	0.4	0.03	0.002	0.08	0.01	0.004	0.009
Cadmium	0.7	0.08	2	0.2	2	0.4	0.03	0.01	0.01	0.09	0.004	0.004
Cobalt	0.6	0.07	1	0.2	4	0.5	0.04	0.01	0.07	0.05	0.02	0.02
Mercury	0.6	0.07	1	0.2	3	0.4	0.04	0.05	0.02	0.09	0.2	0.04
Tributyltin as ion	0.3	0.03	0.7	0.07	4	0.4	0.03	0.007	0.05	0.003	0.2	0.04
Total PCBs[c]	27	3	58	6	214	24	1	13	0.4	0.8	21	8
Total PCBs[d]	8	0.8	17	2	61	7	0.4	4	0.1	0.2	6	2
HIs by Endpoint[e]												
HI for hematological endpoint[f]	**0.3**	**0.05**	**0.8**	**0.1**	**2**	**0.2**	**0.02**	**0.01**	**0.02**	**0.04**	**0.03**	**0.02**
HI for immunological endpoint[g]	**27**	**3**	**59**	**6**	**218**	**24**	**1**	**13**	**0.5**	**0.8**	**21**	**8**
HI for kidney endpoint[h]	**0.8**	**0.1**	**2**	**0.2**	**3**	**0.5**	**0.04**	**0.02**	**0.02**	**0.1**	**0.01**	**0.01**
HI for liver endpoint[i]	**0.06**	**0.008**	**0.1**	**0.02**	**0.3**	**0.04**	**0.003**	**0.007**	**0.006**	**0.004**	**0.01**	**0.008**
HI for neurological endpoint[j]	**28**	**3**	**59**	**6**	**218**	**25**	**1**	**13**	**0.4**	**0.9**	**21**	**8**
HI for endocrine endpoint[k]	**0.6**	**0.08**	**1**	**0.2**	**4**	**0.5**	**0.04**	**0.01**	**0.08**	**0.05**	**0.02**	**0.02**
HI for integumentary endpoint[l]	**28**	**3**	**59**	**6**	**219**	**25**	**1**	**13**	**0.5**	**0.8**	**21**	**8**
HI for digestive system endpoint[m]	**0.5**	**0.06**	**1**	**0.1**	**2**	**0.3**	**0.03**	**0.005**	**0.04**	**0.04**	**0.02**	**0.02**
HI for developmental endpoint[n]	**9**	**0.9**	**18**	**2**	**64**	**7**	**0.4**	**4**	**0.1**	**0.3**	**6**	**2**

a The adult one-meal-per-month scenarios are presented for informational purposes only, and are not used by EPA for risk management decisions.

b Arsenic EPCs and risk estimates are based on inorganic arsenic.

c HQ used for the calculation of the immunological, integumentary, and neurological endpoint hazard indices (Table B.4-1).

d HQ used for the calculation of the developmental endpoint hazard index (Table B.4-1).

Table B.5-48. Summary of estimated non-cancer hazards for the seafood consumption scenarios (cont.)

[e] Total risk values include the risks associated with all COPCs. However, only those COPCs with HQs greater than 1 for at least one scenario are listed in this table.

[f] Hematological endpoint includes the following chemicals: antimony, selenium, and zinc.

[g] Immunological endpoint includes the following chemicals: dibutyltin, total PCBs, and TBT.

[h] Kidney endpoint includes the following chemicals: cadmium, molybdenum, and pentachlorophenol.

[i] Liver endpoint includes the following chemicals: 1,4-dichlorobenzene, alpha-BHC, total chlordane, total DDTs, dieldrin, heptachlor, heptachlor epoxide, mirex, and pentachlorophenol.

[j] Neurological endpoint includes the following chemicals: mercury, total PCBs, and selenium. Neurological effects associated with exposure to lead are discussed in Section B.5.4.

[k] Endocrine endpoint includes the following chemicals: antimony and cobalt.

[l] Integumentary endpoint includes the following chemicals: arsenic, total PCBs, selenium, and vanadium.

[m] Digestive system endpoint includes the following chemicals: chromium and copper.

[n] Developmental endpoint includes the following chemicals: mercury and total PCBs.

API – Asian and Pacific Islanders

BHC – benzene hexachloride

COPC – chemical of potential concern

CT – central tendency

DDT – dichlorodiphenyltrichloroethane

EPA – US Environmental Protection Agency

EPC – exposure point concentration

HI – hazard index

HQ – hazard quotient

nc – not calculated

PCB – polychlorinated biphenyl

RME – reasonable maximum exposure

TBT – tributyltin

In addition to an evaluation of the magnitude of excess cancer risks and non-cancer HQs, the percent contribution of the COPCs to the total excess cancer risk was examined (Section B.5.3.1.3). This evaluation showed that arsenic, cPAH TEQ, and PCBs (as total PCBs or PCB TEQ) were the greatest contributors to the total excess cancer risk estimates for all seafood consumption scenarios. For the three RME scenarios, PCBs were the greatest contributor, accounting for 55 to 70% of the total excess cancer risk. Dioxin/furan TEQ was also an important contributor for some scenarios (contributing up to 9% of the risk). Together, all of the remaining COPCs contributed 5% or less of the total excess cancer risk.

An additional analysis was done for these five COPCs that were identified as having the largest contribution to the risk estimate (arsenic, cPAH TEQ, total PCBs, PCB TEQ, and dioxin/furan TEQ) to identify the seafood consumption categories that contributed the greatest percentage to these risk estimates (Section B.5.3.1.4). The percent contribution of the different seafood categories was variable across scenarios, except for arsenic and cPAH TEQ, for which clam consumption contributed the majority of the risk for all scenarios (over 80%). For PCBs (either as total PCBs or PCB TEQ), the main contributors to risk were benthic fish fillets, perch, and rockfish. Lastly, for dioxin/furan TEQ, the main contributors to risk were clams, crab (both edible meat and whole body), and rockfish.

B.5.6.2 Summary of direct sediment contact and swimming scenarios

The excess cancer risk estimates for the direct sediment exposure scenarios are summarized in Table B.5-49. For the purpose of brevity, chemical-specific risk estimates are provided only for chemicals that had an excess cancer risk estimate greater than 1×10^{-6} for any scenario. However, only those chemicals with excess cancer risks greater than 1×10^{-6} for one of the RME scenarios were designated as COCs. As shown in Table B.5-46, those chemicals designated as COCs for the direct sediment exposure scenarios include arsenic, cPAH TEQ, total TEQ, and total PCBs.

No non-cancer HQs were greater than 1, and thus these values are not summarized in a table. The sum of non-cancer HQs was equal to 0.06 and 0.1 for the netfishing RME and clamming RME scenarios, respectively. The highest HQs were for total PCBs, equal to 0.03 and 0.1 for the netfishing and clamming RME scenarios, respectively. Overall, the sum of non-cancer HQs ranged from 0.009 for the netfishing CT scenario to 0.3 for the tribal clamming 183-day-per-year scenario.

As discussed in Section B.5.3.3, no RME scenario was defined for the swimming scenario, and thus no COCs were identified based on exposure to surface water.

Table B.5-49. Summary of estimated excess cancer risks for direct sediment and water exposure scenarios

Chemical	Estimated Excess Cancer Risk								
	Netfishing		Habitat Restoration Worker	Clamming			Swimming[a]		
	RME	CT		Tribal – 183 Days per Year	Tribal RME	7 Days per Year	High Exposure	Medium Exposure	Low Exposure
Arsenic	3×10^{-6}	7×10^{-7}	5×10^{-7}	2×10^{-5}	1×10^{-5}	4×10^{-7}	4×10^{-7}	4×10^{-8}	3×10^{-10}
cPAH TEQ	3×10^{-6}	2×10^{-7}	7×10^{-7}	3×10^{-5}	2×10^{-5}	5×10^{-7}	na	na	na
Total PCBs	6×10^{-7}	6×10^{-8}	2×10^{-7}	6×10^{-6}	3×10^{-6}	1×10^{-7}	3×10^{-7}	5×10^{-8}	6×10^{-10}
PCB TEQ	3×10^{-7}	4×10^{-8}	5×10^{-8}	2×10^{-6}	1×10^{-6}	3×10^{-8}	9×10^{-6}	2×10^{-6}	2×10^{-8}
Dioxin/furan TEQ	6×10^{-7}	1×10^{-7}	na	2×10^{-6}	1×10^{-6}	4×10^{-8}	na	na	na
Total TEQ excess cancer risk for dioxins/furans and coplanar PCBs	**9×10^{-7}**	**1×10^{-7}**	**na**	**4×10^{-6}**	**2×10^{-6}**	**7×10^{-8}**	**na**	**na**	**na**
Total excess cancer risk (excluding PCB TEQ)[b]	**7×10^{-6}**	**1×10^{-6}**	**1×10^{-6}**	**6×10^{-5}**	**3×10^{-5}**	**1×10^{-6}**	**7×10^{-7}**	**9×10^{-8}**	**9×10^{-10}**
Total excess cancer risk (excluding total PCBs)[b]	**7×10^{-6}**	**1×10^{-6}**	**1×10^{-6}**	**5×10^{-5}**	**3×10^{-5}**	**1×10^{-6}**	**9×10^{-6}**	**2×10^{-6}**	**2×10^{-8}**

[a] For the swimming scenario, the exposure levels are as follows: the high exposure is parameterized as 2.6-hour swim for 24 days per year for 70 years, the medium exposure is parameterized as 1-hour swim for 12 days per year for 30 years, and the low exposure is parameterized as a 0.17-hour swim for 2 days per year for 9 years. Other parameter differences (e.g., exposed skin surface area and incidental water ingestion rate) are presented in Tables B.3-38 to B.3-40.

[b] Total risk values include the risks associated with all COPCs. However, only those COPCs with excess cancer risks greater than 1×10^{-6} for at least one scenario are listed in this table.

COPC – chemical of potential concern

cPAH – carcinogenic polycyclic aromatic hydrocarbon

CT – central tendency

na – not applicable (not a COPC)

PCB – polychlorinated biphenyl

RME – reasonable maximum exposure

TEQ – toxic equivalent

Excess cancer risks for the direct sediment and water exposure pathways were much lower than those for the seafood consumption scenarios. Total excess cancer risk estimates were greater than 1×10^{-6} for both of the RME direct sediment exposure scenarios: equal to 7×10^{-6} for the netfishing RME scenario and equal to 3×10^{-5} for the tribal clamming RME scenario (Table B.5-49). The total excess cancer risk was also greater than 1×10^{-6} for the tribal (183 days per year) clamming scenario (equal to 6×10^{-5} or 5×10^{-5}, depending on the summation approach). Total excess cancer risks for the other sediment exposure scenarios did not exceed 1×10^{-6}. Total excess cancer risks ranged from 9×10^{-10} to 9×10^{-6} for the swimming scenario depending on the exposure level and summation approach (Table B.5-49). No direct sediment or water exposure scenarios had HQs greater than 1 for individual chemicals or generated effect-specific HIs greater than 1, so non-cancer hazards for direct sediment or water exposure scenarios are not included in this summary.

For the RME scenarios, individual COPC risks were greater than 1×10^{-6} for arsenic and cPAH TEQ for the netfishing RME scenario and for arsenic, cPAH TEQ, and total PCBs for the tribal clamming RME scenario. Thus, these three chemicals were identified as COCs. In addition, although neither the PCB TEQ or dioxin/furan TEQ excess cancer risks were greater than 1×10^{-6} for either of the RME scenarios, the sum of these two COPCs was greater than this threshold for the tribal clamming RME scenario (equal to 2×10^{-6}), and thus total TEQ was also identified as a COC for the tribal clamming RME scenario.

As was done for the seafood consumption scenarios, an evaluation of the percent contribution of the COPCs to the total risk was performed for the direct sediment exposure scenarios (Section B.5.3.2.4). The percent contribution by chemical was relatively consistent across scenarios, with arsenic and cPAH TEQ identified as the main contributors to the total excess cancer risk, together contributing over 80% of the total excess cancer risk.

B.5.6.3 Cumulative risk estimates across multiple scenarios

As discussed in Section B.5.2, risks for multiple scenarios were summed to represent the possible exposure of a single individual to EW chemicals from different activities. Summed risks are presented in Table B.5-50 for the various combinations of sediment, surface water, and seafood consumption exposure scenarios. Although some individuals might engage in both netfishing and clamming, risks for these two scenarios were not summed. This is because given the high frequency assumed for each activity (over 100 days per year), engaging in both at the assumed frequency is unlikely. The summed excess cancer risk estimate for each of these three scenarios is the same as their respective estimates for seafood consumption alone after rounding to one significant figure, as recommended by EPA (1989). This analysis demonstrates that the contributions of netfishing, clamming, and swimming to estimated risks are relatively small in comparison with the contributions of seafood consumption and highlights the significance of the seafood consumption exposure pathway for all users of the EW. Excess cancer risks were generally lowest for swimming.

Table B.5-50. Excess cancer risk estimates across related scenarios

Activity	Excess Cancer Risk[a]
Adult Tulalip RME Combination Scenario	
Netfishing RME	7×10^{-6}
Swimming (medium level of exposure)	2×10^{-6}
Adult tribal RME seafood consumption based on Tulalip data	1×10^{-3}
Total	$\mathbf{1 \times 10^{-3}}$
Adult Tulalip CT Combination Scenario	
Netfishing CT	1×10^{-6}
Swimming (low level of exposure)	2×10^{-8}
Adult tribal CT seafood consumption based on Tulalip data	7×10^{-5}
Total	$\mathbf{7 \times 10^{-5}}$
Adult RME Clamming Combination Scenario	
Tribal clamming RME (120 days per year)	3×10^{-5}
Swimming (medium level of exposure)	2×10^{-6}
Adult tribal RME seafood consumption based on Tulalip data	1×10^{-3}
Total	$\mathbf{1 \times 10^{-3}}$

a For seafood consumption and sediment exposure scenarios, total excess cancer risk estimates excluding PCB TEQ were used because these were equal to or higher than total excess cancer risk estimates excluding total PCBs. For swimming, the total excess cancer risk estimates excluding total PCBs were used because they were higher than the total excluding PCB TEQ.

CT – central tendency

PCB – polychlorinated biphenyl

RME – reasonable maximum exposure

TEQ – toxic equivalent

B.5.6.4 Continuum of risk across consumption rates for seafood consumption scenarios

All the risk estimates associated with the seafood consumption scenarios are highly sensitive to consumption rate assumptions. The consumption rates used for the RME scenarios were intended to reflect the 95th percentile of consumption for the population. However, although the consumption rates used for these risk estimates are based on recent consumption studies and direction from EPA (2007b), there is uncertainty related to the application of these rates to groups who use the EW. For example, EPA's tribal seafood consumption framework (2007b) states, "The use of consumption rates of Puget Sound-harvested fish and shellfish derived using Tulalip and Suquamish Tribal data as a surrogate for another Tribe in Puget Sound or the Strait of Georgia could lead to either an overestimate or an underestimate of the actual fish and shellfish consumption rate potentially associated with site releases." Risk estimates would change if consumption rate assumptions were substantially different. To illustrate the relationship between consumption rates and risk estimates, figures were created to show the excess cancer risk estimates for PCB TEQ and total PCBs across a continuum of consumption rates for the different seafood consumption scenarios. Figure B.5-4 shows the total PCB risks for the tribal and API scenarios, and Figure B.5-5 shows the PCB TEQ risks for the tribal and API scenarios. These figures show the direct correlation between changes in consumption rates (assuming the same proportional consumption of different species) and risk estimates for the seafood consumption scenarios.

In addition to these figures (Figure B.5-4 and B.5-5), which show the relationship between consumption rate and risk for the tribal and API scenarios, Figure B.5-6 shows the relationship between consumption rate (number of meals per month) and the total excess cancer risk on a by-species basis, as was done for the one-meal-per-month scenarios. These graphs are intended to aid the public in scaling their risks based on their personal consumption patterns.

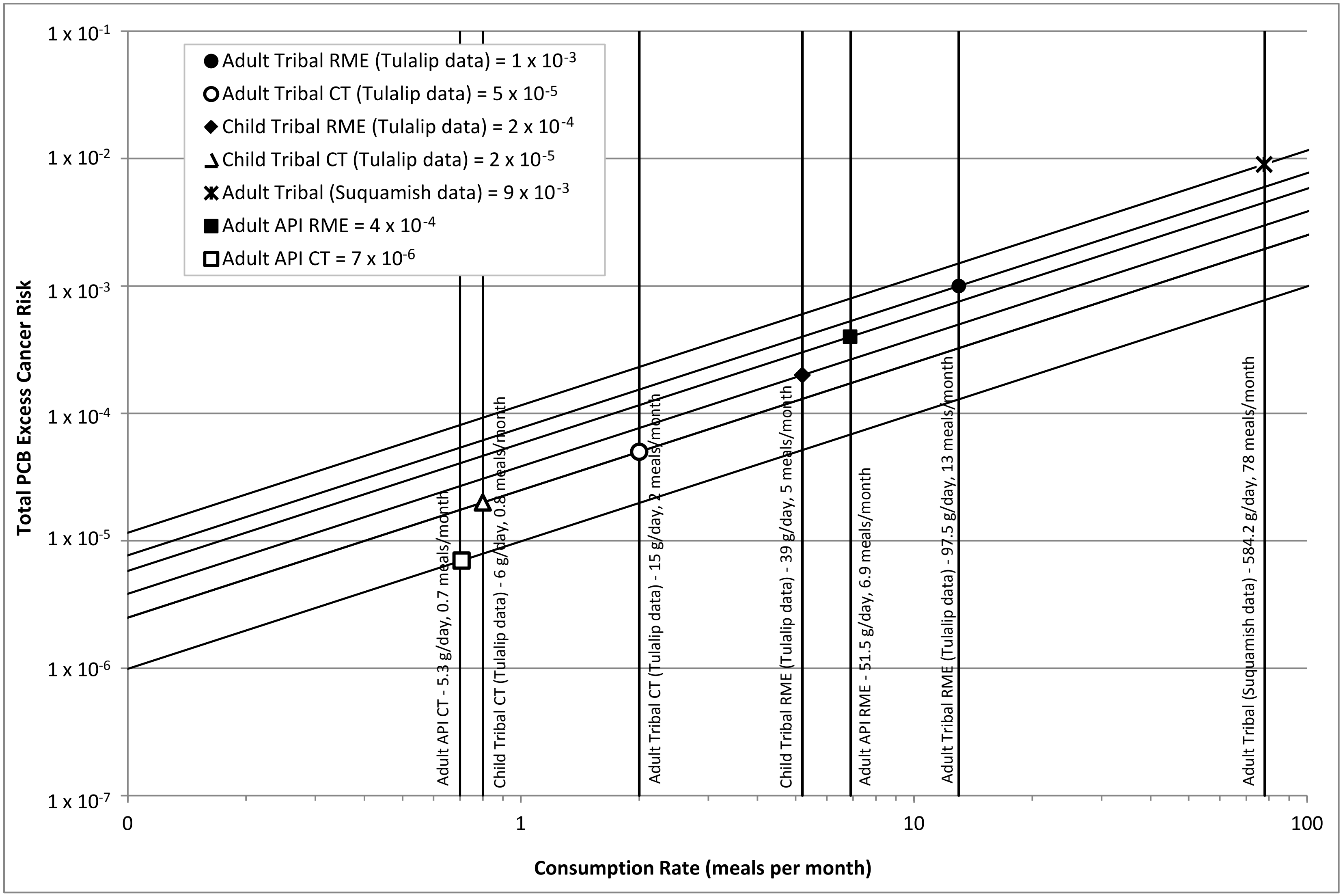

Figure B.5-4. Excess cancer risks from total PCBs for seafood consumption scenarios and rates

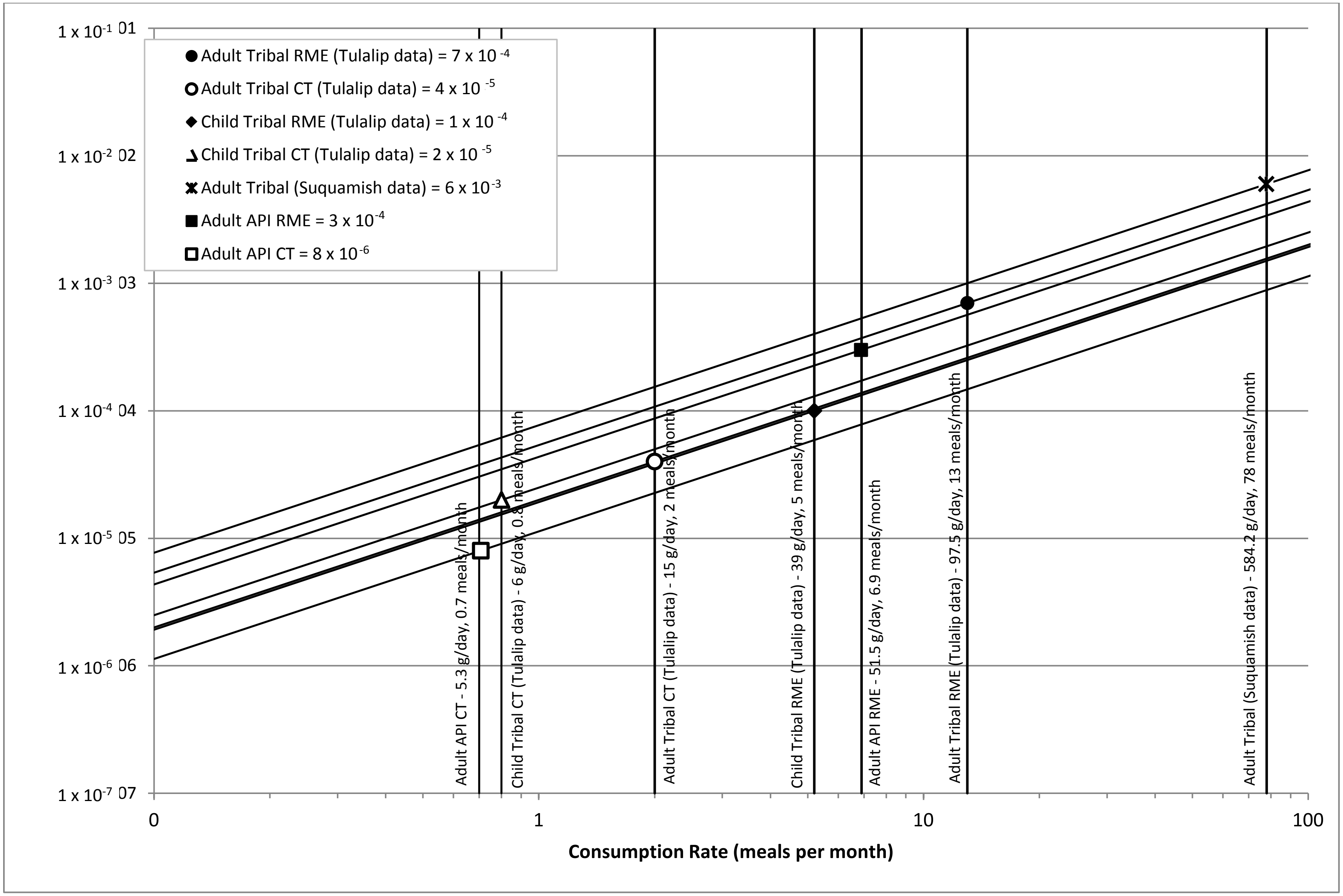

Figure B.5-5. Excess cancer risks from PCB TEQ for seafood consumption scenarios and rates

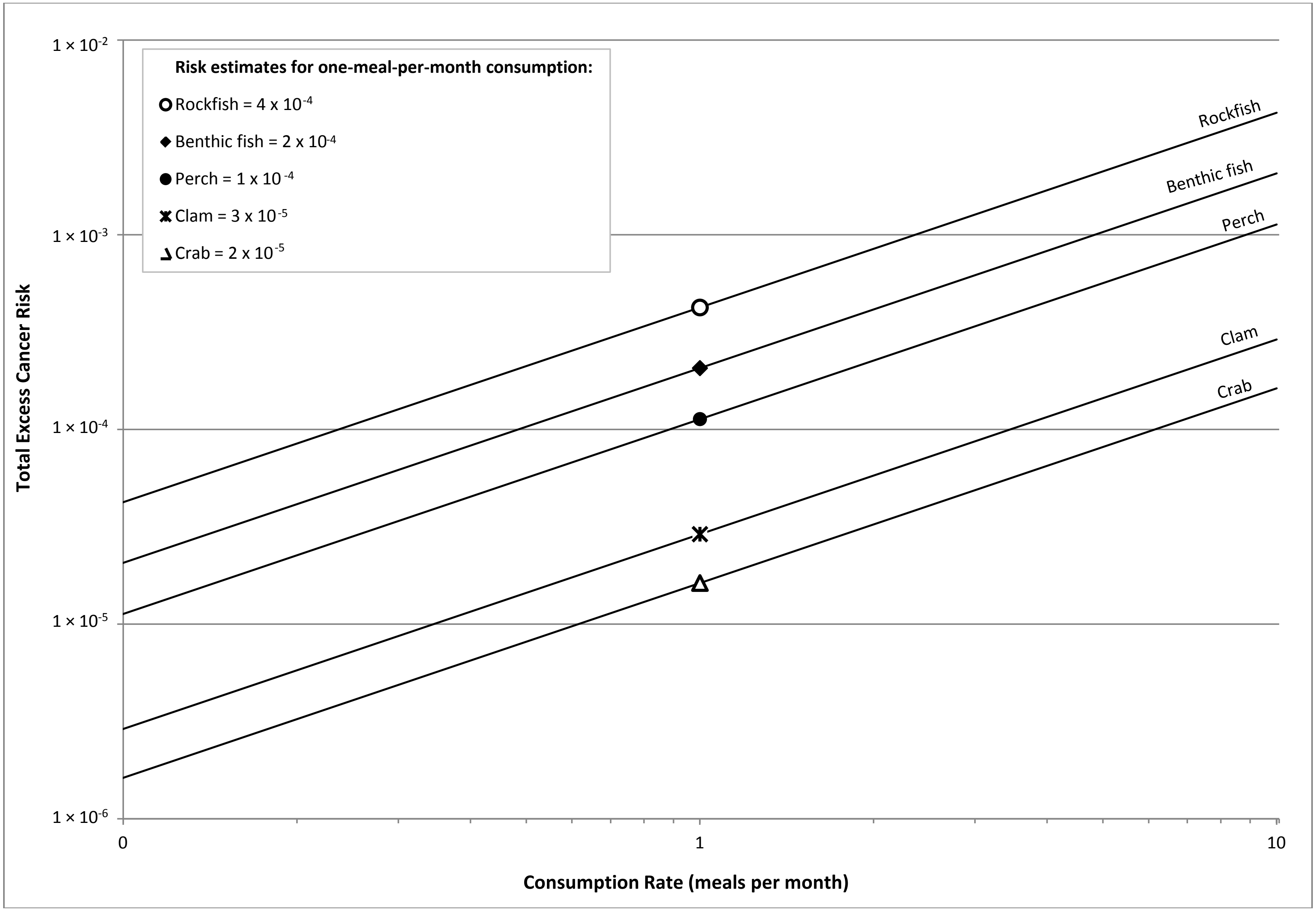

Figure B.5-6. Total excess cancer risk by species across consumption rates

6 Uncertainty Analysis

The exposure and toxicity assumptions used for the EW HHRA, which were based on EPA guidance (incorporating policy decisions), current scientific literature, and best scientific judgment, are inherently uncertain. Therefore, the resulting risk estimates carry a degree of uncertainty. However, it should be noted that the scenario exposure assumptions selected for use in calculating risk estimates were intended to be health-protective. This section discusses some of the key uncertainties in this risk assessment and presents alternative risk estimates based on different hypothetical exposure or toxicity assumptions for many of the exposure scenarios.

Table B.6-1 lists some of the key uncertainties in this baseline HHRA. Each uncertainty is characterized qualitatively as low, medium, or high. Table B.6-1 also characterizes each uncertainty based on the impact that additional information or an alternative analysis may have on the characterization of risk and whether risk estimates included in the risk characterization section are likely to be underestimated or overestimated.

The uncertainties discussed in this section are presented as follows: exposure assessment (Section B.6.1), toxicity assessment (Section B.6.2), and risk characterization (Section B.6.3).

B.6.1 Exposure Assessment

For most HHRAs, including this one, assumptions made during the exposure assessment contribute a level of uncertainty and variability to the risk estimates. Alternative values are possible for all of the exposure parameters described in Section B.3.3, most of which would have a linear effect on the resulting risk estimates.[47] For the exposure frequency and exposure duration parameters for all exposure scenarios, the values selected were based on EPA guidance and professional judgment. For most scenarios, these values were the subject of considerable debate and analysis during the preparation of the baseline LDW HHRA, upon which many of the exposure scenarios presented in this HHRA are based.[48] Thus, these exposure parameters will not be discussed further in this uncertainty analysis. However, there are several other parameters in the exposure assessment for which possible alternative values warrant discussion. These include EPCs and the consideration of non-detected chemicals, seafood consumption rates, incidental sediment ingestion rates, fraction of dose obtained from the EW, and representativeness of existing fish and shellfish data for all potentially exposed populations. Each of these uncertainties is discussed in the following subsections.

[47] Changes to consumption rates for individual seafood categories for scenarios other than the one-meal-per-month scenario would not have a direct linear effect on risk estimates because the CDI is the sum of exposures from the consumption of multiple seafood categories.

[48] This statement applies to only the seafood consumption and direct contact sediment scenarios, not to the swimming scenario. The uncertainties regarding the exposure assumptions for the three levels of exposure evaluated for the swimming scenario are discussed in Section B.5.3.5.

Table B.6-1. Summary of key uncertainties identified in the EW baseline HHRA

Parameter	Level of Uncertainty[a]	Effect of Uncertainty on Risk Estimate	Potential Means to Reduce Uncertainty	Potential Impact on Risk Estimates[b]	Comments
Exposure Assessment (Section B.6.1)					
Exposure Point Concentrations (Section B.6.1.1)					
Statistical approach for the evaluation of infrequently detected COPCs (Section B.6.1.1.1)	low	Risk estimates for chemicals with EPCs based on a single detection and multiple non-detected concentrations are likely to be overestimated.	Perform a more-detailed statistical assessment of data for all COPCs with a low detection frequency or achieve lower reporting limits.	low for total risk estimates	Chemicals affected are minor contributors to total risk. Preliminary explorations of alternative approaches for non-detected concentrations indicate that initial estimates in risk characterization are reasonable.
Consideration of non-detected samples in ProUCL 4 (Section B.6.1.1.2)	low	May have no effect or may overestimate risk if chemicals are not present.	Achieve lower reporting limits.	low for total risk estimates	Risk estimates for chemicals never detected are presented in B.6.3.2.
Calculation methods for total PCBs(Section B.6.1.1.3)	low	Alternative calculation methods could slightly change total PCB concentrations. Total PCBs in the EW HHRA were calculated as the sum of detected Aroclors, but could also be calculated as the sum of detected Aroclors and half RLs for Aroclors detected elsewhere at the site, or as the sum of PCB congeners.	Collect additional PCB Aroclor and congener data for the same samples.	low	Total PCB concentrations were generally similar using different methods for summing PCB Aroclors. Total PCB EPCs based on Aroclors are generally somewhat higher than those based on congeners. The use of Aroclor data is health-protective and maintains consistency with the LDW HHRA.
Use of half-RLs to calculate TEQs (Section B.6.1.1.4)	medium	When calculating cPAH TEQs, PCB TEQs, and dioxin/furan TEQs, one-half of the RL was used for components that were not detected, as described in Section B.2.2.4.	Run additional analyses to lower RLs.	low	The variability in the EPCs was generally low when full RLs or 0 was used in place of the half RLs for calculating TEQ sums. Risks calculated using these alternate EPCs did not change or only slightly changed.

Table B.6-1. Summary of uncertainties identified in the baseline HHRA (cont.)

Parameter	Level of Uncertainty[a]	Effect of Uncertainty on Risk Estimate	Potential Means to Reduce Uncertainty	Potential Impact on Risk Estimates[b]	Comments
EPCs for infrequently detected chemicals (Section B.6.1.1.5)	medium	Alternative methods for calculating EPCs for datasets with fewer than six detected concentrations might lead to different EPCs and risk estimates.	Compare results from multiple calculation methods.	low	EPCs calculated using alternative methods could affect risk estimates for some chemicals but would not affect the chemicals that drive the overall risk conclusions.
EPCs for small datasets (Section B.6.1.1.6)	medium	Unlike datasets that were statistically analyzed (i.e., those with six or more samples that were evaluated using ProUCL 4), 95% UCLs for estimates of the mean for datasets of five or fewer samples were not calculated (except for MIS samples, as discussed in Section B.6.1.1.7). Therefore, uncertainty related to potential EPC underestimation is greater for EPCs derived for small datasets with fewer than six samples, for which ProUCL 4 was not used, as agreed upon with EPA and discussed in Section B.3.3.4. Chemical concentration data are often positively skewed. For small, positively skewed datasets, the true mean may exceed the highest individual sample result. Hence, the selected EPC, which is based on the maximum concentration, may underestimate the mean.	Collect additional data for datasets with small sample sizes (n = 5 or fewer).	low	The majority of tissue and netfishing EPCs were developed from datasets with six or more samples. However, small sample sizes affected EPC development for some tissue EPCs, although not for the COPCs that contribute the majority of the estimated risk. In addition, although there were fewer than six samples for most seafood consumption categories for dioxin/furan TEQ and PCB TEQ (both of which were important contributors to the overall risk), these samples were large composites, thus improving the representativeness of the dataset and reducing the uncertainty.
EPCs for intertidal sediment calculated using MIS samples (Section B.6.1.1.7)	medium	MIS samples, although good for estimating the mean concentration, do not provide information regarding the variance around the mean. Thus, the representativeness of the calculated EPC for upper bound exposures is uncertain.	Collect additional data to evaluate the variance.	medium	The MIS 95% UCL was higher than the maximum concentration in most cases, especially for the MIS 95% UCL calculated for the public-access intertidal area.

Table B.6-1. Summary of uncertainties identified in the baseline HHRA (cont.)

Parameter	Level of Uncertainty[a]	Effect of Uncertainty on Risk Estimate	Potential Means to Reduce Uncertainty	Potential Impact on Risk Estimates[b]	Comments
Spatial bias in sediment EPC estimates (Section B.6.1.1.8)	low	The EPCs calculated in Section B.3.3.4 (using ProUCL) for the netfishing scenario could be overestimated if there is any spatial bias in the underlying data.	A SWAC was calculated for the subtidal portion of the netfishing exposure area for comparison with the arithmetic average concentration.	low	The averages calculated using a SWAC were somewhat lower than the arithmetic averages.
Ingestion Rates (Section B.6.1.2)					
Incidental sediment ingestion rates (Section B.6.1.2.1)	high	The applicability of incidental soil ingestion rates from EPA guidance to sediment exposure scenarios is unknown.	Compare sediment exposure behaviors with behaviors assumed for EPA default soil ingestion rates.	unknown	Ingestion rates would be very difficult to measure, so the assumption that the sediment ingestion rate is equivalent to the soil ingestion rate is based largely on best professional judgment.
EW adult seafood consumption rates assumed for this HHRA (Section B.6.1.2.2)	high	The seafood consumption rates assumed for the adult tribal scenario based on Tulalip data, the adult tribal scenario based on Suquamish data, and the adult API scenario were provided by EPA (EPA 2005a; Kissinger 2005; EPA 2009b) and are based on recent regional seafood consumption studies (EPA 1999a; Suquamish Tribe 2000; Toy et al. 1996). For the tribal scenarios, these rates likely represent a significant overestimate of consumption from the EW because the EW is not the primary fishing area for these groups. Although API community members are known to harvest fish from the EW, it is similarly uncertain to what degree consumption rates from EPA's 1999 API study overestimate EW-specific API consumption rates.	Collect site-specific data that reflects EW resource use by different populations in urban watersheds that have similar habitat to that of the EW but do not have substantial chemical contamination (assuming any such watersheds could be found), which could then be used to draw conclusions about resource use within the EW. However, any EW-specific consumption survey could be affected by concerns regarding chemical contamination, thus resulting in lower consumption rates.	high	Although site use may increase in the future, the degree of future use assumed in this assessment may overestimate risks for most users. However, it should be noted that a small group of individuals could get a significant portion of their fish from the EW.

Table B.6-1. Summary of uncertainties identified in the baseline HHRA (cont.)

