King County Combined Sewer Overflow Water Quality Assessment for the Duwamish River and Elliott Bay

Appendix B: Methods and Results
B4: Aquatic Life Risk Assessment

Prepared by the

Duwamish River and Elliott Bay

Water Quality Assessment Team

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Parametrix, Inc.
5808 Lake Washington Boulevard, NE
Kirkland, Washington, 98033-7350

King County Department of Natural Resources
Wastewater Treatment Division &
Water and Land Resources Division
821 Second Avenue
Seattle, Washington 98104-1598

TABLE OF CONTENTS

Page

TABLE OF CONTENTS (CONTINUED)

TABLE OF CONTENTS (CONTINUED)

LIST OF FIGURES

LIST OF TABLES

Page

LIST OF TABLES (CONTINUED)

Page

LIST OF TABLES (CONTINUED)

Page

LIST OF TABLES (CONTINUED)

Page

LIST OF TABLES (CONTINUED)

Page

LIST OF ACRONYMS

ACR	Acute-chronic ratio
AET	Apparent effects threshold
APHA	American Public Health Association
AQUIRE	AQUatic toxicity Information REtrieval database
BOD	Biological oxygen demand
CLS	Cleanup screening level
COPCs	Constituents of potential concern
CSO	Combined sewer overflow
CV	Coefficient of variation
DO	Dissolved oxygen
ECD	Electron capture detector
EEC	Estimated exposure concentration
EFDC	Environmental Fluid Dynamics Computer Code
EqP	Equilibrium partitioning
ER-L	Effects range—low
ER-M	Effects range—median
GC/MS	Gas chromatography/mass spectrometry
GMAV	Genus mean acute value
GPS	Global positioning system
HPAH	High molecular weight PAH
HQ	Hazard quotient
ICP	Inductively coupled plasma
ILL	Incidence of liver lesions
ITI	Infaunal trophic index
KI	Kellogg Island
LOEC	Lowest observed effect concentration
LPAH	Low molecular weight polycyclic aromatic hydrocarbons
MDL	Method detection limit
MOAB	Mollusk abundance
N/AP	Not applicable
N/AV	Not available
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NOEC	No observed effect concentration
PAH	Polyaromatic hydrocarbons
PCB	Polychlorinated biphenyls
POAB	Polychaete abundance
PSEP	Puget Sound Estuary Program
QSARs	Quantitative structure-activity relationship
SDI	Swartz's Dominance Index
SDN	Specific degenerative/necrotic lesions
SEA	Striplin Environmental Associates
SMS	Sediment Management Standards

LIST OF ACRONYMS (CONTINUED)

SPCC	State Pollution Control Commission
STDS	Sample standard deviation
SQS	Sediment quality standards
TBT	Tributyltin
TOAB	Total abundance
TOC	Total organic carbon
TRV	Toxicity reference value
TSS	Total suspended solids
U.S. EPA	United States Environmental Protection Agency
WAC	Washington Administrative Code
WERF	Water Environment Research Foundation
WQA	Water quality assessment
WSDOE	Washington State Department of Ecology

1. INTRODUCTION

This appendix presents the methods and results of the aquatic life risk assessment portion of King County's Combined Sewer Overflow Water Quality Assessment for the Duwamish River and Elliott Bay. Additional overview and interpretation of the results presented here is provided in Volume 1 – *Overview and Interpretation*. Planning for the aquatic life risk assessments, including identification of types of stressors, identification of aquatic receptors, identification of exposure pathways, and development of conceptual site models, is presented in Appendix A -–*Problem Formulation, Analysis Plan, and Field Sampling Work Plan*.

Specific combinations of stressors and exposure pathways require a variety of approaches to evaluate the potential risks to aquatic life. This aquatic life risk assessment involved four concurrent evaluations: (1) a chemical-specific study of baseline conditions in the study area, without combined sewer overflow (CSO) conditions in the study area, and reference areas; (2) a physical stressor evaluation of the study area; (3) toxicity testing of a CSO discharge and (4) a benthic survey of a CSO sediment footprint. The chemical-specific study was further composed of an examination of exposure of the aquatic community to water column and sediment chemicals, salmonid juveniles to water column and dietary chemicals, and resident flatfish to sediment PAHs. Potential risks to aquatic life from chemicals in the water column were evaluated in two stages, corresponding to Tiers 1 and 3 of the Water Environment Research Foundation (WERF) methodology for aquatic ecological risk assessment (WERF 1996). Tier 2 was omitted due to the availability of site-specific exposure estimates from the Duwamish River and Elliott Bay risk assessment, making it unnecessary. In contrast, chemicals in sediments were assessed using a Tier 1 approach, while physical stressors were evaluated using both quantitative and descriptive approaches.

Five endpoints were selected to represent the aquatic life communities present in the study area (see Appendix A1 - *Problem Formulation* for a description of the selection process). Each assessment endpoint can be exposed to different combinations of study area stressors, requiring that specific evaluation methods be used to determination the level of risk, if any, posed to these receptors (Table 1-1).

Risks from chemical and physical stressors are reported in Sections 2 through 5. Section 2 identifies the toxicity reference values (TRVs) used to evaluate these chemicals in Tiers 1 and 3. This section also discusses the approaches used to determine potential risks from physical stressors and their associated effect thresholds. Section 3 discusses how exposure of aquatic life to chemicals in the water column and sediments was determined, as well as how aquatic life exposure to physical stressors was measured. Section 4 details the risk characterization methods used in the water column Tiers 1 and 3, sediments, and physical stressor assessments, while Section 5 summarizes the results of the risk characterization. Additional assessments of risks were derived from a laboratory toxicity test assessment of Brandon Street CSO effluent (reported in Section 6) and from a survey of the benthic community adjacent to the Duwamish/Diagonal outfall (presented in Section 7).

Table 1-1. Summary of Aquatic Life Assessment Endpoints, Stressors and Evaluation Methods[a]

Assessment Endpoint	Stressor Type	Evaluation Method
Survival and maintenance of aquatic community	Chemical	Tier 1 and Tier 3 aquatic life risk characterization
	Physical	Comparison of conventional water quality parameters to threshold values.
	Chemical	Whole effluent toxicity testing of CSO effluent.
Survival of juvenile salmonids	Chemical	Comparison of water and dietary exposure concentrations to salmonid TRVs
	Physical	Comparison of water velocities to velocity threshold exceedances - displacement
Health of resident flatfish	Chemical	Predict rate of occurrence of sediment polyaromatic hydrocarbon (PAH) exposure biomarkers (liver lesions)
Survival of polychaetes and amphipods; growth of polychaetes	Chemical	Comparison of sediment concentrations to sediment management standards
	Physical	Comparison of scouring and sedimentation rates to thresholds
Abundance and richness of benthic invertebrates	Chemical	Comparison of sediment concentrations to sediment management standards
	Physical	Comparison of scouring and sedimentation rates to thresholds
	All	Benthic community survey

a See Appendix A1 – *Problem Formulation* for further details.

2. AQUATIC LIFE TOXICOLOGICAL EFFECTS CHARACTERIZATION

The aquatic life toxicological effects characterization presents information on the concentrations of chemicals in water, sediment, and dietary items predicted to effect aquatic organisms. Also presented are the levels of changes in physical stressors predicted to effect aquatic organisms. Issue Paper No. 5 – *Physical Stressors* provides additional details on the development of effects levels for physical stressors. Issue Paper No. 6 – *Aquatic and Wildlife Toxicology* provides additional details on aquatic life toxicology. Specifically, the toxicological effects section of this appendix describes:

- Tier 1 surface water TRVs,
- Tier 1 salmon surface water and dietary TRVs,
- Tier 1 sediment TRVs,
- Tier 3 surface water TRVs,
- Physical stressor effects thresholds, and
- Effects characterization uncertainty.

Each of these topics is described in the sections that follow.

2.1 Tier 1 Surface Water TRVs

The Tier 1 surface water TRVs used in this risk assessment represent adverse effects thresholds for either acute or chronic exposures. Conservative estimates of Tier 1 surface water TRVs were developed for all 23 constituents[1] of potential concern (COPCs) identified for evaluation (see Appendix A1 and Appendix B1). The approach followed to select TRVs is presented below, followed by the results.

Marine TRVs were used in this risk assessment because the study area (the Duwamish River downstream of the Norfolk CSO and Elliott Bay) is a marine dominated estuary. Both acute and chronic TRVs are intended to protect the majority of the aquatic community being evaluated. (WAC 173-2010A-040; Stephan et al. 1985). For example, ambient water quality criteria derived by the U.S. EPA are generally designed to protect

1 No aquatic life standard was developed for fecal coliforms because they have no known impact on these organisms.

99 percent of the individuals in 95 percent of the species (Stephan et al. 1985)[2]. The following hierarchy was used in selecting water column TRVs:

1. Water quality standards for waters of the state of Washington
2. Federal ambient water quality criteria for protection of aquatic life
3. Toxicity data from the scientific literature
4. Quantitative structure-activity relationships (QSARs)[3].

Washington State standards and federal ambient water quality criteria were given highest priority because they are typically based on large toxicity databases which can reduce the uncertainty in a TRV, therefore increasing our confidence in the stated level of protection. When State standards or federal criteria were unavailable, TRVs were developed from toxicity studies published in the scientific literature. Toxicity studies identified in the literature were screened versus U.S. EPA guidelines for test acceptability[4] (e.g., Stephan et al. 1985). The lowest toxicity value identified for a given chemical, divided by an uncertainty factor of 20[5], was identified as the TRV when State standards or federal criteria were not available. Freshwater toxicity values were used when no marine/estuarine data were available. Various studies have shown that LC_{50}s for freshwater and saltwater species have indistinguishable distributions (Klapow and Lewis 1979; Suter and Rosen 1986). When neither State standards, federal criteria, nor empirical toxicity data from the literature were available, QSARs were used (where possible). QSARs can be used to estimate the toxicity of organic chemicals (particularly neutral, hydrophobic organics) based on the measured relationship between chemical toxicity, structure, or related properties. The QSARs used in this aquatic ecological risk assessment are based on the relationship of chemical toxicity and the chemical's octanol-

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2 This level of protection is sometimes superceded by the need to protect species of special concern (e.g., endangered) or species of particular commercial value. Thus, concentrations are sometimes set below the 5 percent effect concentration to protect such species.

3 QSARs are measures of the relative toxicity of different compounds based on similarities and differences in their physical and chemical properties.

4 These acceptability criteria focus on the factors such as the quality of the controls as well as the number of replicates.

5 Because limited toxicity data tend to be available for chemicals without State standards or federal criteria, an uncertainty factor of 20 was applied to ensure that potentially more sensitive species that have not been tested are protected. This approach is a modification of that described in the U.S. EPA's (1995a) Great Lakes Initiative, in which successively higher uncertainty factors are applied as the amount of toxicity data decreases. The U.S. EPA recommends uncertainty factors ranging from 1 when toxicity data from several studies are available, up to 21.9, when data for only one species are available. An uncertainty factor of 20 was conservatively applied to all literature-based TRVs used in this aquatic ecological risk assessment regardless of the number of studies available.

water partition coefficient and molecular mass (Clements and Nabholz 1994). TRVs based on QSARs are likely to be the most uncertain because they are not based on empirical data. As with the literature-based TRVs, QSAR results were also divided by an additional uncertainty factor of 20 to provide a conservative approach to the aquatic ecological risk assessment.

For some chemicals (e.g., benzo(a)pyrene and chrysene), chronic toxicity data were unavailable for developing chronic TRVs. In these cases, the chronic TRVs were estimated from acute toxicity values using an acute-chronic ratio (ACR). An ACR is the ratio of the acute LC_{50} for a chemical to its chronic value. The ACR generally can be estimated for one or a few species, and the estimate applied to the acute TRV to estimate the chronic TRV. Most chronic ambient water quality criteria developed by the U.S. EPA were derived using this approach (Stephan et al. 1985).

Of the 23 COPCs evaluated, TRVs for ten were based on State standards or federal criteria, seven were based on toxicity studies in the literature, and six were based on QSARs (Table 2-1). The acute and chronic TRVs selected are shown in Table 2-2 and Table 2-3, respectively. Where appropriate and available, dissolved standards for metals are presented. No acute TRVs were identified for most high molecular weight PAHs (HPAHs) because their low aqueous solubility precludes acute effects (Clements and Nabholz 1994) (i.e., HPAHs in the water column tend to be chronically toxic, but not acutely toxic). Therefore, the lack of acute TRVs for some HPAHs is not considered a significant data gap.

Table 2-1. Sources of Aquatic Life Surface Water Tier 1 TRVs

State Standard/ Federal Criterion	Literature-Based Toxicity Value	QSAR
Inorganics	**Organics**	**Organics**
Arsenic[a] Cadmium[a] Copper[a] Lead[a] Mercury[a] Nickel[a] Zinc[a]	1,4-Dichlorobenzene 4-Methylphenol Benzo(a)anthracene Benzo(a)pyrene Chrysene Fluoranthene Total PCBs	Benzo(b)fluoranthene Benzo(g,h,i)perylene Benzo(k)fluoranthene Dibenzo(a,h)anthracene Indeno(1,2,3-cd)pyrene Pyrene
Organometallics		
Tributyltin[b]		
Organics		
Bis(2-ethylhexyl)phthalate[b] Phenanthrene[b]		

[a] The State standards and federal criteria for these chemicals are equivalent.

[b] Proposed federal criterion.

Table 2-2. Acute Surface Water TRVs (μg/L) Used in Tier 1 of the Aquatic Ecological Risk Assessment

COPC	Total Recoverable	Dissolved	Comment	Reference
Inorganics				
Arsenic	69	N/AV	The TRV is for the more toxic As (III)	WAC 173-201A-040; U.S. EPA (1985a)
Cadmium	43	37.2		WAC 173-201A-040; U.S. EPA (1985b)
Copper	2.9	2.5		WAC 173-201A-040; U.S. EPA (1985c)
Lead	220	151.1		WAC 173-201A-040; U.S. EPA (1991)
Mercury	2.1	N/AV		WAC 173-201A-040; U.S. EPA (1985d)
Nickel	75	71.3		WAC 173-201A-040; U.S. EPA (1986a)
Zinc	95	84.6		WAC 173-2010A-040; U.S. EPA (1987a)
Organometallics				
Tributyltin	0.3674	N/AV	Proposed criterion	U.S. EPA (1997)
Organics				
1,4-Dichlorobenzene	49.75	N/AV	Includes an uncertainty factor	U.S. EPA (1980a)
4-Methylphenol	35	N/AV	Based on freshwater species, includes an uncertainty factor of 20	AQUIRE (1998)
Total PCBs	10	N/AV		WAC 173-2010A-040; U.S. EPA (1980b)
Benzo(a)anthracene	0.25	N/AV	Based on freshwater species, uncertainty factor of 20 applied for literature-based TRV	Trucco et al. (1983)

Table 2-2. Acute Surface Water TRVs (µg/L) Used in Tier 1 of the Aquatic Ecological Risk Assessment (continued)