Parameter	Level of Uncertainty[a]	Effect of Uncertainty on Risk Estimate	Potential Means to Reduce Uncertainty	Potential Impact on Risk Estimates[b]	Comments
Exclusion of adult salmon from overall seafood consumption rates (Section B.6.1.2.2)	low	The overall risk estimate based on resident fish and shellfish is only slightly underestimated by the exclusion of salmon.	Include salmon in consumption rate and risk estimate; applicability to EW sediment-related exposures is uncertain.	low	As was done in the LDW HHRA (Windward 2007c), salmon were excluded from the seafood consumption rate used for tribal and API seafood consumption risk assessments for bioaccumulative chemicals because site-related exposures are likely insignificant compared with exposures that are not site-related.
Child seafood consumption rates (Section B.6.1.2.3)	high	Children's tribal fish consumption rates are generally less well characterized than adult rates. Despite this fact, use of the child-to-adult ratio approach based on the Tulalip data yields a consumption rate that is consistent with the upper percentile children's consumption rates from available studies (CRITFC 1994; EPA 2002b).	Conduct better tribal children's seafood consumption surveys to support consumption rates.	high	Uncertainties for the adult consumption rates also apply to the child rates with additional uncertainty related to the fact that data are more limited for children. The sample size for the Tulalip Tribes study is small, and multiple children from the same households were sampled in the Suquamish study.
Scenario Exposure Duration (Section B.6.1.3)					
Exposure duration for API seafood consumption scenario (B.6.1.3)	medium	The exposure duration for API who use the EW as their primary or exclusive fishing location is unknown. In the exposure assessment, exposure duration was assumed to be 30 years based on EPA's assessment of the 90th percentile for the residence time for the general public in the United States. However, it should be noted that exposure duration based on residence time could underestimate exposure because individuals could relocate over small distances and continue to collect seafood from the same body of water.	Conduct a survey of exposure duration for API who use the EW as their primary or exclusive fishing location. Estimate risks using alternative assumptions of exposure duration.	low	No site-specific survey data were available. An alternative exposure duration assumption of 41 years based on EPA's assessment of national data was used to estimate risk for comparison to the risks presented in Section B.5.

Table B.6-1. Summary of uncertainties identified in the baseline HHRA (cont.)

Parameter	Level of Uncertainty[a]	Effect of Uncertainty on Risk Estimate	Potential Means to Reduce Uncertainty	Potential Impact on Risk Estimates[b]	Comments
Site Use (Section B.6.1.4)					
Fraction of dose obtained from the EW (Section B.6.1.4.1 and B.6.1.4.2)	high	For most individuals, the fraction of Puget Sound- or King County-harvested fish and shellfish obtained from the EW is likely to be moderately to greatly overestimated. This is particularly true for the tribal and API seafood consumption scenarios because all fish consumed are assumed to be from the EW. There may be only a very small population that currently practices subsistence seafood harvesting from the EW. However, because of the presence of other nearby contaminated areas (e.g., Elliott Bay waterfront and the LDW), it is possible that risks may not be overestimated if individuals are fishing mostly from these areas and consuming seafood at or above the rates assumed for this HHRA. The representativeness for a future-use scenario is unknown. For the clamming scenarios, the frequency of exposure and therefore intake from the EW (as opposed to other locations that have been surveyed) is unknown.	Collect additional data that reflects site-specific use and habitat suitability in urban watersheds that have habitat similar to that of the EW but do not have substantial chemical contamination (assuming any such watersheds could be found), which could then be used to draw conclusions about resource use within the EW.	high	Default assumption of 1 (100%) was applied as required by EPA because of a lack of site-specific data. Alternative assumptions of site use equal to 10% and 1% would still result in excess cancer risk estimates greater than 1×10^{-6} for most seafood consumption scenarios.
Dermal Exposure (Section B.6.1.5)					
Chemicals lacking guidance on absorption factors (Section B.6.1.5.1)	medium	The underestimation of dermal risks from metals that lack absorption factors is expected to have a small effect on overall risk estimates.	Evaluate risk estimates using a range of absorption assumptions.	low	Dermal absorption is dependent on the speciation of metals but is typically low. Calculations in this uncertainty analysis in which absorption was assumed to be 0.001 to 0.03 (the range available for metals in documents other than EPA HHRA guidance), indicate that this pathway likely does not contribute significantly to the underestimation of risks.

Table B.6-1. Summary of uncertainties identified in the baseline HHRA (cont.)

Parameter	Level of Uncertainty[a]	Effect of Uncertainty on Risk Estimate	Potential Means to Reduce Uncertainty	Potential Impact on Risk Estimates[b]	Comments
Dermal adherence factors used for sediment exposure scenarios (Section B.6.1.5.2)	medium	May lead to a slight overestimation or underestimation of risks	Evaluate risk estimates using an alternative dermal adherence factor.	low	The use of an alternative larger dermal adherence factor did not significantly change risk estimates.
Representativeness of Data (Section B.6.1.6)					
Representativeness of fish and shellfish COPC data for all potentially exposed populations (Section B.6.1.6.1)	medium	unknown	Collect additional data for different seafood tissue types and analyze for additional chemicals; consider preparation practices in risk calculations. In addition, angler basket surveys could be another method for determining the actual proportions of different species that people may consume from the EW.	low	Changes in chemical concentrations as a result of preparation and cooking were not considered. Food preparation and cooking practices may reduce or increase risks. Given the wide range of cooking practices, it is health-protective to not adjust tissue concentrations for cooking and preparation. Although not all seafood tissue samples were analyzed for all chemicals, particularly dioxins/furans and PCB congeners, the relatively large tissue database used in the HHRA should reasonably approximate the range of chemical concentrations to which seafood consumers might be exposed.
Spatial coverage of sediment chemistry data (Section B.6.1.6.2)	low to medium	The uncertainty is low for most chemicals. If spatial bias exists, risks could be under or overestimated depending on whether areas of higher concentrations were under- or over-sampled.	Compare arithmetic mean concentrations with SWACs to determine if spatial bias exists in the dataset.	unknown	Available information and a comparison of arithmetic means and SWACs does not suggest that there are large sources that have not been characterized, but some minor gaps in spatial coverage may exist for specific exposure areas.

Table B.6-1. Summary of uncertainties identified in the baseline HHRA (cont.)

Parameter	Level of Uncertainty[a]	Effect of Uncertainty on Risk Estimate	Potential Means to Reduce Uncertainty	Potential Impact on Risk Estimates[b]	Comments
Exclusion of King County WQA surface water dataset (Section B.6.1.6.3)	low	Only the SRI surface water dataset was used because it was collected specifically for the purpose of characterizing average water concentrations in the EW. The King County WQA dataset included sampling every week for a 6-month period at one transect location. Statistically, combining these datasets would result in considerable uncertainty.	Evaluate differences in chemical concentrations for the two datasets.	low	Chemicals were more frequently detected and concentrations were generally higher in the EW SRI/FS dataset as compared with the King County WQA dataset. Thus, the use of only the SRI dataset is a health-protective approach.
Calculation of clam EPCs (Section B.6.1.6.4)	low to medium	Uncertainty exists regarding the abundance of different clam species in the EW, differences in concentrations across clam species, and whether groups have certain preferences regarding the harvesting of clams for consumption.	Evaluate differences in chemical concentrations in different clam species and whether different methods could be used to calculate the clam EPC.	low	No clear pattern exists regarding which species had the highest concentrations. In addition, a biomass-weighted approach for developing EPCs did not result in significant changes in EPC values.
Temporal variability in the tissue chemistry dataset (Section B.6.1.6.5)	low	Data included in the tissue chemistry dataset were from six sampling events that occurred in 1995, 1996, 1997, 1998, 2005, and 2008. The 2008 data comprise the majority of the dataset, but uncertainty exists regarding the inclusion of data collected prior to 2008.	Compare EPCs calculated using only data from 2008 (the most recent sampling event) with EPCs calculated using all of the data.	low	Total PCB EPCs (the chemical with the largest historical dataset) calculated using all of the data were generally higher or similar to those calculated using only the 2008 data. The changes in EPCs would result in some changes to risk estimates but would not change risk conclusions.
Health-Protectiveness (Section B.6.1.7 and B.6.1.8)					
Health-protectiveness of sediment and surface water exposure scenarios (Sections B.6.1.7.1 through B.6.1.7.3)	low	Risk estimates for sediment exposure scenarios may be overestimated for current conditions and activities associated with lower levels of exposure.	Separating current from future scenarios could provide more realistic exposure conditions for current conditions.	A current exposure scenario would likely yield lower risk estimates.	The current frequency of sediment and surface water contact activities has not been well quantified. However, reasonable future exposure must be considered in CERCLA risk assessments. Quantifying future exposure frequencies is very difficult and highly uncertain.

Table B.6-1. Summary of uncertainties identified in the baseline HHRA (cont.)

Parameter	Level of Uncertainty[a]	Effect of Uncertainty on Risk Estimate	Potential Means to Reduce Uncertainty	Potential Impact on Risk Estimates[b]	Comments
Basis for fish and shellfish tissue screening levels (Section B.6.1.8)	low	The use of RSLs based on the adult tribal RME scenario based on Tulalip data for developing the COPC list is health-protective for most scenarios but could result in an underestimation of total risks for the adult tribal scenario based on Suquamish data (i.e., several additional chemicals would screen in as COPCs if Suquamish-based RSLs were used, but risks for these chemicals would be low).The consumption rate based on Suquamish data is approximately 6 times higher than that based on Tulalip data.	Use scenario-specific RSLs	low	Four additional chemicals would screen in as COPCs if RSLs were modified based on the Suquamish tribal parameterization. However, total risk estimates for the adult tribal scenario based on Suquamish data would not change given the number of significant figures presented for the risk estimates (i.e., risks for the additional chemicals that would screen in are much lower than the total risk for this scenario).
Toxicity Assessment (Section B.6.2)					
Chemicals without toxicity benchmarks and RSLs (Section B.6.2.1)	low	For chemicals that were detected in EW samples but lacked RSLs, risks could be underestimated to an unknown degree for chemicals lacking toxicity benchmarks.	unknown	medium	Chemicals could not be screened against RSLs, and risks could not be estimated for chemicals for which toxicity data are inadequate to develop toxicity values (i.e., RfDs or SFs).
Total PCBs (Section B.6.2.2)	low	Risk may be moderately overestimated based on the selection of the PCB cancer SF. However, in other settings, bioaccumulation and environmental weathering have been demonstrated to alter the components of PCB mixtures, resulting in greater a toxicity of the mixture compared with that of commercial PCB mixtures from which the SF was derived.	unknown	low	Environmental mixtures of PCBs differ from Aroclor formulations. The most-health-protective SF derived based on Aroclors 1254 and 1260 is probably not representative of the toxicity of all PCB Aroclors and may overestimate carcinogenicity of lower-chlorinated Aroclors. However, because the Aroclors detected are predominantly the more highly chlorinated Aroclors, uncertainty is low.

Table B.6-1. Summary of uncertainties identified in the baseline HHRA (cont.)

Parameter	Level of Uncertainty[a]	Effect of Uncertainty on Risk Estimate	Potential Means to Reduce Uncertainty	Potential Impact on Risk Estimates[b]	Comments
PCB TEQs (Section B.6.2.3)	medium	unknown	unknown	high	PCB TEFs used to calculate PCB TEQs are based primarily on structure activity relationships rather than direct toxicity data. The calculation of PCB TEQ excess cancer risks uses a dioxin SF that is highly uncertain and undergoing review.
TEQ approach for sediment and water (Section B.6.2.3)	high	Toxic equivalency approach involves uncertainty because it estimates the toxicity of congeners relative to 2,3,7,8-TCDD, and in this HHRA, it was assumed that bioavailability was equal for all matrices. In sediment, this approach likely overestimates the bioavailability of PCB, dioxin/furan congeners because they tend to bind tightly to sediment particles. Similarly, bioavailability in water is not well understood. Thus, uncertainty related to exposure affects the applicability of the toxicity metric.	Obtain congener-specific bioavailability estimates to adjust the TEFs.	high	The TEQ approach is most appropriately applied to tissue matrices; bioavailability estimates for sediment and water TEQ would be uncertain.
SF used to calculate PCB and dioxin/furan TEQ risks (Section B.6.2.3)	medium	Risks may be the same, or may be moderately over- or under-estimated based on the selected SF for 2,3,7,8-TCDD.	unknown	low	A review of other dioxin SFs indicated that the uncertainty associated with the excess cancer risks calculated using the selected SF of 150,000 mg/kg-day^{-1} is low.

Table B.6-1. Summary of uncertainties identified in the baseline HHRA (cont.)

Parameter	Level of Uncertainty[a]	Effect of Uncertainty on Risk Estimate	Potential Means to Reduce Uncertainty	Potential Impact on Risk Estimates[b]	Comments
Chromium speciation (Section B.6.2.4)	medium	Risks may be moderately overestimated because the RfD for hexavalent chromium (the most toxic species) was used for total chromium rather than the less toxic form of trivalent chromium.	Collect additional data on chromium species present in sediment and tissue.	low	Chromium risks do not exceed acceptable risk levels even with this health-protective (i.e., conservative) risk assumption. In addition, it should be noted that EPA is currently reviewing the carcinogenicity of hexavalent chromium via oral exposure. If chromium is found to be carcinogenic via oral exposure, excess cancer risks could be underestimated.
Mercury speciation (Section B.6.2.5)	medium	Risks may be moderately overestimated because the RfD for methylmercury (the most toxic form) is used for total mercury rather than the less toxic form of elemental mercury. This assumption is more reasonable for fish tissue than for shellfish tissue or sediment because the percentage of mercury that is methylated approaches 100% in fish, while the percentage is more variable for shellfish.	Collect additional data on mercury species present in sediment and tissue.	low	Mercury risks do not exceed acceptable risk levels for any RME seafood consumption scenarios (the adult tribal scenario based on Suquamish data has an HQ of 3) or any sediment exposure scenarios, even with this health-protective risk assumption.
Arsenic speciation (Section B.6.2.6)	low	Risks may be slightly underestimated because the organic portion of total arsenic is not assessed for risk based on seafood consumption. However, this is unlikely because in many studies, organic arsenic has been shown to be metabolically inert and non-toxic.	Determine toxicity for organic arsenic and calculate organic arsenic risks. These could then be added to the risks based on inorganic arsenic.	low	The evaluation of only inorganic arsenic was done per EPA guidance based on the fact that in general, organic arsenic compounds have been shown to be metabolically inert and non-toxic. An evaluation of HQs calculated using the provisional RfD for DMA indicate that this uncertainty is low. .

Table B.6-1. Summary of uncertainties identified in the baseline HHRA (cont.)

Parameter	Level of Uncertainty[a]	Effect of Uncertainty on Risk Estimate	Potential Means to Reduce Uncertainty	Potential Impact on Risk Estimates[b]	Comments
Risk Characterization (Section B.6.3)					
Inclusion of PCBs in estimates of the total excess cancer risk (Section B.6.3.1)	medium	Reporting totals with only total PCBs or only PCB TEQ may underestimate total excess cancer risk.	Develop an adjustment process to avoid double-counting when summing total PCB and PCB TEQ risks.	high	The approach used in this HHRA has been used at other Superfund sites, although other approaches have been provided in EPA guidance and used in other risk assessments. Risk characterization of environmental PCB mixtures using toxicity estimates derived for commercial PCB mixtures is problematic. Alternate methods to calculate the risk from total PCBs could result in higher risk estimates.
Risk calculations for non-detected chemicals (Section B.6.3.2)	medium	Greatly overestimated if non-detected COPCs are not present; uncertain if these COPCs are present at concentrations below the RLs.	Collect additional data with lower RLs, if analytically possible. Conduct risk calculations using RL data to bound potential risks.	low	Many of the chemicals that were never detected have no known EW source, so lower RLs may not be helpful. However, it is possible that some of these chemicals could have sources upstream of the EW (e.g., in the LDW). Calculations conducted assuming that non-detected chemicals were present at the RL resulted in relatively low estimates for all but a few non-detected chemicals that had high RLs and thus high risk estimates.

Table B.6-1. Summary of uncertainties identified in the baseline HHRA (cont.)

Parameter	Level of Uncertainty[a]	Effect of Uncertainty on Risk Estimate	Potential Means to Reduce Uncertainty	Potential Impact on Risk Estimates[b]	Comments
Calculation of combined risks for adults and children (Section B.6.3.3)	low	As was done in the LDW HHRA, excess cancer risks were calculated in two parts for the tribal seafood consumption scenario based on Tulalip data: for children (aged 0 to 6 years), and for adults (70-year duration). However, risks could be higher if the lifetime exposure period was used (i.e., aged 0 to 70 years), particularly for chemicals with mutagenic modes of action, such as cPAHs.	Risks could be calculated for the lifetime exposure period (from age 0 to 70).	low	Total lifetime excess cancer risks presented in this uncertainty analysis indicate that risks are not greatly underestimated in Section B.5.

[a] Level of uncertainty: low = large and relevant dataset; medium = small dataset or limited information; high = very limited data or no site-specific information.

[b] Potential impact: low = additional data or analysis unlikely to result in a change in the determination of whether a chemical exceeds acceptable risk levels (i.e., HQ greater than 1 or cumulative excess cancer risk greater than 1×10^{-6}) or the identification of a pathway of concern; medium = additional data or analysis could result in a change in the determination of whether a chemical exceeds acceptable risk levels or the identification of a pathway of concern; high = additional data or analysis likely to result in a change in the determination of whether a chemical exceeds acceptable risk levels or the identification of a pathway of concern.

CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act
COPC – chemical of potential concern
cPAH – carcinogenic polycyclic aromatic hydrocarbon
CT – central tendency
DMA – dimethyl arsenic acid
EPA – US Environmental Protection Agency
EPC – exposure point concentration
EW – East Waterway
HHRA – human health risk assessment
HPLC – high-performance liquid chromatography
HQ – hazard quotient
LDW – Lower Duwamish Waterway
MIS – multi-increment sampling
PCB – polychlorinated biphenyl
RfD – reference dose
RL – reporting limit
RME – reasonable maximum exposure
RSL – regional screening level
SF – slope factor
SRI – supplemental remedial investigation
SWAC – spatially weighted average concentration
TEQ – toxic equivalent
TCDD – tetrachlorodibenzo-*p*-dioxin
TEF – toxic equivalency factor
UCL – upper confidence limit on the mean
WQA – water quality assessment

B.6.1.1 Exposure point concentrations

EPCs were calculated for each medium in order to calculate risks, as described in Section B.3.3.4 and summarized below:

- **Tissue –** For each COPC, an EPC was calculated for each of the 10 seafood consumption categories (Section B.3.3.4.1).
- **Sediment –** For each COPC, an EPC was calculated for the site-wide intertidal exposure area using three MIS samples, for the public access intertidal area using one public access MIS sample, and for the site-wide exposure area using both the intertidal and subtidal samples (Section B.3.3.4.2).
- **Water –** For each COPC, an EPC was calculated using the site-wide surface water dataset (Section B.3.3.4.3). The surface water dataset included samples collected from 1 m below the water surface but not samples collected from 1 m above the sediment.

The following subsections discuss the uncertainties associated with these EPCs.

B.6.1.1.1 Statistical approach for the evaluation of infrequently detected COPCs

ProUCL software was used to develop 95% UCLs of COPCs. ProUCL first evaluates the distribution of the data and then recommends a statistical approach and provides an estimated 95% UCL (EPA 2007d). ProUCL 4 software provides defensible statistical methods and does not rely on simple substitutions for non-detected data points. ProUCL 4 software allows for a parametric and non-parametric analysis of both uncensored datasets (i.e., all detected concentrations) and those that contain non-detects to determine a distribution from which a 95% UCL may be calculated. Some of the methods (e.g., Kaplan-Meier method) are able to handle datasets that have multiple detection limits (EPA 2007d). The more accurate methods for identifying distributions make it possible to better define the appropriate 95% UCL for use in risk assessment.

The ProUCL software generally determined that non-parametric statistics were most appropriate for 95% UCL calculations with more than 50% non-detected values and frequently recommended very conservative statistical approaches, such as the 95^{th} percentile Chebyshev, for the 95% UCL (Tables B.3-42 through B.3-47). In some cases, the recommended 95% UCLs were close to the maximum for these COPCs and sometimes exceeded the maximum. In all cases, the recommended 95% UCL from ProUCL was used in the risk equations.

The 95% UCL calculation methods were intended to provide reasonably health-protective estimates of EPCs for the large number of infrequently detected chemicals present in seafood and sediment. For the seafood consumption scenarios, many chemicals identified as exceeding acceptable risk levels (i.e., upper-bound excess cancer risk estimate > 1×10^{-6} or non-cancer HQ > 1) were less frequently detected, or had only non-detected values, in at least one seafood category. Although they exceeded risk thresholds, many of these chemicals had excess cancer risk estimates that were an order of magnitude or more below estimates for the most significant risk contributors (i.e., arsenic, cPAH TEQ, PCB TEQ, total PCBs) (Table B.5-47), which were detected in nearly all seafood categories in which they were analyzed (Table B.3-42). Thus, the uncertainty surrounding the presence and concentrations in seafood tissue of chemicals with lower risk estimates is greater than the uncertainty associated with the chemicals with the highest risk estimates. This was less of an issue for the direct sediment exposure scenarios because all chemicals that exceeded acceptable risk levels had detection frequencies greater than 50%, and none of them were tentatively identified (i.e., N-qualified).

For the intertidal sediment EPCs that were based on the MIS samples (Section B.3.3.4.2), COPCs were generally detected in all of the samples. If a COPC was not detected in a particular sample, half of the RL was used for that sample in the EPC calculations. The resulting EPCs may underestimate or overestimate the concentration of these chemical depending on whether a chemical is present at, above, or below half of the RL. Overall, the impact of this uncertainty on risk estimates is expected to be low.

B.6.1.1.2 Consideration of non-detects in ProUCL 4

The ProUCL 4 software, which was used to calculate all EPCs (for which there were a sufficient number of detected values [see Section B.3.3.4]), has the capability of assigning a hypothetical interpolated result for non-detects based on the distribution of detected concentrations, as explained further in this section. Given the statistical treatment of non-detects for EPC calculation in this software, the resulting EPC calculations are not necessarily biased either high or low. The uncertainty associated with these calculations is relatively low.

For chemicals that were detected five or fewer times within a dataset used for EPC calculation (other than for MIS datasets), the EPC chosen was the higher of either the maximum detected concentration or half the maximum RL (see Section B.6.1.1.6 for a discussion of the uncertainty associated with EPCs for datasets with fewer than six samples). The use of only a single maximum sample concentration to represent the EPC is associated with some uncertainty. This is because the use of a detected concentration does not take into account any of the other data that may indicate lower concentrations or the absence of the chemical above the RL. The resulting EPC is likely to be an overestimate of the "true" 95% UCL (the typical statistic used for the EPC), but there is no statistically reliable means to estimate the 95% UCL. To highlight this uncertainty in the risk estimates for the seafood consumption scenarios, chemicals were footnoted in the risk characterization tables if greater than 50% of the estimated excess cancer risk or non-cancer hazard (based on total dose) was attributable to seafood categories (e.g., crab whole body, clams) with no detected concentrations. This uncertainty may have no effect or may somewhat overestimate risks if chemicals are not present. Overall, the uncertainty associated with the handling of non-detects in this HHRA is expected to be low for total risk estimates.

It should be noted that this issue does not apply to chemicals that were never detected because they were evaluated only in the uncertainty analysis (see Section B.6.3.2), rather than in the risk characterization.

B.6.1.1.3 Calculation methods for total PCBs

The concentration of total PCBs in a sample may be calculated as the sum of Aroclors or the sum of PCB congeners. This section explores differences in total PCB concentrations between the sum of Aroclors and the sum of congeners. Aroclor data were available for many more tissue samples (n = 124) than were congener data (n = 28). For the sediment dataset, equal numbers of intertidal MIS samples were available for Aroclor and congener data (n = 3). For subtidal sediments, 13 composite samples were analyzed for PCB congeners, and 237 grab samples were analyzed for Aroclors. Because of the larger dataset available for Aroclors, and to be consistent with the LDW HHRA, total PCBs in tissue and sediment were assessed as the sum of Aroclors in the risk characterization. The sample size affects the calculation of 95% UCLs (for EPCs) in that ProUCL 4 attempts to compensate for the uncertainty of having fewer measurements by selecting calculation approaches that may lead to higher 95% UCL estimates.

Total PCBs in sediment and tissue were calculated as described in Section B.2.2.4 in accordance with SMS (WAC 173-204), as was done in the LDW HHRA (Windward 2007c). That method sums only detected Aroclor concentrations or assigns a value equal to the highest Aroclor RL if all Aroclors are non-detected. The same summing approach was applied for congeners. At other Superfund sites, different methods for calculating total PCBs have been used. For example, EPA is developing guidance for assessing human health risks from total PCBs that includes summing detected Aroclor concentrations and one-half the RL for particular Aroclors, if they were detected in a significant number of samples found elsewhere at the site. This is the approach being used at the Portland Harbor Superfund site.

Table B.6-2 presents a comparison of total PCB concentrations in EW tissue samples calculated by the primary method used elsewhere in this document and the alternative method mentioned above. The differences in the calculated tissue concentrations between the two methods are small. For fish consumption categories, which have the highest total PCB concentrations, the percentage difference is 0% (i.e., the average tissue concentrations for the two methods are equal). Average concentrations were also generally similar for the shellfish consumption categories, with average percent differences of 0 to 6%. The one exception is mussels, for which the percent difference of the average concentrations was 46%, with higher concentrations calculated using the alternative method that includes one-half RLs for Aroclors detected in samples from elsewhere in the EW site.

Table B.6-2. Comparison of two different methods for calculating total PCBs in EW tissue samples

Consumption Category	Average Total PCB Concentration (µg/kg ww)		Absolute Difference Between Methods	Percent Difference Between Methods
	Primary Method[a]	Alternative Method[b]		
Benthic fish, fillet	1,720	1,720	0	0%
Benthic fish, WB	3,160	3,160	0	0%
Clams	56.3	56.8	0.5	0.9%
Crab, EM	126	134	8	6%
Crab, WB	306	321	15	5%
Geoduck, EM	19.0	20.0	1	5%
Geoduck, WB	28.0	28.0	0	0%
Mussels	26.0	38.0	12	46%
Pelagic fish, perch	998	998	0	0%
Pelagic fish, rockfish	2,020	2,020	0	0%

a Primary method (described in Section B.2.2.4): sum only detected Aroclor concentrations or assign a value equal to the highest Aroclor RL if all Aroclors undetected.

b Alternative method: sum detected Aroclor concentrations and one-half the RL for undetected Aroclors that were detected elsewhere at the site. The alternative method does not include seven samples from 1998 that had detected concentrations of Aroclor 1016/1242. The analyst could not distinguish between these two Aroclor patterns. The maximum concentration was 13 µg/kg ww. Neither Aroclor was detected in any other sample in any sampling event in the LDW or the EW. For EW tissue samples, Aroclors 1254 and 1260 were detected frequently, so the total PCB sums presented above include detected concentrations and one-half RLs for those two Aroclors.

EW – East Waterway
EM – edible meat
LDW – Lower Duwamish Waterway
PCB – polychlorinated biphenyl
RL – reporting limit
WB – whole body
ww – wet weight

Overall, as indicated by the similar average total PCB concentrations in Table B.6-2, both total PCB calculation methods would yield very similar risk estimates for seafood consumption. Given the small difference between the two methods for summing PCBs, and the complexity associated with combining information from two datasets, it was felt that the approach outlined in WAC 173-204 should be applied in this risk assessment. The same approach was used in the EW ERA and the LDW HHRA.

In addition to the different summation methods for calculating total PCBs based on Aroclors, a total PCB sum can be calculated using congener data. A comparison of total PCBs as Aroclors and total PCBs as congeners in tissue samples is presented here to facilitate an assessment of the uncertainty associated with characterizing PCB risks based on the Aroclor data. Both Aroclor and congener data are available for each of the 10 seafood consumption categories. Table B.6-3 presents PCB EPCs for three datasets:

- Sum of congeners (n = 28 for tissue; n = 13 for subtidal composite surface sediment; and n = 4 for intertidal MIS composite surface sediment)

- Sum of Aroclors using only those data that were also analyzed for congeners (n = 28 for tissue; n = 13 for subtidal composite surface sediment; and n = 4 for intertidal MIS composite surface sediment)
- Sum of Aroclors using the entire HHRA dataset (i.e., the same EPCs presented in Section B.3.3.4; n = 124 for tissue; n = 237 for subtidal grab surface sediment; and n = 4 for intertidal MIS composite surface sediment)

Table B.6-3. EPCs for total PCBs based on PCB congener and Aroclor sums

	Total PCBs					
	Samples Analyzed for Both Congeners and Aroclors				Sum of Aroclors Using Entire HHRA Dataset (as Presented in Section B.3)	
	Sum of Congeners		Sum of Aroclors			
Consumption Category or Exposure Area Type	Number of Samples	EPC (mg/kg ww)	Number of Samples	EPC (mg/kg ww)	Number of Samples	EPC (mg/kg ww)
Seafood Consumption Category						
Benthic fish, fillet	3	1.2[a]	3	1.6[a]	20	2.4[b]
Benthic fish, whole body	3	2.4[a]	3	2.0[a]	13	4.1[b]
Clams	3	0.072[a]	3	0.082[a]	11	0.069[b]
Crab, edible meat	3	0.11[a]	3	0.093[a]	12	0.16[b]
Crab, whole body	3	0.30[a]	3	0.41[a]	9	0.45[b]
Geoduck, edible meat	3	0.026	3	0.024	6	0.022
Geoduck, whole body	1	0.032	1	0.025	4	0.034
Mussels	nd	nd	nd	nd	17	0.031
Pelagic fish, perch	3	0.76[a]	3	1.1[a]	17	1.6[b]
Pelagic fish, rockfish	6	3.9	6	3.4	15	4.0
Sediment Exposure Area						
Site-wide	3 intertidal;[c] 13 subtidal	1.2	3 intertidal;[c] 13 subtidal	0.75	3 intertidal;[c] 237 subtidal	0.79
Site-wide intertidal area	3[c]	1.4	3[c]	1.9	3[c]	1.9
Public access intertidal area	1[c]	1.4	1[c]	2.0	1[c]	2.0

a This EPC is based on data for supercomposite samples (see Attachment 1).

b The supercomposite Aroclor data were not included in this EPC (consistent with Section B.3 and the risk characterization).

c These samples are the intertidal MIS composite samples, three of which were used to represent the larger accessible intertidal area (i.e., not under piers, accessible by boat or shoreline access).A fourth sample (not included in the site-wide intertidal dataset in this table) was used to represent only the public access intertidal area.

EPC – exposure point concentration

nd – no data

PCB – polychlorinated biphenyl

ww – wet weight

It should be noted that for the same sample type, the 95% UCLs based on congeners, which were analyzed for in a smaller number of samples than those analyzed for Aroclors, may be different than the 95% UCLs based on Aroclors, because of differences in the treatment of the data by ProUCL 4. However, the main differences between EPCs based on Aroclors and those for congeners are likely the result of the differences in analytical methods and analytical variability, rather than the treatment of the data by ProUCL.

Aroclors were summed according to the primary method described above and in Section B.2.2.4. Total PCBs based on PCB congeners were calculated as the sum of detected congeners in each sample. The tissue samples analyzed for PCB congeners were analyzed for all 209 congeners, with an average detection frequency of 84% (i.e., in each of the 28 samples, an average of 84% of the 209 congeners tested were detected).

In general, EPCs based on the three datasets presented in Table B.6-3 were relatively similar, both for the seafood consumption categories and sediment exposure areas. Differences between the EPCs based on the sum of congeners and the EPCs based on sum of Aroclors calculated using the same datasets (the first two EPC columns presented in the table) could result from differences in analytical methods or from analytical variability. With the exception of the site-wide exposure area EPCs, these values (i.e., the first two EPC columns) are generally similar. The last column of the table, which presents the EPCs based on the sum of Aroclors using the full dataset used to calculate EPCs for the EW HHRA, has been included to enable an evaluation of the differences between the smaller subset of data analyzed for congeners and the larger dataset analyzed for Aroclors that was used for the risk characterization. The two Aroclor-based EPCs are generally similar, with the greatest differences being for the benthic fish and crab EPCs.

Excess cancer risks and non-cancer hazards for the adult tribal RME seafood consumption scenario based on Tulalip data were estimated using EPCs from both total PCBs as congeners and total PCBs as Aroclors (Table B.6-4). The excess cancer risk estimate and HQ based on the sum of congeners were 10 to 20% lower than the estimates based on the sum of Aroclors for the equivalent dataset and were 30 to 40% lower than the excess cancer risk estimate and HQ based on the sum of Aroclors using the full HHRA dataset. This indicates that the use of the full Aroclor dataset in the risk characterization resulted in higher and more health-protective risk estimates.

Table B.6-4. Excess cancer risks and non-cancer hazards for PCBs for select RME exposure scenarios

Chemical	Excess Cancer Risk	Hazard Quotient
Adult Tribal RME Seafood Consumption Scenario Based on Tulalip Data		
Total PCBs (sum of congeners)[a]	7×10^{-4}	16
Total PCBs (sum of Aroclors)[a]	8×10^{-4}	20
Total PCBs (sum of Aroclors, as reported in risk characterization section for all data)[b]	1×10^{-3}	27
Netfishing RME Direct Sediment Exposure Scenario		
Total PCBs (sum of congeners)[a]	9×10^{-7}	0.04
Total PCBs (sum of Aroclors)[a]	6×10^{-7}	0.03

Table B.6-4. Excess cancer risks and non-cancer hazards for PCBs for select RME exposure scenarios (cont.)

Chemical	Excess Cancer Risk	Hazard Quotient
Total PCBs (sum of Aroclors, as reported in risk characterization section for all data)[b]	6×10^{-7}	0.03
Tribal Clamming RME Direct Sediment Exposure Scenario		
Total PCBs (sum of congeners)[a]	3×10^{-6}	0.08
Total PCBs (sum of Aroclors)[a]	3×10^{-6}	0.1
Total PCBs (sum of Aroclors, as reported in risk characterization section for all data)[b]	3×10^{-6}	0.1

a Excess cancer risks and hazard quotients are based on EPCs for 28 samples (see Table B.6-3).

b Excess cancer risks and hazard quotients are based on EPCs for 124 samples (see Table B.6-3).

CDI – chronic daily intake

PCB – polychlorinated biphenyl

RME – reasonable maximum exposure

B.6.1.1.4 Use of half RLs to calculate TEQs

As discussed in Section B.2.2.4, one-half of the RL was used for non-detected component chemicals when calculating the cPAH TEQs, PCB TEQs, and dioxin/furan TEQs. However, TEQs could also be calculated by using 0 or full RLs to represent the non-detected components chemicals. Table B.6-5 presents detection frequencies and EPCs for seafood consumption categories.

Table B.6-5. Comparison of EPCs for TEQs calculated using full RLs, half RLs, or 0 for non-detected components

Seafood Consumption Category	cPAH TEQ EPCs[a] (µg/kg ww)				PCB TEQ EPCs[a] (ng/kg ww)				Dioxin/Furan TEQ EPCs[a] (ng/kg ww)			
	DF[b]	RL = Full	RL = Half	RL = 0	DF[b]	RL = Full	RL = Half	RL = 0	DF[b]	RL = Full	RL = Half	RL = 0
Benthic fish, fillet	3/11	0.80	0.42	0.21	3/3	15	15	15	3/3	0.91	0.79	0.76
Benthic fish, WB	9/11	5.1	6.8	3.9	3/3	37	37	37	3/3	1.9	1.9	1.9
Clam	11/11	27	27	27	3/3	0.75	0.73	0.72	3/3	0.66	0.38	0.10
Crab, EM	6/9	1.3	1.1	0.8	3/3	2.1	1.7	1.7	3/3	0.58	0.49	0.46
Crab, WB	7/7	1.2	1.1	1.0	3/3	5.6	5.6	5.6	3/3	1.3	1.3	1.2
Geoduck, EM	6/6	2.3	2.2	2.1	3/3	0.21	0.19	0.18	3/3	0.49	0.25	0.02
Geoduck, WB	4/4	4.1	4.1	4.1	1/1	0.24	0.23	0.22	1/1	0.29	0.20	0.12
Mussel	16/17	76	59	73	nd	nd	nd	nd	nd	nd	nd	nd
Pelagic fish, perch	6/8	2.1	1.6	1	3/3	14	14	14	3/3	1.4	1.4	1.3
Pelagic fish, rockfish	0/12	0.6	0.29	0	6/6	41	40	39	6/6	3.0	2.8	2.6

a Although EPC values are generally highest when the full RL is used for TEQ summation and lowest when 0 is used in place of the RL for TEQ summation, this pattern does not always hold true because of the use of ProUCL to calculate EPCs with six or more detected values (i.e., the use of one of the assumptions for non-detects could result in a broader or narrower variance among the samples, thus resulting in a higher or lower EPC than expected).

b Detection frequencies are for the TEQ sum, not for the components (if one or more component chemical was detected, the sum is considered to be detected). Thus, even for TEQs that are considered detected, the value may change as a result of using a different RL assumption since one or more components may not have been detected.