COPC	Total Recoverable	Dissolved	Comment	Reference
Benzo(a)pyrene	25	N/AV	Uncertainty factor of 20 applied for literature-based TRV	Rossi and Neff (1978)
Benzo(b)fluoranthene	N/AP	N/AV	TRV not available from literature and acutely toxic concentration exceeds aqueous solubility	Clements and Nabholz (1994)
Benzo(g,h,i)perylene	N/AP	N/AV	TRV not available from literature and acutely toxic concentration exceeds aqueous solubility	Clements and Nabholz (1994)
Benzo(k)fluoranthene	N/AP	N/AV	TRV not available from literature and acutely toxic concentration exceeds Aqueous solubility	Clements and Nabholz (1994)
Bis(2-ethylhexyl)phthalate	400	N/AV	Proposed criterion	U.S. EPA (1987b)
Chrysene	25	N/AV	Based on freshwater species, uncertainty factor of 20 applied for literature-based TRV	Rossi and Neff (1978)
Dibenzo(a,h)anthracene	N/AP	N/AV	TRV not available from literature and acutely toxic concentration exceeds aqueous solubility	Clements and Nabholz (1994)

Table 2-2. Acute Surface Water TRVs (µg/L) Used in Tier 1 of the Aquatic Ecological Risk Assessment (continued)

COPC	Total Recoverable	Dissolved	Comment	Reference
Fluoranthene	1	N/AV	Based on freshwater species, uncertainty factor of 20 applied for literature-based TRV	U.S. EPA (1980c)
Indeno(1,2,3-cd)pyrene	N/AP	N/AV	TRV not available from literature and acutely toxic concentration exceeds aqueous solubility	Clements and Nabholz (1994)
Phenanthrene	7.7	N/AV	Proposed criterion	U.S. EPA (1988)
Pyrene	N/AP	N/AV	TRV not available from literature and acutely toxic concentration exceeds aqueous solubility	Clements and Nabholz (1994)

N/AP = Not applicable (acute QSARs not applicable for chemicals with a log $K_{ow} > 5$)

N/AV = Not available

K_{ow} = Octanol-water partition coefficient

Table 2-3. Chronic Surface Water TRVs (µg/L) Used in Tier 1 of the Aquatic Ecological Risk Assessment

COPC	Total Recoverable	Dissolved	Comment	Reference
Inorganics				
Arsenic	36	N/AV	The chronic TRV is for the more toxic As (III)	WAC 173-201A-040; U.S. EPA (1985a)
Cadmium	9.3	8		WAC 173-201A-040; U.S. EPA (1985b)
Copper	2.9	N/AV	Same as acute because the lowest acute values are from tests with embryos and larvae of mollusks and embryos of summer flounders, which are possibly the most sensitive life stages of these species. Therefore, concentrations that do not cause acute lethality to these organisms probably are not chronically toxic either.	U.S. EPA (1985c)
Lead	8.5	5.8		WAC 173-201A-040; U.S. EPA (1991)
Mercury	1.1	N/AV[a]		U.S. EPA (1985d)
Nickel	8.3	7.9		WAC 173-201A-040; U.S. EPA (1986a)
Zinc	86	76.6		WAC 173-201A-040; U.S. EPA (1987a)
Organometallics				
Tributyltin	0.01	N/AV	Proposed criterion	U.S. EPA (1997)

Table 2-3. Chronic Surface Water TRVs (µg/L) Used in Tier 1 of the Aquatic Ecological Risk Assessment (continued)

COPC	Total Recoverable	Dissolved	Comment	Reference
Organics				
1,4-Dichlorobenzene	19.15	N/AV	Chronic TRV estimated from acute using ACR of 5.2, uncertainty factor of 20 applied for literature-based TRV	U.S. EPA (1980a)
4-Methylphenol	7	N/AV	Based on freshwater species, uncertainty factor of 20 applied for literature-based TRV	AQUIRE (1998)
Total PCBs	0.0049	N/AV[b]	Chronic TRV estimated from acute using ACR of 8.6, uncertainty factor of 20 applied for literature-based TRV	U.S. EPA (1980b)
Benzo(a)anthracene	0.11	N/AV	Based on freshwater species, chronic TRV estimated from acute using ACR of 4.73, uncertainty factor of 20 applied for literature-based TRV	Trucco et al. (1983)
Benzo(a)pyrene	11	N/AV	Chronic TRV estimated from acute using ACR of 4.73, uncertainty factor of 20 applied for literature-based TRV	Rossi and Neff (1978)
Benzo(b)fluoranthene	0.2	N/AV	Based on freshwater species, chronic TRV estimated from acute using QSAR, uncertainty factor of 20 applied for literature-based TRV	Clements and Nabholz (1994)
Benzo(g,h,i)perylene	0.05	N/AV	Based on freshwater species, chronic TRV estimated from acute using QSAR, uncertainty factor of 20 applied for literature-based TRV	Clements and Nabholz (1994)
Benzo(k)fluoranthene	0.2	N/AV	Chronic TRV estimated from acute using QSAR, uncertainty factor of 20 applied for literature-based TRV	Clements and Nabholz (1994)

Table 2-3. Chronic Surface Water TRVs (µg/L) Used in Tier 1 of the Aquatic Ecological Risk Assessment (continued)

COPC	Total Recoverable	Dissolved	Comment	Reference
Bis(2-ethylhexyl) phthalate	360	N/AV	Proposed	U.S. EPA (1987b)
Chrysene	11	N/AV	Chronic TRV estimated from acute using ACR of 4.73, uncertainty factor of 20 applied for literature-based TRV	Rossi and Neff (1978)
Dibenzo(a,h)anthracene	0.2	N/AV	Freshwater QSAR, uncertainty factor of 20 applied for literature-based TRV	Clements and Nabholz (1994)
Fluoranthene	0.8	N/AV	Uncertainty factor of 20 applied for literature-based TRV	U.S. EPA (1980c)
Indeno(1,2,3-cd)pyrene	0.05	N/AV	Freshwater QSAR, uncertainty factor of 20 applied for literature-based TRV	Clements and Nabholz (1994)
Phenanthrene	4.6	N/AV	Proposed	U.S. EPA (1988)
Pyrene	2.1	N/AV	Freshwater QSAR, uncertainty factor of 20 applied for literature-based TRV	Clements and Nabholz (1994)

[a] Washington State chronic standard for mercury not included because it is based on protection of human health for fish consumption.

[b] Washington State chronic standard for Total PCBs not included because it is based on residual concentrations in fish tissues.

N/AV = Not available

ACR = Acute-chronic ratio

QSAR = Quantitative structure-activity relationship

2.2 Tier 1 Salmon Surface Water and Dietary TRVs

Toxicity data for juvenile salmonids were identified for those surface water chemicals exceeding TRVs in Tier 1 (described below in Section 5.1). Toxicity data for salmon (e.g., chinook, coho) were not always available, so data for other salmonids (e.g., rainbow trout) were selected where available. Toxicity data for salmonids could not be identified for three chemicals with surface water chemicals exceeding TRVs in Tier 1, including: benzo(a)anthracene, benzo(g,h,i)perylene, and fluoranthene. All toxicity values for salmonids used in the risk characterization are shown in Table 2-4. Additionally, juvenile salmon can be exposed to chemicals in the Duwamish River and Elliott Bay from consuming prey that have accumulated concentrations in their tissues. Dietary effect levels for juvenile salmon were taken from an U.S. EPA toxicity database (AQUIRE) and the scientific literature (Table 2-5). Studies were screened for usability in this risk assessment, and only studies satisfying the AQUIRE data quality 1 or 2 criteria[6] were used in this project (Chemical Information Systems, Inc. 1991).