Table B.6-5. Comparison of EPCs for TEQs calculated using full RLs, half RLs, or 0 for non-detected components (cont.)

cPAH – carcinogenic polycyclic aromatic hydrocarbon
DF – detection frequency (ratio)
EM – edible meat
EPC – exposure point concentration
nd – no data
RL – reporting limit
TEQ – toxic equivalent
WB – whole body
ww – wet weight

As can be seen in Table B.6-5, the EPCs can be quite variable, particularly for the less-frequently detected consumption categories. For example, the cPAH TEQ EPC for benthic fish fillets ranged from 0.80 using the full RL to 0.21 using 0 for non-detected results. For PCB TEQ and dioxin/furan TEQ, detection frequencies were higher, so the EPCs are generally less variable. However, it should be noted that the detection frequencies presented in this table are for the TEQ sum, not for the components (if one or more component chemical was detected, the sum is considered to be detected). Thus, although the PCB TEQs and dioxin/furan TEQs are shown as having 100% detection frequencies, some variability in the EPC exists because not all of the components were detected in every sample. Using the EPCs presented in Table B.6-5, excess cancer risks were calculated for the adult tribal RME scenario based on Tulalip data to evaluate the impact of this uncertainty on risk estimates. For all three of these TEQs, risk estimates either did not change (for cPAH TEQ and PCB TEQ) or changed only slightly (for dioxin/furan TEQ, the risk estimate did not change when the full RL was used but decreased from 1×10^{-4} to 8×10^{-5} when 0 was used in place of the RL). This limited impact on risk estimates was expected because the use of different assumptions for the non-detected components resulted in relatively small changes in the EPCs that contributed the most to the TEQ risk estimates (clams for cPAH TEQ, benthic fish fillet/perch/rockfish for PCB TEQ, and clams/crab edible meat/whole body crab/rockfish for dioxin/furan TEQ). Thus, the overall impact on risk estimates is low.

B.6.1.1.5 EPCs for infrequently detected chemicals

Although most HHRA datasets had a sufficient number of samples and detected concentrations to allow the use of ProUCL in calculating the EPC, there were some cases in which ProUCL could not be used to develop EPCs because there were fewer than six detected values. Developing 95% UCL EPCs for datasets with very few detected concentrations is not statistically feasible. As summarized in Figure B.3-3, a decision was made for the EW HHRA, based both on ProUCL guidance (EPA 2009c) and discussions with EPA, to set the EPC at the higher of one-half the maximum RL or the maximum detected value for datasets with fewer than six detected values. In these cases, the "true" EPC may be overestimated because the complete dataset, including detected and non-detected values that are lower than the maximum, is not considered in the calculation. The EPC may also be underestimated because the true mean may be higher than the maximum value of a small dataset because the distribution for most chemical contamination data may be positively skewed. This uncertainty is unavoidable when only a few samples are available to characterize exposure (see also Section B.6.1.1.6). However, it should be noted that in some cases, when there were fewer than six detected values, it was necessary to use the maximum value as the EPC for detected COPCs. This was necessary for 1 of 9 (11%) EPCs for COPCs in subtidal sediment and for 174 of 278 (63%) EPCs for COPC-seafood consumption category combinations (over half of these cases for seafood COPCs were for pesticides or TEQs, both of which were typically analyzed for in five or fewer samples for each seafood consumption category). For detected COPCs, subtidal sediment data and surface water datasets contained a sufficient number of detected values for the use of ProUCL in the calculation of EPCs (i.e., it was not necessary to use maximum values). Overall, the impact of this uncertainty on total risk estimates is expected to be low because a sufficient number of detected concentrations were available for the chemicals that drive overall risk conclusions for this HHRA.

B.6.1.1.6 EPCs for small datasets

The approach for EPC estimation based on dataset size and number of detected concentrations is presented in Section B.3.3.4. ProUCL 4 was used to develop EPCs for datasets with six or more detected samples. ProUCL 4 takes into account sample size and distribution in its recommendation of a 95% UCL. In some cases, the recommended 95% UCL is greater than the maximum sample concentration. For datasets with fewer than six samples, a decision was made, in consultation with EPA and for consistency with the LDW HHRA (Windward 2007c), to use the maximum detected concentration or one-half the maximum RL, whichever was higher. Therefore, uncertainty related to potential EPC underestimation is greater for EPCs derived for datasets with fewer than six samples. Uncertainties related to infrequently detected chemicals were discussed in Section B.6.1.1.5.

For some datasets, the available sample size was very small (irrespective of the number of detections). The following describes the EPCs based on small sample sizes:

- **Tissue EPCs –** Of the over 500 tissue EPCs, approximately 140 EPCs were developed based on five or fewer samples. The majority of these datasets with five or fewer samples were for PCB congeners, dioxins/furans, pesticides, bis(2-ethylhexyl) phthalate (BEHP), or pentachlorophenol (see discussion in Attachment 1).
- **Intertidal sediment EPCs –** EPCs for the intertidal sediment dataset were based on three site-wide intertidal MIS samples (used in the tribal clamming and habitat restoration worker scenarios) or on the one public access area MIS sample (used in the clamming – 7 days per year scenario). ProUCL was not used to determine EPCs for these intertidal samples; uncertainty regarding the calculation of the MIS 95% UCLs is discussed in Section B.6.1.1.7.
- **Subtidal sediment EPCs –** All subtidal EPCs were based on six or more samples, for which ProUCL was used. These EPCs were used along with the intertidal sediment EPCs to evaluate risks associated with netfishing.

- **Surface water EPCs –** All EPCs were based on six or more samples, for which ProUCL was used.

When only five or fewer samples were available, the EPCs used in the risk calculation were based on a maximum detected concentration or one-half the maximum RL, except in the case of the MIS samples (see Section B.6.1.1.7). For datasets with five or fewer values, the uncertainty about whether the EPC is equal to or exceeds the mean is much higher than that for datasets with six or more samples, for which ProUCL 4 was used to estimate the EPC. As mentioned, for datasets with fewer than six samples (other than the MIS datasets) a decision for EPC selection was made and no 95% UCL was estimated or incorporated into the EPC. Because there is some unknown probability that the true mean is actually greater than the maximum detected value, it is possible that the use of the maximum value for the EPC for datasets with fewer than six detects may underestimate the mean in some cases. Because the likelihood of the maximum value being less than the mean for small datasets is unknown, the potential magnitude of this uncertainty in terms of its effect on the risk estimates is also unknown. However, because this uncertainty was generally not applicable for the COCs with the highest risk estimates, the overall impact of this uncertainty is considered to be low.

B.6.1.1.7 EPCs intertidal sediment calculated using MIS samples

For the clamming and habitat restoration worker scenarios, which assumed exposure to only intertidal sediment, MIS samples were collected to estimate chemical concentrations. Each MIS sample was composed of 30 grab samples from throughout the intertidal areas shown on Map B.2-2. For the MIS samples, 95% UCLs were calculated using the methodology described briefly below. Additional information on the methodology was provided in Section B.3.3.4.2. For the site-wide intertidal exposure area (used for the tribal clamming scenarios, the habitat restoration worker scenario, and the intertidal portion of the netfishing scenario, as discussed in Section B.3.3.4.2), the mean and standard error of the three MIS samples were used to calculate the 95% UCL. For the public access intertidal area (used for the clamming – 7 days per year scenario), the single public access MIS sample concentration was used along with the standard deviation of the three site-wide MIS samples to calculate a 95% UCL, as described in Section B.3.3.4.2.

Although the MIS samples were designed to represent average concentrations across the intertidal area, the variability across the three MIS samples resulted in 95% UCLs that were higher than the maximum MIS sample concentration for all COPCs, particularly for the public-access intertidal area 95% UCLs (Table B.6-6). This health-protective approach may overestimate the true mean value for the intertidal exposure area and thus could overestimate risk. However, because information regarding the true variability of the intertidal sediment concentrations is unknown, the magnitude of this uncertainty is unknown. The overall potential impact on the risk estimates was categorized as medium.

Table B.6-6. Range of concentrations and 95% UCLs for MIS datasets

Chemical	Unit	Site-Wide Intertidal Area		Public Access Intertidal Area	
		Concentration Range	95% UCL	Concentration	95% UCL
Arsenic	mg/kg dw	7.9 – 13.3	15	7.7	16
Cobalt	mg/kg dw	5.7 – 6.8	7.1	5.0	6.6
Vanadium	mg/kg dw	34.3 – 45.5	51	31 J	48
cPAH TEQ	µg/kg dw	450 – 1,900	2,300	390	2,600
PCB TEQ	ng/kg dw	3.3 – 6.31	7.2	1.4	6.1
Total PCBs	µg/kg dw	540 – 1,590	1,900	370	2,000
Dioxin/furan TEQ	ng/kg dw	9.19 – 13.8	16	8.5	16

cPAH – carcinogenic polycyclic aromatic hydrocarbon
dw – dry weight
EPC – exposure point concentration
J – estimated concentration
MIS – multi-increment sampling
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
UCL – upper confidence limit on the mean

B.6.1.1.8 Spatial bias in sediment EPC estimates

Sediment samples used to calculate EPCs for the subtidal portion of the netfishing scenario were collected from throughout the EW. Map B.2-1 shows the distribution of grab samples, and Map B.2-2 shows the distribution of samples and areas for composite samples (individual grab samples were analyzed for some chemicals; composite samples were analyzed for other chemicals, as discussed in Attachment 1). As can be seen on these maps, the distribution of samples is relatively consistent throughout the EW, and thus there is no suspected spatial bias in the use of arithmetic EPCs.

To evaluate the uncertainty associated with spatial bias, the average chemical concentration in the subtidal portion of the EW was calculated using both the arithmetic mean of all samples and the spatially weighted average concentration (Table B.6-7). As presented in this table, the netfishing EPCs based on the subtidal area average (i.e., arithmetic mean approach) were 8 to 23% higher than the EPCs based on the spatially weighted average (i.e., a spatially weighted average concentration [SWAC] approach) for arsenic, cPAH TEQ, and total PCBs. Thus, if spatial weighting were to be taken into account for the calculation of EPCs, risks could be slightly lower, although using spatially weighted EPCs would not change the risk conclusions.

Table B.6-7. Comparison of EPCs for the netfishing RME scenario using various calculation methods

COPC	Detection Frequency	Unit	Arithmetic Mean Approach		SWAC Approach	
			Subtidal Mean[a]	Site-Wide EPC[b]	Subtidal SWAC[c]	Site-Wide EPC[b]
Arsenic	157/227	mg/kg dw	10	10	8.3	8.5
cPAH TEQ	229/237	µg/kg dw	500	510	340	390
Total PCBs	223/237	µg/kg dw	530	540	460	500

a The subtidal mean concentration was calculated as a average of all samples.

b The netfishing RME scenario assumes a site-wide exposure. To calculate the site-wide EPC, the subtidal EPC and intertidal EPC were weighted based on the percentage of area for each, as discussed in Section B.3.3.4.2. In this table, the intertidal EPC was not changed. The approach refers to the subtidal EPC, which made up 97.3% of the weighted total.

c The subtidal SWAC was calculated using the IDW approach.

COPC – chemical of potential concern
cPAH – carcinogenic polycyclic aromatic hydrocarbon
dw – dry weight
EPC – exposure point concentration
IDW – inverse distance weighting
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
SWAC – spatially weighted average concentration
TEQ – toxic equivalent

B.6.1.2 Ingestion rates

B.6.1.2.1 Incidental sediment ingestion rates

Incidental sediment ingestion rates for the netfishing, habitat restoration, and clamming scenarios were evaluated using soil ingestion rates identified in EPA guidance. This approach is commonly used in HHRAs, but it is not clear to what extent incidental soil ingestion rates are applicable to the evaluation of incidental sediment ingestion. For example, the amount of sediment transferred to a fisherman's hands when handling monofilament gill nets is not known.

B.6.1.2.2 EW adult seafood consumption rates assumed for this HHRA

EW-specific estimates of seafood consumption rates were not available (e.g., for Muckleshoot and Suquamish Indian Tribe members, API, or recreational users. However, it has been documented that some anglers (many of whom are API) use the Spokane Street Bridge as a fishing location (King County 1999a) and that tribal commercial netfishing occurs on the EW.

As described in Section B.3.3.1, the seafood consumption rates assumed for the adult tribal scenario based on Tulalip data, adult tribal scenario based on Suquamish data, and adult API scenario were provided by EPA (EPA 2005a; Kissinger 2005; EPA 2009b) and were based on recent regional seafood consumption studies (EPA 1999a; Suquamish Tribe 2000; Toy et al. 1996). In addition, the ingestion rates used in this document are consistent with those used in the LDW HHRA (Windward 2007c, 2009c). The child tribal consumption rate based on Tulalip data presented in the risk characterization was derived, in part, from a ratio applied to the adult tribal consumption rate based on Tulalip data, as described in Section B.3.3.1. The uncertainties specifically associated with the child tribal consumption rate based on Tulalip data and the development of a child tribal consumption rate based on Suquamish data are discussed in Section B.6.1.2.3. The adult seafood consumption rates were based on surveys that appeared to fairly represent the populations that were interviewed (EPA 1999a; Suquamish Tribe 2000; Toy et al. 1996).

The groups whose surveys were used to develop the rates used in this HHRA, do not use the EW as their primary fishing area, and thus the degree to which the rates represent people who currently or may in the future consume fish and shellfish from the EW is not known. It is possible that some individuals could get a significant portion of their seafood from the EW and may consume seafood at high rates. The behavior of these individuals, those who may be using the EW for subsistence fishing, might not be captured by most seafood consumption surveys, even if the surveys were focused on the EW (e.g., because subsistence fishers may collect seafood during non-business hours when surveys are usually conducted, individuals may not want to participate, and/or language barriers may limit participation). The consumption rates for individuals who may be engaging in subsistence fishing on the EW is unknown but could be similar to upper percentile tribal or API consumption rates (EPA 1999a; Suquamish Tribe 2000; Toy et al. 1996). The consumption continuum figures presented in Section B.5.6.4 illustrate the risks associated with different seafood consumption rates. These figures illustrate that assumptions regarding which consumption rates are appropriate for a given scenario can have significant effects on risk estimates. Additionally, it is important to recognize that the EW is within the larger Duwamish River and Elliott Bay environment. All individual sites within a larger water body must be appropriately addressed to reduce risks associated with the consumption of contaminated seafood.

EPA's interpretation of the seafood consumption studies required numerous assumptions to develop consumption rates. For example, the total seafood consumption rate was allocated among 10 seafood categories based on the reported mean consumption of each seafood category, regardless of the source of the seafood (i.e., regardless of whether it was self-caught, store-bought, or from some other source (EPA 2005a; Kissinger 2005)). For the tribal populations evaluated in this HHRA, this assumption is reasonable because the majority of the seafood consumed by these populations is self-caught. However, based on the survey of the API population (EPA 1999a), the majority of their consumed seafood is purchased in stores. Less than a quarter of the overall fish consumption reported in the API survey was self-harvested (EPA 1999a). The initial total API seafood consumption rate used in the risk calculations was developed using demographically weighted data for consumers of King County species and was intended to reflect the 95th percentile of API consumption of seafood from only King County (as described in Section B.3.3.1.3) (Kissinger 2005). The percentages of consumption for the different seafood categories were derived using the same data (demographically weighted) as that used for the consumption of only King County seafood. However information about the preparation style for the crab (whole body vs. edible meat) and benthic fish (whole body vs. fillet) in the dataset did not distinguish between King County seafood and seafood from other sources (such as store-bought seafood). Thus, the crab and benthic fish apportionments have uncertainty as to how well they reflect the consumption of EW seafood. In addition, as requested by EPA (2006b), the consumption of freshwater fish reported in the survey was reapportioned to other marine categories, and it was assumed that there was no freshwater fish consumption in the scenarios used in the risk assessments (see Section B.3.3.1.3). There is also uncertainty related to how much of the reported King County-harvested seafood is harvested in the EW.

Another uncertainty regarding the interpretation of the API survey relates to the difficulties of characterizing consumption for the many diverse ethnic groups included in the study. As discussed in Section B.3.3.1, the sample sizes for the 10 ethnic groups included in the study were generally not demographically representative of the API population in King County. Survey results were adjusted statistically to make them representative of the King County population (Kissinger 2005). Despite this adjustment, several ethnic groups were represented by small sample sizes (n = 10 or fewer). Defining a consumption rate for a large population that includes several groups represented by a small number of survey respondents may result in substantial uncertainty. For example, many individuals in the API survey reported no consumption of King County seafood during the interviews, while others reported very high percentages (EPA 1999a). The reported estimate for 50th percentile consumption was 5.8 g/day, as compared with the 95th percentile estimate of 57.1 g/day, which was used for risk estimates in this document (Kissinger 2005).[49] The uncertainties related to the characterization of a single seafood consumption rate to represent the many diverse API ethnic groups included in the survey are reflected in the wide range of the upper and lower confidence bounds for the estimate of total King County 95th percentile consumption (approximately 25 g/day to approximately 80 g/day) (Kissinger 2005) and should be considered in interpreting the API risk estimates.

The seafood consumption rates from the Suquamish (2000) and Tulalip Tribes (Toy et al. 1996) studies were based on surveys of tribal members who consume seafood from outside the EW. The Suquamish and Tulalip Tribes consume seafood from Puget Sound habitats that differ considerably in terms of quality and quantity from those of the EW. Consequently, it is not known how well these tribal seafood consumption rates apply as surrogates for tribal seafood consumption rates specific to the EW, particularly for clams. The consumption rates used in calculating the CDI equate to approximately 900 and 11,000 clams per year for the adult tribal RME scenario based on Tulalip data and the adult tribal scenario based on Suquamish data, respectively, for each individual who consumes clams, assuming a weight of approximately 15 g ww for each clam.

[49] As described in Section B.3.4.1.3, the anadromous fish portion of consumption rate was not included in the adult API RME and CT exposure scenarios.

As part of the LDW HHRA (Windward 2007c), EPA acknowledged the importance of habitat quality in selecting seafood consumption rates that should be applied to the LDW (EPA 2007b): "As a policy decision, for sites in the Puget Sound and Strait of Georgia that lack extensive intertidal habitat, the consumption rate derived by EPA from data from the Tulalip Tribes represents a sustainable consumption rate." Based on these considerations, EPA selected the consumption rates based on the Tulalip Tribes as most appropriate for the LDW (EPA 2005a, 2009b). Furthermore, as stated in the LDW application of the EPA tribal seafood consumption framework (EPA 2005a), "EPA believes that use of Suquamish exposure parameters will not provide the best estimate of LDW tribal seafood consumption risks due to the degraded habitat in the LDW and questions whether the high Suquamish shellfish consumption rate could be sustained." This same rationale is considered to be applicable to the EW, which has an even smaller fraction of quality habitat than does the LDW.

As with the LDW, the ability of EW habitats to support the clam populations that would be necessary to sustainably achieve the clam consumption rates that have been assumed in the EW HHRA is unknown. Clam habitat in the EW is limited as a result of the presence of steep banks of riprap, concrete, and other construction materials. Future habitat improvements may increase the quantity of harvestable clams, but some physical constraints are likely to remain.

The seafood consumption rates used in this HHRA do not include salmon, as explained in Sections B.3.3.1. Some of the chemicals detected in adult salmon that return to the Duwamish River may have originated in the EW during juvenile outmigration. However, given the great size difference between juvenile and adult salmon, a significant growth dilution occurs during the lifetime of these fish. In addition, most of the salmon's life is spent outside the EW, such that the body burden of bioaccumulative chemicals in an adult salmon is largely attributed to contaminant uptake that occurred outside the EW. An example calculation presented in Section B.2.1.2 of the LDW HHRA (Windward 2007c) suggests that the fraction of the PCB body burden in an adult Chinook salmon that can be attributed to direct exposure within the EW during juvenile outmigration is less than 1%. Recent studies indicate that adult salmon have higher PCB body burdens as a result of time spent in Puget Sound (PSAT 2007; Missildine et al. 2005). The transport of contaminants from the EW to Elliott Bay or other areas of Puget Sound could also result in the indirect uptake of site-related contaminants during residence within Puget Sound. The exclusion of salmon from seafood consumption scenarios and ingestion rates will underestimate site-related contaminant exposures and overall seafood consumption risks. However, because the portion of total body burden that results from the uptake of contaminants from the EW is expected to be small, the effect on risk estimates of excluding salmon from the seafood consumption scenarios is thought to be negligible.

B.6.1.2.3 *Child seafood consumption rates*

There are a number of uncertainties associated with the child tribal seafood consumption information. In general, the regional tribal seafood consumption surveys included smaller numbers of children than adults and had a higher percentage of children reported as non-consumers than adults (Table B.6-8). Therefore, estimates of children's consumption have additional uncertainties beyond many of those described above for the adult seafood consumption scenarios. Risks presented in Section B.5 for the child seafood consumption scenario based on Tulalip data were calculated using consumption rates derived as a percentage of the adult tribal consumption rate based on Tulalip data, rather than the child consumption rates from the Tulalip Tribes survey (Toy et al. 1996). Because there are uncertainties in this approach and because actual child consumption rate data are available, an alternative approach to assessing child seafood consumption risks using actual Tulalip child consumption rate data is presented in this section.

Table B.6-8. Number of participants in child and adult tribal seafood surveys

Tribe	No. of Children Surveyed	No. of Children Who Consumed Seafood	No. of Adults Surveyed	No. of Adults Who Consumed Seafood	Source
Tulalip Tribes[a]	21	15	73	73	Toy et al. (1996)
Squaxin Island Tribe[a]	48	36	117	117	Toy et al. (1996)
Suquamish Tribe[b]	31	31	92	92	Suquamish Tribe (2000)
Nez Perce, Yakama, Warm Springs, Umatilla	194	153	513	477	CRITFC (1994)

a Less than 1% of those (adults) contacted were excluded due to non-consumption of fish.

b All (adult) respondents consumed at least one type of fish or shellfish. Thus, no respondents were excluded because of non-consumption.

This section also presents a child tribal scenario based on Suquamish data and the associated risk estimates. As has been noted previously in this risk assessment regarding the evaluation of seafood consumption for individual API ethnic groups, drawing conclusions from small numbers of individuals creates uncertainty. In addition, as for any non-observational survey of young children, children's consumption rates were obtained by interviewing adults in the same household. Finally, for the Suquamish survey, multiple children may have been selected from the same household, leading to a lack of independence in the recorded data. This issue is discussed further as part of the child tribal seafood consumption scenario based on Suquamish data.

In comparing upper percentiles of children's seafood consumer-only consumption rates, the child rates for the Tulalip Tribes are lower than those reported for other tribes. Hence, use of the child rates for the Tulalip Tribes may underestimate children's exposures for other tribes. For this reason, 40% of the adult Tulalip Tribes 95th percentile consumption rate (194 g/day),[50] or 77.6 g/day, was used to assess tribal children's seafood consumption risks in the risk characterization section. The rate of 77.6 g/day falls within the 95th percentiles of child tribal seafood consumption rates estimated from other studies (Table B.6-9).

Table B.6-9. Child tribal seafood consumption rates

Tribe	Child Seafood Consumption (g/day)		Source
	90th Percentile	95th Percentile	
Tulalip Tribes[a]	13.3[a]	20.4[b]	Toy et al. (1996)
Squaxin Island Tribe[b]	42.4[a]	115[a]	Toy et al. (1996)
Suquamish Tribe	50.6	122.2	Suquamish Tribe (2000)
Nez Perce, Yakama, Warm Springs, Umatilla	53.2[c]	71.6[c]	CRITFC (1994)

a Based on a re-analysis of the original study data (EPA 2006c).

b The 95th percentile was computed using a lognormal distribution fit to the Tulalip children's consumption data (EPA 2006c; Kissinger 2007b).

c Consumption rates derived using data for only consumers (CRITFC 1994).

Child Tribal 95th Percentile Seafood Consumption Scenario Based on Tulalip Data

Several approaches to developing child tribal consumption rates based on Tulalip data have been proposed and discussed in this HHRA. In the risk characterization, a ratio approach recommended by EPA (EPA 2006c) as part of the LDW HHRA (Windward 2007c, 2009c) was used to develop the child tribal RME exposure scenario based on Tulalip data (Section B.3.3.1.2). Despite the issues related to the small sample size noted above, the children's data for the Tulalip Tribes are a measure of existing tribal children's seafood consumption. Thus, the Tulalip Tribes child seafood consumption data (Toy et al. 1996) were used to provide an alternative estimate of children's seafood consumption risks.

As part of the LDW HHRA (Windward 2007c, 2009c), EPA (2006c) calculated a 95th percentile consumer-only seafood consumption rate for children of 20.4 g/day based on seafood consumption reported in the Tulalip Tribes survey (Kissinger 2007b).[51] This rate reflects the total reported seafood consumption from any source. The apportionment of this rate among seafood categories, based on the Tulalip child data (Toy et al. 1996), is presented in Tables B.6-10 through B.6-12. When data from the children's survey were available, apportionment was based on the children's consumption data. When children's data were lacking, apportionment was based on Tulalip Tribes adult consumption data (Toy et al. 1996). The approach for the

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[50] Forty percent of the adult seafood consumption rate is an option provided in EPA's draft Framework for Selecting and Using Tribal Fish and Shellfish Consumption Rates for Risk-Based Decision Making at CERCLA and RCRA Cleanup Sites in Puget Sound and the Strait of Georgia (EPA 2007b).

[51] The Tulalip Tribes survey included 21 children aged 0 to 5 years, although only 15 consumed seafood (Toy et al. 1996).

apportionment was the same as that used for the adult tribal RME scenario based on Tulalip data, as described in Section B.3.3.1.1. The total consumption was first broken down into broad seafood groups and then into edible-meat and whole-body portions, as applicable.

Table B.6-10. Percentages and rates associated with different seafood categories for the child tribal 95th percentile seafood consumption scenario based on Tulalip data

Seafood Category	Percentage of Total Seafood Consumption[a]	Consumption Rate (g/day)[b]
Anadromous fish[c]	28	5.7
Pelagic fish	18	3.7
Benthic fish	1	0.2
Shellfish	53	10.8

a Calculated from average consumption rates by seafood category based on the Tulalip child data (Toy et al. 1996).

b Calculated by multiplying the 95th percentile of total seafood consumption (20.4 g/day) (Kissinger 2007b) by the percentage of consumption by Tulalip children for the various seafood categories using data for children from Toy et al. (1996).

c The consumption of anadromous fish was not included in this HHRA.

HHRA – human health risk assessment

LDW – Lower Duwamish Waterway

Table B.6-11. Consumption of shellfish for the child tribal 95th percentile seafood consumption scenario based on Tulalip data

Shellfish Type	Percentage of Total Shellfish Consumption[a]	Consumption Rate (g/day)[b]
Crabs	42	4.5
Clams[c]	48	5.2
Mussels	1	0.1
Geoduck	9	1

a These are the same consumption percentages as those used for the adult tribal RME scenario based on Tulalip data (Table B.3-6).

b Calculated by multiplying the child tribal shellfish consumption rate based on Tulalip data (10.8 g/day [Table B.6-10]) by the percentage of adult Tulalip shellfish consumption for each category.

c Includes Manila/littleneck clams, horse clams, butter clams, cockles, oysters, and scallops.

RME – reasonable maximum exposure

Table B.6-12. Portions of pelagic fish, crab, and geoduck consumed for the child tribal 95th percentile scenario based on Tulalip data

Species or Tissue Type	Percentage of Consumption[a]	Consumption Rate (g/day)[b]
Pelagic Fish		
Perch	88	3.3
Rockfish	12	0.4
Crab		
Edible meat	76	3.4
Whole body	24	1.1
Geoduck		
Edible meat	88	0.9
Whole body	12	0.1

[a] These are the same consumption percentages as those used for the adult tribal RME scenario based on Tulalip data (Tables B.3-7 and 3-8).

[b] Calculated by multiplying the child tribal consumption rate based on Tulalip data by the percentage of adult Tulalip consumption for each category.

RME – reasonable maximum exposure

The child tribal 95th percentile consumption rates based on Tulalip data were used with EPCs for the different seafood categories (Section B.3.3.4) to develop high-end excess cancer risk and non-cancer hazard estimates (Table B.6-13). In this uncertainty analysis, estimated child tribal 95th percentile consumption rates based on Tulalip data (Table B.6-10 to B.6-12) were used with other exposure parameters (e.g., body weight, exposure duration) from Table B.3-17. As shown in Table B.6-13, the total excess cancer risk estimate based on the 95th percentile consumption exceeded 1×10^{-6} but was approximately half the total excess cancer risk estimates for the child tribal RME scenario based on Tulalip data using the 40% adult ratio (Table B.5-3). The most significant contributors to the excess cancer risk were PCBs (total PCBs and PCB TEQ) and cPAH TEQ. The HQs for the child tribal 95th percentile scenario based on Tulalip data (Table B.6-14) were approximately one-half of the estimates for the child tribal RME scenario based on Tulalip data using the 40% adult ratio presented in Table B.5-11. Only total PCBs had an HQ that exceeded 1 for the child tribal 95th percentile scenario based on Tulalip data (Table B.6-14).

Table B.6-13. Excess cancer risk estimates for the child tribal 95th percentile seafood consumption scenario based on Tulalip data

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Receptor population: Tribal fish and shellfish consumers
Receptor age: Child

Chemical	EPC (mg/kg ww)[a]	Cancer CDI (mg/kg-day)	Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk
Arsenic[b]	Table B.3-42	8.3×10^{-6}	1.5	1×10^{-5}
cPAH TEQ	Table B.3-42	9.0×10^{-7}	7.3	3×10^{-5}
1,4-Dichlorobenzene[c]	Table B.3-42	2.1×10^{-5}	0.0054	1×10^{-7}
Pentachlorophenol[c]	Table B.3-42	1.2×10^{-6}	0.12	1×10^{-7}
Total PCBs	Table B.3-42	5.0×10^{-5}	2	1×10^{-4}
PCB TEQ[d]	Table B.3-42	4.6×10^{-10}	150,000	7×10^{-5}
Total DDTs	Table B.3-42	3.8×10^{-7}	0.34	1×10^{-7}
alpha-BHC[c]	Table B.3-42	3.9×10^{-8}	6.3	2×10^{-7}
beta-BHC[c]	Table B.3-42	4.0×10^{-8}	1.8	7×10^{-8}
Dieldrin[c]	Table B.3-42	4.1×10^{-8}	16	6×10^{-7}
Total chlordane	Table B.3-42	3.6×10^{-7}	0.35	1×10^{-7}
Heptachlor[c]	Table B.3-42	1.7×10^{-8}	4.5	8×10^{-8}
Heptachlor epoxide[c]	Table B.3-42	1.8×10^{-8}	9.1	2×10^{-7}
Mirex[c]	Table B.3-42	1.8×10^{-8}	18	3×10^{-7}
Dioxin/furan TEQ[d]	Table B.3-42	6.3×10^{-11}	150,000	1×10^{-5}
Total TEQ excess cancer risk for dioxins/furans and coplanar PCBs				**8×10^{-5}**
Total excess cancer risk (excluding PCB TEQ)				**2×10^{-4}**
Total excess cancer risk (excluding total PCBs)				**1×10^{-4}**

a An EPC for each seafood category was calculated in the exposure section.
b Arsenic EPCs and risk estimates are based on inorganic arsenic.
c Greater than 50% of the risk associated with this chemical is derived from seafood categories with non-detected concentrations.
d No mussel data were available for this chemical. When the CDI and risk values were calculated, the portion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

BHC – benzene hexachloride
CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
ww – wet weight

Table B.6-14. Non-cancer hazard estimates for the child tribal 95th percentile seafood consumption scenario based on Tulalip data

Scenario timeframe: Current/future **Medium:** Sediment **Exposure medium:** Fish and shellfish tissue **Receptor population:** Tribal fish and shellfish consumers **Receptor age:** Child

Chemical	EPC (mg/kg ww)[a]	Non-Cancer CDI (mg/kg-day)	Reference Dose (mg/kg-day)	Hazard Quotient
Antimony	Table B.3-42	1.2×10^{-5}	0.0004	0.03
Arsenic[b]	Table B.3-42	9.6×10^{-5}	0.0003	0.3
Cadmium	Table B.3-42	5.2×10^{-4}	0.001	0.5
Chromium	Table B.3-42	4.0×10^{-4}	0.003	0.1
Cobalt	Table B.3-42	1.3×10^{-4}	0.0003	0.4
Copper	Table B.3-42	9.5×10^{-3}	0.04	0.2
Mercury	Table B.3-42	4.8×10^{-5}	0.0001	0.5
Molybdenum	Table B.3-42	4.8×10^{-4}	0.005	0.1
Selenium	Table B.3-42	6.5×10^{-4}	0.005	0.1
Vanadium	Table B.3-42	3.2×10^{-4}	0.009	0.04
Zinc	Table B.3-42	3.0×10^{-2}	0.3	0.1
Dibutyltin as ion	Table B.3-42	6.5×10^{-6}	0.0003	0.02
Tributyltin as ion	Table B.3-42	4.5×10^{-5}	0.00015	0.3
1,4-Dichlorobenzene	Table B.3-42	2.5×10^{-4}	0.07	0.004
Pentachlorophenol	Table B.3-42	1.5×10^{-5}	0.03	0.0005
Total PCBs	Table B.3-42	5.8×10^{-4}	0.00002	29[c]
			0.00007	8[d]
Total DDTs	Table B.3-42	4.4×10^{-6}	0.0005	0.009
alpha-BHC	Table B.3-42	4.5×10^{-7}	0.008	0.00006
Dieldrin	Table B.3-42	4.7×10^{-7}	0.00005	0.009
Total chlordane	Table B.3-42	4.2×10^{-6}	0.0005	0.008
Heptachlor	Table B.3-42	2.0×10^{-7}	0.0005	0.0004
Heptachlor epoxide	Table B.3-42	2.2×10^{-7}	0.000013	0.02
Mirex	Table B.3-42	2.1×10^{-7}	0.0002	0.001
Hazard indices by effect:				
Hazard index for hematological endpoint[e]				**0.2**
Hazard index for immunological endpoint[f]				**29**
Hazard index for kidney endpoint[g]				**0.6**
Hazard index for liver endpoint[h]				**0.05**
Hazard index for neurological endpoint[i]				**30**
Hazard index for endocrine endpoint[j]				**0.4**
Hazard index for integumentary endpoint[k]				**29**
Hazard index for digestive system endpoint[l]				**0.3**

Table B.6-14. Non-cancer hazard estimates for the child tribal 95th percentile seafood consumption scenario based on Tulalip data (cont.)

Chemical	EPC (mg/kg ww)[a]	Non-Cancer CDI (mg/kg-day)	Reference Dose (mg/kg-day)	Hazard Quotient
Hazard index for development endpoint[m]				**9**

[a] An EPC for each seafood category was calculated in the exposure section.

[b] Arsenic EPCs and risk estimates are based on inorganic arsenic.

[c] HQ used for the calculation of the immunological, integumentary, and neurological endpoint hazard indices (Table B.4-1).

[d] HQ used for the calculation of the developmental endpoint hazard index (Table B.4-1).

[e] Hematological endpoint includes the following chemicals: antimony, selenium, and zinc.

[f] Immunological endpoint includes the following chemicals: dibutyltin, total PCBs, and TBT.

[g] Kidney endpoint includes the following chemicals: cadmium, molybdenum, and pentachlorophenol.

[h] Liver endpoint includes the following chemicals: 1,4-dichlorobenzene, alpha-BHC, total chlordane, total DDTs, dieldrin, heptachlor, heptachlor epoxide, mirex, and pentachlorophenol.

[i] Neurological endpoint includes the following chemicals: mercury, total PCBs, and selenium. Neurological effects associated with exposure to lead are discussed in Section B.5.4.

[j] Endocrine endpoint includes the following chemicals: antimony and cobalt.

[k] Integumentary endpoint includes the following chemicals: arsenic, total PCBs, selenium, and vanadium.

[l] Digestive system endpoint includes the following chemicals: chromium and copper.

[m] Developmental endpoint includes the following chemicals: mercury and total PCBs.