2.3 Tier 1 Sediment TRVs

Potential risks to aquatic life from exposures to chemicals in sediment were assessed by comparing sediment concentrations to Tier 1 bulk sediment TRVs. Sediment TRVs were identified from the following sources:

- Washington State Sediment Management Standards (WSDOE 1995a);
- U.S. EPA Sediment Quality Criteria (U.S. EPA 1993a,b);
- Scientific literature studies (Long et al. 1995, Weston 1996);
- Calculation methods for untested sediment chemicals-Ecotox Threshold (Ecotox 1996) and equilibrium partitioning (Di Toro et al. 1991, U.S. EPA 1993c).

The bases behind the different sediment guideline values are described for each source below.

2.3.1 Overview of TRV Development Methodologies

The State of Washington has developed sediment quality standards designed to result in no adverse effects on biological resources (WAC 173-204-320). The standards were developed using a biological effects-based approach that generally uses the lowest

Table 2-4. Acute and Chronic Surface Water TRVs for Salmonids (µg/L) (Total Recoverable Concentrations)

COPC	Salmonid Species	Acute[a]	Chronic	Reference
Arsenic	*Oncorhynchus mykiss* (rainbow trout)	6,670	N/AP	U.S. EPA (1985a)
	Salvelinus fontinalis (brook trout)	7,480	N/AP	U.S. EPA (1985a)
Copper	*Oncorhynchus kisutch* (coho salmon)	124	88[b]	U.S. EPA (1985c)
	Oncorhynchus tshawytscha (chinook salmon)	75	53[b]	U.S. EPA (1985c)
	Oncorhynchus nerka (sockeye salmon)	414	293[b]	U.S. EPA (1985c)
	Salvelinus fontinalis (brook trout)	196	55	U.S. EPA (1985c)
	Salmo salar (Atlantic salmon)	195	138[b]	U.S. EPA (1985c)
	Oncorhynchus mykiss (rainbow trout)	74	65	U.S. EPA (1985c)
	Salmo clarkii (cutthroat trout)	117	83[b]	U.S. EPA (1985c)
	Salmo trutta (brown trout)	N/AV	105	U.S. EPA (1985c)
	Salvelinus namaycush (lake trout)	N/AV	104	U.S. EPA (1985c)
Lead	*Oncorhynchus mykiss* (rainbow trout)	5,653	368	U.S. EPA (1985e)
	Salvelinus fontinalis (brook trout)	11,144	452	U.S. EPA (1985e)
Nickel	*Oncorhynchus mykiss* (rainbow trout)	21,574	288	U.S. EPA (1986a)
Zinc	*Oncorhynchus kisutch* (coho salmon)	2,577	N/AP	U.S. EPA (1987a)
	Oncorhynchus tshawytscha (chinook salmon)	707	N/AP	U.S. EPA (1987a)
	Oncorhynchus nerka (sockeye salmon)	2,377	N/AP	U.S. EPA (1987a)
	Oncorhynchus mykiss (rainbow trout)	1,091	N/AP	U.S. EPA (1987a)
	Salvelinus fontinalis (brook trout)	3,324	N/AP	U.S. EPA (1987a)
	Salmo salar (Atlantic salmon)	3,444	N/AP	U.S. EPA (1987a)
TBT	*Oncorhynchus tshawytscha* (chinook salmon)	N/AV	0.10[c]	U.S. EPA (1997)
	Oncorhynchus mykiss (rainbow trout)	N/AV	0.31[c]	U.S. EPA (1997)
	Salvelinus namaycush (lake trout)	N/AV	0.87[c]	U.S. EPA (1997)
PCB	*Salvelinus fontinalis* (brook trout)	N/AV	1	U.S. EPA (1980b)

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[6] These indicate data reliability criteria established by U.S. EPA, such as adequate controls, measured toxicant concentrations, and adequate methods descriptions. Specific criteria are presented in Chemical Information Systems, Inc. (1991).

[a] Genus Mean Acute Value (GMAV) divided by 2.

[b] Chronic value estimated from GMAV using ACR of 2.823 (U.S. EPA 1985c).

[c] Chronic value estimated from GMAV using ACR of 14.69 (U.S. EPA 1997).

N/AP = Not applicable (not a COPC)

N/AV = Not available

Table 2-5. Acute and Chronic Dietary TRVs for Juvenile Salmonids

Chemical Name	Test Species	Effect	Wet Weight Conc.	Reference
Aroclor 1254	*Salvelinus namaycush*	Growth	0.72 µg/g	Mac and Seelye (1981)
	Salvelinus namaycush	Mortality	0.72 µg/g	Mac and Seelye (1981)
	Salvelinus namaycush	Growth	0.72 µg/g	Mac and Seelye (1981)
	Oncorhynchus mykiss	Mortality	>1.5 g/kg	Mayer et al. (1977)
	Oncorhynchus mykiss	Mortality	>1.5 g/kg	Mayer et al. (1977)
Copper	*Oncorhynchus mykiss*	Mortality	37 mg/kg	Lanno et al. (1985)
	Oncorhynchus mykiss	Mortality	83 mg/kg	Lanno et al. (1985)
	Oncorhynchus mykiss	Mortality	132 mg/kg	Lanno et al. (1985)
	Oncorhynchus mykiss	Mortality	171 mg/kg	Lanno et al. (1985)
	Oncorhynchus mykiss	Mortality	258 mg/kg	Lanno et al. (1985)
	Oncorhynchus mykiss	Mortality	403 mg/kg	Lanno et al. (1985)
	Oncorhynchus mykiss	Mortality	511 mg/kg	Lanno et al. (1985)
	Oncorhynchus mykiss	Growth	511 mg/kg	Lanno et al. (1985)
	Oncorhynchus mykiss	Mortality	664 mg/kg	Lanno et al. (1985)
	Oncorhynchus mykiss	Growth	664 mg/kg	Lanno et al. (1985)
	Oncorhynchus mykiss	Mortality	730 mg/kg	Lanno et al. (1985)
	Oncorhynchus mykiss	Mortality	796 mg/kg	Lanno et al. (1985)
	Oncorhynchus mykiss	Growth	796 mg/kg	Lanno et al. (1985)
	Oncorhynchus mykiss	Mortality	1,585 mg/kg	Lanno et al. (1985)
	Oncorhynchus mykiss	Mortality	3,088 mg/kg	Lanno et al. (1985)
	Oncorhynchus mykiss	Growth	13 µg/g	Miller et al. (1993)

	Oncorhynchus mykiss	Growth	684 µg/g	Miller et al. (1993)
Lead	*Oncorhynchus mykiss*	Growth	7,040 µg/g	Goettl and Davies (1976)

apparent effects threshold (AET) values of four biological indicators. An AET is the chemical concentration in sediments above which a particular biological effect is observed. The four biological indicators for which AETs have been developed are 1) amphipod (*Rhepoxynius abronius*) mortality, 2) bivalve (*Crassostrea gigas*) larval abnormality, 3) Microtox® (*Photobacterium phosphoreum*) bacterial luminescence bioassay endpoints, and 4) abundances of major taxa of indigenous benthic infauna. However, for phenanthrene, Washington State used equilibrium partitioning[7] to develop a criterion.

U.S. EPA sediment quality criteria were available for two of the chemicals being evaluated: fluoranthene (U.S. EPA 1993a) and phenanthrene (U.S. EPA 1993b). These criteria are derived by the EqP approach (U.S. EPA 1993a,b,c):

$$SQC_{oc} = \text{Chronic WQC} \times K_{oc} \qquad \text{Equation 2-1}$$

Where:

SQC_{oc}	=	the sediment quality criterion on a total organic carbon basis,
WQC	=	water quality criterion,
K_{oc}	=	organic carbon-normalized sediment water partition coefficient.

To adjust SQC_{oc} for the organic carbon at the site, it is multiplied by the fraction of organic carbon at the site:

$$\text{Site} - \text{specific SQC} = SQC_{oc} \times F_{oc} \qquad \text{Equation 2-2}$$

Where: F_{oc} is the fraction organic carbon in the sediment (assumed to be 1.06 percent for the study area).

The EqP approach is based on three observations (U.S. EPA 1993a,b,c):

- The concentrations of nonionic chemicals[8] in sediments, expressed on an organic carbon basis, and in pore waters correlate to observed biological effects in sediment-dwelling organisms across a range of sediment types.

[7] Equilibrium partitioning theory states that organic chemicals tend to preferentially bind to the organic fraction of sediment, where an assumed "equilibrium" concentration is achieved between the bound chemical fraction (sediment organic carbon) and the unbound dissolved chemical phase (interstitial water chemical concentrations) over time.

[8] A nonionic chemical is one that does not have an electronic charge. All organic chemicals in this risk assessment are nonionic.

- Partitioning models can relate sediment concentrations for nonionic organic chemicals on an organic carbon basis to freely dissolved concentrations in pore water.

- The distribution of benthic and water column organism's sensitivities to chemicals are similar; thus, the currently established WQC final chronic value (FCV) can be used to define the acceptable effects concentration of a chemical freely dissolved in porewater.

Long et al. (1995) calculated effects range-low (ER-L) and effects range-median (ER-M) values based on a biological effects sediment database. Data from EqP modeling, laboratory spiked-sediment bioassays, and field studies of sediment toxicity and benthic community composition were reviewed, and those meeting specified criteria were used to derive the ER-L and ER-M values. Adverse biological effects included in the database were measures of altered benthic communities, significantly or elevated sediment toxicity, histopathological disorders in demersal fish, EC_{50} and LC_{50} values from laboratory experiments with sediments spiked with a single chemical, and predicted toxicity from EqP models. The effects data for a chemical were then arranged in ascending order, with ER-L value being defined as the lower 10th percentile of the effects data and the ER-M value being defined as the median (50th percentile) of the effects data.

Other sediment guidelines used to assess potential risks to aquatic life in sediment were derived from EPA's Ecotox (1996) database and EqP. The Ecotox database consists of thresholds designed for screening purposes and are generally based on EqP for organic compounds. For metals, Ecotox thresholds are equivalent to ER-L values derived by Long et al. (1995) (see above). Finally, sediment guidelines were also derived using the EqP approach based on the surface water TRVs and literature-based sediment-water partition coefficients where possible.

A proposed sediment toxicity value was developed for TBT based on the EqP approach (Weston 1996). Various approaches for deriving screening values were evaluated by an interagency work group comprising the U.S. EPA Region X, Washington State Departments of Ecology and Natural Resources, U.S. Army Corps of Engineers, National Oceanic and Atmospheric Administration, EVS Environment Consultants, and Roy F. Weston, Inc. The work group was unable to identify data correlating TBT concentrations in field sediments with observed biological effects. Moreover, the work group identified limited data on laboratory toxicity studies of TBT in sediment. The work group recommended a sediment guideline using the EqP approach (discussed above). Using a mean K_{oc} value of 25,100 L/kg from a study by Meador et al. (1996) and the proposed chronic water quality criterion of 0.010 µg/L, a sediment guideline of 0.251 µg TBT/g organic carbon was calculated. Assuming an average organic carbon content of 1.06 percent in the Duwamish River and Elliott Bay translates this into a bulk sediment guideline of 0.00266 µg TBT/g. For comparison, an "in-house" sediment guideline of 0.0047 µg TBT/g was derived based on the same approach except an average K_{oc} value of 44,330 L/kg was used (Springborn 1995; Unger et al. 1988; and Meador et al. 1996).

2.3.2 Tier 1 Sediment TRVs Selection Process

Tier 1 Sediment TRVs were selected following the hierarchy presented in Table 2-6. Sediment TRVs were available from Washington State Management Standards (WSDOE 1995b) or Long et al. (1995) for all COPCs evaluated except TBT and benzo(e)pyrene. For some sediment chemicals, the U.S. EPA has published criteria developed using the

Table 2-6. Sediment TRV Selection Hierarchy

First choice	Washington State Sediment Management Standards (Title 173-204 WAC), (WSDOE 1995a) **or**
Second choice	Long et al. (1995)**, or**
Third choice	Ecotox Threshold (1996)**, or**
Fourth choice	Use acute water Tier 1 TRV and EqP to develop sediment TRV (Di Toro et al. 1991; U.S. EPA 1993c).

Ecotox Threshold process (Ecotox 1996). If no sediment criteria were available for a nonionic organic chemical, then one was calculated using the equilibrium partitioning (EqP) approach (Di Toro et al. 1991). Sediment TRVs were developed for TBT and benzo(e)pyrene using the equilibrium partitioning approach. The Tier 1 sediment TRVs selected for use in this risk assessment are presented in Table 2-7.

2.4 Tier 3 Surface Water TRVs

Those COPCs identified in the aquatic risk characterization(Section 5.1.1 below) as having surface water concentrations exceeding Tier 1 TRVs were further evaluated in Tier 3 of the aquatic ecological risk assessment. Chemicals without an identifiable Tier 1 acute criterion were not evaluated further, nor were chemicals identified in the aquatic risk characterization (Section 5.1.1 below) as not having surface water concentrations exceeding Tier 1 TRVs. The COPCs exceeding TRVs were:

- arsenic (acute[9]);
- benzo(a)anthracene (acute);
- benzo(g,h,i)perylene (chronic);
- copper (acute and chronic);
- fluoranthene (acute);

9 The type of TRVs exceeded by each COPC are listed following the chemical name.

- lead (acute and chronic);
- nickel (acute and chronic);
- tributyltin (TBT) (chronic);
- total PCBs (chronic); and
- zinc (acute).

Table 2-7. Sediment TRVs (mg/kg dry weight) Used in Tier 1 of the Aquatic Ecological Risk Assessment

COPC	WA SQS	U.S. EPA SQC	Long et al. (1995)		Ecotox	EqP[a]
			ERL	ERM		
Inorganics						
Arsenic	57	-	8.2	70	8.2	-
Cadmium	5.1	-	1.2	9.6	1.2	-
Copper	390	-	34	270	34	-
Lead	450	-	46.7	218	47	-
Mercury	0.41	-	0.15	0.71	0.15	-
Nickel	-	-	20.9	51.6	21	-
Zinc	410	-	150	410	150	-
Organometallics						
Tributyltin	-	-	-	-	-	0.0027[b] 0.0047[c]
Organics						
1,4-Dichlorobenzene[d]	0.0331	-	-	-	-	0.345
4-Methylphenol	0.67	-	-	-	-	-
Benzo(a)anthracene[d]	1.166	-	0.261	1.6	-	0.0466
Benzo(b&k)fluoranthene[d,e]	2.438	-	-	-	-	-
Benzo(a)pyrene[d]	1.049	-	0.43	1.6	0.430	804
Benzo(g,h,i)perylene[d]	0.329	-	-	-	-	0.848
Bis(2-ethylhexyl)phthalate[d]	0.498	-	-	-	-	382

Chrysene[d]	1.166	-	0.384	2.8	-	34.4
Dibenzo(a,h)anthracene[d]	0.127	-	0.0634	0.26	-	2.82
Fluoranthene[d]	1.696	14.2	0.6	5.1	1.5	14.2

Table 2-7. Sediment TRVs (mg/kg dry weight) Used in Tier 1 of the Aquatic Ecological Risk Assessment (continued)

COPC	WA SQS	U.S. EPA SQC	Long et al. (1995)		Ecotox	EqP[a]
			ERL	ERM		
Indeno(1,2,3-cd)pyrene[d]	0.36	-	-	-	-	3.75
Phenanthrene[d]	1.06	2.54	0.24	1.5	1.2	2.54
Pyrene[d]	10.6	-	0.665	2.6	0.66	1.55
Total PCBs[d]	0.127	-	0.0227	0.18	0.023	0.00239

[a] All guidelines in this column derived "in-house" using literature-based K_{oc} values unless otherwise noted.