BHC – benzene hexachloride
CDI – chronic daily intake
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
PCB – polychlorinated biphenyl
TBT – tributyltin
ww – wet weight

Child Tribal Scenario Based on Suquamish Data

Estimates of risks for the child tribal scenario based on Suquamish data were not provided in the risk characterization section of this document because of the high uncertainty associated with this scenario and EPA's assessment that the Tulalip Tribes' consumption rates were more appropriate for the EW than the Suquamish consumption rates (EPA 2005a). This approach was also consistent with that used in the LDW HHRA (Windward 2007c). The Suquamish Tribe survey (Suquamish Tribe 2000) included 31 children from 0 to 6 years old from 21 different households and provided a 95th percentile estimate of children's seafood consumption[52] equal to 122.2 g/day.[53] The survey report also provided data on the categories of seafood Suquamish children consume.

Child-specific rates appropriate for the apportionment of total seafood consumption to different seafood categories were based on information on categories of seafood consumed by Suquamish children, as presented in the consumption survey by the Suquamish Tribe (2000). Children's seafood consumption is potentially influenced by

[52] The Suquamish seafood consumption study included 31 children aged 0 to 6 years. However, the survey included responses from only 21 households. The 95th percentile rate was provided for consumers and non-consumers (combined), but all children were reported to consume seafood (Suquamish Tribe 2000).

[53] The total consumption rate was calculated by multiplying the 95th percentile consumption rate of 7.272 g/kg/day by the average children's body weight of 16.8 kg (Suquamish Tribe 2000).

the consumption patterns of adults living in the same household. Multiple children from the same household were selected for the Suquamish survey. Consequently, consumption data for child participants living in the same household are not independent. The effect of the lack of independence with respect to children's consumption rates on the overall calculated consumption rate is unclear.

Suquamish child total consumption data was apportioned into seafood categories using the same basic approach as that used above for the child tribal 95th percentile scenario based on Tulalip data (and described in detail in Section B.3.3.1.2) (Tables B.6-15 and B.6-16). Again, the health protective assumption was made that all seafood consumed by children was from the EW. The survey did not report the portion of children's seafood consumption from Puget Sound versus other sources. For adults, an average of 19% or more consumption for each of the major seafood categories (e.g., anadromous, shellfish) was reported as being from sources other than "caught in Puget Sound" (Suquamish Tribe 2000). The apportionment approach involved first dividing the total seafood consumption into broad categories and then dividing the shellfish portion into the specific types of shellfish consumed. No children's benthic fish consumption other than fillet (e.g., organs or whole fish) was reported (Suquamish Tribe 2000), so all benthic fish consumption was assumed to be fillet. Similarly, no children's consumption of crab other than edible meat was reported (Suquamish Tribe 2000), so all crab consumption was assumed to be edible meat. For chemicals with higher concentrations in crab hepatopancreas than in crab edible meat, risks for children could be higher if the crab hepatopancreas was found to be consumed by children. The apportionment of consumption for the scenario based on Suquamish children's consumption rates was very similar to that of the scenario based on Suquamish adult consumption rates (Section B.3.3.1.1).

Table B.6-15. Percentages and rates associated with different seafood categories for the child tribal seafood consumption scenario based on Suquamish data

Seafood Category	Percentage of Total Seafood Consumption[a]	Consumption Rate (g/day)[b]
Anadromous fish[c]	21.9	26.8[c]
Pelagic fish	10.9 (99% perch, 1% rockfish)	13.3 (13.2 g perch, 0.1 g rockfish)
Benthic fish	2.4	3.0[d]
Shellfish	64.8	79.1

[a] Calculated from average children's consumption rates by seafood category (Suquamish Tribe 2000).

[b] Calculated by multiplying the 95th percentile of child Suquamish total seafood consumption (122.2 g/day) by the percentage of consumption of the various seafood categories.

[c] The consumption of anadromous fish was not included in this HHRA.

[d] No children's consumption of benthic fish other than fillet was reported (Suquamish Tribe 2000), so all benthic fish consumption was assumed to be benthic fish fillet.

HHRA – human health risk assessment

Table B.6-16. Consumption of shellfish for the child tribal seafood consumption scenario based on Suquamish data

Shellfish Type	Percentage of Total Shellfish Consumption[a]	Consumption Rate (g/day)[b]
Crabs	40.3	31.9[c]
Clams[d]	43.3	34.5
Mussels	0.1	0.08
Geoduck	16 (95% edible meat, 5% whole body)	12.6 (12 g edible meat, 0.6 g whole body)

[a] Calculated from average children's consumption rates by seafood category (Suquamish Tribe 2000).

[b] Calculated by multiplying the consumption percentages by total shellfish consumption (79.1 g/day from Table B.6-15).

[c] Adults were asked about children's consumption of crab parts. No children's consumption of whole crab or crab butter (i.e., hepatopancreas) was reported (Suquamish Tribe 2000), so all crab consumption was assumed to be edible meat.

[d] Includes Manila/littleneck clams, horse clams, butter clams, cockles, oysters, and scallops.

The child tribal consumption rates based on Suquamish data were used with EPCs for the different seafood categories (Section B.3.3.4) to develop excess cancer risk and non-cancer hazard estimates. Consumption rates from Tables B.6-15 and B.6-16 were used with other exposure parameters (e.g., exposure duration) from Table B.3-17, with one exception. The reported average body weight for Suquamish children (16.8 kg) from the Suquamish survey (Suquamish Tribe 2000) was used.

Cancer risk and non-cancer hazard estimates for this scenario are presented in Tables B.6-17 and B.6-18, respectively. Excess cancer risk estimates were greater than 1×10^{-6}, with PCBs and cPAH TEQ contributing the most to the risk. HQs were greater than 1 for five individual chemicals and for most endpoints. Excess cancer risk estimates and non-cancer hazards were less than those for the adult tribal scenario based on Suquamish data (Tables B.5-5 and B.5-13), but they were higher than those for the child tribal 95th percentile scenario based on Tulalip data (Tables B.6-13 and B.6-14) and the child tribal RME scenario based on Tulalip data (using the 40% adult ratio [Tables B.5-3 and B.5-11]).

Table B.6-17. Excess cancer risk estimates for the child tribal seafood consumption scenario based on Suquamish data

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Receptor population: Tribal fish and shellfish consumers
Receptor age: Child

Chemical	EPC (mg/kg ww)[a]	Cancer CDI (mg/kg-day)	Cancer Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk
Arsenic[b]	Table B.3-42	4.9×10^{-5}	1.5	7×10^{-5}
cPAH TEQ	Table B.3-42	5.2×10^{-6}	7.3	2×10^{-4}
1,4-Dichlorobenzene[c]	Table B.3-42	1.0×10^{-4}	0.0054	6×10^{-7}
Pentachlorophenol[c]	Table B.3-42	1.2×10^{-5}	0.12	1×10^{-6}
Total PCBs	Table B.3-42	1.9×10^{-4}	2	4×10^{-4}
PCB TEQ[d]	Table B.3-42	1.6×10^{-9}	150,000	2×10^{-4}
Total DDTs	Table B.3-42	1.4×10^{-6}	0.34	5×10^{-7}
alpha-BHC[c]	Table B.3-42	2.6×10^{-7}	6.3	2×10^{-6}
beta-BHC[c]	Table B.3-42	2.8×10^{-7}	1.8	5×10^{-7}
Dieldrin[c]	Table B.3-42	2.3×10^{-7}	16	4×10^{-6}
Total chlordane	Table B.3-42	1.4×10^{-6}	0.35	5×10^{-7}
Heptachlor[c]	Table B.3-42	1.0×10^{-7}	4.5	5×10^{-7}
Heptachlor epoxide[c]	Table B.3-42	1.0×10^{-7}	9.1	1×10^{-6}
Mirex[c]	Table B.3-42	1.0×10^{-7}	18	2×10^{-6}
Dioxin/furan TEQ[d]	Table B.3-42	2.7×10^{-10}	150,000	4×10^{-5}
Total TEQ excess cancer risk for dioxins/furans and coplanar PCBs				**2×10^{-4}**
Total excess cancer risk (excluding PCB TEQ)				**7×10^{-4}**
Total excess cancer risk (excluding total PCBs)				**5×10^{-4}**

a An EPC for each seafood category was calculated in the exposure section.
b Arsenic EPCs and risk estimates are based on inorganic arsenic.
c Greater than 50% of the risk associated with this chemical is derived from seafood categories with non-detected concentrations.
d No mussel data were available for this chemical. When the CDI and risk values were calculated, the portion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

BHC – benzene hexachloride
CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
ww – wet weight

Table B.6-18. Non-cancer hazard estimates for the child tribal seafood consumption scenario based on Suquamish data

Scenario timeframe: Current/future
Medium: Sediment
Exposure medium: Fish and shellfish tissue
Receptor population: Tribal fish and shellfish consumers
Receptor age: Child

Chemical	EPC (mg/kg ww)[a]	Non-Cancer CDI (mg/kg-day)	Reference Dose (mg/kg-day)	Hazard Quotient
Antimony	Table B.3-42	7.0×10^{-5}	0.0004	0.2
Arsenic[b]	Table B.3-42	5.8×10^{-4}	0.0003	2
Cadmium	Table B.3-42	2.1×10^{-3}	0.001	2
Chromium	Table B.3-42	2.3×10^{-3}	0.003	0.8
Cobalt	Table B.3-42	7.8×10^{-4}	0.0003	3
Copper	Table B.3-42	5.6×10^{-2}	0.04	1
Mercury	Table B.3-42	2.6×10^{-4}	0.0001	3
Molybdenum	Table B.3-42	3.1×10^{-3}	0.005	0.6
Selenium	Table B.3-42	3.9×10^{-3}	0.005	0.8
Vanadium	Table B.3-42	1.8×10^{-3}	0.009	0.2
Zinc	Table B.3-42	1.8×10^{-1}	0.3	0.6
Dibutyltin as ion	Table B.3-42	3.3×10^{-5}	0.0003	0.1
Tributyltin as ion	Table B.3-42	2.1×10^{-4}	0.00015	1
1,4-Dichlorobenzene	Table B.3-42	1.2×10^{-3}	0.07	0.02
Pentachlorophenol	Table B.3-42	1.4×10^{-4}	0.03	0.005
Total PCBs	Table B.3-42	2.2×10^{-3}	0.00002	109[c]
			0.00007	31[d]
Total DDTs	Table B.3-42	1.6×10^{-5}	0.0005	0.03
alpha-BHC	Table B.3-42	3.0×10^{-6}	0.008	0.0004
Dieldrin	Table B.3-42	2.7×10^{-6}	0.00005	0.05
Total chlordane	Table B.3-42	1.6×10^{-5}	0.0005	0.03
Heptachlor	Table B.3-42	1.2×10^{-6}	0.0005	0.002
Heptachlor epoxide	Table B.3-42	1.2×10^{-6}	0.000013	0.09
Mirex	Table B.3-42	1.2×10^{-6}	0.0002	0.006
Hazard Indices by Endpoint:				
Hazard index for hematological endpoint[e]				**2**
Hazard index for immunological endpoint[f]				**110**
Hazard index for kidney endpoint[g]				**3**
Hazard index for liver endpoint[h]				**0.2**
Hazard index for neurological endpoint[i]				**113**
Hazard index for endocrine endpoint[j]				**3**
Hazard index for integumentary endpoint[k]				**112**
Hazard index for digestive system endpoint[l]				**2**
Hazard index for developmental endpoint[m]				**112**

Table B.6-18. Non-cancer hazard estimates for the child tribal seafood consumption scenario based on Suquamish data (cont.)

[a] An EPC for each seafood category was calculated in the exposure section.
[b] Arsenic EPCs and risk estimates are based on inorganic arsenic.
[c] HQ used for the calculation of the immunological, integumentary, and neurological endpoint hazard indices (Table B.4-1).
[d] HQ used for the calculation of the developmental endpoint hazard index (Table B.4-1).
[e] Hematological endpoint includes the following chemicals: antimony, selenium, and zinc.
[f] Immunological endpoint includes the following chemicals: dibutyltin, total PCBs, and TBT.
[g] Kidney endpoint includes the following chemicals: cadmium, molybdenum, and pentachlorophenol.
[h] Liver endpoint includes the following chemicals: 1,4-dichlorobenzene, alpha-BHC, total chlordane, total DDTs, dieldrin, heptachlor, heptachlor epoxide, mirex, and pentachlorophenol.
[i] Neurological endpoint includes the following chemicals: mercury, total PCBs, and selenium. Neurological effects associated with exposure to lead are discussed in Section B.5.4.
[j] Endocrine endpoint includes the following chemicals: antimony and cobalt.
[k] Integumentary endpoint includes the following chemicals: arsenic, total PCBs, selenium, and vanadium.
[l] Digestive system endpoint includes the following chemicals: chromium and copper.
[m] Developmental endpoint includes the following chemicals: mercury and total PCBs.

BHC – benzene hexachloride
CDI – chronic daily intake
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
PCB – polychlorinated biphenyl
TBT – tributyltin
ww – wet weight

B.6.1.3 Exposure duration for API seafood consumption scenario

Uncertainty regarding the ingestion rate for the API RME scenario was discussed above (Section B.6.1.2.2). There is also uncertainty surrounding the value selected for the exposure duration for the API RME scenario. EPA (1997a) has calculated a 90th percentile residence time in the same household of approximately 30 years for the general US population. However, the residence time of API in the vicinity of the EW may be different from that of the general population. The mobility of API individuals who may use the EW as a primary or exclusive fishing resource is unknown. There are two main sources of uncertainty regarding exposure duration of API individuals who consume seafood from the EW. First, it is possible that API residents remain in areas near the EW for longer than 30 years; and second, it is possible that even when they move away from the EW, they may return to the EW to maintain connection with their communities and to catch seafood in the waterway.

No studies that analyze residence time of API in neighborhoods bordering the EW have yet been completed, and there are no known studies of similar settings or populations that could be used as a surrogate for the API population. A modeling effort was conducted for the Hudson River HHRA (TAMS and Gradient 2000) using data on residence time and population mobility to examine residence time in the five counties directly adjacent to the Hudson River. The 90th percentile value for residence time in that analysis was 40 years. However, because of differences between the Hudson River and EW, including scale (i.e., the Hudson study included five counties and 40 miles of river; the EW includes a single county and approximately 1.5 miles of river), the 40-year exposure duration identified in the Hudson River HHRA may not be applicable to the

EW. However, this modeling effort indicates that exposure durations may differ from the EPA-recommended default value for some populations.

In the absence of site-specific information or appropriate surrogate information, EPA's *Exposure Factors Handbook* (1997a) reported (in Table 15-167) that the general US population has a 95th percentile residence time of 41 years. In order to investigate the effect of assumptions of longer exposure durations on the risk estimates for the API scenario, risks were calculated using an exposure duration of 41 years. This exposure duration is significantly longer than the current exposure duration of 30 years for the API RME scenario. A similar evaluation was performed in the HHRA uncertainty analysis for the LDW (Windward 2007c), which is much longer (~4 miles) than the EW. Table B.6-19 presents the results of the CDI and excess cancer risk calculations derived using this longer exposure duration, compared with the 30-year duration presented previously in the risk characterization.

Table B.6-19. Comparison of API RME excess cancer risks using 30-year and 41-year exposure durations

Chemical	Adult API RME with 30-Year Exposure Duration		Adult API RME with 41-Year Exposure Duration	
	Cancer CDI (mg/kg-day)	Excess Cancer Risk	Cancer CDI (mg/kg-day)	Excess Cancer Risk
Arsenic[a]	5.0×10^{-5}	8×10^{-5}	6.8×10^{-5}	1×10^{-4}
cPAH TEQ	7.3×10^{-6}	5×10^{-5}	1.0×10^{-5}	7×10^{-5}
1,4-Dichlorobenzene	6.5×10^{-5}	4×10^{-7}	8.9×10^{-5}	5×10^{-7}
Pentachlorophenol	2.3×10^{-6}	3×10^{-7}	3.1×10^{-6}	4×10^{-7}
Total PCBs	2.0×10^{-4}	4×10^{-4}	2.8×10^{-4}	6×10^{-4}
PCB TEQ	2.1×10^{-9}	3×10^{-4}	2.7×10^{-9}	4×10^{-4}
Total DDTs	1.6×10^{-6}	6×10^{-7}	2.2×10^{-6}	8×10^{-7}
Dieldrin	1.5×10^{-7}	2×10^{-6}	2.0×10^{-7}	3×10^{-6}
Total chlordane	1.9×10^{-6}	7×10^{-7}	2.6×10^{-6}	9×10^{-7}
Dioxin/furan TEQ	2.6×10^{-10}	4×10^{-5}	3.3×10^{-10}	5×10^{-5}
alpha-BHC	1.5×10^{-7}	9×10^{-7}	2.1×10^{-7}	1×10^{-6}
beta-BHC	1.5×10^{-7}	3×10^{-7}	2.0×10^{-7}	4×10^{-7}
Heptachlor	7.0×10^{-8}	3×10^{-7}	9.5×10^{-8}	4×10^{-7}
Heptachlor epoxide	8.1×10^{-8}	7×10^{-7}	1.1×10^{-7}	1×10^{-6}
Mirex	7.6×10^{-8}	1×10^{-6}	1.0×10^{-7}	2×10^{-6}
Total excess cancer risk (excluding PCB TEQ)		**6×10^{-4}**		**8×10^{-4}**
Total excess cancer risk (excluding total PCBs)		**5×10^{-4}**		**6×10^{-4}**

[a] Arsenic risk estimates are based on inorganic arsenic.

API – Asian and Pacific Islander
BHC – benzene hexachloride
CDI – chronic daily intake
cPAH – carcinogenic polycyclic aromatic hydrocarbon
DDT – dichlorodiphenyltrichloroethane
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent

As shown in Table B.6-19, increasing the exposure duration from 30 to 41 years increases the total excess cancer risk from 6×10^{-4} to 8×10^{-4} when the total risk across all pathways excluding PCB TEQ is summed. When the excess cancer risks are summed excluding total PCBs, the total risks associated with the longer exposure duration are increased from 5×10^{-4} for an exposure duration of 30 years to 6×10^{-4} for an exposure duration of 41 years. The increase in estimated risk is not exactly proportional to the 37% increase in the exposure duration because the excess cancer risk is reported to only one significant figure. Although the appropriate exposure duration for API users of the EW is unknown, these results provide an upper-bound risk estimate for members of the general population who may be less mobile or who move away but choose to continue to use the EW as their harvesting resource.

B.6.1.4 Fraction of dose obtained from the EW

The fractional intake (FI) obtained from the EW, which is equivalent to a site use factor, was set at 1 by default for all exposure pathways, as was done in the LDW HHRA (Windward 2007c, 2009c). This assumption may be appropriate for the netfishing scenario, which could occur primarily within the EW. However, for the seafood consumption scenario, there is considerably more uncertainty regarding the degree to which seafood consumers would use only the EW for the collection of fish and shellfish, and thus there is uncertainty in the selection of an appropriate FI value. For the clamming scenarios, it is also possible that intertidal areas outside the EW would be used.

B.6.1.4.1 Fraction of dose obtained from site for seafood consumption scenarios

There are a number of factors to consider in the selection of an FI for the seafood consumption scenario. As discussed in EPA's framework for tribal seafood consumption rates (EPA 2007b), all of the seafood consumed by the Tulalip or Suquamish tribal members is assumed to be from Puget Sound (i.e., FI = 1). Based on EPA guidance and for consistency with the LDW HHRA (Windward 2007c), an FI value of 1 was selected for all seafood consumption scenarios in this HHRA because site-specific data are insufficient to derive specific quantitative estimates of FI values that are applicable to the RME individuals within the tribal or API consumer groups. The applied FI of 1 likely overestimates current exposures associated with the EW for most individuals, specifically those individuals who consume a portion of their seafood intake from outside the EW or whose seafood intake is partly made up of species not found in the EW. However, it should be noted that a creel study conducted by King County found that some individuals reported fishing as frequently as every day from the Spokane Street Bridge (1999a). EPA required the use of a source fraction of 1 for the evaluation of RME scenarios. In addition, this approach accounts for the possibility that when not consuming seafood from the EW, individuals may be consuming seafood from other nearby contaminated sites (e.g., the Elliott Bay waterfront and the LDW).

Another important factor in selecting an FI value is the consideration of future resource quality. In support of the LDW HHRA (Windward 2007c), EPA stated that "…use of

Suquamish exposure parameters will not provide the best estimate of LDW tribal seafood consumption risks due to the degraded habitat in the LDW and questions whether the high Suquamish shellfish consumption rate could be sustained" (EPA 2005a). This rationale indicates that perhaps the FI for the Suquamish scenario would be less than 1. This same rationale can be applied to the EW because the EW is both a smaller site and has fewer intertidal areas to provide habitat for species such as clams. Although it is possible that resource availability and use could increase in the future following remediation, it is unknown whether this will affect the quantity of harvestable seafood and, in turn, make the FI of 1 more realistic. For the one-meal-per-month scenarios, an FI of 1 for the EW is generally appropriate because that consumption rate is likely to be sustainable wholly within the EW, and thus the FI was not evaluated for these scenarios.

Because the use of an FI of 1 may overestimate risks for many site users, and to offer different perspectives for risk management decisions, order-of-magnitude variations (i.e., 0.1 and 0.01, which correspond to 10% and 1% site use, respectively) of the default FI value of 1 were evaluated for the tribal and API seafood consumption scenarios. However, it should be noted that although the same FI was applied to the entire market basket, the fraction of seafood taken from the EW is likely to vary by species. Even at an FI of 0.01, the combined excess cancer risk estimates from all chemicals were greater than 1×10^{-6} for the three RME scenarios and the Suquamish scenario (Table B.6-20). At an FI of 0.01, no total excess cancer risks were greater than the upper end of EPA's identified range of acceptable risks (10^{-4}). In addition, it should be noted that some chemicals would no longer exceed acceptable risk levels if an FI of 0.1 or 0.01 was assumed. Overall, the potential impact of this uncertainty on risk estimates is high (i.e., risks could be overestimated).

Table B.6-20. Excess cancer risk estimates for seafood consumption scenarios using alternative assumptions for the fraction of dose obtained from the site

Seafood Consumption Scenario	Total Excess Cancer Risk[a]		
	Fractional Intake = 1	Fractional Intake = 0.1	Fractional Intake = 0.01
Adult tribal RME (Tulalip data)	1×10^{-3}	1×10^{-4}	1×10^{-5}
Adult tribal CT (Tulalip data)	7×10^{-5}	7×10^{-6}	7×10^{-7}
Child tribal RME (Tulalip data)	4×10^{-4}	4×10^{-5}	4×10^{-6}
Child tribal CT (Tulalip data)	4×10^{-5}	4×10^{-6}	4×10^{-7}
Adult tribal (Suquamish data)	1×10^{-2}	1×10^{-3}	1×10^{-4}
Adult API RME	6×10^{-4}	6×10^{-5}	6×10^{-6}
Adult API CT	1×10^{-5}	1×10^{-6}	1×10^{-7}

[a] Excess cancer risk estimates represent totals for all COPCs, excluding PCB TEQ.

API – Asian and Pacific Islander
COPC – chemical of potential concern
CT – central tendency
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent

B.6.1.4.2 Fraction of dose obtained from site for direct sediment exposure scenarios

As with the seafood consumption scenarios, alternatives to the default FI value of 1 (i.e., 0.5 and 0.1, which correspond to 50 and 10% site use, respectively) were evaluated for the netfishing, habitat restoration worker, and clamming scenarios (Table B.6-21). For the netfishing and clamming scenarios, at an FI of 0.5, the highest excess cancer risk estimate was 3×10^{-5}; and at an FI of 0.1, all excess cancer risk estimates for the clamming scenarios were less than 1×10^{-5}, and all excess cancer risk estimates for the netfishing and habitat restoration worker scenarios were below 1×10^{-6}. The risk estimates for netfishing, habitat restoration, and clamming, would be altered if the intertidal areas visited (for habitat restoration and clamming) or areas used (for netfishing) within the site were different from those assumed for this risk assessment. Overall, the potential impact of this uncertainty on risk estimates is high (i.e., risks could be overestimated). However, it is also important to consider that the EW is within the larger Elliott Bay and Duwamish River area, and that direct contact exposure with contaminants throughout this area is of concern.

Table B.6-21. Excess cancer risk estimates for direct sediment exposure scenarios using alternative assumptions for the fraction of dose obtained from the site

Sediment Exposure Scenario	Total Excess Cancer Risk[a]		
	Fractional Intake = 1	Fractional Intake = 0.5	Fractional Intake = 0.1
Netfishing RME	7×10^{-6}	4×10^{-6}	7×10^{-7}
Netfishing CT	1×10^{-6}	5×10^{-7}	1×10^{-7}
Habitat restoration worker	1×10^{-6}	5×10^{-7}	1×10^{-7}
Tribal clamming RME (120 days per year)	3×10^{-5}	2×10^{-5}	3×10^{-6}
Tribal clamming – 183 days per year	6×10^{-5}	3×10^{-5}	6×10^{-6}
Clamming – 7 days per year	1×10^{-6}	5×10^{-7}	1×10^{-7}

[a] Excess cancer risk estimates represent totals for all chemicals, excluding PCB TEQ.

CT – central tendency

PCB – polychlorinated biphenyl

RME – reasonable maximum exposure

TEQ – toxic equivalent

B.6.1.5 Dermal exposure

B.6.1.5.1 Chemicals lacking guidance on absorption factors

Dermal exposure to three metals identified as COPCs for the direct sediment exposure scenarios (antimony, cobalt, and vanadium) was not evaluated because these chemicals lacked dermal absorption factors. EPA guidance states that "for inorganics, the speciation of the compound is critical to the dermal absorption and there are too little data to extrapolate a reasonable default value" (EPA 2004b). Therefore, only incidental ingestion for these three metals was considered in the risk characterization (Section B.5.3.2). To investigate whether this approach may have resulted in a significant underestimation of risk from exposure to these metals, risk estimates were calculated for these metals using the Model Toxics Control Act (MTCA) default dermal absorption value of 0.01 for inorganics (Ecology 2001). None of the HQs exceeded 1 for any scenario when this value was applied to these three chemicals (none of which are carcinogenic).

In addition, risks were calculated for one of the metals assuming several different absorption factors to assess the impact of different assumptions for dermal absorption values on risk estimates. Vanadium was selected because of its high detection frequency (antimony was infrequently detected) and because of high maximum concentrations relative to RSLs (over 7-fold higher). In addition, vanadium has one of the lowest RfDs of metals that lack an absorption factor (when multiplied by the oral absorption adjustment as shown in Table B.3-36). For these reasons, the inclusion of dermal exposure to vanadium would be expected to have a more significant impact on sediment risk estimates than the inclusion of dermal exposure to antimony or cobalt. Only non-cancer hazards were considered because vanadium, as with other metals that lack absorption factors, has not been demonstrated to cause cancer.

Table B.6-22 presents hypothetical HQs for the sediment exposure scenarios assuming a range of possible dermal absorption factors for vanadium. Specifically, EPA guidance provides dermal absorption factors for only two metals (0.03 for arsenic and 0.001 for cadmium) (EPA 2004b). In addition, a value of 0.01 has been presented by Cal EPA (2005) and is the MTCA default dermal absorption value for inorganics (Ecology 2001). Exposure via incidental sediment ingestion is also included in Table B.6-22 so that total risks associated with direct sediment exposure could be assessed (see Sections B.3.1 and B.3.3.2 for details on incidental sediment ingestion risk estimates).

Assuming the highest proportion of dermal absorption recommended by EPA (2004b) for any metal (0.03 for arsenic), hypothetical HQs were still less than 1 for all sediment exposure scenarios, although the dermal absorption HQ was higher than the incidental ingestion HQ for all scenarios. At an assumed dermal absorption factor of 0.01, which was the dermal absorption factor recommended in EPA dermal guidance prior to the current guidance (EPA 2004b), the dermal absorption contribution to the HQ was greater than the contribution from incidental sediment ingestion for all scenarios except the netfishing CT scenario (Table B.6-22).

Thus, risk estimates for those chemicals that lack dermal absorption factors are somewhat uncertain because it is not possible to quantitatively identify the risk without a dermal absorption factor. However, the hypothetical dermal absorption factors assumed in Table B.6-22 provide boundaries for the range of possible risks associated with exposure to vanadium. This analysis indicates that this uncertainty (i.e., lack of dermal absorption factors for some metals) is unlikely to affect the overall conclusions about risks associated with dermal exposure to metals in sediment.

Table B.6-22. Hypothetical non-cancer hazard estimates for vanadium using three dermal absorption factors for the direct sediment exposure scenarios

Dermal Absorption Factor[a]	CDI (mg/kg-day)		Hazard Estimate		
	Incidental Ingestion	Dermal Absorption	Incidental Ingestion	Dermal Absorption[b]	Total
Netfishing RME					
0.03	1.2×10^{-5}	5.1×10^{-6}	0.001	0.02	0.02
0.01	1.2×10^{-5}	1.7×10^{-6}	0.001	0.007	0.008
0.001	1.2×10^{-5}	1.7×10^{-7}	0.001	0.0007	0.002
Netfishing CT					
0.03	6.0×10^{-6}	2.6×10^{-7}	0.0007	0.001	0.002
0.01	6.0×10^{-6}	8.7×10^{-8}	0.0007	0.0004	0.001
0.001	6.0×10^{-6}	8.7×10^{-9}	0.0007	0.00004	0.0007
Habitat Restoration Worker					
0.03	2.9×10^{-6}	1.1×10^{-6}	0.0003	0.005	0.005
0.01	2.9×10^{-6}	3.5×10^{-7}	0.0003	0.002	0.002
0.001	2.9×10^{-6}	3.5×10^{-8}	0.0003	0.0002	0.0005
Clamming – 7 days per year					
0.03	1.3×10^{-6}	4.6×10^{-7}	0.0001	0.002	0.002
0.01	1.3×10^{-6}	1.5×10^{-7}	0.0001	0.0007	0.0008
0.001	1.3×10^{-6}	1.5×10^{-8}	0.0001	0.00007	0.0002
Tribal Clamming RME (120 days per year)					
0.03	2.0×10^{-5}	7.4×10^{-6}	0.002	0.03	0.03
0.01	2.0×10^{-5}	2.5×10^{-6}	0.002	0.01	0.01
0.001	2.0×10^{-5}	2.5×10^{-7}	0.002	0.001	0.003
Tribal Clamming – 183 days per year					
0.03	3.1×10^{-5}	1.1×10^{-5}	0.003	0.05	0.05
0.01	3.1×10^{-5}	3.8×10^{-6}	0.003	0.02	0.02
0.001	3.1×10^{-5}	3.8×10^{-7}	0.003	0.002	0.005

Note: The incidental sediment ingestion estimates were presented in the risk characterization (Section B.5) and are included here for completeness.

[a] EPA guidance provides a dermal absorption factor for only two metals (0.03 for arsenic and 0.001 for cadmium). The hypothetical dermal absorption factors assumed in this table (0.03, 0.01, and 0.001) were selected to represent a range of possible values.

[b] Oral adjustment factor for vanadium = 0.026 (EPA 2004a). Dermal HQ = CDI/(RfD × 0.026) (EPA 2004b).

CDI – chronic daily intake
CT – central tendency
EPA – US Environmental Protection Agency
HQ – hazard quotient
RME – reasonable maximum exposure

B.6.1.5.2 Dermal adherence factors used for sediment exposure scenarios

Dermal adherence factors are used to estimate the amount of sediment that adheres to exposed skin in the assessment of risks posed by dermal exposure to sediment. EPA

(2004b) recommends a dermal adherence value of 0.2 mg/cm^2 as a default health-protective factor for the exposure of children and adults to moist soil. There are three main sources of uncertainty surrounding the use of this parameter in risk assessment scenarios that involve marine sediments. The first source of uncertainty is related to the limited amount of data on dermal adherence values for sediment from field studies used as the basis for the EPA recommendation. Nearly all the studies used by EPA in determining dermal adherence factors focused on exposure to terrestrial soil. Direct sediment exposure data were derived from only two studies: an investigation of sediment adherence for adults gathering reeds in marine sediment and an inland study of children playing in mud along the shoreline of a lake (EPA 2004b).

The second main source of uncertainty regarding the dermal adherence factor relates to the differences in the particulate make-up of soil and sediment. Marine sediment generally has a higher sand fraction than do freshwater sediments and may potentially have a greater percentage of larger particles, which are less prone to dermal adherence than are small particles. However, the higher moisture content in sediment, the third source of uncertainty, will likely increase the adherence of particles of all sizes. Also important in the discussion of particle size and skin adherence is the concept of mono-layer loading of the skin surface. As sediment loading of the skin surface increases, the fraction of chemical that is available to be absorbed remains constant until all of the skin is covered by a thin layer of sediment (known as the mono-layer) (Duff and Kissel 1996). Once the monolayer threshold is crossed, the fraction of chemical that can be absorbed will decrease because not all of the sediment is in constant, direct contact with skin. Both the amount of sediment required to form the mono-layer and the associated adherence ability of the soil depend directly on the size of the sediment particles and the moisture content of the sediment. In general, larger, drier particles will have a lower adherence factor than smaller, more moist particles.

Since the publication of the EPA guidance for dermal risk assessment (2004b), additional studies that focus specifically on dermal adherence of marine sediment during clamming activities (for adults) have been conducted (Shoaf et al. 2005a, b). Both of these newer studies included unscripted clamming activities and identified post-exposure dermal sediment loadings. Surface-area-weighted sediment dermal adherence factors were calculated from the body-part-specific sediment loadings presented in these studies. For adults digging in a clam flat, the sediment dermal adherence factor was 0.3 mg/cm^2, quite similar to EPA's recommended value of 0.2 mg/cm^2. This value was also similar to that presented in the study that investigated individuals gathering reeds in marine sediment in Washington State (Kissel et al. 1996). Table B.6-23 summarizes the effect on total risk estimates when the adult exposure dermal adherence factor of 0.3 mg/cm^2 is used instead of the value of 0.2 mg/cm^2. There are only slight changes in the excess cancer risk estimates and non-cancer hazards, and therefore it can be concluded that the effect of higher skin adherence on the overall risk estimates for netfishing and clamming is not significant.

Table B.6-23. Effect of increased dermal adherence factors on risk estimates for sediment exposure scenarios

Dermal Adherence Factor	Summation Approach	Hypothetical Risk by Exposure Scenario					
		Netfishing			Clamming		
		RME	CT[a]	Habitat Restoration Worker	Tribal – 183 Days per Year	Tribal RME	7 Days per Year
Total Excess Cancer Risk							
0.2	total risk excluding PCB TEQ	7×10^{-6}	2×10^{-6}	1×10^{-6}	6×10^{-5}	3×10^{-5}	1×10^{-6}
0.3	total risk excluding PCB TEQ	8×10^{-6}	2×10^{-6}	2×10^{-6}	6×10^{-5}	4×10^{-5}	1×10^{-6}
Total Non-Cancer Hazard[b]							
0.2	na	0.06	0.02	0.01	0.2	0.1	0.01
0.3	na	0.07	0.02	0.02	0.3	0.2	0.01

a To characterize risks for the netfishing CT scenario, a dermal adherence factor of 0.02 was used in Section B.5 (specified as the default value for a CT industrial worker in EPA (2004b)). However, for consistency, the values of 0.2 and 0.3 were used in this table.

b Estimates for total non-cancer hazards are provided as HQs. Non-cancer hazards do not include estimates of dermal risk from metals other than arsenic and cadmium because of the lack of dermal absorption factors for all other metals (see Section B.6.1.5).

CT – central tendency
EPA – US Environmental Protection Agency
HQ – hazard quotient
na – not applicable
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent

B.6.1.5.3 Cumulative effects on risk estimates of alternative dermal absorption and alternative dermal adherence factors used for sediment exposure scenarios

Risks were estimated using alternative dermal absorption factors and high-end dermal adherence factors to assess the potential cumulative effects of the uncertainties associated with these parameters. Excess cancer risks and non-cancer HQs were calculated using the default MTCA dermal absorption factor of 0.01 for inorganic chemicals that lack such values. A high-end dermal adherence value of 0.3 was used for the clamming and netfishing scenarios (increased from 0.2 for the netfishing RME, habitat restoration worker, and clamming scenarios and from 0.02 for the clamming CT scenario). These are the same values that were used in Section B.6.1.5.2.

Overall, changes to the risk conclusions presented in Section B.5 were minor. As was determined in Section B.5, no netfishing, habitat restoration, or clamming scenario HQs exceeded 1. With use of the higher dermal adherence factor, no new chemical-scenario combinations had an excess cancer risk estimates greater than the 1×10^{-6} threshold.

B.6.1.6 Representativeness of data

B.6.1.6.1 Representativeness of fish and shellfish COPC data for all potentially exposed populations

Tissue data from over 100 composite tissue samples of fish, crabs, clams, mussels, and geoduck, representing 12 different species were available (see Table B.2-3 for details). As discussed in the Section B.3.3.1, 10 seafood categories were assumed to reasonably characterize the consumption habits of the several diverse groups of consumers. In the surveys used to develop the consumption rates for this risk assessment, the consumption of several dozen different seafood species was reported. These were assigned to a handful of seafood categories based on the initial consumption studies. Ten consumption categories were used in this HHRA, with tissue data assumed to be representative of each category. For example, what was reported in a survey as benthic fish consumption might have included English sole and flounder. In this HHRA, only English sole data were available to estimate risks from this category. This uncertainty may have led to either the overestimation or underestimation of risk.

The tissue samples used in this HHRA were uncooked portions of the total organism (e.g., whole body and fillets for benthic fish, hepatopancreas and muscle meat for crab). These portions represent the consumption habits of many, but not all, of the potentially exposed populations. For example, most people cook fish or shellfish before eating them. Data from uncooked or raw tissue samples were used in this HHRA because most chemistry data were collected for this type of sample. There is no standard cooking preparation that is used for environmental investigations. The King County WQA (King County 1999a) included an analysis of two LDW composite samples of cooked crab tissue and two LDW composite samples of uncooked crab tissue. Mean concentrations of arsenic and PCBs, which are two COCs identified in the risk characterization section, were 9.95 mg/kg and 156 µg/kg, respectively, in the uncooked samples of crab and 4.84 mg/kg and 89.5 µg/kg, respectively, in the cooked samples of crab. In the same assessment, risk estimates for PCBs were approximately double for cooked sole compared with those of uncooked sole, and risk estimates for PCBs were approximately one-half for cooked crab compared with those for uncooked crab. For arsenic, risks associated with cooked sole were only slightly higher than those for raw sole and were three times lower for cooked crab compared with raw crab (King County 1999a). Preparation and cooking practices that remove fats (e.g., filleting the fish) will actually increase the per gram mercury concentration while decreasing the concentration of lipophilic bioaccumulative chemicals such as PCBs (EPA 2000c). This occurs because mercury is concentrated in muscle tissue; lipophilic bioaccumulative chemicals are concentrated in fatty tissue. Thus, risk estimates may increase or decrease when cooking is considered. Because there are no standard cooking practices, the assumption that risks would be uniformly reduced by cooking is inappropriate. For example, the preparation of soups or stews from seafood would not likely reduce chemical concentrations to the same degree as broiling, during which fats drip away. Given the uncertainties associated with both the change in chemical concentration associated with different cooking practices and the cooking practices employed by different groups, uncooked tissue samples were used for risk assessment purposes.

In addition to uncertainties related to cooking, there are also uncertainties related to other preparation methods. Many individuals depurate clams (i.e., hold clams alive in water to remove the sediment in the clam digestive system) prior to consumption. The clams used for the development of clam EPCs were not depurated. This is an uncertainty and may lead to an overestimation or underestimation of risk, depending on whether chemical concentrations in clam gut contents are higher or lower than chemical concentrations in clam tissue and on how the clams are actually prepared prior to consumption (i.e., depurated or not). However, an analysis done as part of the LDW RI (Windward 2010g) found that the depuration of clams did not consistently increase or decrease arsenic (total and inorganic) or PCB concentrations in clam tissue.

B.6.1.6.2 Spatial coverage of sediment chemistry data

As described in Section B.2.1.1, the sediment chemistry database is thought to be representative of both site-related contamination and human use patterns. Sampling was designed to provide good spatial coverage of the EW. For the subtidal sediment dataset, numerous grab sediment samples collected throughout the waterway were analyzed for most chemicals, except for PCB congeners and dioxins/furans, which were analyzed for in 13 composite subtidal sediment samples rather than in individual grab samples. The PCB congener and dioxin/furan composite samples were each made up of numerous individual grab samples distributed throughout the waterway to ensure good spatial coverage. For intertidal sediment, composite samples were used for all chemicals to ensure good overall spatial coverage. The intertidal sediment was collected using the MIS technique; three composite samples represented site-wide intertidal concentrations, and one sample represented only the public-access portion of the intertidal sediment.

Although the spatial sampling density and overall spatial coverage of EW sediment were good, arithmetic means of site-wide concentrations were compared with spatially weighted averages in Section B.6.1.1.8 to further evaluate the spatial coverage of the sediment data. This comparison showed that concentrations based on arithmetic means were only slightly higher than those based on SWACs, indicating that these two calculation methods produced similar average concentrations.

B.6.1.6.3 Exclusion of King County WQA surface water dataset

As discussed in the EW HHRA technical memorandum (Windward 2010f) and a September 2010 memorandum to EPA (Windward 2010i), there were two potential surface water chemistry datasets for use in the evaluation of the swimming scenario for the EW HHRA. These included the dataset collected specifically for the SRI/FS (Anchor and Windward 2008) and the King County WQA dataset (King County 1999a).

This HHRA used only the EW SRI/FS dataset (Windward 2010i), which represents the most recent data and was collected from locations distributed throughout the EW.

These data were collected from September through February to capture different seasons (28 samples) and are therefore considered representative of potential exposure conditions in the EW. The SRI/FS water data were collected from five locations during five sampling periods in 2008-2009 (Map B.6-1). Surface water sampling for the SRI/FS was designed to represent a variety of environmental conditions (i.e., slips and channel, seasons, depths, and flow rates) in the EW. While the King County WQA dataset provided a significant number of samples (n = 102), spanning nine months, it did not include the analysis of PCBs, and SVOCs were rarely detected. Furthermore, the King County WQA dataset focused on wet season conditions in one area (sampling was conducted along a transect across the waterway from October to June) and did not include sampling in either Slip 27 or Slip 36 or the area upstream of the bridges at the head of the waterway (Map B.6-1).

Both the EW SRI/FS and King County WQA datasets had relatively high detection frequencies for metals. However, significant advances in the development of high sensitivity analyses for water samples had been made in the time period between 1999 and 2008, which resulted in significant differences between the two datasets with respect to the detection limits for many SVOCs and PCBs. In the King County WQA dataset, SVOCs were rarely detected (BEHP and benzoic acid were the most commonly detected organic chemicals). Organic chemicals were detected more frequently in the EW SRI/FS dataset because of the higher sensitivity of the analytical methods available. Table B.6-24 provides a comparison of concentrations and EPCs for detected surface water COPCs using the EW SRI/FS dataset alone (as done in this HHRA) and using the both the SRI/FS dataset and the KC WQA dataset combined.

As presented in Table B.6-24, mean and maximum concentrations in the combined dataset (King County WQA and SRI/FS data) were equal to or lower than those in the EW SRI/FS dataset, except in the case of lead, for which the maximum detection was highest in the combined dataset because of a single high detected concentration in a King County WQA sample. Based on the outlier test available through ProUCL, this high detected concentration of 0.00804 mg/L is a statistical outlier. The next highest detected concentration was 0.00239 mg/L.

The two datasets were also compared to determine which had the highest surface water concentration (detected concentration or RL, whichever was higher) for each analyte. EW SRI/FS maximum detected surface water concentrations or RLs were highest for 86 out of 102 analytes. For the other 15 chemicals, only lead had a detected concentration that was higher than that in the EW SRI/FS dataset. The other 14 chemicals were highest in the King County WQA dataset based on high RLs for non-detected samples. It should be noted that the mean lead concentration in the EW SRI/FS dataset was higher than the mean lead concentration in the King County WQA dataset (1 µg/L vs. 0.362 µg/L, respectively [Table B.6-24]).

Table B.6-24. Comparison of COPC concentrations in surface water from the two available datasets

COPC	Dataset[a]	Detection Frequency (ratio)	Concentration (mg/L)			
			Mean Value	Maximum Detection	Maximum RL	EPC
Arsenic (total)	EW SRI/FS only	28/28	0.0011	0.0016 J	na	0.0012
	EW SRI/FS and KC WQA	112/112	0.001	0.00161 J	na	0.0011
Chromium (total)	EW SRI/FS only	19/28	0.00091	0.0036 J	0.0024 U	0.0011
	EW SRI/FS and KC WQA	97/106	0.00055	0.00361 J	0.00236 U	0.0006
Lead (total)[b]	EW SRI/FS only	19/28	0.0010	0.00239	0.0068 U	0.0034
	EW SRI/FS and KC WQA	106/115	0.00053	0.00804 J	0.0068 U	0.00054
Naphthalene	EW SRI/FS only	8/28	0.00045	0.012	0.000042 U	0.0049
	EW SRI/FS and KC WQA	8/48	0.00034	0.012	0.00038 U	0.0029
PCB TEQ	EW SRI/FS only	28/28	5.80×10^{-10}	6.9×10^{-10} J	na	6.1×10^{-10}
	EW SRI/FS and KC WQA	ne	ne	ne	ne	ne
Total PCBs[c]	EW SRI/FS only	28/28	1.20×10^{-6}	5.8×10^{-6} J	na	1.5×10^{-6}
	EW SRI/FS and KC WQA	ne	ne	ne	ne	ne
Vanadium (total)	EW SRI/FS only	27/28	0.0022	0.0093	0.000080 UJ	0.0039
	EW SRI/FS and KC WQA	93/94	0.0015	0.00929	0.00008 UJ	0.0018

a Only those samples collected from 1 m below the water surface were included in each dataset.

b Lead was not a COPC based on the EW SRI/FS dataset alone. However, lead was included in this comparison table because the maximum detected value for the KC WQA dataset (and thus for the combined datasets) was greater than the RSL.

c Total PCBs were evaluated as total of PCB congeners. PCB congener data were not available from the KC WQA dataset (King County 1999a).

COPC – chemical of potential concern
EW – East Waterway
FS – feasibility study
J – estimated concentration
KC – King County
na – not applicable (no data available)
nd – not detected
ne – not evaluated (no data available for KC WQA dataset)
PCB – polychlorinated biphenyl
SRI – supplemental remedial investigation
TEQ – toxic equivalent
WQA – water quality assessment

This analysis indicates that the uncertainty associated with the exclusion of the King County WQA dataset from the risk assessment is relatively low. In addition, risks based on the EW SRI/FS dataset alone would be higher than those based on the combined dataset and thus would be more health protective.

B.6.1.6.4 Calculation of clam EPCs

As discussed in Section B.3.3.4.1, the tissue samples from the four species of clams collected from the EW were used together to calculate the clam EPCs. These species were butter clams, cockles, eastern soft-shell clams, and native littleneck clams. The approach used to calculate clam EPCs (i.e., the inclusion of all samples in one EPC without any weighting by species) is advantageous in that it does not make any assumptions regarding the assemblage of clams available from the EW, differences in contaminant concentrations across species, or the eating preferences of a particular consumer group.

However, it is useful to evaluate alternate approaches to determine the level of uncertainty surrounding the clam EPC. As discussed in EPA's tribal seafood consumption framework (2007b), several approaches may be used when information regarding preferred species for consumption is not available:

- 95% UCL across all species (as was used to calculate risks in Section B.5)
- Highest concentration for an individual species
- Biomass-weighted approach in which weighting factors are developed based on the relative biomass of the different species. This approach assumes that clams are consumed at a rate proportional to their abundance.

Table B.6-25 presents a comparison of these approaches.

Table B.6-25. Comparison of potential clam EPCs

COPC	Average Concentration in Clams [mg/kg ww] (number of composite samples)				Clam EPC (mg/kg ww)	
	Butter Clam	Cockle	Eastern Soft-Shell Clam	Native Littleneck Clam	Overall EPC (as used in Section B.5)	Biomass-Weighted EPC
Arsenic[a]	0.11 (7)	0.22 (2)	0.44 (1)	0.16 (2)	0.22	0.19
cPAH TEQ	0.011 (7)	0.015 (2)	0.0024 (1)	0.063 (1)	0.027	0.019
Total PCBs	0.056 (7)	0.074 (2)	0.0047 (1)	0.072 (1)	0.069	0.066
PCB TEQ[b]	5.0×10^{-7} (2)	2.1×10^{-7} (1)	nd	nd	7.3×10^{-7}	6.3×10^{-7}
Dioxin/furan TEQ[b]	3.1×10^{-7} (2)	2.3×10^{-7} (1)	nd	nd	3.8×10^{-7}	3.5×10^{-7}

[a] Arsenic EPCs and are based on inorganic arsenic.

[b] Only butter clam and cockle data were available for these chemicals. For the biomass-weighted approach, the apportionment was adjusted assuming only these two clam species: butter clam (79%) and cockle (21%).

COPC – chemical of potential concern
cPAH – carcinogenic polycyclic aromatic hydrocarbon
nd – no data
EPC – exposure point concentration
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
ww – wet weight

As can be seen in Table B.6-25, there was no clear pattern regarding which species had the highest concentrations; arsenic was highest in eastern soft-shell clams, cPAH TEQ was highest in native littleneck clams, total PCBs were highest in native littleneck clams and cockles, and PCB TEQ and dioxin/furan TEQ were highest in butter clams (although no data were available for eastern soft-shell and native littleneck for these TEQs). In addition, as indicated in the table, there were only 1 or 2 composite samples available for the majority of the species-COPC combinations, and thus this approach does not allow for the calculation of a 95% UCL for most species-COPC combinations.

The biomass-weighted clam EPCs presented in Table B.6-25 were calculated by developing a weighting scheme based on the proportion of the total biomass sampled represented by each clam species. Butter clams comprised 69% of the total clam biomass, littleneck clams comprised 4% of the total clam biomass, cockles comprised 18% of the total clam biomass, and soft-shell clams comprised 9% of the total clam biomass (Windward 2008d). This approach is described further in Attachment 6.

Table B.6-25 provides a comparison of the clam EPCs used in the risk characterization section of this HHRA and those calculated using the weighted biomass approach described here for selected COPCs. As can be seen in Table B.6-25, there is little difference in the EPCs calculated using the different approaches, and thus the uncertainty related to the clam EPCs is low.

B.6.1.6.5 Temporal variability in the tissue chemistry dataset

Data included in the tissue chemistry dataset for the EW included samples collected during 6 years (1995, 1996, 1997, 1998, 2005, and 2008). The majority of the data (114 of 145 samples) were collected in 2008. Non-2008 data were available for 6 of the 10 seafood consumption categories, including benthic fish fillet (9 samples), benthic fish whole body (2 samples), crab edible meat (3 samples), mussels (6 samples), and both types of pelagic fish (perch and rockfish [9 and 2 samples, respectively]). Table B.6-26 provides a summary of the sample counts available for the chemicals identified as the main contributors to the seafood consumption scenario risks. Note that only 2008 data were available for inorganic arsenic, dioxin/furan TEQ, and PCB TEQ.

Table B.6-26. Number of samples by year for select COPCs

Consumption Category	Number of Samples by Year										
	Arsenic (inorganic)	cPAH TEQ			Dioxin/ Furan and PCB TEQ	Total PCBs					
	2008	1996	1997	2008	2008	1995	1996	1997	1998	2005	2008
Benthic fish, fillet	11	0	0	11	3	3	0	0	0	6	11
Benthic fish, whole body	11	0	0	11	3	0	0	0	0	2	11
Clam	12	0	0	11	3	0	0	0	0	0	11
Crab, edible meat	9	0	0	9	3	0	0	0	3	0	9
Crab, whole body	9	0	0	9	3	0	0	0	0	0	9
Geoduck clam, edible meat	6	0	0	6	3	0	0	0	0	0	6
Geoduck clam, whole body	4	0	0	4	1	0	0	0	0	0	4
Mussels	11	3	3	11	0	0	3	3	0	0	11
Pelagic fish, perch	8	0	0	8	3	0	0	0	6	3	8
Pelagic fish, rockfish	13	0	0	13	6	0	0	0	0	2	13

COPC – chemical of potential concern
cPAH – carcinogenic polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

As presented in Table B.6-26, nearly all of the data for the top contributors to risks from the consumption of seafood are from 2008. The total PCB dataset has the largest proportion of pre-2008 samples, and thus the impact of historical data on the total PCB EPCs was evaluated. Table B.6-27 presents a comparison of EPCs calculated using all data (as used in Section B.5) and EPCs calculated using only the 2008 data.

Table B.6-27. Comparison of total PCB EPCs based on all tissue data and only tissue data from 2008

Consumption Category	All Data				2008 Data Only			
	No. of Samples	Concentration (mg/kg ww)			No. of Samples	Concentration (mg/kg ww)		
		Range	Mean	EPC		Range	Mean	EPC
Benthic fish, fillet	20	0.409 – 5.7	1.7	2.4	11	0.53 – 2	1.1	1.3
Benthic fish, whole body	13	1.46 – 7.9 J	3.2	4.1	11	1.46 – 5	2.7	3.3
Clam	11	0.0047 JN – 0.082	0.056	0.069	all samples are from 2008 (no change)			
Crab, edible meat	12	0.048 J – 0.21 J	0.13	0.16	9	0.048 J – 0.21 J	0.11	0.14
Crab, whole body	9	0.18 J – 0.86	0.3	0.45	all samples are from 2008 (no change)			
Geoduck clam, edible meat	6	0.014 – 0.024 JN	0.019	0.022	all samples are from 2008 (no change)			
Geoduck clam, whole body	4	0.025 J – 0.034 JN	0.028	0.034	all samples are from 2008 (no change)			
Mussels	17	0.013 U – 0.044	0.026	0.031	11	0.019 JN – 0.044 J	0.030	0.033
Pelagic fish, perch	17	0.104 – 5.4	1	1.6	8	0.38 JN – 1.24	0.95	1.1
Pelagic fish, rockfish	15	0.4 J – 6.2	2	4.0	13	0.4 J – 4.3	1.6	3.2

EPC – exposure point concentration
J – estimated concentration
N – tentative identification
PCB – polychlorinated biphenyl
U – not detected at given concentration
ww – wet weight

As shown in Table B.6-27, no new EPCs were calculated for the four consumption categories for which no pre-2008 data were available for inclusion in the tissue dataset (i.e., clams, whole-body crab, geoduck edible meat, and whole-body geoduck). EPCs based on only 2008 data for the remaining six consumption categories were lower than those calculated using all data, except for mussels, for which the 2008-only EPC was slightly higher than the EPC calculated using all data. EPCs calculated using only the 2008 data (as compared with those calculated using the full dataset) were lower because the 2005 data had higher PCB concentrations for benthic fish (both fillet and whole body) and pelagic fish (both perch and rockfish) (Figure B.6-1).

Although risks for current conditions may be slightly overestimated through the inclusion of the higher concentrations in some samples from the 2005 dataset, risk conclusions for the seafood consumption scenario would be unlikely to change if the 2008 data alone were used.

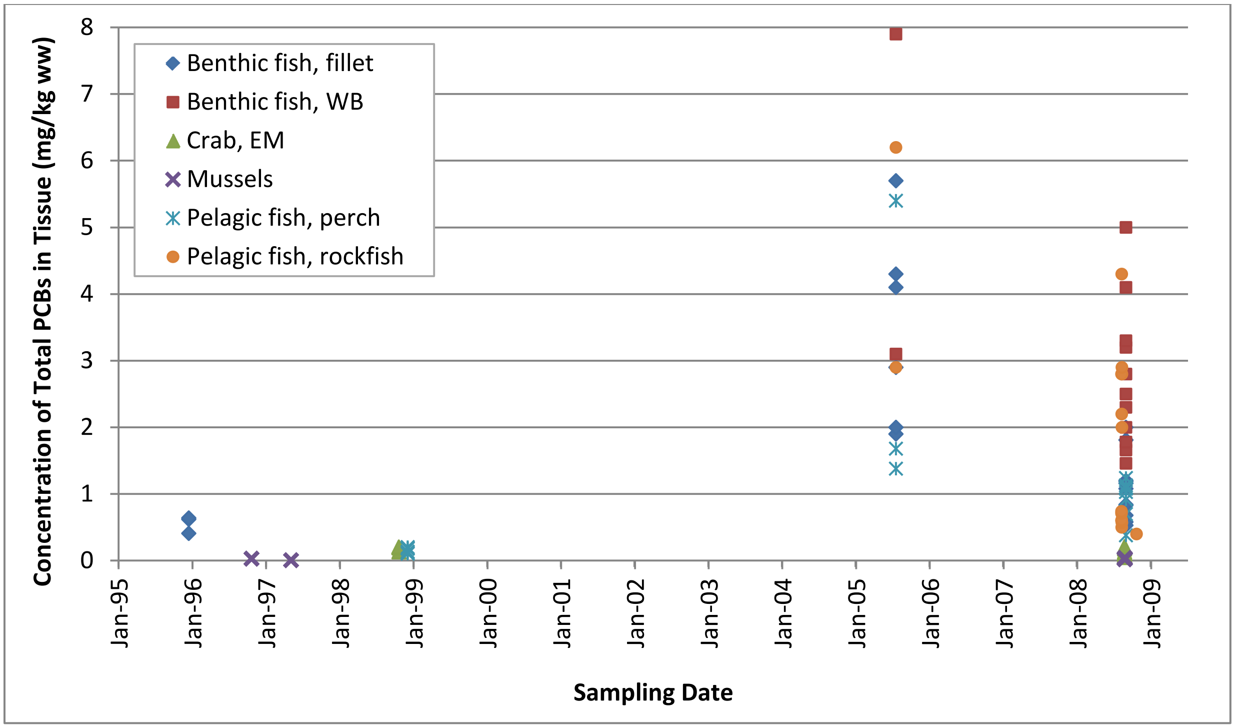

Figure B.6-1. Temporal variability in tissue concentrations of total PCBs

B.6.1.7 Health-protectiveness of sediment and surface water exposure scenarios

The sediment exposure scenarios summarized in Section B.3.1 and used for risk characterization in this HHRA were selected because they represent activities that may commonly occur in the EW or could commonly occur in the future. They were also selected to represent activities that result in a relatively higher amount of exposure than do other activities. Other activities that might occur in the EW and could result in contact with sediment and surface water, such as kayaking, fishing/crabbing, and occupational exposure associated with specific industrial or commercial facilities, are not explicitly discussed in this HHRA. The risks from these other activities are expected to be lower than those for the scenarios that were quantified. These other activities are described briefly below. Note that dog-walking, which was discussed in the LDW HHRA (Windward 2007c), is not expected to occur with any regularity in the EW because of the relatively small beach or intertidal areas that are accessible to the public.

B.6.1.7.1 Kayaking

Although recreational boating is not common along the EW, individuals may kayak along the waterway and occasionally pull boats out along the shore, particularly at public access areas. Exposure to surface water may occur during kayaking, as may exposure to intertidal surface sediment if stops are made at intertidal areas along the EW. The level of exposure from kayaking-related activities is expected to be significantly lower than that for other scenarios quantified in this HHRA. Surface water exposure from swimming would be much higher, as would sediment exposure during clamming.

B.6.1.7.2 Fishing or crabbing

Individuals may come into contact with EW surface water while fishing (e.g., from the Spokane Street Bridge) or contact sediment while retrieving crab traps. The level of exposure from these activities would be low relative to that for scenarios quantified in this HHRA. Exposure to surface water from swimming (Section B.5.3.3) would be much higher than the exposure that would occur from fishing or crab trap retrieval. Similarly, exposure to sediment from netfishing, habitat restoration work, or clamming (Section B.5.3.2) would be higher than the exposure that would occur during the retrieval of crab traps.

B.6.1.7.3 Occupational exposure

Daily work occurs at the terminals and industrial facilities along the EW, and thus workers are present at these terminals every day. However, exposure to EW contaminants is not expected to occur on a daily basis because the work occurs on the piers along the waterway (e.g., container offloading). However, there could be occasional occupational exposure to sediment or water at specific facilities if a worker were to fall into the water or do maintenance work that involved walking on the sediment. This type of exposure is expected to be very low compared with the occupational exposure of netfishing (for sediment) or swimming (for surface water) that was quantitatively evaluated in the HHRA.

B.6.1.8 Basis for fish and shellfish tissue screening levels

As discussed in Section B.3.2.2, COPCs for fish and shellfish tissue were identified by comparing concentrations in EW tissue with RSLs that had been modified based on the parameters developed for the adult tribal RME scenario based on Tulalip data. The parameters for this scenario were used because this RME parameterization resulted in the most health-protective (i.e., lowest) screening values of the three RME scenarios and for consistency with the LDW HHRA (Windward 2007c).

However, it would also be possible to identify COPCs separately for each scenario based on the scenario-specific parameters. In other words, the RSLs could be modified separately for each scenario to develop scenario-specific RSLs. For most scenarios, this would result in a smaller COPC list than that identified in this HHRA (i.e., the RSLs based on the adult tribal RME scenario based on Tulalip data are more health protective). The one exception is the adult tribal scenario based on Suquamish data, for which scenario-specific RSLs are lower than those used in this HHRA. Suquamish tribal parameters were used to develop Suquamish scenario-specific RSLs, which resulted in two additional chemicals (1-methylnaphthalene and gamma-BHC) identified as having detected concentrations greater than screening levels, and two additional chemicals (3,3′-dichlorobenzidine and isophorone) identified as having RLs greater than screening levels. The details of this modified screen and the Suquamish-based RSLs are presented in Attachment 6.

To further evaluate the uncertainty associated with these four chemicals, risks were calculated for the adult tribal scenario based on Suquamish data. Detection frequencies and EPCs are presented in Table B.6-28 (with additional details provided in Attachment 6), and excess cancer risks and HQs are presented in Table B.6-29.

Table B.6-28. EPCs for Suquamish-based RSL evaluation

Consumption Category	gamma-BHC		1-Methyl-naphthalene		3,3′-Dichloro-benzidine		Isophorone	
	DF	EPC[a]	DF	EPC[a]	DF	EPC[a]	DF	EPC[a]
Benthic fish, fillet	0/1	0.00022	7/11	0.00057[b]	0/11	0.85	0/11	0.17
Benthic fish, WB	0/1	0.00023	4/11	0.1	0/11	0.5	0/11	0.1
Clam	0/6	0.00022	11/11	0.015[b]	0/10	0.75	0/10	0.15
Crab, EM	0/1	0.00021	1/9	0.0013	0/9	0.85	0/9	0.17
Crab, WB	0/1	0.00011	6/9	0.0011[b]	0/9	0.43	0/9	0.085
Geoduck, EM	1/1	0.00018[c]	0/6	0.00025	0/5	0.27	0/6	0.055
Geoduck, WB	1/1	0.00023[c]	0/4	0.0002	0/3	0.14	0/4	0.031
Mussel	0/1	0.00024	11/11	0.00072[b]	0/10	0.5	0/17	0.1
Pelagic fish, perch	0/1	0.00022	8/8	0.0024[b]	0/8	3.4	0/8	0.65
Pelagic fish, rockfish	0/9	0.00024	12/13	0.0025[b]	0/12	0.85	0/13	0.17

[a] All EPCs are presented in mg/kg ww and are equal to the one-half of the maximum RL unless otherwise noted. Summary statistics and additional details regarding the EPCs are provided in Attachment 6.

[b] EPC was calculated using ProUCL because the dataset contained six or more detected values.

[c] EPC is equal to the maximum detected concentration.

BHC – benzene hexachloride
DF – detection frequency
EM – edible meat
EPC – exposure point concentration
RL – reporting limit
RSL – regional screening level
WB – whole body
ww – wet weight

Table B.6-29. Risk estimates for chemicals with detected concentrations greater than Suquamish-based RSLs or reporting limits for non-detected chemicals greater than Suquamish-based RSLs

Chemical	Cancer Risks for the Adult Tribal Scenario Based on Suquamish Data			Non-Cancer Hazards for the Adult Tribal Scenario Based on Suquamish Data		
	CDI (mg/kg-day)	Slope Factor $(mg/kg\text{-}day)^{-1}$	Excess Cancer Risk	CDI (mg/kg-day)	RfD (mg/kg-day)	Hazard Quotient
Detected Chemicals						
gamma-BHC	1.6×10^{-6}	1.1	2×10^{-6}	1.6×10^{-6}	0.0003	0.005
1-Methylnaphthalene	8.2×10^{-5}	0.029	2×10^{-6}	8.2×10^{-5}	0.07	0.001
Non-Detected Chemicals[a]						
3,3’-Dichlorobenzidine	5.3×10^{-3}	0.45	2×10^{-3}	na	na	na
Isophorone	1.1×10^{-3}	0.00095	1×10^{-6}	1.1×10^{-3}	0.2	0.005

[a] Risks calculated for non-detected chemicals were based on reporting limits and are thus uncertain, as discussed in Section B.6.3.2.

BHC – benzene hexachloride
CDI – chronic daily intake
RfD – reference dose
RSL – regional screening level

The total excess cancer risk calculated in Section B.5 for the adult tribal scenario based on Suquamish data was equal to 1×10^{-2} (Table B.5-5), and hazard indices ranged from 0.4 to 219 (Table B.5-13). For the detected chemicals shown in Table B.6-29, both excess cancer risks and non-cancer HQs were well below these values and would not change these sums. The risks for the non-detected chemicals in Table B.6-29 are somewhat higher than those for the detected chemicals; but as discussed in Section B.6.3.2, risks for chemicals that were never detected are highly uncertain.

B.6.2 Toxicity Assessment

Three topics related to uncertainty in the toxicity assessment are discussed here in greater detail: toxicity benchmarks, the PCB toxicity assessment, and the chromium toxicity assessment.

B.6.2.1 Chemicals without toxicity benchmarks and RSLs

The toxicity benchmarks used in this HHRA are based on the most recent guidance provided by EPA. They are health-protective in that they include uncertainty factors or extrapolations to account for sensitive subpopulations or other limitations of the toxicity data on which they are based. The toxicity benchmarks presented in Section B.4 are based on many different studies using both animals and human populations. The RfDs published by EPA included consideration of data available at that time for effects on children (based in some cases on developmental effects in animal studies), particularly the developing fetus. The RfDs are designed to be protective of sensitive sub-populations, but the inherent uncertainty may span one or more orders of magnitude. For example, the RfD for methylmercury, which was used as a surrogate for mercury in this HHRA, is based on developmental effects on children following exposure during gestation. EPA's RfD for methylmercury has been extensively peer-reviewed and is thought to be sufficiently health-protective for children (NRC 2000).

Some chemicals may have developmental effects, but other effects were used by EPA to develop the RfDs. For example, several studies have documented developmental effects from the exposure of pregnant women to PCBs through fish consumption (Fein et al. 1984; Jacobson and Jacobson 1996, 1997), but the RfD published in IRIS is based on an immunological effect because it was considered to be more health-protective than the developmental effect (i.e., the effect occurs at a lower dose level). Studies published since the publication of the PCB RfD have investigated possible reproductive effects and neurotoxic effects in children. It is unclear whether consideration of these more recent neurotoxicity studies would result in a change to the current PCB RfD, which is based on immunotoxicity. Similarly, arsenic may have some developmental effects at sufficiently high dose levels (ATSDR 2005), but the critical study described in IRIS documenting dermal and cardiovascular effects was used to set the RfD because EPA considered these effects to be more health-protective than the developmental effect.

Of all the chemicals that were analyzed in EW samples, 21 do not have sediment RSLs, 28 do not have tissue RSLs, and 10 do not have water RSLs. Table B.6-30 lists these chemicals and indicates whether they have been detected. More information on the results of the analysis of these chemicals in EW tissue is provided in Attachment 2.

Table B.6-30. Chemicals in EW samples without toxicity benchmarks

Chemical Type	Sediment	Tissue	Surface Water
Detected chemicals	Magnesium Manganese Monobutyltin as ion Acenaphthylene Benzo(g,h,i)perylene Phenanthrene Dimethyl phthalate Di-n-octyl phthalate 1,3-Dichlorobenzene Carbazole TPH – oil and grease	Monobutyltin as ion Acenaphthylene Benzo(g,h,i)perylene Dibenzofuran Perylene Phenanthrene 2-Methylphenol	Thallium[a] Monobutyltin as ion Phenanthrene
Non-detected chemicals	Thallium[a] 2-Nitrophenol 3-Nitroaniline 4-Bromophenyl phenyl ether 4-Chlorophenyl phenyl ether delta-BHC	Thallium[a] Dimethyl phthalate Di-n-butyl phthalate Di-n-octyl phthalate 1,3-Dichlorobenzene 2-Nitrophenol 3,3'-Dichlorobenzidine 3-Nitroaniline 4-Bromophenyl phenyl ether 4-Chloro-3-methylphenol 4-Chlorophenyl phenyl ether 4-Methylphenol 4-Nitrophenol bis(2-Chloroisopropyl)ether Caffeine Carbazole Coprostanol delta-BHC Chlorpyrifos Isodrin Octachlorostyrene	Benzo(g,h,i)perylene Dimethyl phthalate Di-n-octyl phthalate 2-Nitrophenol 3-Nitroaniline 4-Bromophenyl phenyl ether 4-Chlorophenyl phenyl ether Carbazole

Note: Chemicals included in this table are only those without toxicity information that are not already accounted for as part of a sum or TEQ (as indicated in Attachment 2).

[a] When the screening for this HHRA was conducted, no RSL was available for thallium (the May 2010 version EPA's RSL tables (EPA 2010b)). Since this time, RSLs based on toxicity data for "thallium (soluble salts)" have become available. Based on the new RfD, thallium would be identified as a non-detect COPC for tissue (seafood consumption scenarios) and for sediment (netfishing and habitat restoration worker scenarios) but would not be a COC.

BHC – benzene hexachloride
COC – chemical of concern
COPC – chemical of potential concern
EW – East Waterway
HHRA – human health risk assessment
RfD – reference dose
RSL – regional screening level
TEQ – toxic equivalent
TPH – total petroleum hydrocarbons

Toxicity information for these chemicals is not provided in the EPA's RSL tables for sediment and surface water or in EPA's screening level calculator used to calculate fish tissue RSLs for this risk assessment (EPA 2009d). Toxicity benchmarks could be developed for these chemicals by the National Center for Exposure Assessment if a review was requested, as indicated in EPA guidance. However, inasmuch as these chemicals were not identified as COPCs through screening (because of inadequate toxicity information, and thus a lack of screening values), they were not included in the risk estimates. Overall risks may be underestimated if there are significant toxic effects associated with these chemicals at the concentrations present in the EW.

B.6.2.2 Total PCBs

One uncertainty associated with the PCB risk assessment is the difference between PCB mixtures found in the environment and those used in laboratory toxicity studies. As discussed by Cogliano (1998), the commercial PCB mixtures released into the environment may be altered by volatilization, vaporization, differential sorption, bacterial degradation, photolysis, and metabolism and elimination. In particular, differential bioaccumulation of more highly chlorinated PCB congeners may alter the toxicity of the bioaccumulated mixtures relative to unweathered Aroclors. PCB mixtures that have been altered by bioaccumulation processes are more toxic to mink than are unweathered Aroclors (EPA 1996b). In addition, bioaccumulated PCB mixtures might contain other bioaccumulative compounds that could modify the toxicity of PCBs.

Individual PCB congeners have a range of toxic effects (e.g., cancer, immune system, neurodevelopmental, cardiovascular, reproductive, endocrine/thyroid, dermal), and it is expected that environmental mixtures that differ in congener composition from unweathered Aroclors would also exhibit differences in toxicity. This uncertainty was considered in EPA's dose response evaluation and in their recommendation of a PCB RfD and SF (EPA 1996b).

EPA (1996b) has recommended a tiered approach for establishing the most appropriate SF for assessing excess cancer risks from PCBs. The PCB cancer SF associated with high risk and persistence was used for the seafood consumption scenarios. It is intended that this SF be applied to total PCBs rather than to any specific Aroclor mixture (EPA 1996b). Although alternative SFs for PCBs do exist, the application of an alternative SF for the seafood consumption scenarios would not be appropriate because of the highly persistent nature of many of the PCB congeners that bioaccumulate in fish (Lake et al. 1995) and sediment (Cogliano 1998). EPA derived a range of upper-end SFs, with greater potency reported for the more highly chlorinated Aroclors (EPA 1996b). The SF of 2 $(mg/kg\text{-}day)^{-1}$ was derived based on carcinogenicity data for Aroclor 1260 and 1254, which were the most frequently detected Aroclors in EW samples. Ultimately, rather than using toxicity data on unweathered Aroclor mixtures to predict the toxicity of environmental mixtures, it might be helpful to use toxicity data based on direct toxicity studies on relevant environmental mixtures to reflect the enrichment of persistent congeners, including dioxin-like PCBs. Such studies are outside the scope of the EW RI/FS. The SF for PCBs is based on a study of carcinogenicity data for Aroclors 1260 and 1254 (EPA 1996b), with the estimated SF for Aroclor 1260 being higher than that for Aroclor 1254. However, there are some uncertainties related to the toxicity evaluation of Aroclor 1254 from that EPA study. The Aroclor 1254 mixture evaluated initially differed from the formulation of most Aroclor 1254 that was produced in that it had a higher proportion of PCB 126 and PCDFs. To make the formulation more like the standard Aroclor 1254 formulation, 99% of the PCDFs in the mixture was removed, as well as some portion of the PCB 126, prior to the study (Mayes et al. 1998). However, the amount of PCB 126 in the mixture used was still three to five times greater than in the standard Aroclor 1254 formulation. The use of an atypical and altered formulation might have influenced the toxicity results, as compared with the results of a study conducted using the standard Aroclor 1254 formulation. If the toxicity of Aroclor 1254 was significantly affected by the different formulations, the cancer SF could potentially be affected as well.