[b] Derived by Roy F. Weston, Inc. for the U.S. EPA Region X (see text).

[c] Derived by Parametrix using an alternative K_{oc} values for TBT (see text).

[d] Washington State Sediment Quality Standard for these chemicals are normalized to an organic carbon content of 1.06 percent.

[e] The Washington State Sediment Quality Standard is for the sum of benzo(b)fluoranthene, benzo(j)fluoranthene, and benzo(k)fluoranthene. However, benzo(j)fluoranthene was not evaluated in this aquatic ecological risk assessment, nor was it available form the EFDC model.

Not available

WA SQS = Washington State Sediment Quality Standards (WSDOE 1995a)

U.S. EPA SQC = United States Environmental Protection Agency Sediment Quality Criteria (U.S. EPA 1993a,b)

ERL = Effects Range-Low

ERM = Effects Range-Median

EqP = Equilibrium Partitioning

= shaded values were selected for use as the Tier 1 sediment TRVs

Arsenic, copper, lead, nickel, zinc and TBT were evaluated in more detail in Tier 3 of the aquatic ecological risk assessment by comparing the spatial distribution of exposure concentrations to the species distribution of toxicity values for that COPC. Total PCBs and the three PAHs identified in Tier 1 could not be evaluated by the Tier 3 method because toxicity data were not available for an adequate number of saltwater species.

In the Tier 3 effects characterization, acute and chronic toxicity values[10] are identified for multiple aquatic species. Toxicity data that met the Stephan et al. (1985) guidelines for test acceptability[11] were taken from U.S. EPA's ambient water quality criteria documents, U.S. EPA's AQUIRE database, and the scientific literature. For both acute and chronic effects, organism sensitivities to COPCs were expressed at the genus[12] level. This parallels the process used to derive water quality criteria (Stephan et al. 1985). The geometric mean of all appropriate acute toxicity data (e.g., LC_{50} values[13]) for a given genus (termed the Genus Mean Acute Value or GMAV) was used for acute effects. Following U.S. EPA guidance, GMAVs (which are based on LC_{50} values) were divided by two to predict a conservative acute effects threshold. Following the U.S. EPA's approach in developing chronic water quality criteria, most of the chronic toxicity data were estimated from GMAVs using a chemical-specific acute-chronic ratio. For TBT, a combination of measured chronic values and estimated chronic values was used because the ACR did not adequately reflect the chronic toxicity of TBT to certain sensitive species. Chronically sensitive species include larvae of certain bivalves (e.g., *Mercenaria mercenaria, Crassostrea gigas*, and *Ostrea edulis*). No ACRs were available for sensitive bivalves, so the chronic toxicity of TBT to these could be under estimated.

Moving from Tier 1 to Tier 3 in the WERF method involves using more detailed information to move from conservative assumptions to more realistic values without reducing the margin of safety in the risk assessment. For metals, this involved using total recoverable concentrations in Tier 1 and dissolved concentrations in Tier 3. This approach was possible because the Environmental Fluids Dynamic Computer Code (EFDC) water quality model predicted both total recoverable and dissolved metals concentrations. However, the effects concentrations are invariably expressed as the total recoverable metal. Consequently, it was necessary to convert the total recoverable effects concentrations for the Tier 1 metal COPCs (arsenic, copper, lead, nickel and zinc) to dissolved effects concentrations in the Tier 3 analysis. This represents a more realistic representation of the aquatic life risks because the dissolved fraction more closely approximates the bioavailable fraction of metals in the water column (Prothro 1993). We multiplied the total recoverable concentrations by the U.S. EPA-developed metal-specific conversion factors to convert total recoverable effect concentrations to dissolved metal effect concentrations (U.S. EPA 1996). These conversion factors are shown in Table 2-8.

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[10] These TRVs were selected from tests that were screened for criteria such as an acceptable number of controls, exposure duration and suitable endpoint (Chemical Information Systems, Inc. 1991).

[11] Stephan et al. (1985) requires data be available for eight genera representing multiple levels of taxonomy for a criterion to be developed. This approach has been adopted in WERF Methodology (WERF 1996).

[12] A genus is a group used in classifying organisms that consists of one or more similarly related species.

[13] The LC_{50} is the chemical concentration that resulted in the mortality of 50 percent of the organisms in a toxicity experiment.

Conversion factors for saltwater have been determined for acute toxicity tests, but not for chronic. We assumed that the marine water chronic conversion factors would be the

Table 2-8. Saltwater Conversion Factors for Dissolved Metals

Metal	Conversion Factor
Arsenic	1.000
Copper	Not Required[a]
Lead	0.951
Nickel	0.990
Zinc	0.946

[a] In the Draft Ambient Water Quality Criteria – Saltwater Copper Addendum (U.S. EPA 1995), toxicity values were already expressed as dissolved copper. The total recoverable value was used in Tier 1 (U.S. EPA 1985c), and the dissolved value (U.S. EPA 1995) was used in Tier 3.

same as the acute, based on the observation that the acute and chronic conversion factors for these same metals in freshwater were virtually identical.

The toxicity data identified for each COPC evaluated in Tier 3 are presented in Table 2-9 through Table 2-14. The distributions of available marine toxicity data for each COPC are shown graphically in Figure 2-1 through Figure 2-6. Freshwater toxicity data for salmonids (e.g. salmon and trout) are also noted in Figure 2-1 through Figure 2-6.

Table 2-9. Marine Acute Toxicity Values Identified for Arsenic (Dissolved)

Species	Common Name	Genus Mean Acute Value (µg/L)	Reference
Cancer magister	Dungeness crab	232	U.S. EPA (1985a)
Acartia clausi	Copepod	508	U.S. EPA (1985a)
Crassostrea sp.	Oyster	1,564	U.S. EPA (1985a)
Mysidopsis bahia	Mysid	1,740	U.S. EPA (1985a)
Mytilus trossulus	Blue mussel	>3,000	U.S. EPA (1985a)
Argopecten irradians	Bay scallop	3,490	U.S. EPA (1985a)
Ampelisca abdita	Amphipod	8,227	U.S. EPA (1985a)
Neanthes arenaceodentata	Polychaete	10,120	U.S. EPA (1985a)
Cyprinodon variegatus	Sheepshead minnow	12,700	U.S. EPA (1985a)
Apeltes quadracus	Fourspine stickleback	14,950	U.S. EPA (1985a)

Menidia menidia	Atlantic silverside	16,030	U.S. EPA (1985a)
Corophium volutator	Amphipod	60,000	AQUIRE (1998)

Figure 2-1. Acute EEC and Marine Toxicity Distributions for Dissolved Arsenic

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Table 2-10. Marine Acute and Chronic Toxicity Values Identified for Copper (Dissolved)