Although the use of the most health-protective SF (derived based on Aroclors 1254 and 1260) may overestimate the toxicity of all PCB Aroclors, the uncertainty for the overall risk estimates associated with the PCB SF is low because the more highly chlorinated Aroclors (1254 and 1260) are the most frequently detected in EW samples.

B.6.2.3 PCB and dioxin/furan TEQs

To address the toxicity associated with dioxin-like PCB congeners, excess cancer risk was evaluated based on PCB TEQ exposure and the cancer SF associated with 2,3,7,8-TCDD. The use of toxic equivalency introduces an additional level of uncertainty because the TEFs used to calculate the PCB TEQ are estimates of congener toxicity relative to TCDD and have been rounded to a value of 1 or 3, regardless of the order of magnitude (Van den Berg et al. 2006). The PCB TEQ is then multiplied by the 2,3,7,8-TCDD cancer SF to calculate an excess cancer risk estimate. Excess cancer risk estimates based on PCB TEQ were the same or lower for most seafood consumption scenarios compared with excess cancer risk estimates based on total PCBs (see Table B.5-47). The implications of these two methods for the calculation of excess cancer risk associated with exposure to PCBs in the risk characterization step are discussed in Section B.6.3.1.

The TEQ approach is widely used in risk assessments for both dioxin/furan and dioxin-like PCB congeners. In a recent WHO re-evaluation of TEFs, it was noted that the "present TEF scheme (see Table 1)[54] and TEQ methodology are primarily meant for estimating exposure via dietary intake situations because present TEFs are based largely on oral uptake studies often through diet" (Van den Berg et al. 2006). The application of the TEQ approach based on oral TEFs to environmental matrices such as sediment or water for which exposures are largely dermal may greatly overestimate the potential toxicity of the mixture because the highly hydrophobic PCBs, PCDDs, and PCDFs bind

[54] Refers to Table 1 in Van den Berg et al. (2006), which is not included in this document.

strongly to particles, thereby significantly reducing their bioavailability to living organisms. The bioavailability of these chemicals is largely dependent upon the organic carbon content and age of the particles. This problem could be reduced if the degree of absorption of specific PCBs, PCDDs, and PCDFs was considered for direct contact sediment or water exposure assessments. Available information regarding the bioavailability of dioxins/furans in soils indicates that a more realistic estimation of bioavailability could be 40 to 60% (Ecology 2007), rather than 100% as assumed in this HHRA. If these lower bioavailability assumptions were used, dioxin/furan TEQ risk estimates in this HHRA would be approximately half of their current levels for the direct sediment exposure scenarios (i.e., ranging from 2×10^{-8} to 1×10^{-6} instead of 4×10^{-8} to 3×10^{-6}).

Toxicological studies using abiotic matrices with dioxin-like compounds that would allow the development of sediment-based TEFs are almost nonexistent (Van den Berg et al. 2006). Likewise, there is little information for the development of water-based TEFs. Thus, it is not possible to estimate the degree of overestimation included in the EW risk estimates for dioxin/furan and PCB congeners via the direct sediment or water exposure pathways.

In addition to uncertainties associated with the applicability of the TEQ approach to the direct sediment and water exposure pathways, there is some uncertainty associated with the 2,3,7,8-TCDD SF of 150,000 (mg/kg-day)$^{-1}$ that was used to calculate the TEQ risk estimates. Although now withdrawn, EPA's D*raft Recommended Interim Preliminary Remediation Goals for Dioxin in Soil at CERCLA and RCRA Sites* (EPA 2009a) presented a range of SFs that could be used to evaluate uncertainty in the SF used to calculate risks in Section B.5. The 2,3,7,8-TCDD SFs presented in this document include the following:

- EPA's Office of Health and Environmental Assessment – 156,000 (mg/kg-day)$^{-1}$
- EPA's HEAST – 150,000 (mg/kg-day)$^{-1}$
- Cal EPA – 130,000 (mg/kg-day)$^{-1}$
- Michigan Department of Environmental Quality – 75,000 (mg/kg-day)$^{-1}$
- Minnesota Department of Health – 1,400,000 (mg/kg-day)$^{-1}$

However, EPA determined that the SFs developed by the Michigan and Minnesota agencies were not considered appropriate for use in CERCLA (or RCRA) risk assessments because they did not meet EPA's evaluation criteria (EPA 2009a). The remaining alternate values were used to evaluate the uncertainty associated with the SF of 150,000 (mg/kg-day)$^{-1}$ that was used in Section B.5. Table B.6-31 presents excess cancer risks calculated using the alternate SFs of 130,000 and 156,000 (mg/kg-day)$^{-1}$ for the seafood consumption and direct sediment contact RME scenarios. As can be seen in this table, the resulting risk estimates are the same as or slightly lower than those presented in Section B.5, indicating that the uncertainty associated with the selected SF for 2,3,7,8-TCDD is relatively low.

Table B.6-31. Excess cancer risks for dioxin/furan TEQ calculated using different slope factors for the seafood consumption RME scenarios

Scenario	Excess Cancer Risk Calculated Using Different Slope Factors		
	As Presented in Section B.5 (SF of 150,000 mg/kg-day^{-1})	Cal EPA SF (130,000 mg/kg-day^{-1})	EPA Office of Health and Environmental Assessment SF (156,000 mg/kg-day^{-1})
Seafood Consumption RME Scenarios			
PCB TEQ			
Adult tribal RME scenario (Tulalip data)	7×10^{-4}	6×10^{-4}	7×10^{-4}
Child tribal RME scenario (Tulalip data)	1×10^{-4}	1×10^{-4}	1×10^{-4}
Adult API RME scenario	3×10^{-4}	3×10^{-4}	3×10^{-4}
Dioxin/Furan TEQ			
Adult tribal RME scenario (Tulalip data)	1×10^{-4}	9×10^{-5}	1×10^{-4}
Child tribal RME scenario (Tulalip data)	2×10^{-5}	2×10^{-5}	2×10^{-5}
Adult API RME scenario	4×10^{-5}	3×10^{-5}	4×10^{-5}
Total TEQ for Dioxins/Furans and Coplanar PCBs			
Adult tribal RME scenario (Tulalip data)	8×10^{-4}	7×10^{-4}	8×10^{-4}
Child tribal RME scenario (Tulalip data)	1×10^{-4}	1×10^{-4}	1×10^{-4}
Adult API RME scenario	3×10^{-4}	3×10^{-4}	3×10^{-4}
Direct Sediment Contact RME Scenarios			
PCB TEQ			
Netfishing RME	3×10^{-7}	3×10^{-7}	3×10^{-7}
Tribal clamming RME	1×10^{-6}	9×10^{-7}	1×10^{-6}
Dioxin/Furan TEQ			
Netfishing RME	6×10^{-7}	4×10^{-7}	6×10^{-7}
Tribal clamming RME	1×10^{-6}	1×10^{-6}	1×10^{-6}
Total TEQ for Dioxins/Furans and Coplanar PCBs			
Netfishing RME	9×10^{-7}	7×10^{-7}	9×10^{-7}
Tribal clamming RME	2×10^{-6}	2×10^{-6}	2×10^{-6}

API – Asian and Pacific Islander
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
SF – slope factor
TEQ – toxic equivalent

B.6.2.4 Chromium speciation

The available chromium data for sediment, tissue, and water are based on total chromium. However, the RfD used for chromium in this HHRA is based on hexavalent chromium, for which the RfD is orders of magnitude lower than the RfD for chromium III and which would likely make up only a portion of the total chromium. This health-protective assumption overestimates the risks from chromium, because chromium VI is unlikely to be present in any substantial quantity in a marine environment. However, because chromium was not identified as exceeding acceptable risk levels (i.e., HQ was not greater than 1) for any scenario, the overall impact to the risk conclusions is low.

In addition, it should be noted that EPA is currently reviewing whether chromium VI is carcinogenic via oral exposure. This risk assessment does not evaluate excess cancer risks for chromium because no cancer slope factor is currently available for oral exposure. If a slope factor were available, the determination of the portion of total chromium in EW media that is chromium III and chromium VI could become more important.

B.6.2.5 Mercury speciation

The EPC for total mercury was used as a surrogate for methylmercury, the toxic form of mercury for which the RfD was developed. Sediment and tissue data collected from the EW were not analyzed for methylmercury, consistent with EPA guidance (EPA 2000d). For the seafood consumption scenarios, concentrations of total mercury and methylmercury in fish tissue were expected to be similar (i.e., methylmercury is close to 100% of the total mercury concentration) based on the data available for English sole fillets from the LDW collected as part of the 1996 Elliott Bay/Duwamish River fish tissue study (Battelle 1996; Frontier Geosciences 1996), as discussed in the LDW HHRA (Windward 2007c, Section B.2.1.2). The ratios of methylmercury to total mercury appear to be more variable for invertebrates as compared with those for fish. For clams, methylmercury can account for 72 to 95% of the total mercury (Trombini et al. 2003) and in a study on blue crabs, methylmercury accounted for 35% of total mercury (Ward et al. 1979). Thus, despite the lack of tissue methylmercury data, there is relatively low uncertainty associated with the risk estimates for mercury in the seafood consumption scenarios, and if anything, this approach is health-protective (i.e., conservative). For all seafood consumption scenarios except the adult tribal scenario based on Suquamish data, HQs for mercury were less than or equal to 1. In addition, the child tribal seafood consumption scenario based on Suquamish data (evaluated in Section B.6.1.2.3) had a mercury HQ greater than 1.

No site-specific data for methylmercury in EW sediment have been collected. Based on total mercury concentrations and the toxicity benchmark for methylmercury, mercury was not identified as a COPC for the sediment exposure scenarios. Total mercury and methylmercury data from other estuaries suggest that methylmercury makes up a very small fraction of total mercury in sediment (Mason and Lawrence 1999); thus, it is likely that the risks for the sediment exposure scenarios are overestimated in this HHRA. However, because risks from mercury were low (i.e., mercury was not identified as a COC for either the seafood consumption scenarios or the direct sediment exposure scenarios), the overall impact of the lack of methylmercury data on the risk conclusions is very low.

B.6.2.6 Arsenic speciation

EPA guidance recommends the use of inorganic arsenic tissue concentration data for the purposes of evaluating human health risks based on seafood consumption (EPA 2000c). Organic arsenic (the other portion of total arsenic present in tissue) "has been shown in numerous studies to be metabolically inert and nontoxic" (EPA 2000c). Based on this information, EPA's IRIS database provides a toxicity value based on inorganic arsenic because it is known to be highly toxic. Thus, for the purpose of evaluating risks based on seafood consumption, both the tissue data and the toxicity information are based on inorganic arsenic. Although it is possible that this may slightly underestimate arsenic risks because organic arsenic is not being evaluated, the associated uncertainty is very low. However, it should be noted that research regarding the toxicity associated with organic forms of arsenic is ongoing. EPA recently developed a provisional RfD for dimethyl arsenic acid (DMA), which is a form of organic arsenic (EPA 2011b). Arsenosugars (another form of organic arsenic) are generally metabolized to DMA in humans, and thus DMA may account for a significant portion of the organic arsenic in humans.

To evaluate the hazards associated with DMA, HQs were calculated using a provisional RfD of 0.014 mg/kg-day for DMA (for reference, the RfD for inorganic arsenic is 0.0003 mg/kg-day, approximately 50 times lower [i.e., more toxic] than the provisional value for DMA). DMA was not analyzed in tissue samples collected from the EW, and thus literature information was used to convert total arsenic to DMA. An Ecology study regarding arsenic concentrations in Puget Sound reported the DMA and total arsenic concentrations for various fish and crab species (Ecology 2002). Using these data, the percent of total arsenic that is DMA was calculated to range from 0.2% to 13% across the various seafood types evaluated in the study.[55] As a health-protective assumption, HQs were calculated assuming the high-end of this range (13%), which resulted in HQs ranging from 0.02 to 0.08 for DMA for the seafood consumption RME scenarios (i.e., an order of magnitude lower than the HQs calculated for inorganic arsenic in Section B.5). This further supports the conclusion that the use of inorganic arsenic to evaluate arsenic in the HHRA is health-protective.

In addition to a consideration of the toxicity of organic forms of arsenic, the percentage of total arsenic that is inorganic was examined. Inorganic arsenic has been reported to generally make up between less than 1 and 20% of total arsenic in seafood tissue (EPA 2000c). Data collected from the EW largely follow this pattern, with inorganic arsenic percentages ranging from 0.1 to 3.2% for fish, 0.4 to 1.3% for crab, 1.3 to 4.9% for geoduck, 4.5% to 15.7% for mussels, and 3.4 to 47% for clams. Only the range for clams was outside of the range specified by EPA. The percent inorganic arsenic for clams was

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[55] The five species for which total arsenic and DMA data was available in the Ecology report included the following: English sole (average of 0.3% DMA), quillback rockfish (average of 0.6% DMA), Dungeness crab (average of 3% DMA), Coho salmon (average of 7% DMA), and Pacific herring (average of 10% DMA).

highly variable by species, with cockles and Eastern soft-shell clams containing considerably more inorganic arsenic (18 and 21% for the two cockle composite samples and 47% for the single Eastern soft-shell clam composite sample). The percent inorganic arsenic for butter clams and native littleneck ranged from 3.4 to 8.5%, well within the range specified by EPA. Additional analysis of the percentage of total arsenic that is inorganic is presented in the RI.

B.6.3 Risk Characterization

In addition to the uncertainties related to exposure and toxicity, the risk characterization step can also have uncertainty. The first area of uncertainty discussed in this section relates to the total excess cancer risk estimates for multiple chemicals, particularly PCBs. Uncertainties related to potential health risks associated with chemicals that were never detected in tissue or sediment samples are also evaluated.

B.6.3.1 Inclusion of PCBs in estimates of the total excess cancer risk

PCBs consist of 209 individual congeners. Aroclors are commercial mixtures of PCB congeners that contain a large number of individual congeners. The different Aroclors contain many of the same congeners and vary mostly in terms of the relative abundance of specific congeners. After a commercial mixture is released into the environment, the original congener composition of the commercial PCB mixture changes over time through various processes (e.g., partitioning between environmental media, chemical transformation, and preferential bioaccumulation) (Cogliano 1998). The assessment of cancer risks for environmental PCB mixtures is complicated in that carcinogenicity data are available for commercial but not environmental mixtures. Consequently, the carcinogenicity of commercial mixtures must be used to estimate the toxicity of environmental mixtures that may have a different congener composition than that of the Aroclors used to develop the carcinogenicity data. Cancer risks for environmental PCB mixtures may be estimated on the basis of either: 1) commercial Aroclor toxicity (hereafter referred to as total PCB risks), or 2) the toxicity of specific components of Aroclor mixtures (i.e., co-planar PCB congeners that have a mode of toxicity similar to that of dioxin [hereafter referred to as PCB TEQ risks]). Total PCB cancer risks are computed by multiplying the total PCB CDI by the SF for PCBs (as Aroclors). As discussed in Section B.4, after PCB TEQs were calculated by applying the TEFs to the individual dioxin-like PCB congeners, PCB TEQ cancer risks were computed by multiplying the PCB TEQ CDI by the dioxin SF.

Challenges exist in using total PCB and PCB TEQ cancer risk estimates to represent the true risks posed by environmental PCB mixtures. As will be subsequently noted in examples from guidance and site-specific risk assessments, the cancer risks posed by environmental PCB mixtures are bounded on the low end by total PCB or PCB TEQ cancer risk estimates and bounded on the high end by summing total PCB and PCB TEQ cancer risk estimates. There are issues with these approaches for generating lower and upper estimates of PCB cancer risks. Environmental processes (e.g., bioaccumulation) may increase levels of more highly chlorinated and potentially more toxic congeners (e.g., co-planar PCBs with dioxin-like toxicity) relative to those found in commercial PCB mixtures (EPA 1996b). Hence, using either total PCB or PCB TEQ cancer risk estimates alone to describe overall environmental PCB cancer risks may underestimate the true risk posed by an environmental PCB mixture. However, adding total PCB and PCB TEQ cancer risks may overestimate the true risk posed by an environmental PCB mixture. Co-planar PCBs were present in the commercial mixtures used to derive the Aroclor SF; hence, there is a likely potential for "double counting" the risk posed by the co-planar PCBs when adding total PCB and PCB TEQ cancer risks.

A further uncertainty is the degree to which potential co-planar PCB enrichment in environmental vs. commercial PCB mixtures is the primary cause for enhanced carcinogenicity in environmental PCB mixtures. The EPA Science Advisory Board cited the van der Plas et al. (2000) study of rats exposed to Aroclor 1260, which suggests that most of the tumor promotion potential of PCB mixtures is attributable to the non-dioxin-like fraction (EPA 2001a). Because this fraction is not included in the TEQ calculation, van der Plas et al. (2000) concluded that the tumor promotion potential of PCBs might be underestimated by the TEQ approach alone. This is also supported by estimates of TEQs for the different Aroclors. Although the cancer SF included the consideration of several Aroclors, the SFs for 1260, followed by 1254, were the highest in the studies evaluated and were used for the development of the SF for total PCBs (EPA 1996b). However, the TEQ potency for Aroclor 1260 on a mass basis is lower than the potencies for several other Aroclors (Rushneck et al. 2004; Van den Berg et al. 2006). This comparison also suggests that some of its carcinogenic potency is not attributable to dioxin-like PCB congeners.

However, there is uncertainty related to the carcinogenic potency of non-dioxin-like PCB congeners. Knerr and Schrenk (2006) reviewed the carcinogenicity of non-dioxin-like PCB congeners across numerous studies and concluded that in most cases, dioxin-like PCB congeners were more potent tumor promoters than non-dioxin-like congeners. However, they also stated that a weak carcinogenic potency of some non-dioxin-like congeners could not be excluded. In the case of the van der Plas (2000) study, Knerr and Schrenk (2006) asserted that the purity data provided in that study was not enough to exclude the potential contribution of some dioxin-like congeners to the observed toxicity.

Several approaches are available to address the fact that commercial Aroclor mixtures contain PCB congeners that have dioxin-like activity (including those listed below), although the benefits and limitations of these approaches are still being evaluated. These approaches include recommendations made in EPA's *Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures* (EPA 2000e), an example given in EPA's *PCBs: Cancer Dose-Response Assessment and Application to Environmental Mixtures* (EPA 1996b), PCB and PCB TEQ approaches discussed in the risk assessment for the Housatonic Superfund site (Weston Solutions 2005), and the analysis of PCB and PCB TEQ risks from Columbia Basin fish (EPA 2000e). The implications of using different approaches to address this issue were explored quantitatively in order to estimate the PCB cancer risk for the adult tribal RME seafood consumption scenario based on Tulalip data. As recommended by EPA Region 10 (EPA 2006b) in comments on the LDW HHRA (Windward 2007c), several options were explored:

- Aroclor sum excess cancer risk alone
- PCB TEQ excess cancer risk alone
- Aroclor sum excess cancer risk plus PCB TEQ excess cancer risk
- PCB TEQ excess cancer risk plus excess cancer risk computed using the sum of Aroclor mass minus the mass of dioxin-like PCB congeners

Options 1 and 2 may lead to an underestimation of risk, and Option 3 may overestimate risk because the mass and toxicity of PCB congeners may be double-counted, as mentioned previously. The rationale for Option 4 is to prevent the double-counting of the mass of the dioxin-like PCB congeners. However, this approach does not address the potential double-counting of the toxicity of dioxin-like PCB congeners that were present in the Aroclor test material used to generate the PCB SF.

The sum of Aroclor risks was calculated using the total PCB SF, and PCB TEQ risks were calculated using the dioxin SF. The EPCs, CDIs, and risk estimates for the adult tribal RME seafood consumption scenario based on Tulalip data needed for Options 1, 2, and 3 and the first part of Option 4 were previously presented in Sections B.3 and B.5. To calculate the sum of Aroclor mass minus the mass of the dioxin-like PCB congeners (for the second part of Option 4), the difference between the sum of Aroclors and the mass of dioxin-like PCB congeners was calculated for each sample, and then the EPCs for each seafood consumption category were recalculated. The resulting EPCs are presented in Table B.6-32.

Table B.6-32. EPCs for sum of Aroclor mass minus the mass of dioxin-like PCB congeners

Seafood Category[a]	Detection Frequency (Ratio)[b]	Concentrations (mg/kg ww)			Statistic Used	EPC (mg/kg ww)
		Mean Value	Maximum Detect	Maximum RL		
Benthic fish, fillet	3/3	1.15	1.59	na	maximum detect	1.6
Benthic fish, whole body	3/3	2.39	2.59	na	maximum detect	2.6
Clams	3/3	0.0640	0.0805	na	maximum detect	0.081
Crab, edible meat	3/3	0.0651	0.0873	na	maximum detect	0.087
Crab, whole body	3/3	0.373	0.393	na	maximum detect	0.39
Geoduck, edible meat	3/3	0.021	0.023	na	maximum detect	0.023
Geoduck, whole body	1/1	0.024	0.024	na	maximum detect	0.024
Pelagic fish, perch	3/3	0.954	1.10	na	maximum detect	1.1
Pelagic fish, rockfish	6/6	2.01	3.76	na	95% Student's-t UCL	3.1

[a] No PCB congener data were available for mussels. Therefore, the EPC for total PCBs (Aroclors) was used for mussels in CDI and risk estimations.

Table B.6-31. EPCs for sum of Aroclor mass minus the mass of dioxin-like PCB congeners

[b] The total number of samples with PCB congener data (n = 28) available for this analysis and for the PCB TEQ EPCs presented in Section B.3.3.4 were fewer than the total number of samples with Aroclor data (n = 124) available for the total PCB EPC presented in Section B.3.3.4.

CDI – chronic daily intake
EPC – exposure point concentration
na – not applicable
PCB – polychlorinated biphenyl
RL – reporting limit
TEQ – toxic equivalent
UCL – upper confidence limit on the mean
ww – wet weight

Table B.6-33 presents the excess cancer risk estimates calculated for each of the four options using the EPCs presented in Section B.3.3.4 (for Options 1 through 3 and the first part of Option 4) and from Table B.6-33 (for the second part of Option 4) and the appropriate SFs (presented in Section B.4). Option 2, which was presented in Section B.5, has the lowest risk estimates of the four options evaluated (7×10^{-4}). The risk estimates for the other three options are similar (ranging from 1×10^{-3} to 2×10^{-3}, Table B.6-33). Note that Option 1 was also presented in Section B.5.

Table B.6-33. PCB excess cancer risk estimates calculated using various methods

Risk Calculation Option	PCB Type	Adult Tribal RME Scenario Based on Tulalip Data	
		Cancer CDI (mg/kg-day)	Excess Cancer Risk
1. Aroclor sum excess cancer risk alone	total PCBs based on Aroclors	5.4×10^{-4}	1×10^{-3}
2. PCB TEQ excess cancer risk alone	PCB TEQ	4.6×10^{-9}	7×10^{-4}
3. Aroclor sum excess cancer risk plus PCB TEQ excess cancer risk	total PCBs based on Aroclors and PCB TEQ	na	2×10^{-3}
4. PCB TEQ excess cancer risk plus excess cancer risk computed using sum of Aroclor mass minus mass of dioxin-like PCB congeners	Aroclor mass minus mass of dioxin-like PCB congeners (calculated using total PCBs SF)	3.9×10^{-4}	8×10^{-4}
	PCB TEQ	4.6×10^{-9}	7×10^{-4}
	Total estimate for Option 4	na	2×10^{-3}

CDI – chronic daily intake
na – not applicable
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
SF – slope factor
TEQ – toxic equivalent

As can be seen in Table B.6-33, the excess cancer risks for total PCBs can be quite variable depending on the risk calculation method selected. Thus, the potential impact on risk estimates was classified as high. However, it should be noted that one issue with the comparison of risks associated with these four options is that different datasets with different numbers of samples were available for different portions of the analysis. The total PCBs data as sum of Aroclors were available for a larger dataset (sample size = 124, used for Option 1 and the first part of Option 3). A smaller number of samples were analyzed for PCB congeners (sample size = 28, used for Option 2, the second part of Option 3, and Option 4). The larger dataset was used for risk characterization in this HHRA because it was expected to more accurately represent EW PCB risks than did the smaller PCB congener dataset. Excess cancer risk estimates for total PCBs (without consideration of PCB TEQ) for the larger dataset and the congener subset are not equivalent, with risk estimates for the total PCBs sum based on the sum of congeners being approximately 30 to 40% lower than the risks for the total PCBs sum based on the sum of Aroclors (see Section B.6.1.1.3). Differences in EPCs for total PCBs based on Aroclors and congeners may exist because different samples were analyzed for these data (fewer samples were analyzed for congeners, as discussed in Attachment 1) as well as differences in the number of samples (i.e., smaller sample sizes contribute to lower confidence in estimates of the mean and therefore higher 95% UCLs and EPCs).

B.6.3.2 Risk calculations for non-detected chemicals

As indicated in Section B.5.2, risks were characterized only for those chemicals that were detected in the medium specific to that exposure scenario (i.e., sediment, tissue, or water). Several chemicals in each scenario were never detected, but a sufficient number of sample RLs exceeded the applicable RSLs, and these non-detected chemicals were therefore identified as COPCs (see Section B.3.2).[56] Hypothetical EPCs were calculated for these non-detected chemicals and are presented in Table B.6-34 for seafood consumption scenarios, in Table B.6-35 for direct sediment exposure scenarios, and in Table B.6-36 for the swimming scenario. The hypothetical EPCs correspond to one-half the highest RL for that chemical.

[56] A total of 26 of 54 COPCs in tissue were non-detects in all seafood categories, 0 of 9 COPCs in sediment for the netfishing scenario were non-detects, 0 of 5 COPCs in sediment for the habitat restoration worker scenario were non-detects, 3 of 11 COPCs in sediment for the clamming scenarios were non-detects, and 9 of 15 COPCs in surface water were non-detects.

Table B.6-34. EPCs and summary statistics for non-detected COPCs in tissue

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used	EPC (mg/kg ww)
BEHP					
Benthic fish, fillet	0/3	0.035	0.10 U	one-half maximum RL	0.050
Benthic fish, whole body	0/3	0.026	0.066 U	one-half maximum RL	0.033
Clam	0/10	0.044	0.29 U	one-half maximum RL	0.15
Crab, edible meat	0/3	0.025	0.057 U	one-half maximum RL	0.029
Crab, whole body	0/3	0.015	0.041 UM	one-half maximum RL	0.021
Geoduck clam, edible meat	0/6	0.018	0.060 U	one-half maximum RL	0.03
Geoduck clam, whole body	0/4	0.015	0.055 UM	one-half maximum RL	0.028
Mussels	0/9	0.025	0.20 UJ	one-half maximum RL	0.10
Pelagic fish, perch	0/3	0.093	0.24 U	one-half maximum RL	0.12
Pelagic fish, rockfish	0/13	0.038	0.15 U	one-half maximum RL	0.075
Butyl benzyl phthalate					
Benthic fish, fillet	0/11	0.17	0.33 U	one-half maximum RL	0.17
Benthic fish, whole body	0/11	0.10	0.20 U	one-half maximum RL	0.10
Clam	0/10	0.15	0.30 U	one-half maximum RL	0.15
Crab, edible meat	0/9	0.11	0.33 U	one-half maximum RL	0.17
Crab, whole body	0/9	0.067	0.17 UM	one-half maximum RL	0.085
Geoduck clam, edible meat	0/6	0.034	0.11 U	one-half maximum RL	0.055
Geoduck clam, whole body	0/4	0.026	0.062 UM	one-half maximum RL	0.031
Mussels	0/17	0.068	0.20 U	one-half maximum RL	0.10
Pelagic fish, perch	0/8	0.65	1.3 U	one-half maximum RL	0.65
Pelagic fish, rockfish	0/13	1.3	4.9 U	one-half maximum RL	2.5
1,2,4-Trichlorobenzene					
Benthic fish, fillet	0/11	0.17	0.33 U	one-half maximum RL	0.17
Benthic fish, whole body	0/11	0.10	0.20 U	one-half maximum RL	0.10
Clam	0/10	0.15	0.30 U	one-half maximum RL	0.15
Crab, edible meat	0/9	0.11	0.33 U	one-half maximum RL	0.17
Crab, whole body	0/9	0.067	0.17 UM	one-half maximum RL	0.085
Geoduck clam, edible meat	0/6	0.034	0.11 U	one-half maximum RL	0.055
Geoduck clam, whole body	0/4	0.026	0.062 UM	one-half maximum RL	0.031
Mussels	0/17	0.068	0.20 U	one-half maximum RL	0.10
Pelagic fish, perch	0/8	0.65	1.3 U	one-half maximum RL	0.65
Pelagic fish, rockfish	0/13	0.17	0.33 U	one-half maximum RL	0.17
1,2-Diphenylhydrazine[b]					
Mussels	0/6	0.027	0.027	one-half maximum RL	0.027
2,4,6-Trichlorophenol					
Benthic fish, fillet	0/11	0.85	1.7 U	one-half maximum RL	0.85
Benthic fish, whole body	0/11	0.50	1.0 U	one-half maximum RL	0.50
Clam	0/10	0.73	1.5 U	one-half maximum RL	0.75
Crab, edible meat	0/9	0.54	1.7 U	one-half maximum RL	0.85

Table B.6-34. EPCs and summary statistics for non-detected COPCs in tissue (cont.)

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used	EPC (mg/kg ww)
Crab, whole body	0/9	0.33	0.85 UM	one-half maximum RL	0.43
Geoduck clam, edible meat	0/6	0.17	0.53 U	one-half maximum RL	0.27
Geoduck clam, whole body	0/4	0.13	0.31 UM	one-half maximum RL	0.16
Mussels	0/17	0.34	1.0 U	one-half maximum RL	0.50
Pelagic fish, perch	0/8	3.3	6.7 U	one-half maximum RL	3.4
Pelagic fish, rockfish	0/13	0.85	1.7 U	one-half maximum RL	0.85
2,4-Dichlorophenol					
Benthic fish, fillet	0/11	0.85	1.7 U	one-half maximum RL	0.85
Benthic fish, whole body	0/11	0.50	1.0 U	one-half maximum RL	0.50
Clam	0/10	0.73	1.5 U	one-half maximum RL	0.75
Crab, edible meat	0/9	0.54	1.7 U	one-half maximum RL	0.85
Crab, whole body	0/9	0.33	0.85 UM	one-half maximum RL	0.43
Geoduck clam, edible meat	0/6	0.17	0.53 U	one-half maximum RL	0.27
Geoduck clam, whole body	0/4	0.13	0.31 UM	one-half maximum RL	0.16
Mussels	0/17	0.33	1.0 U	one-half maximum RL	0.50
Pelagic fish, perch	0/8	3.3	6.7 U	one-half maximum RL	3.4
Pelagic fish, rockfish	0/13	0.85	1.7 U	one-half maximum RL	0.85
2,4-Dinitrophenol					
Benthic fish, fillet	0/11	1.7	3.3 U	one-half maximum RL	1.7
Benthic fish, whole body	0/11	1.0	2.0 U	one-half maximum RL	1.0
Clam	0/10	1.5	3.0 U	one-half maximum RL	1.5
Crab, edible meat	0/9	1.1	3.3 U	one-half maximum RL	1.7
Crab, whole body	0/9	0.67	1.7 UM	one-half maximum RL	0.85
Geoduck clam, edible meat	0/6	0.34	1.1 U	one-half maximum RL	0.55
Geoduck clam, whole body	0/4	0.26	0.62 UM	one-half maximum RL	0.31
Mussels	0/17	0.66	2.0 U	one-half maximum RL	1.0
Pelagic fish, perch	0/8	6.5	13 U	one-half maximum RL	6.5
Pelagic fish, rockfish	0/13	1.7	3.3 U	one-half maximum RL	1.7
2,4-Dinitrotoluene					
Benthic fish, fillet	0/11	0.85	1.7 U	one-half maximum RL	0.85
Benthic fish, whole body	0/11	0.50	1.0 U	one-half maximum RL	0.50
Clam	0/10	0.73	1.5 U	one-half maximum RL	0.75
Crab, edible meat	0/9	0.54	1.7 U	one-half maximum RL	0.85
Crab, whole body	0/9	0.33	0.85 UM	one-half maximum RL	0.43
Geoduck clam, edible meat	0/6	0.17	0.53 U	one-half maximum RL	0.27
Geoduck clam, whole body	0/4	0.13	0.31 UM	one-half maximum RL	0.16
Mussels	0/17	0.33	1.0 U	one-half maximum RL	0.50
Pelagic fish, perch	0/8	3.3	6.7 U	one-half maximum RL	3.4
Pelagic fish, rockfish	0/13	0.85	1.7 U	one-half maximum RL	0.85
2,6-Dinitrotoluene					
Benthic fish, fillet	0/11	0.85	1.7 U	one-half maximum RL	0.85
Benthic fish, whole body	0/11	0.50	1.0 U	one-half maximum RL	0.50
Clam	0/10	0.73	1.5 U	one-half maximum RL	0.75

Table B.6-34. EPCs and summary statistics for non-detected COPCs in tissue (cont.)

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used	EPC (mg/kg ww)
Crab, edible meat	0/9	0.54	1.7 U	one-half maximum RL	0.85
Crab, whole body	0/9	0.33	0.85 UM	one-half maximum RL	0.43
Geoduck clam, edible meat	0/6	0.17	0.53 U	one-half maximum RL	0.27
Geoduck clam, whole body	0/4	0.13	0.31 UM	one-half maximum RL	0.16
Mussels	0/17	0.33	1.0 U	one-half maximum RL	0.50
Pelagic fish, perch	0/8	3.3	6.7 U	one-half maximum RL	3.4
Pelagic fish, rockfish	0/13	0.85	1.7 U	one-half maximum RL	0.85
2-Nitroaniline					
Benthic fish, fillet	0/11	0.85	1.7 U	one-half maximum RL	0.85
Benthic fish, whole body	0/11	0.50	1.0 U	one-half maximum RL	0.50
Clam	0/10	0.73	1.5 U	one-half maximum RL	0.75
Crab, edible meat	0/9	0.54	1.7 U	one-half maximum RL	0.85
Crab, whole body	0/9	0.33	0.85 UM	one-half maximum RL	0.43
Geoduck clam, edible meat	0/6	0.17	0.53 U	one-half maximum RL	0.27
Geoduck clam, whole body	0/4	0.13	0.31 UM	one-half maximum RL	0.16
Mussels	0/17	0.34	1.0 U	one-half maximum RL	0.50
Pelagic fish, perch	0/8	3.3	6.7 U	one-half maximum RL	3.4
Pelagic fish, rockfish	0/13	0.85	1.7 U	one-half maximum RL	0.85
4,6-Dinitro-o-cresol					
Benthic fish, fillet	0/11	1.7	3.3 U	one-half maximum RL	1.7
Benthic fish, whole body	0/11	1.0	2.0 U	one-half maximum RL	1.0
Clam	0/10	1.5	3.0 U	one-half maximum RL	1.5
Crab, edible meat	0/9	1.1	3.3 U	one-half maximum RL	1.7
Crab, whole body	0/9	0.67	1.7 UM	one-half maximum RL	0.85
Geoduck clam, edible meat	0/6	0.34	1.1 U	one-half maximum RL	0.55
Geoduck clam, whole body	0/4	0.26	0.62 UM	one-half maximum RL	0.31
Mussels	0/17	0.66	2.0 U	one-half maximum RL	1.0
Pelagic fish, perch	0/8	6.5	13 U	one-half maximum RL	6.5
Pelagic fish, rockfish	0/13	1.7	3.3 U	one-half maximum RL	1.7
4-Chloroaniline					
Benthic fish, fillet	0/11	0.85	1.7 U	one-half maximum RL	0.85
Benthic fish, whole body	0/11	0.50	1.0 U	one-half maximum RL	0.50
Clam	0/10	0.73	1.5 U	one-half maximum RL	0.75
Crab, edible meat	0/9	0.54	1.7 U	one-half maximum RL	0.85
Crab, whole body	0/9	0.33	0.85 UM	one-half maximum RL	0.43
Geoduck clam, edible meat	0/5	0.17	0.53 U	one-half maximum RL	0.27
Geoduck clam, whole body	0/3	0.12	0.27 UM	one-half maximum RL	0.14
Mussels	0/10	0.50	1.0 U	one-half maximum RL	0.50
Pelagic fish, perch	0/8	3.3	6.7 U	one-half maximum RL	3.4
Pelagic fish, rockfish	0/12	0.85	1.7 U	one-half maximum RL	0.85
4-Nitroaniline					
Benthic fish, fillet	0/11	0.85	1.7 U	one-half maximum RL	0.85
Benthic fish, whole body	0/11	0.50	1.0 U	one-half maximum RL	0.50

Table B.6-34. EPCs and summary statistics for non-detected COPCs in tissue (cont.)