Species	Common Name	Genus Mean Acute Value (µg/L)	Estimated Chronic Value[a] (µg/L)	Reference
Mytilus trossulus	Blue mussel	9.63	3.08	U.S. EPA (1995b)
Paralichthys dentatus	Summer flounder	11.6	3.70	U.S. EPA (1995b)
Mulinia lateralis	Coot clam	17.7	5.66	U.S. EPA (1995b)
Crassostrea sp.	Oyster	21.4	6.84	U.S. EPA (1995b)
Arbacia punctulata	Sea urchin	21.4	6.84	U.S. EPA (1995b)
Mya arenaria	Soft-shell clam	35.1	11.2	U.S. EPA (1995b)
Acartia sp.	Copepod	36.0	11.5	U.S. EPA (1995b)
Cancer magister	Dungeness crab	44.1	14.1	U.S. EPA (1995b)
Haliotis sp.	Abalone	59.0	18.9	U.S. EPA (1995b)
Homarus americanus	American lobster	62.4	19.9	U.S. EPA (1995b)
Pseudopleuronectes americanus	Winter flounder	107	34.2	U.S. EPA (1995b)
Phyllodoce maculata	Polychaete	108	34.5	U.S. EPA (1995b)
Menidia sp.	Silverside	116	37.2	U.S. EPA (1995b)
Pseudolaptomus coronatus	Copepod	124	39.7	U.S. EPA (1995b)
Mysidopsis sp.	Mysid	136	43.3	U.S. EPA (1995b)
Neanthes arenaceodentata	Polychaete	151	48.2	U.S. EPA (1995b)
Tigriopus californica	Copepod	212	67.9	U.S. EPA (1995b)
Atherinops affinis	Topsmelt	219	69.9	U.S. EPA (1995b)
Leiostomus xanthurus	Spot	252	80.6	U.S. EPA (1995b)
Nereis sp.	Polychaete	>260	>83.2	U.S. EPA (1995b)
Cyprinodon variegatus	Sheepshead minnow	305	97.7	U.S. EPA (1995b)
Trachinotus carolinus	Florida pompano	371	118	U.S. EPA (1995b)
Eurytemora affinis	Copepod	473	151	U.S. EPA (1995b)
Carcinus maenus	Green crab	540	173	U.S. EPA (1995b)
Fundulus heteroclitus	Mummichog	1,391	445	U.S. EPA (1995b)
Rangia cuneata	Common rangia	6,925	2,215	U.S. EPA (1995b)

[a] The chronic value was estimated from the genus mean acute value using an acute-chronic ratio of 3.127 (U.S. EPA 1995).

Figure 2-2. Acute and Chronic EEC and Marine Toxicity Distributions for Dissolved Copper

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Table 2-11. Marine Acute and Chronic Toxicity Values Identified for Lead (Dissolved)

Species	Common Name	Genus Mean Acute Value (µg/L)	Estimated Chronic Value[a] (µg/L)	Reference
Fundulus heteroclitus	Mummichog	300	5.84	U.S. EPA (1985e)
Mytilus trossulus	Blue mussel	453	8.83	U.S. EPA (1985e)
Ampelisca abdita	Amphipod	520	10.1	U.S. EPA (1985e)
Cancer magister	Dungeness crab	547	10.7	U.S. EPA (1985e)
Acartia tonsa	Copepod	635	12.4	U.S. EPA (1985e)
Mercenaria mercenaria	Quahog clam	742	14.5	U.S. EPA (1985e)
Crassostrea gigas	Pacific oyster	1,296	25.3	U.S. EPA (1985e)
Capitella capitata	Polychaete	2,853	56	AQUIRE (1998)
Mysidopsis bahia	Mysid	2,977	58.0	U.S. EPA (1985e)
Cyprinodon variegatus	Sheepshead minnow	2,986	58.2	U.S. EPA (1985e)
Perna viridis	Mussel	4184	82	AQUIRE (1998)
Spisula solidissima	Surf clam	5,135	100	AQUIRE (1998)
Menidia sp.	Silverside	5,329	104	U.S. EPA (1985e)
Argopecten irradians	Bay scallop	8,178	159	AQUIRE (1998)
Mya arenaria	Soft-shell clam	25,677	501	U.S. EPA (1985e)
Paralichthys olivaceus	Flounder	28,530	556	AQUIRE (1998)
Ophryotrocha diadema	Polychaete	95,100	1,854	AQUIRE (1998)

[a] The chronic value was estimated from the genus mean acute value using an acute-chronic ratio of 51.29. (U.S. EPA 1985e)

Figure 2-3. Acute and Chronic EEC and Marine Toxicity Distributions for Dissolved Lead

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Table 2-12. Marine Acute and Chronic Toxicity Values Identified for Nickel (Dissolved)

Species	Common Name	Genus Mean Acute Value (µg/L)	Estimated Chronic Value[a] (µg/L)	Reference
Heteromysis formosa	Mysid	150	8.35	U.S. EPA (1986a)
Mercenaria mercenaria	Quahog clam	307	17.1	U.S. EPA (1986a)
Mysidopsis sp.	Mysid	562	31.2	U.S. EPA (1986a)
Crassostrea virginica	Eastern oyster	1,168	64.9	U.S. EPA (1986a)
Metapenaeus ensis	Greasyback shrimp	1,267	70.4	AQUIRE (1998)
Acartia clausi	Copepod	3,431	191	U.S. EPA (1986a)
Nitocra spinipes	Copepod	5,940	330	U.S. EPA (1986a)
Eurytemora affinis	Copepod	11,128	619	U.S. EPA (1986a)
Monhystera disjuncta	Nematode	14,850	825	AQUIRE (1998)
Ctenodrilus serratus	Polychaete	16,830	936	U.S. EPA (1986a)
Menidia sp.	Silverside	17,216	957	U.S. EPA (1986a)
Corophium volutator	Amphipod	18,761	1,043	U.S. EPA (1986a)
Morone saxatilis	Striped bass	20,790	1,156	U.S. EPA (1986a)
Neries sp.	Polychaete	34,650	1,926	U.S. EPA (1986a)
Pagurus longicarpus	Hermit crab	46,530	2,586	U.S. EPA (1986a)
Liza vaigiensis	Square tail mullet	46,646	2,593	AQUIRE (1998)
Capitella capitata	Polychaete	49,500	2,752	U.S. EPA (1986a)
Leiostomus xanthurus	Spot	69,300	3,852	U.S. EPA (1986a)
Nassarius obsoletus	Mud snail	71,280	3,962	U.S. EPA (1986a)
Fundulus heteroclitus	Mummichog	148,401	8,249	U.S. EPA (1986a)
Asterias forbesii	Starfish	148,500	8,255	U.S. EPA (1986a)
Macoma balthica	Clam	291,555	16,207	U.S. EPA (1986a)
Mya arenaria	Soft-shell clam	316,800	17,610	U.S. EPA (1986a)

[a] The chronic value was estimated from the genus mean acute value using an acute-chronic ratio of 17.99. (U.S. EPA 1986a)

Figure 2-4. Acute and Chronic EEC and Marine Toxicity Distributions for Dissolved Nickel

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Table 2-13. Marine Acute Toxicity Values Identified for Zinc (Dissolved)