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used	EPC (mg/kg ww)
Clam	0/10	0.73	1.5 U	one-half maximum RL	0.75
Crab, edible meat	0/9	0.54	1.7 U	one-half maximum RL	0.85
Crab, whole body	0/9	0.33	0.85 UM	one-half maximum RL	0.43
Geoduck clam, edible meat	0/5	0.17	0.53 U	one-half maximum RL	0.27
Geoduck clam, whole body	0/3	0.12	0.27 UM	one-half maximum RL	0.14
Mussels	0/16	0.33	1.0 U	one-half maximum RL	0.50
Pelagic fish, perch	0/8	3.3	6.7 U	one-half maximum RL	3.4
Pelagic fish, rockfish	0/12	0.85	1.7 U	one-half maximum RL	0.85
Aniline					
Benthic fish, fillet	0/10	0.17	0.33 U	one-half maximum RL	0.17
Benthic fish, whole body	0/10	0.10	0.20 U	one-half maximum RL	0.10
Clam	0/10	0.15	0.30 U	one-half maximum RL	0.15
Crab, edible meat	0/9	0.11	0.33 U	one-half maximum RL	0.17
Crab, whole body	0/9	0.067	0.17 UM	one-half maximum RL	0.085
Geoduck clam, edible meat	0/5	0.035	0.11 U	one-half maximum RL	0.055
Geoduck clam, whole body	0/3	0.024	0.055 UM	one-half maximum RL	0.028
Mussels	0/16	0.072	0.20 U	one-half maximum RL	0.10
Pelagic fish, perch	0/8	0.65	1.3 U	one-half maximum RL	0.65
Pelagic fish, rockfish	0/12	0.17	0.33 UJ	one-half maximum RL	0.17
Bis(2-chloroethoxy)methane					
Benthic fish, fillet	0/11	0.17	0.33 U	one-half maximum RL	0.17
Benthic fish, whole body	0/11	0.10	0.20 U	one-half maximum RL	0.10
Clam	0/10	0.15	0.30 U	one-half maximum RL	0.15
Crab, edible meat	0/9	0.11	0.33 U	one-half maximum RL	0.17
Crab, whole body	0/9	0.067	0.17 UM	one-half maximum RL	0.085
Geoduck clam, edible meat	0/6	0.034	0.11 U	one-half maximum RL	0.055
Geoduck clam, whole body	0/4	0.026	0.062 UM	one-half maximum RL	0.031
Mussels	0/17	0.069	0.20 U	one-half maximum RL	0.10
Pelagic fish, perch	0/8	0.65	1.3 U	one-half maximum RL	0.65
Pelagic fish, rockfish	0/13	0.17	0.33 U	one-half maximum RL	0.17
Bis(2-chloroethyl)ether					
Benthic fish, fillet	0/11	0.17	0.33 U	one-half maximum RL	0.17
Benthic fish, whole body	0/11	0.10	0.20 U	one-half maximum RL	0.10
Clam	0/10	0.15	0.30 U	one-half maximum RL	0.15
Crab, edible meat	0/9	0.11	0.33 U	one-half maximum RL	0.17
Crab, whole body	0/9	0.067	0.17 UM	one-half maximum RL	0.085
Geoduck clam, edible meat	0/6	0.034	0.11 U	one-half maximum RL	0.055
Geoduck clam, whole body	0/4	0.026	0.062 UM	one-half maximum RL	0.031
Mussels	0/17	0.068	0.20 U	one-half maximum RL	0.10
Pelagic fish, perch	0/8	0.65	1.3 U	one-half maximum RL	0.65
Pelagic fish, rockfish	0/13	0.17	0.33 U	one-half maximum RL	0.17
Hexachlorobenzene					
Benthic fish, fillet	0/11	0.0025	0.0050 U	one-half maximum RL	0.0025

Table B.6-34. EPCs and summary statistics for non-detected COPCs in tissue (cont.)

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used	EPC (mg/kg ww)
Benthic fish, whole body	0/11	0.0024	0.0050 U	one-half maximum RL	0.0025
Clam	0/10	0.0024	0.0050 U	one-half maximum RL	0.0025
Crab, edible meat	0/9	0.0025	0.0050 U	one-half maximum RL	0.0025
Crab, whole body	0/9	0.0013	0.0034 UM	one-half maximum RL	0.0017
Geoduck clam, edible meat	0/6	0.0025	0.0050 U	one-half maximum RL	0.0025
Geoduck clam, whole body	0/4	0.0013	0.0025 UM	one-half maximum RL	0.0013
Mussels	0/17	0.0044	0.016 U	one-half maximum RL	0.008
Pelagic fish, perch	0/8	0.0025	0.0050 U	one-half maximum RL	0.0025
Pelagic fish, rockfish	0/13	0.0025	0.0050 U	one-half maximum RL	0.0025
Hexachlorobutadiene					
Benthic fish, fillet	0/11	0.0025	0.0050 U	one-half maximum RL	0.0025
Benthic fish, whole body	0/11	0.0024	0.0050 U	one-half maximum RL	0.0025
Clam	0/10	0.15	0.30 U	one-half maximum RL	0.15
Crab, edible meat	0/9	0.0025	0.0050 U	one-half maximum RL	0.0025
Crab, whole body	0/9	0.0013	0.0034 UM	one-half maximum RL	0.0017
Geoduck clam, edible meat	0/6	0.0025	0.0050 U	one-half maximum RL	0.0025
Geoduck clam, whole body	0/4	0.0013	0.0025 UM	one-half maximum RL	0.0013
Mussels	0/17	0.0063	0.027 U	one-half maximum RL	0.014
Pelagic fish, perch	0/8	0.0025	0.0050 U	one-half maximum RL	0.0025
Pelagic fish, rockfish	0/13	0.0032	0.017 U	one-half maximum RL	0.0085
Hexachlorocyclopentadiene					
Benthic fish, fillet	0/11	0.85	1.7 U	one-half maximum RL	0.85
Benthic fish, whole body	0/11	0.50	1.0 U	one-half maximum RL	0.50
Clam	0/10	0.73	1.5 U	one-half maximum RL	0.75
Crab, edible meat	0/9	0.54	1.7 U	one-half maximum RL	0.85
Crab, whole body	0/9	0.33	0.85 UM	one-half maximum RL	0.43
Geoduck clam, edible meat	0/6	0.17	0.53 U	one-half maximum RL	0.27
Geoduck clam, whole body	0/4	0.13	0.31 UM	one-half maximum RL	0.16
Mussels	0/17	0.33	1.0 U	one-half maximum RL	0.50
Pelagic fish, perch	0/8	3.3	6.7 U	one-half maximum RL	3.4
Pelagic fish, rockfish	0/13	0.85	1.7 U	one-half maximum RL	0.85
Hexachloroethane					
Benthic fish, fillet	0/11	0.17	0.33 U	one-half maximum RL	0.17
Benthic fish, whole body	0/11	0.10	0.20 U	one-half maximum RL	0.10
Clam	0/10	0.15	0.30 U	one-half maximum RL	0.15
Crab, edible meat	0/9	0.11	0.33 U	one-half maximum RL	0.17
Crab, whole body	0/9	0.067	0.17 UM	one-half maximum RL	0.085
Geoduck clam, edible meat	0/6	0.0025	0.005 U	one-half maximum RL	0.0025
Geoduck clam, whole body	0/4	0.0013	0.0025 UM	one-half maximum RL	0.0013
Mussels	0/17	0.064	0.20 U	one-half maximum RL	0.10
Pelagic fish, perch	0/8	0.65	1.3 U	one-half maximum RL	0.65
Pelagic fish, rockfish	0/13	0.17	0.33 U	one-half maximum RL	0.17

Table B.6-34. EPCs and summary statistics for non-detected COPCs in tissue (cont.)

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used	EPC (mg/kg ww)
Nitrobenzene					
Benthic fish, fillet	0/11	0.17	0.33 U	one-half maximum RL	0.17
Benthic fish, whole body	0/11	0.10	0.20 U	one-half maximum RL	0.10
Clam	0/10	0.15	0.30 U	one-half maximum RL	0.15
Crab, edible meat	0/9	0.11	0.33 U	one-half maximum RL	0.17
Crab, whole body	0/9	0.067	0.17 UM	one-half maximum RL	0.085
Geoduck clam, edible meat	0/6	0.034	0.11 U	one-half maximum RL	0.055
Geoduck clam, whole body	0/4	0.026	0.062 UM	one-half maximum RL	0.031
Mussels	0/17	0.069	0.20 U	one-half maximum RL	0.10
Pelagic fish, perch	0/8	0.65	1.3 U	one-half maximum RL	0.65
Pelagic fish, rockfish	0/13	0.17	0.33 U	one-half maximum RL	0.17
n-Nitrosodimethylamine[c]					
Benthic fish, fillet	0/11	0.85	1.7 U	one-half maximum RL	0.85
Benthic fish, whole body	0/11	0.50	1.0 U	one-half maximum RL	0.50
Crab, edible meat	0/9	0.54	1.7 U	one-half maximum RL	0.85
Crab, whole body	0/9	0.33	0.85 UM	one-half maximum RL	0.43
Geoduck clam, edible meat	0/6	0.17	0.53 U	one-half maximum RL	0.27
Geoduck clam, whole body	0/4	0.13	0.31 UM	one-half maximum RL	0.16
Mussels	0/17	0.34	1.0 U	one-half maximum RL	0.50
Pelagic fish, perch	0/8	3.3	6.7 U	one-half maximum RL	3.4
Pelagic fish, rockfish	0/13	0.85	1.7 U	one-half maximum RL	0.85
n-Nitroso-di-n-propylamine					
Benthic fish, fillet	0/11	0.85	1.7 U	one-half maximum RL	0.85
Benthic fish, whole body	0/11	0.50	1.0 U	one-half maximum RL	0.50
Clam	0/10	0.73	1.5 U	one-half maximum RL	0.75
Crab, edible meat	0/9	0.54	1.7 U	one-half maximum RL	0.85
Crab, whole body	0/9	0.33	0.85 UM	one-half maximum RL	0.43
Geoduck clam, edible meat	0/6	0.17	0.53 U	one-half maximum RL	0.27
Geoduck clam, whole body	0/4	0.13	0.31 UM	one-half maximum RL	0.16
Mussels	0/17	0.33	1.0 U	one-half maximum RL	0.50
Pelagic fish, perch	0/8	3.3	6.7 U	one-half maximum RL	3.4
Pelagic fish, rockfish	0/13	0.85	1.7 U	one-half maximum RL	0.85
n-Nitrosodiphenylamine					
Benthic fish, fillet	0/11	0.17	0.33 U	one-half maximum RL	0.17
Benthic fish, whole body	0/11	0.10	0.20 U	one-half maximum RL	0.10
Clam	0/10	0.15	0.30 U	one-half maximum RL	0.15
Crab, edible meat	0/9	0.11	0.33 U	one-half maximum RL	0.17
Crab, whole body	0/9	0.067	0.17 UM	one-half maximum RL	0.085
Geoduck clam, edible meat	0/6	0.034	0.11 U	one-half maximum RL	0.055
Geoduck clam, whole body	0/4	0.026	0.062 UM	one-half maximum RL	0.031
Mussels	0/17	0.069	0.20 U	one-half maximum RL	0.10
Pelagic fish, perch	0/8	0.65	1.3 U	one-half maximum RL	0.65
Pelagic fish, rockfish	0/13	0.17	0.33 U	one-half maximum RL	0.17

Table B.6-34. EPCs and summary statistics for non-detected COPCs in tissue (cont.)

Consumption Category	Detection Frequency	Mean Value (mg/kg ww)	Maximum RL (mg/kg ww)	Statistic Used	EPC (mg/kg ww)
Toxaphene					
Benthic fish, fillet	0/11	0.25	0.50 U	one-half maximum RL	0.25
Benthic fish, whole body	0/11	0.24	0.50 U	one-half maximum RL	0.25
Clam	0/10	0.24	0.50 U	one-half maximum RL	0.25
Crab, edible meat	0/9	0.25	0.50 U	one-half maximum RL	0.25
Crab, whole body	0/9	0.13	0.34 UM	one-half maximum RL	0.17
Geoduck clam, edible meat	0/6	0.25	0.50 U	one-half maximum RL	0.25
Geoduck clam, whole body	0/4	0.13	0.25 UM	one-half maximum RL	0.13
Mussels	0/14	0.19	0.49 U	one-half maximum RL	0.25
Pelagic fish, perch	0/8	0.25	0.50 U	one-half maximum RL	0.25
Pelagic fish, rockfish	0/13	0.25	0.50 U	one-half maximum RL	0.25
Aldrin					
Benthic fish, fillet	0/1	0.00044	0.00088 U	one-half maximum RL	0.00044
Benthic fish, whole body	0/1	0.00046	0.00092 U	one-half maximum RL	0.00046
Clam	0/6	0.00043	0.00087 U	one-half maximum RL	0.00044
Crab, edible meat	0/1	0.00042	0.00083 U	one-half maximum RL	0.00042
Crab, whole body	0/1	0.00022	0.00043 UM	one-half maximum RL	0.00022
Geoduck clam, edible meat	0/1	0.00044	0.00088 U	one-half maximum RL	0.00044
Geoduck clam, whole body	0/1	0.00022	0.00044 UM	one-half maximum RL	0.00022
Mussels	0/1	0.00047	0.00094 U	one-half maximum RL	0.00047
Pelagic fish, perch	0/1	0.00043	0.00086 U	one-half maximum RL	0.00043
Pelagic fish, rockfish	0/9	0.00045	0.00094 U	one-half maximum RL	0.00047

[a] Whole-body crab and geoduck samples were calculated using relative weights and concentrations of the edible meat and hepatopancreas for crabs or edible meat and gut ball for geoduck. Details regarding this approach are provided in Table B.2-3 and in Attachment 1.

[b] Only mussel data were available for 1,2-diphenylhydrazine. When the CDI and risk values were calculated, the entire seafood consumption was assumed to be mussels.

[c] No clam data were available for this chemical. When the CDI and risk values were calculated, seafood consumption that had been assigned to clams was divided proportionally among the remaining shellfish consumption categories.

BEHP – bis(2-ethylhexyl) phthalate
CDI – chronic daily intake
COPC – chemical of potential concern
EPC – exposure point concentration
J – estimated concentration
M – calculated concentration
RL – reporting limit
U – not detected at given concentration
ww – wet weight

Table B.6-35. EPCs and summary statistics for non-detected COPCs in sediment for exposure scenarios using only intertidal sediment data

COPC	Intertidal Exposure Area	Detection Frequency	Mean Value (mg/kg dw)	Maximum RL (mg/kg dw)	Standard Deviation (mg/kg dw)	Standard Error (mg/kg dw)	Statistic Used	EPC (mg/kg dw)
Antimony	public access intertidal[a]	0/1	3	6 UJ	nc	nc	one-half maximum RL	3
	site-wide intertidal[b]	0/3	3	6 UJ	nc	nc	one-half maximum RL	3
n-Nitrosodi-methylamine	public access intertidal[a]	0/1	0.015	0.030 U	nc	nc	one-half maximum RL	0.015
	site-wide intertidal[b]	0/3	0.015	0.030 U	nc	nc	one-half maximum RL	0.015
Toxaphene	public access intertidal[a]	0/1	0.27	0.54 U	nc	nc	one-half maximum RL	0.65
	site-wide intertidal[b]	0/3	0.29	0.84 U	nc	nc	one-half maximum RL	0.50

[a] The public access intertidal area was the exposure area used to evaluate the 7-days-per-year clamming scenario.

[b] The site-wide intertidal area was the exposure area used to evaluate the tribal clamming scenarios and the habitat restoration worker scenario.

COPC – chemical of potential concern

dw – dry weight

EPC – exposure point concentration

nc – not calculated

TEQ – toxic equivalent

U – not detected at given concentration

Table B.6-36. EPCs and summary statistics for non-detected COPCs in surface water

COPC	Detection Frequency	Mean Value (mg/L)	Maximum RL (mg/L)	Statistic Used	EPC (mg/L)
Benzo(a)pyrene	0/28	0.000076	0.0010 UJ	one-half maximum RL	0.00050
2,4-Dinitrotoluene	0/28	0.0025	0.0050 U	one-half maximum RL	0.0025
3,3'-Dichlorobenzidine	0/28	0.0025	0.0050 U	one-half maximum RL	0.0025
4,6-Dinitro-o-cresol	0/28	0.005	0.010 U	one-half maximum RL	0.0050
4-Chloroaniline	0/28	0.0025	0.0050 U	one-half maximum RL	0.0025
Bis(2-chloroethyl)ether	0/28	0.0005	0.0010 U	one-half maximum RL	0.0005
Hexachlorobenzene	0/28	0.0005	0.0010 U	one-half maximum RL	0.0005
n-Nitroso-di-n-propylamine	0/28	0.0025	0.0050 U	one-half maximum RL	0.0025
n-Nitrosodimethylamine	0/28	0.0021	0.0050 U	one-half maximum RL	0.0025

COPC – chemical of potential concern
EPC – exposure point concentration
J – estimated concentration
PCB – polychlorinated biphenyl
RL – reporting limit
TEQ – toxic equivalent
U – not detected at given concentration

Risks calculated using one-half RL values overestimate risks if these COPCs are not present (or are present only at concentrations lower than one-half the highest RL), or underestimate risks if the COPCs are present at an average concentration greater than one-half the RL. Laboratory RLs and the degree of spatial coverage of the samples are important factors to consider in determining whether the lack of detection truly indicates that a substance is not present. Information on possible chemical sources and environmental conditions (e.g., that affect the transport or speciation of chemicals) is also useful. These issues are discussed after the presentation of the risks and hazards associated with non-detected chemicals. If these COPCs are truly present in the samples, then the effect of using one-half the RL in the risk analysis is uncertain because the true concentration could be anywhere between zero and the RL.

Similar to the risks for detected carcinogenic COPCs in tissue, all non-detected carcinogenic COPCs in tissue had hypothetical excess cancer risk estimates greater than or equal to 1×10^{-6} for the adult tribal RME seafood consumption scenario based on Tulalip data (Table B.6-37). In addition, many of the risk estimates for non-detected chemicals were above this excess cancer risk for the child tribal RME scenario based on Tulalip data, adult API RME scenario, and one-meal-per-month seafood consumption scenarios. For the adult tribal CT scenario based on Tulalip data, child tribal CT scenario based on Tulalip data, and adult API CT scenario, only a few of the non-detected chemicals exceed the 1×10^{-6} risk threshold.

Table B.6-37. Summary of hypothetical excess cancer risk estimates for seafood consumption scenarios for COPCs that were never detected in EW tissue samples

Non-Detected COPC	Hypothetical Excess Cancer Risk by Seafood Consumption Scenario											
								Adult One Meal per Month[a]				
	Adult Tribal RME (Tulalip Data)	Adult Tribal CT (Tulalip Data)	Child Tribal RME (Tulalip Data)	Child Tribal CT (Tulalip Data)	Adult Tribal (Suquamish Data)	Adult API RME	Adult API CT	Benthic Fish	Clam	Crab	Pelagic Fish, Rockfish	Pelagic Fish, Perch
BEHP	1×10^{-6}	4×10^{-8}	3×10^{-7}	2×10^{-8}	1×10^{-5}	5×10^{-7}	6×10^{-9}	3×10^{-8}	9×10^{-8}	2×10^{-8}	5×10^{-8}	8×10^{-8}
1,2-Diphenylhydrazine[b]	3×10^{-5}	2×10^{-6}	5×10^{-6}	7×10^{-7}	2×10^{-4}	7×10^{-6}	2×10^{-7}	nd	nd	nd	nd	nd
2,4,6-Trichlorophenol	1×10^{-5}	7×10^{-7}	2×10^{-6}	3×10^{-7}	6×10^{-5}	3×10^{-6}	8×10^{-8}	4×10^{-7}	4×10^{-7}	4×10^{-7}	4×10^{-7}	2×10^{-6}
2,4-Dinitrotoluene	3×10^{-4}	2×10^{-5}	6×10^{-5}	8×10^{-6}	2×10^{-3}	8×10^{-5}	2×10^{-6}	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}	5×10^{-5}
2,6-Dinitrotoluene	7×10^{-4}	4×10^{-5}	1×10^{-4}	2×10^{-5}	4×10^{-3}	2×10^{-4}	5×10^{-6}	3×10^{-5}	2×10^{-5}	3×10^{-5}	3×10^{-5}	1×10^{-4}
4-Nitroaniline	2×10^{-5}	1×10^{-6}	4×10^{-6}	5×10^{-7}	1×10^{-4}	5×10^{-6}	1×10^{-7}	8×10^{-7}	7×10^{-7}	8×10^{-7}	8×10^{-7}	3×10^{-6}
Aniline	5×10^{-6}	7×10^{-8}	9×10^{-7}	3×10^{-8}	4×10^{-5}	2×10^{-6}	9×10^{-9}	4×10^{-8}	4×10^{-7}	4×10^{-8}	4×10^{-8}	2×10^{-7}
Bis(2-chloroethyl)ether	9×10^{-4}	1×10^{-5}	2×10^{-4}	6×10^{-6}	9×10^{-3}	4×10^{-4}	2×10^{-6}	8×10^{-6}	7×10^{-5}	8×10^{-6}	8×10^{-6}	3×10^{-5}
Hexachlorobenzene	5×10^{-6}	3×10^{-7}	9×10^{-7}	1×10^{-7}	3×10^{-5}	2×10^{-6}	4×10^{-8}	2×10^{-7}	2×10^{-7}	2×10^{-7}	2×10^{-7}	2×10^{-7}
Hexachlorobutadiene	6×10^{-5}	4×10^{-7}	1×10^{-5}	2×10^{-7}	6×10^{-4}	2×10^{-5}	7×10^{-8}	9×10^{-9}	5×10^{-6}	9×10^{-9}	3×10^{-8}	9×10^{-9}
Hexachloroethane	1×10^{-5}	2×10^{-7}	2×10^{-6}	8×10^{-8}	1×10^{-4}	4×10^{-6}	2×10^{-8}	1×10^{-7}	9×10^{-7}	1×10^{-7}	1×10^{-7}	4×10^{-7}
n-Nitroso-di-n-propylamine	8×10^{-3}	4×10^{-4}	1×10^{-3}	2×10^{-4}	4×10^{-2} [d]	2×10^{-3}	5×10^{-5}	3×10^{-4}	2×10^{-4}	3×10^{-4}	3×10^{-4}	1×10^{-3}
n-Nitrosodimethylamine[c]	5×10^{-2} [d]	3×10^{-3}	9×10^{-3}	1×10^{-3}	2×10^{-1} [d]	1×10^{-2} [d]	2×10^{-4}	2×10^{-3}	nd	2×10^{-3}	2×10^{-3}	8×10^{-3}
n-Nitrosodiphenylamine	4×10^{-6}	6×10^{-8}	8×10^{-7}	3×10^{-8}	4×10^{-5}	2×10^{-6}	7×10^{-9}	4×10^{-8}	3×10^{-7}	4×10^{-8}	4×10^{-8}	1×10^{-7}
Aldrin	8×10^{-6}	5×10^{-7}	2×10^{-6}	2×10^{-7}	5×10^{-5}	3×10^{-6}	8×10^{-8}	3×10^{-7}	3×10^{-7}	3×10^{-7}	4×10^{-7}	3×10^{-7}
Toxaphene	3×10^{-4}	2×10^{-5}	6×10^{-5}	9×10^{-6}	2×10^{-3}	9×10^{-5}	3×10^{-6}	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}
Total excess cancer risk	6×10^{-2}	3×10^{-3}	1×10^{-2}	1×10^{-3}	3×10^{-1}	1×10^{-2}	3×10^{-4}	2×10^{-3}	3×10^{-4}	2×10^{-3}	2×10^{-3}	9×10^{-3}

[a] The adult one-meal-per-month scenarios are presented for informational purposes only, and are not used by EPA for risk management decisions.

[b] Only mussel data were available for 1,2-diphenylhydrazine. When the risk estimated was calculated, 100% of the ingestion rate was assigned to mussels.

[c] No clam data were available for n-nitrosodimethylamine. When the risk estimates were calculated, the portion of seafood consumption that had been assigned to clams was distributed among the other shellfish consumption categories (crab, geoduck, and mussels).

[d] Because the excess cancer risk was greater than 1×10^{-2}, risk was calculated using the exponential equation in EPA (1989) rather than the linear equation used to calculate the majority of the excess cancer risks.

API – Asian and Pacific Islander
BEHP – bis(2-ethylhexyl) phthalate
CT – central tendency
EPA – US Environmental Protection Agency
nd – no data
RME – reasonable maximum exposure

The highest hypothetical excess cancer risk estimates were for n-nitrosodimethylamine (2×10^{-1} for the adult tribal scenario based on Suquamish data and 5×10^{-2} for the adult tribal RME scenario based on Tulalip data). The total hypothetical excess cancer risk estimates for these COPCs that were never detected were approximately two orders of magnitude higher than the excess cancer risk estimates for detected tissue COPCs. However, the quantifiable presence or absence of non-detected chemicals in tissue is unknown. Consequently, it would not be appropriate to sum excess cancer risk estimates from detected COPCs with hypothetical excess cancer risk estimates from non-detected COPCs.

Two chemicals were never detected in EW tissue samples but had hypothetical non-cancer HQs greater than 1 for the adult tribal RME seafood consumption scenario based on Tulalip data (Table B.6-38). Non-cancer HQs were greater than 1 for at least one non-detected COPC for each scenario. HQs for n-nitrosodimethylamine were the highest (ranging from 5 to 471) for all scenarios, with the exception of the clam one-meal-per-month scenario for which no n-nitrosodimethylamine data were available.

Table B.6-38. Summary of hypothetical non-cancer hazard estimates for seafood consumption scenarios for COPCs that were never detected in EW tissue samples

Non-Detected COPC	Hypothetical Non-Cancer Hazard by Seafood Consumption Scenario											
								Adult One Meal per Month[a]				
	Adult Tribal RME (Tulalip Data)	Adult Tribal CT (Tulalip Data)	Child Tribal RME (Tulalip Data)	Child Tribal CT (Tulalip Data)	Adult Tribal (Suquamish Data)	Adult API RME	Adult API CT	Benthic Fish	Clam	Crab	Pelagic Fish, Rockfish	Pelagic Fish, Perch
BEHP	0.005	0.0003	0.01	0.001	0.04	0.004	0.0002	0.0003	0.001	0.0001	0.0004	0.001
Butyl benzyl phthalate	0.001	0.0002	0.003	0.0003	0.01	0.001	0.0001	0.0001	0.0001	0.0001	0.001	0.0003
1,2,4-Trichlorobenzene	0.02	0.003	0.05	0.006	0.1	0.01	0.001	0.002	0.002	0.002	0.002	0.007
2,4,6-Trichlorophenol	1	0.1	2	0.3	5	0.6	0.06	0.09	0.08	0.09	0.09	0.4
2,4-Dichlorophenol	0.4	0.05	0.8	0.1	2	0.2	0.02	0.03	0.03	0.03	0.03	0.1
2,4-Dinitrophenol	1	0.1	2	0.3	5	0.6	0.06	0.09	0.08	0.09	0.09	0.3
2,4-Dinitrotoluene	0.5	0.07	1	0.2	3	0.3	0.03	0.04	0.04	0.04	0.04	0.2
2,6-Dinitrotoluene	1	0.1	2	0.3	5	0.6	0.06	0.09	0.08	0.09	0.09	0.4
2-Nitroaniline	0.1	0.02	0.2	0.03	0.5	0.06	0.006	0.009	0.008	0.009	0.009	0.04
4,6-Dinitro-o-cresol	27	4	57	8	132	15	1	2	2	2	2	9
4-Chloroaniline	0.3	0.04	0.6	0.08	1	0.2	0.02	0.02	0.02	0.02	0.02	0.09
4-Nitroaniline	0.3	0.04	0.6	0.08	1	0.2	0.01	0.02	0.02	0.02	0.02	0.09
Aniline	0.1	0.004	0.3	0.009	1	0.1	0.002	0.002	0.02	0.002	0.002	0.01
Bis(2-chloroethoxy)methane	0.3	0.01	0.6	0.02	3	0.2	0.004	0.006	0.05	0.006	0.006	0.02
Hexachlorobenzene	0.004	0.001	0.009	0.001	0.03	0.003	0.0003	0.0003	0.0004	0.0003	0.0003	0.0003
Hexachlorobutadiene	0.7	0.01	2	0.02	8	0.7	0.007	0.0003	0.2	0.0003	0.001	0.0003
Hexachlorocyclopentadiene	0.2	0.02	0.4	0.05	0.9	0.1	0.009	0.02	0.01	0.02	0.02	0.06
Hexachloroethane	0.9	0.03	2	0.06	8	0.7	0.01	0.02	0.2	0.02	0.02	0.07
n-Nitrosodimethylamine[b]	116	14	249	31	471	58	5	11	nd	11	11	44
Nitrobenzene	0.4	0.02	0.9	0.03	4	0.4	0.006	0.009	0.08	0.009	0.009	0.03
Aldrin	0.02	0.003	0.04	0.005	0.1	0.01	0.001	0.002	0.002	0.001	0.002	0.001

[a] The adult one-meal-per-month scenarios are presented for informational purposes only, and are not used by EPA for risk management decisions.

[b] No clam data were available for n-nitrosodimethylamine. When the risk estimates were calculated, the portion of seafood consumption that had been assigned to clams was distributed among the other shellfish consumption categories (crab, geoduck, and mussels).

API – Asian and Pacific Islander
BEHP – bis(2-ethylhexyl) phthalate
CT – central tendency
EPA – US Environmental Protection Agency
nd – no data
HQ – hazard quotient
RME – reasonable maximum exposure

Hypothetical excess cancer risk and non-cancer hazard estimates for COPCs that were never detected in EW sediments or surface water are summarized in Tables B.6-39 and B.6-40, respectively. No non-detected COPCs were identified for the netfishing or habitat restoration worker scenarios. Total excess cancer risks were less than or equal to 2×10^{-6} for the clamming scenarios, and HQs were less than or equal to 0.4. Total hypothetical excess cancer risk estimates and HQs for non-detected chemicals were lower than total excess cancer risk estimates for detected chemicals (Section B.5.3.2). For the swimming scenario at each exposure level (i.e., high, medium, and low), total hypothetical excess cancer risks and the sum of HQs for the non-detected COPCs were higher than those for detected chemicals (Section B.5.3.3)

Table B.6-39. Summary of hypothetical excess cancer risks and non-cancer hazards for direct sediment exposure scenarios for COPCs that were never detected in EW sediment samples

Non-Detected COPC	Excess Cancer Risk or Non-Cancer Hazard		
	Tribal Clamming RME	Tribal Clamming – 183 Days per Year	Clamming – 7 Days per Year
Excess Cancer Risk			
Toxaphene	4×10^{-7}	7×10^{-7}	2×10^{-8}
n-Nitrosodimethylamine	6×10^{-7}	1×10^{-6}	2×10^{-8}
Total excess cancer risk	**1×10^{-6}**	**2×10^{-6}**	**4×10^{-8}**
Non-Cancer Hazard			
Antimony	0.2	0.4	0.02
n-Nitrosodimethylamine	0.002	0.003	0.0001

Note: No chemicals were identified as non-detected COPCs for the netfishing scenarios or the habitat restoration worker scenarios.

COPC – chemical of potential concern

EW – East Waterway

HQ – hazard quotient

RME – reasonable maximum exposure

Table B.6-40. Summary of hypothetical excess cancer and non-cancer hazards for surface water exposure scenarios for COPCs that were never detected in EW surface water samples

COPC	Excess Cancer Risk or Non-Cancer Hazard		
	Swimming – High Exposure	Swimming – Medium Exposure	Swimming – Low Exposure
Excess Cancer Risk			
Benzo(a)pyrene	1×10^{-4}	3×10^{-5}	3×10^{-7}
2,4-Dinitrotoluene	3×10^{-7}	3×10^{-8}	3×10^{-10}
3,3'-Dichlorobenzidine	9×10^{-7}	2×10^{-7}	2×10^{-9}
Bis(2-chloroethyl)ether	1×10^{-7}	2×10^{-8}	1×10^{-10}
Hexachlorobenzene	6×10^{-6}	1×10^{-6}	1×10^{-8}
n-Nitroso-di-n-propylamine	5×10^{-6}	6×10^{-7}	6×10^{-9}
n-Nitrosodimethylamine	2×10^{-5}	2×10^{-6}	9×10^{-9}
Total excess cancer risk	1×10^{-4}	3×10^{-5}	3×10^{-7}
Non-Cancer Hazard			
2,4-Dinitrotoluene	0.0004	0.0001	0.000004
4,6-Dinitro-o-cresol[a]	0.01	0.001	0.00002
4-Chloroaniline[a]	0.0001	0.00001	0.0000002
Hexachlorobenzene	0.004	0.002	0.00008
n-Nitrosodimethylamine	0.06	0.009	0.0002

a No absorption factor was available for this chemical, and thus the HQ presented here is based only on incidental ingestion. See Section B.6.1.5.1 for a discussion of this uncertainty.

COPC – chemical of potential concern

EW – East Waterway

HQ – hazard quotient

Hypothetical non-cancer HQs for the COPCs that were never detected in sediment or surface water were low (0.4 or lower) for all assessed direct sediment exposure scenarios (Table B.6-39) and surface water exposure scenarios (Table B.6-40). Non-cancer HIs for all direct sediment exposure and surface water exposure scenarios, based on both hypothetical hazard estimates for non-detected chemicals (Table B.6-39 and B.6-38) and hazard estimates for detected chemicals (Tables B.5-26, B.5-27, B.5-29, B.5-33 to B.5-35, and B.5-38), were all below 1. The hypothetical non-cancer HIs for non-detected chemicals were many times lower than those based on detected chemicals and indicate that these non-detected chemicals would not pose non-cancer health hazards in the EW at the RLs used in this assessment.

The sample-specific RL is based on the lowest point of the calibration curve associated with each analytical batch of samples. The most common reason for elevated RL values is sample extract dilution. For example, elevated RLs for some chemicals in some areas reflect the greater degree of analytical dilution required for the quantification of other analytes, such as PCBs. In addition, there is a group of analytes that are known to be analytically difficult to quantify. These compounds tend to have chemical characteristics that differ from those of other analytes that are analyzed using the same method. For example, phenols, and n-nitrosodiphenylamine are all more chemically reactive than the other SVOCs analyzed by EPA (EPA 2003a). More reactive compounds can be difficult to extract and often degrade during analysis. The group of analytically difficult compounds included the following chemicals: chlorobenzenes, phenol, methyl phenols, pentachlorophenol, benzoic acid, benzyl alcohol, hexachlorobutadiene, hexachlorobenzene, and n-nitrosodiphenylamine. These compounds are analytically difficult to quantify at the concentrations required for comparison with risk-based analytical concentration goals and are very rarely detected.

Although efforts to identify current and historical sources of contamination to the EW are not yet complete, it does not appear that the major industries that used these non-detected chemicals are or have been present along the EW area. Common manufacturing uses for several of these non-detected chemicals include:

- Dinitrotoluenes may enter the environment during the production of polyurethane foams used by building furniture and bedding manufacturers.
- Small amounts of n-nitroso-di-n-propylamine may be released into the environment during manufacturing processes associated with the production of weed killers or rubber.
- n-Nitrosodimethylamine can be released during the manufacture of pesticides, rubber tires, alkylamines, and dyes and also may form under natural conditions in air, water, and soil as a result of chemical, photochemical, and biological processes.

It is possible that these products were used or handled by industries along the EW, but research to date has not identified any reason to believe that there are concentrations of these non-detected chemicals approaching the RLs in the EW. For n-nitrosodimethylamine, no additional analytical techniques are available to achieve lower RLs for this chemical. The lack of a known or suspected source is important and puts the uncertainty associated with the relatively high risks estimates for n-nitrosodimethylamine into perspective (Tables B.6-37 and B.6-38).

In addition, it is useful to consider whether there may be a source of these chemicals upstream of the EW (e.g., in the LDW). The following bullets summarize to what extent non-detected COPCs for the EW were detected in the applicable LDW media:

Tissue – Of the 26 non-detected tissue COPCs, 5 were detected infrequently in LDW tissue samples (i.e., BEHP, butyl benzyl phthalate, bis(2-chloroethoxy)methane, hexachlorobenzene, and aldrin). Detection frequencies for these five chemicals were 15% or lower in LDW tissue samples.

Sediment – Of the three non-detected sediment COPCs, both toxaphene and antimony were detected in LDW sediment. In addition, it should be noted that antimony was detected in EW subtidal sediment (antimony was a detected COPC for the netfishing scenario but a non-detected COPC for the clamming scenarios).

Surface water – None of the nine non-detected COPCs were detected in samples collected from the LDW as part of King County's WQA (King County 1999c).

B.6.3.3 Calculation of combined risks for adults and children

As was done in the LDW HHRA (Windward 2007c), risks were calculated two ways for the tribal seafood consumption scenario based on Tulalip data: a 6-year exposure duration for children (aged 0 to 6 years) and a 70-year exposure duration for adults (i.e., risks were calculated using adult exposure parameters to estimate exposure assuming 70 adult years). However, it is possible that risks could be higher if a lifetime exposure period were used (i.e., risks were calculated based on exposure for one individual from age 0 to 70 years, thus including the most sensitive developmental life stages, rather than based on 70 years adult-only exposure). This is particularly important for chemicals with mutagenic modes of action, such as cPAHs, for which EPA guidance (2005d) directs that the slope factor for children aged 0 to 16 should be adjusted upward in risk calculations to account for the higher susceptibility of children to these chemicals, as described in Section B.4.2 and B.5.1.1.

To evaluate the impact of calculating risks for the lifetime exposure period (aged 0 to 70 years), risk were calculated for the following three age groups and then summed to calculate the lifetime excess cancer risk. A brief description of the exposure parameters used for these three age groups is provided below, and details regarding the risk calculations are provided in Attachment 6:

- **Young children (aged 0 to 6) –** Parameters for the child tribal RME scenario based on Tulalip data were used to calculate risks for this portion of the lifetime risk scenario.
- **Older children (aged 7 to 16) –** Parameters for the adult and child tribal RME scenarios based on Tulalip data were averaged and used to calculate risks for this portion of the lifetime risk scenario. Note that only ingestion rates, exposure duration, and body weight were different for the adult and child scenarios.[57]
- **Adults (aged 17 to 70) –** Parameters for the adult tribal RME scenario based on Tulalip data were used to calculate risks for this portion of the lifetime risk scenario.

The excess cancer risks for these three groups are presented in Table B.6-41, along with the total lifetime risk estimate, for three example COCs for which risk estimates were highest in Section B.5. For comparison purposes, this table also presents the adult and child tribal RME seafood consumption scenario excess cancer risk estimates from Section B.5. It should be noted that non-cancer HQs are not presented on a lifetime exposure basis (i.e., averaged over a 70-year exposure duration) because HQs are highest for the most sensitive life stage (i.e., for young children).

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[57] For calculating risks for the older child portion of the lifetime risk estimate, a body weight of 48.5 kg and a total ingestion rate of 68.2 g/day were used.

Table B.6-41. Excess cancer risk estimates for select COCs calculated based on lifetime exposure

Select COCs	Lifetime Risks for the Tribal RME Scenario based on Tulalip Data				Tribal RME Scenario based on Tulalip Data (Section B.5)	
	0 to 6 Years	7 to 16 Years	17 to 70 Years	Total[a]	Adult	Child
Arsenic	4×10^{-5}	3×10^{-5}	2×10^{-4}	2×10^{-4}	2×10^{-4}	4×10^{-5}
cPAH TEQ	1×10^{-4}	5×10^{-5}	8×10^{-5}	2×10^{-4}	1×10^{-4}	1×10^{-4}
Total PCBs	2×10^{-4}	2×10^{-4}	8×10^{-4}	1×10^{-3}	1×10^{-3}	2×10^{-4}

[a] Intermediate excess cancer risks (i.e., those for the three different age groups) were not rounded. Thus, the total may not appear to match the sum of the three intermediate excess cancer risks because of additional significant figures, which are not shown here. Details regarding these calculations are presented in Attachment 6.

COC – chemical of concern

cPAH – carcinogenic polycyclic aromatic hydrocarbon

HHRA – human health risk assessment

PCB – polychlorinated biphenyl

RME – reasonable maximum exposure

TEQ – toxic equivalent

As can be seen in Table B.6-41, the total lifetime excess cancer risks are similar to those for the adult tribal RME seafood consumption scenario based on Tulalip data. The risk estimates for arsenic and total PCBs are equal, while the total lifetime risk estimate for cPAH TEQ is slightly higher than the adult tribal RME scenario risk estimate (2×10^{-4} as compared with 1×10^{-4}), primarily because of the age-specific slope factor adjustments that are done for chemicals with mutagenic modes of action (Section B.5.1.1). Risk estimates for the child tribal RME scenario based on Tulalip data are lower than the total lifetime risks for all three example chemicals. Thus, as shown by the example calculations presented in Table B.6-41, the total excess cancer risk for some chemicals may be slightly underestimated by calculating the adult and child risks separately (e.g., for chemicals with age-specific adjustment factors, such as cPAH TEQ), but the potential impact on the overall risk estimates is low.

B.7 Identification of Risk Drivers

This section presents the rationale for the identification of chemicals as "risk drivers" (EPA 1999b) based on estimated human health risks in the EW. The risk drivers from both this HHRA and the ERA (Appendix A) will be the focus of remedial analyses in the FS. Chemicals considered to be risk drivers are a subset of the COCs.[58] COCs were identified for the seafood consumption and direct sediment exposure RME scenarios (Table B.5-46). It should be noted that no RME was defined for the swimming scenario, and thus no COCs or risk drivers were identified for that scenario.[59] The following criteria were used for identifying risk drivers: the relative percentage that the COC contributes to the total human health risk, the absolute magnitude of the risk associated with the COC (including a consideration of background concentrations, if applicable), the frequency of detection of the COC, and the level of uncertainty in the risk estimate. These criteria are consistent with those used to identify risk drivers in the LDW HHRA (Windward 2007c).

All COCs will be mapped and discussed in the RI (although the RI will provide greater detail for the risk drivers). In the FS, the development and evaluation of alternatives will focus on the risk drivers, inasmuch as these chemicals account for over 80% of the total risk. COCs not selected as risk drivers in the EW HHRA will be evaluated using the same approach as the LDW RI/FS. This evaluation will include a follow-up check for the non-risk-driver COCs to ensure that sediment with elevated levels of these COCs will be included in the remedial footprint of the remedial alternatives evaluated in the FS. Furthermore, all COCs (risk driver and non-risk driver) will be included in the long-term monitoring plan for the EW.

B.7.1 Seafood Consumption Scenarios

Table B.7-1 summarizes the rationale for the selection of risk drivers for the seafood consumption scenarios based primarily on risk magnitude, percent contribution of total risk, and detection frequency in EW seafood tissue. Additional discussion is provided in the following subsections for each chemical or group of chemicals. Based on the analysis presented in this section, three COCs (total PCBs, cPAH TEQ, and dioxin/furan TEQ) were identified as risk drivers based on the three RME seafood consumption scenarios (adult and child tribal seafood consumption rates based on Tulalip data and adult API).

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[58] A COC is a chemical that has an excess cancer risk greater than 1×10^{-6} and/or a non-cancer HQ greater than 1 for one or more RME scenarios. Table B.5-46 presents the COCs by scenario, and Tables B.5-47 through B.5-49 present a summary of risk estimates for all scenarios.

[59] For the swimming scenario, one COPC (PCB TEQ) had a risk greater than 10^{-6} for the medium level of exposure (1 hour/event, 12 days/year, 30 years) and high level of exposure (2.6 hours/event, 24 days/year, 70 years). However, because of the low water temperatures and high level of boat traffic in the EW (Section B.5.3.3), a swimming scenario is not considered to be realistic for either current or future use in an industrial waterway such as the EW. Thus, no RME level of swimming exposure was defined, and PCB TEQ was not selected as a COC for the swimming scenario.

Table B.7-1. Summary of criteria for the identification of COCs as risk drivers for the RME seafood consumption scenarios

COC	Excess Cancer Risk or HQ (Percent Contribution to the Total Excess Cancer Risk)[a]			Detection Frequency in EW Seafood Tissue
	Adult Tribal RME (Tulalip Data)	Child Tribal RME (Tulalip Data)	Adult API RME	
Risk Drivers				
cPAH TEQ	1×10^{-4} (7%)	1×10^{-4} (27%)	5×10^{-5} (9%)	71%
Total PCBs[b]	1×10^{-3} (70%) HQ = 27	2×10^{-4} (55%) HQ = 58	4×10^{-4} (69%) HQ = 24	98%
Dioxin/furan TEQ	1×10^{-4} (7%)	2×10^{-5} (5%)	4×10^{-5} (7%)	100%
Other COCs				
Metals				
Arsenic	2×10^{-4} (14%)	4×10^{-5} (11%)	8×10^{-5} (14%)	88%
Cadmium	HQ = 0.7	HQ = 2	HQ = 0.4	58%
SVOCs				
Pentachlorophenol	2×10^{-6} (< 1%)	4×10^{-7} (< 1%)	3×10^{-7} (< 1%)	4%
Pesticides				
alpha-BHC	4×10^{-6} (< 1%)	7×10^{-7} (< 1%)	9×10^{-7} (< 1%)	17%
Dieldrin	8×10^{-6} (< 1%)	1×10^{-6} (< 1%)	2×10^{-6} (< 1%)	48%
Total chlordane	2×10^{-6} (< 1%)	3×10^{-7} (< 1%)	7×10^{-7} (< 1%)	70%
Heptachlor epoxide	2×10^{-6} (< 1%)	4×10^{-7} (< 1%)	7×10^{-7} (< 1%)	9%
Mirex	4×10^{-6} (< 1%)	8×10^{-7} (< 1%)	1×10^{-6} (< 1%)	43%

[a] The percent contribution to the total is provided only for excess cancer risks and is based on the sum excluding PCB TEQ (total PCBs included).

[b] Risk estimates for total PCBs are greater than those for PCB TEQ, and thus risks for total PCBs are presented here. The risk estimate for PCB TEQ ranged from 1×10^{-4} to 7×10^{-4} for the three RME scenarios.

BHC – benzene hexachloride
COC – chemical of concern
cPAH – carcinogenic polycyclic aromatic hydrocarbon
EW – East Waterway
HQ – hazard quotient
na – not applicable
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
SVOC – semivolatile organic compound
TEQ – toxic equivalent

B.7.1.1 cPAH TEQ, PCBs, and dioxin/furan TEQ

Risk estimates for cPAH TEQ, PCBs, and dioxin/furan TEQ together contributed 84% or more to the total excess cancer risk for the three RME seafood consumption scenarios.

Risk estimates for PCBs[60] were greater than 1×10^{-4} (the upper end of EPA's acceptable risk range) for one or more of the RME seafood consumption scenarios and were thus

[60] Risks for both total PCBs and PCB TEQ were greater than 1×10^{-4} for one or more of the RME seafood consumption scenarios. However, risks for total PCBs were higher for all scenarios, and thus these risks are discussed here.

above EPA's acceptable risk range of 10^{-4} to 10^{-6}. PCBs contributed 55% or more to the total excess cancer risk for the RME scenarios, and thus were selected as a risk driver.

Excess cancer risks for cPAH TEQ were within EPA's acceptable risk range, ranging from 5×10^{-5} to 1×10^{-4} for the three RME seafood consumption scenarios. For cPAH TEQ, the percent contribution to the total excess cancer risk ranged from 7 to 27%, with the high end of this range reflecting the higher toxicity of chemicals with mutagenic modes of action to children (Table B.7-1). Because of the magnitude of the excess cancer risk, the percent contribution of total risk (especially for children), and the relatively high detection frequency (71%), cPAHs were retained as a risk driver.

For dioxin/furan TEQ, excess cancer risks were also within EPA's acceptable risk range, ranging from 2×10^{-5} to 1×10^{-4} for dioxin/furan TEQ for the three RME seafood consumption scenarios. The percent contribution to the total excess cancer risk for dioxin/furan TEQ (5 to 7%) was lower than that for total PCBs or cPAH TEQ. However, because of the magnitude of the excess cancer risk, the high detection frequency (100%), and the fact that the total TEQ excess cancer risk (i.e., the sum of PCB TEQ and dioxin/furan TEQ risks) was greater than or equal to 1×10^{-4} for all three RME scenarios, dioxin/furan TEQ was also selected as a risk driver.

B.7.1.2 Arsenic

Excess cancer risks for arsenic ranged from 4×10^{-5} to 2×10^{-4} for the three RME seafood consumption scenarios and contributed 11 to 14% of the total excess cancer risk. However, the evaluation of background concentrations presented in Section B.5.5.1.2 indicated that EW concentrations were similar to or lower than those in samples collected from background areas in Puget Sound. Incremental risk estimates (i.e., background risks subtracted from site risks) were equal to or less than 1×10^{-6}. Thus, arsenic was not designated as a risk driver based on the seafood consumption scenarios.

B.7.1.3 Cadmium

Cadmium had an HQ equal to 2 for the child tribal RME scenario based on Tulalip data (HQs were less than 1 for the other two RME seafood consumption scenarios). Because of the considerable uncertainty associated with this scenario (as discussed in Section B.6.1.2.3) and because the HQ was only slightly greater than 1 (and more than an order of magnitude less than the HQ for total PCBs), cadmium was not designated as a risk driver.

B.7.1.4 Pentachlorophenol

Pentachlorophenol had an excess cancer risk estimate greater than 1×10^{-6} for only one of the three RME scenarios; the adult tribal seafood consumption RME scenario based on Tulalip data had an excess cancer risk of 2×10^{-6}. In addition, pentachlorophenol was a minor contributor to the total excess cancer risk (contributing less than 1%) and was detected infrequently in EW tissue samples (4% across tissue types). For these reasons, pentachlorophenol was not identified as a risk driver.

B.7.1.5 Pesticides

Five pesticides (alpha-BHC, dieldrin, total chlordane, heptachlor epoxide, and mirex) had excess cancer risks greater than 1×10^{-6} for at least one of the three RME seafood consumption scenarios, although risks for all five of these chemicals were less than 1×10^{-5} (Table B.7-1). Each of the pesticides was a minor contributor to total excess cancer risks, contributing less than 1% (with a contribution of less than 1.5% for the five chemicals combined). Therefore, none of these pesticides were selected as risk drivers.

B.7.2 Direct Sediment Exposure Scenarios

Table B.7-2 summarizes the rationale for the selection of risk drivers for the direct sediment exposure RME scenarios (netfishing RME and tribal clamming RME). There were no COCs based on non-cancer hazards so only COCs based on excess cancer risks are discussed here for the selection of risk drivers.

Table B.7-2. Summary of criteria for the identification of COCs as risk drivers for the RME direct sediment exposure scenarios

COC	Excess Cancer Risk (Percent Contribution to the Total)[a]		Detection Frequency in Surface Sediment Samples[b]
	Netfishing RME	Tribal Clamming RME	
Risk Drivers			
Arsenic	3×10^{-6} (42%)	1×10^{-5} (29%)	70%
cPAH TEQ	3×10^{-6} (42%)	2×10^{-5} (59%)	97%
Other COCs			
Total PCBs[c]	6×10^{-7} (8%)	3×10^{-6} (9%)	94%
Total TEQ (sum of PCB TEQ and dioxin/furan TEQ)[c, d]	9×10^{-7} (13%)	2×10^{-6} (6%)	100%

a The percent contribution to the total excess cancer risk is based on the total risk excluding PCB TEQ (but including total PCBs), except for total TEQ, for which the percent contribution is based on the total risk excluding total PCBs (but including PCB TEQ).

b Detection frequency is for all EW surface sediment data (subtidal and intertidal), not just the data used for a particular scenario.

c Total PCBs and total TEQ were not COCs for the netfishing RME scenario (i.e., excess cancer risks were not greater than 1×10^{-6}).

d Independently, excess cancer risks for either PCB TEQ or dioxin/furan TEQ were not greater than 1×10^{-6}.

COC – chemical of concern
cPAH – carcinogenic polycyclic aromatic hydrocarbon
EW – East Waterway
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent

As shown in Table B.7-2, two COCs (arsenic and cPAH TEQ) were identified as risk drivers for the direct sediment exposure RME scenarios. Arsenic and cPAH TEQ were the greatest contributors to the total excess cancer risk and both have risks equal to or greater than 1×10^{-5} for the tribal clamming RME scenario (risks were lower but still greater than 1×10^{-6} for the netfishing RME scenario). Total PCBs were identified as a

COC for the tribal clamming RME scenario (but not the netfishing RME scenario), but because of the lower magnitude of total PCB risks (compared with arsenic and cPAH TEQ) and the relatively low percentage of contribution to the total excess cancer risk (9%), total PCBs were not identified as a risk driver for the direct sediment exposure RME scenarios. However, as noted in Section B.7.1, PCBs were identified as a risk driver for the seafood consumption scenarios and thus will be a focus of remedial activities in the EW. Additionally, residual risks from PCBs for direct sediment exposure scenarios will be evaluated in the FS.

Dioxin/furan TEQ, although identified as a risk driver for the seafood consumption scenarios, was not identified as a COC independently for either of the direct sediment exposure RME scenarios because excess cancer risks did not exceed the 1×10^{-6} threshold. However, when summed with PCB TEQ, the total TEQ excess cancer risk estimate was equal to 2×10^{-6} for the tribal clamming RME scenario, and thus total TEQ was identified as a COC for this scenario. Based on the relatively low excess cancer risk and percent contribution to the total risk, total TEQ was not identified as a risk driver. However, as noted in Section B.7.1, dioxins/furans were identified as a risk driver for the seafood consumption scenarios and thus will be a focus of remedial activities in the EW.

B.8 Conclusions

The baseline HHRA presents risk estimates for various scenarios whereby people (e.g., tribal members or members of the general public) could be exposed to COPCs found in fish and shellfish tissue, sediment, and surface water in the EW. This HHRA will also be used to support risk management decisions and the evaluation of remedial options related to the EW. In addition, this HHRA serves to inform the public of the health risks that could result from engaging in different activities associated with the EW (e.g., consumption of EW seafood, netfishing, habitat restoration, clamming, and swimming). A variety of different exposure scenarios were evaluated to provide a range of risk estimates. Individuals may evaluate their own risks by comparing their behavior with the assumptions included in each of the exposure scenarios. A summary of the risk estimates and uncertainties associated with these estimates is provided in Section B.5.6 and Table B.6-1, respectively.

Risks were greatest from the seafood consumption pathway as compared with the sediment or surface water exposure pathways. The total excess cancer risk was 1×10^{-3} for the adult tribal RME seafood consumption scenario based on Tulalip data (which assumed a consumption rate of 97.5 g/day of resident species of fish and shellfish [i.e., not including salmon], or approximately 13 meals per month, for 70 years). For the other two RME seafood consumption scenarios, risks were 4×10^{-4} for the child tribal RME scenario based on Tulalip data and 6×10^{-4} for the adult API RME scenario. The majority of the estimated excess cancer risks for the seafood consumption scenarios were attributable to three COCs that were designated as risk drivers (total PCBs, cPAH TEQ, and dioxin/furan TEQ), as summarized in Section B.7. These risk driver chemicals combined account for 84% or more of the excess cancer risk for the RME seafood consumption scenarios. Other COCs identified for these RME scenarios based on their excess cancer risks included arsenic, pentachlorophenol, alpha-BHC, dieldrin, total chlordane, heptachlor epoxide, and mirex. Based on non-cancer hazards, total PCBs was identified as a COC (HQs for all three RME seafood consumption scenarios were greater than 1), as was cadmium (the HQ for the child tribal RME seafood consumption scenario based on Tulalip data was greater than 1). Hazard indices for the immunological endpoint (24 to 59), neurological endpoint (25 to 59), integumentary endpoint (25 to 59), and developmental endpoint (7 to 18) were greater than 1 for all three RME scenarios. In addition, the hazard index for the kidney endpoint was greater than 1 (equal to 2) for the child tribal RME seafood consumption scenario based on Tulalip data.

All non-RME scenarios had total excess cancer risks greater than 1×10^{-6} (ranging from 1×10^{-5} for the adult API CT scenario to 1×10^{-2} for the adult tribal scenario based on Suquamish data). Non-cancer HQs for total PCBs were greater than 1 for all scenarios except the adult API CT scenario and the crab and clam adult one-meal-per-month scenarios,[61] for which no COPCs had HQs greater than 1. In addition, HQs were greater than 1 for arsenic (HQ of 4), cadmium (HQ of 2), cobalt (HQ of 4), mercury (HQ of 3), and TBT (HQ of 4) for the adult tribal scenario based on Suquamish data (Table B.5-48). For this scenario, the hazard indices based on the hematological, immunological, kidney, neurological, endocrine, integumentary, digestive system, and developmental endpoints were all greater than 1 (HIs ranged from 2 to 218).

The total excess cancer risks for the two direct sediment exposure RME scenarios were 7×10^{-6} for the netfishing RME scenario and 3×10^{-5} for the tribal clamming RME scenario. The majority of the total excess cancer risk (over 80%) for the direct sediment exposure RME scenarios was attributable to the two COCs designated as risk drivers (arsenic and cPAH TEQ). Other COCs included total PCBs and total TEQ (the sum of PCB TEQ and dioxin/furan TEQ) for the tribal clamming RME scenario.[62] No chemicals had non-cancer HQs greater than 1 for either of the two direct sediment exposure RME scenarios, and therefore non-cancer hazards are not expected. For the non-RME scenarios, total excess cancer risks were greater than 1×10^{-6} for the tribal clamming 183-days-per-year scenario (equal to 6×10^{-5}) but less than or equal to 1×10^{-6} for the netfishing CT scenario, habitat restoration worker scenario, and the 7-day-per-year clamming scenario. No non-cancer HQs were greater than 1 for any of the non-RME scenarios.

Risks from exposure to surface water while swimming were calculated for three levels of exposure (high, medium, and low),[63] although none of these exposure levels was defined as an RME, as discussed in Section B.5.3.3. Only one COPC had an excess cancer risk greater than the 1×10^{-6} risk threshold: PCB TEQ at the high and medium levels of exposure had excess cancer risk estimate of 9×10^{-6} and 2×10^{-6}, respectively. The PCB TEQ risk did not exceed the risk threshold at the low level of exposure. As discussed in Section B.5.3.3, the PCB TEQ risk estimate is considered highly uncertain based on both on current or anticipated future site use and on the uncertainty associated with the application of the dioxin-like TEQ approach for dermal exposure (Section B.6.2.3), which contributed nearly all (over 99%) of PCB TEQ swimming risk (as compared with the incidental ingestion of water). No other COPCs (including total PCBs) had excess cancer risks greater than 1×10^{-6}, and no non-cancer HQs were greater than 1 for any COPC-exposure level combination.

[61] The adult one-meal-per-month scenarios are presented for informational purposes only, and are not used by EPA for risk management decisions.

[62] Individually, PCB TEQ and dioxin/furan TEQ did not exceed the excess cancer risk threshold of 1×10^{-6}. However, when summed as total TEQ, the excess cancer risk was equal to 2×10^{-6}, and therefore, total TEQ was designated as a COC for the tribal clamming RME scenario.

[63] The three levels of exposure evaluated for swimming were high (which assumed a 2.6-hour swim, 24 days per year for 70 years), medium (which assumed a 1-hour swim, 12 days per year for 30 years), and low (which assumed a 10-minute swim, 2 days per year for 9 years).

Risks attributable to the other chemicals found in the EW are considerably lower than the risks attributable to the risk drivers because the concentrations of other chemicals are relatively low and/or because the other chemicals are not particularly toxic. For all COCs not designated as risk-drivers,[64] risk estimates for the RME scenarios were less than the upper end of EPA's acceptable risk range (1×10^{-4}). As noted above, the total excess cancer risk from exposure to either sediment or surface water was much lower than the total risk associated with seafood consumption. Total risks from exposure to all chemicals in sediment and surface water are within or less than EPA's acceptable risk range of 10^{-6} to 10^{-4}, whereas risks from seafood consumption are higher than the risk range. However, there is considerable uncertainty about the site-specific applicability of some of the seafood consumption rates used in this HHRA, particularly for clams, given the low quality and small quantity of shellfish habitat in the EW. However, it is also important to consider that the EW is within the larger Elliott Bay and Duwamish River area, and that direct contact exposure to contaminants throughout this area is of concern.

Table B.8-1 presents the chemicals selected as risk drivers for the seafood consumption and sediment exposure scenarios, and provides a summary of the rationale for the selection or exclusion of these chemicals as risk drivers. A summary of risks for each COC, as well as a more detailed discussion of the selection of risk drivers, is presented in the subsections that follow Table B.8-1.

Table B.8-1. Summary of risk driver selection

COC	Selection as Risk Driver and Summary of Rationale	
	Seafood Consumption RME Scenarios	Direct Sediment Exposure RME Scenarios
Arsenic	**NO** – risks greater than the upper end of EPA's acceptable risk range (1×10^{-4}); however, incremental risks equal to or less than 1×10^{-6} because concentrations in seafood are similar to or lower than those in samples collected from background areas	**YES** – risk greater than the 10^{-6} threshold, percent contribution to the total risk (29 to 43%), and high detection frequency (70%)
cPAH TEQ	**YES** – risks equal to the upper end of EPA's acceptable risk range (1×10^{-4}), percent contribution to the total risk (7 to 27%), and high detection frequency (71%)	**YES** – risks greater than the 10^{-6} threshold, percent contribution to the total risk (42 to 63%), and high detection frequency (97%)
Total PCBs	**YES** – risks greater than the upper end of EPA's acceptable risk range (1×10^{-4}), percent contribution to the total risk (55 to 70%), and high detection frequency (98%)	**NO** – risks were slightly greater than the 1×10^{-6} threshold, and had a relatively lower contribution to the total risk (8 to 9%)

[64] As discussed in Section B.7, all COCs (both risk drivers and non-risk drivers) will be mapped and discussed in the RI (although the RI will provide greater detail for the risk drivers). In addition, although the risk drivers will be the focus of the development and evaluation of alternatives in the FS, COCs not selected as risk drivers will be evaluated qualitatively in the FS and will be included in the long-term monitoring plan for the EW.

COC	Selection as Risk Driver and Summary of Rationale	
	Seafood Consumption RME Scenarios	Direct Sediment Exposure RME Scenarios
Dioxin/furan TEQ	**YES** – risks equal to the upper end of EPA's acceptable risk range (1×10^{-4}) and high detection frequency (100%)	**NO** – not a COC
Total TEQ (sum of PCB TEQ and dioxin/furan TEQ)	na[a]	**NO –** risks were slightly greater than the 1×10^{-6} threshold and had a relatively lower contribution to the total risk (6 to 13%)
Cadmium	**NO** – HQ equal to 2 for the child tribal RME scenario based on Tulalip data, but considerable uncertainty is associated with this scenario and HQs for total PCBs were over an order of magnitude higher	**na –** not a COPC
Pentachlorophenol	**NO** – risk slightly greater than the 1×10^{-6} threshold for one of the three RME scenarios; contribution to the total excess cancer risk was less than 1%, and chemical was detected in less than 4% of EW samples	**na –** not a COPC
Pesticides[b]	**NO** – risks less than 1×10^{-5}, and each chemical contributed less than 1% to the total excess cancer risk (combined contribution was less than 1.5% of the total)	**na –** not a COPC

[a] Total TEQ was considered only when neither PCB TEQ nor dioxin/furan TEQ independently qualified as a COC. Note that PCB TEQ and dioxin/furan TEQ will be addressed both together and separately in the FS as part of the evaluation of non-risk driver COCs.

[b] A total of five pesticides were identified as COCs for the seafood consumption scenarios: alpha-BHC, dieldrin, total chlordane, heptachlor epoxide, and mirex.

COC – chemical of concern
COPC – chemical of potential concern
cPAH – carcinogenic polycyclic aromatic hydrocarbon
EW – East Waterway
na – not applicable
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent

B.8.1 ARSENIC

Excess cancer risks for inorganic arsenic associated with the consumption of seafood from the EW were greater than the acceptable risk threshold range for all RME seafood consumption scenarios, and therefore arsenic was identified as a COC. The highest excess cancer risk associated with inorganic arsenic for an RME seafood consumption scenario of 2×10^{-4} was calculated for the adult tribal RME scenario based on Tulalip data. Risks from arsenic were attributable predominantly to the consumption of clams (Section B.5.3.1.4). However, as shown in Section B.5.5.1.2, incremental risks were equal to or less than 1×10^{-6} (i.e., concentrations in tissue samples collected from the EW are similar to or lower than those in samples collected from background areas). Thus, arsenic was not identified as a risk driver for the seafood consumption RME scenarios

Although not identified as a risk driver based on seafood consumption, arsenic was identified as a risk driver for the direct sediment exposure scenarios based on the magnitude of the excess cancer risk (3×10^{-6} to 1×10^{-5}), percent contribution to the total excess cancer risk (29 to 43%), and high detection frequency in EW sediments (70%).

Although much lower than the risks associated with inorganic arsenic for the seafood consumption scenarios (the highest excess cancer risk associated with inorganic arsenic for an RME scenario was 1×10^{-5} for the tribal clamming RME scenario), excess cancer risk estimates were greater than the threshold of 1×10^{-6} for the two direct sediment exposure RME scenarios. When compared with upstream sediment samples (i.e., those collected from upstream of the LDW Superfund site), concentrations in the EW samples were higher, although a portion of the arsenic concentrations in the EW are likely attributable to upstream or background sources that are unrelated to the site, as discussed in Section B.5.5.1.1.

B.8.2 cPAH TEQ

Excess cancer risks for cPAH TEQ from the consumption of EW seafood exceeded the risk threshold of 1×10^{-6} for all RME seafood consumption scenarios and therefore cPAH TEQ was identified as a COC. The highest excess cancer risk estimate of 1×10^{-4} was calculated for the adult and child tribal RME scenarios based on Tulalip data. Most of the risk can be attributed to the consumption of clams (Section B.5.3.1.4). Thus, cPAH TEQ was selected as a risk driver for the seafood consumption RME scenarios based on the excess cancer risk magnitude (5×10^{-5} to 1×10^{-4}), percent contribution to the total excess cancer risk (7 to 27%) and high detection frequency in EW tissue samples (71%).

In addition, cPAH TEQ was identified as a risk driver for the direct sediment exposure RME scenarios based on the magnitude of the excess cancer risk (3×10^{-6} to 2×10^{-5}), percent contribution to the total excess cancer risk (42 to 59%), and high detection frequency in EW sediments (97%). The highest cPAH TEQ excess cancer risk estimate for an RME sediment exposure scenario was 2×10^{-5} for the tribal clamming RME scenario.

B.8.3 PCBs

Excess cancer risks and non-cancer hazards for total PCBs from the consumption of seafood from the EW exceeded the upper end of EPA's acceptable risk range (1×10^{-4}), and HQs were greater than 1 for non-cancer hazards for all RME seafood consumption scenarios. Note that risks for total PCBs are higher than those for PCB TEQ, and thus total PCB risks are discussed here. Of the RME scenarios, the highest excess cancer risk associated with total PCBs of 1×10^{-3} was estimated for the adult tribal RME scenario based on Tulalip data. PCBs were identified as a risk driver for the seafood consumption scenarios, based on the magnitude of the excess cancer risk (2×10^{-4} to 1×10^{-3}), percent contribution to the total excess cancer risk (55 to 70%), and high detection frequency in EW tissue samples (98%). Most of the risk from PCBs can be attributed to the consumption of benthic fish fillets (English sole) and/or pelagic fish (perch or rockfish), depending on the seafood consumption scenario evaluated (Section B.5.3.1.4).

Risks from sediment contact were much lower than risks associated with seafood consumption. Total PCBs, although identified as a COC for sediment contact, was not identified as a risk driver for sediment because of the relatively low magnitude of the excess cancer risk (which was less than 1×10^{-6} for the netfishing RME scenario and only slightly greater than the threshold of 1×10^{-6} for the tribal clamming RME scenario) and because the contribution to the total excess cancer risk for the direct sediment exposure RME scenarios was relatively low (8 to 9%). The total PCB excess cancer risk for the tribal clamming RME scenario was equal to 3×10^{-6}. Note that risks for PCB TEQ for the direct sediment exposure RME scenarios did not exceed 1×10^{-6}.

B.8.4 DIOXIN/FURAN TEQ

Excess cancer risk estimates for dioxin/furan TEQ exceeded the risk threshold of 1×10^{-6} for all RME seafood consumption scenarios. The highest excess cancer risk estimate of 1×10^{-4} was calculated for the adult tribal RME scenarios based on Tulalip data. Hence, dioxin/furan TEQ, a COC for seafood consumption, was identified as a risk driver for the seafood consumption RME scenarios based on the magnitude of the excess cancer risk (2×10^{-5} to 1×10^{-4}) and high detection frequency in EW tissue samples (100%). Risks from dioxins/ furan TEQ were mostly attributable to the consumption of clams, crabs (edible meat and whole body), and pelagic fish (rockfish), depending on the seafood consumption scenario evaluated (Section B.5.3.1.4).

As discussed in Section B.7, dioxin/furan TEQ independently was not a COC for the sediment exposure scenarios because excess cancer risks for the RME scenarios were not greater than 1×10^{-6}. However, when summed with PCB TEQ, the total TEQ excess cancer risk estimate was equal to 2×10^{-6} for the tribal clamming RME scenario, and thus was identified as a COC. Based on the relatively low magnitude of the excess cancer risk (2×10^{-6} for the tribal clamming RME scenario and 9×10^{-7} for the netfishing RME scenario) and low percent contribution to the total excess cancer risk (6 to 13%), total TEQ was not identified as a risk driver.

B.8.5 CADMIUM

The non-cancer HQ for cadmium was equal to 2 for the child tribal RME scenario based on Tulalip data (HQs were less than 1 for the other RME seafood consumption scenarios). However, cadmium was not identified as a risk driver because of the uncertainty associated with this scenario (as discussed in Section B.6.1.2.3) and because the HQ was only slightly greater than 1 (and more than an order of magnitude less than the HQ for total PCBs).

Cadmium was not identified as a COPC based on the direct sediment exposure RME scenarios.

B.8.6 PENTACHLOROPHENOL

The excess cancer risk for pentachlorophenol was slightly greater than the 1×10^{-6} threshold for one of the three RME scenarios (equal to 2×10^{-6} for the adult tribal RME scenario based on Tulalip data). Pentachlorophenol was not selected as a risk driver because of the low percent contribution to the total excess cancer risk (less than 1%) and because it was detected infrequently in EW tissue samples (detection frequency of less than 4%).

Pentachlorophenol was not identified as a COPC based on the direct sediment exposure RME scenarios.

B.8.7 PESTICIDES

The excess cancer risks for five pesticides (alpha-BHC, dieldrin, total chlordane, heptachlor epoxide, and mirex) were greater than 1×10^{-6} for at least one of the three RME seafood consumption scenarios. Risks for all five of these pesticides were greater than this threshold for the adult tribal RME scenario based on Tulalip data, with excess cancer risks ranging from 2×10^{-6} to 8×10^{-6}. None of these pesticides had excess cancer risks greater than 1×10^{-6} for the child tribal RME scenario based on Tulalip data, and only one of these pesticides had an excess cancer risks greater than 1×10^{-6} for the adult API RME scenario (dieldrin had an excess cancer risk equal to 2×10^{-6}).

None of these fives pesticides were selected as a risk driver based on their relatively low excess cancer risks (less than 1×10^{-5}), low percent contribution to the total excess cancer risk (less than 1.5% combined for all five pesticides).

Additionally, none of these pesticides were identified as COPCs based on the direct sediment exposure RME scenarios. It should also be noted that there is no evidence of historical use or manufacture of these pesticides in the EW.

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