Species	Common name	Genus Mean Acute Value (µg/L)	Reference
Scorpaenichthys marmoratus	Cabezon	181	U.S. EPA (1987a)
Mercenaria mercenaria	Quahog clam	184	U.S. EPA (1987a)
Crassostrea sp.	Oyster	234	U.S. EPA (1987a)
Homarus americanus	American lobster	360	U.S. EPA (1987a)
Pagurus longicarpus	Hermit crab	378	U.S. EPA (1987a)
Morone saxatilis	Striped bass	407	U.S. EPA (1987a)
Mysidopsis sp.	Mysid	454	U.S. EPA (1987a); AQUIRE (1998)
Cancer magister	Dungeness crab	554	U.S. EPA (1987a)
Acartia sp.	Copepod	630	U.S. EPA (1987a)
Carcinus maenus	Green crab	946	U.S. EPA (1987a)
Nitocra spinipes	Copepod	1,050	U.S. EPA (1987a); AQUIRE (1998)
Neanthes arenaceodentata	Polychaete	1,204	U.S. EPA (1987a)
Ophryotrocha diadema	Polychaete	1,324	U.S. EPA (1987a)
Monhystera disjuncta	Nematode	1,797	AQUIRE (1998)
Loligo opalescens	Squid	1,816	U.S. EPA (1987a)
Allorchestes compressa	Amphipod	1,892	AQUIRE (1998)
Capitella capitata	Polychaete	2,327	U.S. EPA (1987a); AQUIRE (1998)
Spisula solidissima	Surf clam	2,791	AQUIRE (1998)
Mytilus trossulus	Blue mussel	3,722	U.S. EPA (1987a)
Eurytemora affinis	Copepod	3,854	U.S. EPA (1987a)
Menidia sp.	Silverside	4,271	U.S. EPA (1987a)
Corophium volutator	Amphipod	4,430	U.S. EPA (1987a)
Mya arenaria	Soft-shell clam	5,986	U.S. EPA (1987a)
Ctenodrilus sp.	Polychaete	6,717	U.S. EPA (1987a)
Pseudopleuronectes americanus	Winter flounder	8,956	U.S. EPA (1987a)
Nereis sp.	Polychaete	9,649	U.S. EPA (1987a); AQUIRE (1998)
Palaemonetes pugio	Grass shrimp	10,690	AQUIRE (1998)
Fundulus heteroclitus	Mummichog	34,652	U.S. EPA (1987a)
Leiostomus xanthurus	Spot	35,948	U.S. EPA (1987a)
Asterias forbesii	Starfish	36,894	U.S. EPA (1987a)
Nassarius obsoletus	Mud snail	47,300	U.S. EPA (1987a)
Macoma balthica	Clam	303,098	U.S. EPA (1987a)

[a] The chronic value was estimated from the genus mean acute value using an acute-chronic ratio of 2.208. (U.S. EPA 1987a)

Figure 2-5. Acute and Chronic EEC and Marine Toxicity Distributions for Dissolved Zinc

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Table 2-14. Marine Acute and Chronic Toxicity Values Identified for TBT

Species	Common Name	Genus Mean Acute Value (µg/L)	Estimated Chronic Value[a] (µg/L)	Reference
Mercenaria mercenaria	Quahog clam	---[b]	0.010[c]	U.S. EPA (1997)
Crassostrea gigas	Oyster	---[b]	0.014[c]	U.S. EPA (1997)
Acartia tonsa	Copepod	---[b]	0.016[c]	U.S. EPA (1997)
Nucella lapillus	Snail	---[b]	0.016[c]	Harding et al. (1995)
Ostrea edulis	European oyster	---[b]	0.020[c]	U.S. EPA (1997)
Dendraster sp.	Sand dollar	0.465	0.0317	Parametrix (1995)
Acanthomysis sculpta	Mysid	0.506	0.0345	U.S. EPA (1997)
Metamysidopsis elongata	Mysid	0.973	0.0662	U.S. EPA (1997)
Gammarus sp.	Amphipod	1.30	0.0885	U.S. EPA (1997)
Mytilus sp.	Bay oyster	1.43	0.0977	U.S. EPA (1997); Battelle (1990)
Oncorhynchus tshawytscha	chinook salmon	1.46	0.0994	U.S. EPA (1997)
Mysidopsis bahia	Mysid	1.69	0.115	U.S. EPA (1997)
Homarus americanus	American lobster	1.75	0.119	U.S. EPA (1997)
Eohaustorius estuarius	Amphipod	1.79	0.122	Meador et al.(1993); Meador (1993)
Nitocra spinipes	Copepod	1.91	0.130	U.S. EPA (1997)
Eurytemora affinis	Copepod	1.97	0.134	U.S. EPA (1997)
Arenicola cristata	Lugworm	5.03	0.342	U.S. EPA (1997)
Menidia sp.	Silverside	5.17	0.352	U.S. EPA (1997)
Brevoortia tyrannus	Atlantic menhaden	5.20	0.354	U.S. EPA (1997)
Neanthes arenaceodentata	Polychaete	6.81	0.464	U.S. EPA (1997)
Cyprinodon variegatus	Sheepshead minnow	9.04	0.615	U.S. EPA (1997)
Carcinus maenas	Shore crab	9.73	0.662	U.S. EPA (1997)
Branchiostoma caribaeum	Amphioxis	10.0	0.681	U.S. EPA (1997)
Palaemonetes pugio	Grass shrimp	11.2	0.765	Khan et al. (1993)
Orchestia traskiana	Amphipod	14.6	0.994	U.S. EPA (1997)
Fundulus heteroclitus	Mummichog	21.2	1.45	U.S. EPA (1997)
Rhithropanopeus harrisii	Mud crab	34.9	2.38	U.S. EPA (1997)
Rhepoxynius abronius	Amphipod	47.7	3.25	Meador et al.(1993)
Hemigrapsus nudus	Shore crab	83.3	5.67	U.S. EPA (1997)

[a] Unless otherwise noted, the chronic value was estimated from the genus mean acute value using an acute-chronic ratio of 14.69. (U.S. EPA 1997)

[b] Not needed as actual chronic values were available and no exceedances of TBT acute criteria were observed.

[c] This value is a measured chronic value (i.e., it was not estimated using an acute-chronic ratio).

Figure 2-6. Chronic EEC and Marine Toxicity Distributions for TBT

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To probabilistically estimate these risks requires fitting a probability distribution to the toxicity data for each chemical. This probability distribution is a mathematical model that describes the inter-genera variability in toxicity across all aquatic species. Research has indicated that the logistic regression model is one of the most appropriate distributions for these types of data (Aldenberg and Slob 1993, WERF 1996) and this model was used to fit the toxicity data.

The distributions of toxicity data for each chemical are shown in Figure 2-1 through Figure 2-6. Figure 2-1 through Figure 2-6 may be interpreted as showing the percent of aquatic species that are effected for any particular chemical concentration. For example, Figure 2-1 shows that at 1,000 μg/L arsenic, about 35 percent of marine species are predicted to be acutely effected.

2.3 Physical Stressor Effect Thresholds

Physical stressors evaluated for risks to aquatic life included sediment effects (TSS, sedimentation, and scouring), salinity effects, dissolved oxygen (DO), pH, temperature, and displacement caused by increases in water velocity.

2.3.1 Sediment Effects

Several studies were reviewed to determine total suspended solids (TSS) and sedimentation effects on fish and invertebrate communities and individual species. Each study was evaluated for appropriateness and quality for all data. For example, data were rejected for inadequate documentation of the health of control specimens. Studies passing this data review were used to derive effects criteria for TSS and sedimentation (see Issue Paper No. 5 - *Physical Stressors* in Appendix C for further details). Based on the distribution of sensitivities to sedimentation rates and TSS, the lowest TSS concentrations or sedimentation rates expected to protect 95, 90, 85, and 75 percent of the exposed aquatic species were calculated. In order to overcome differences in test durations, a stress index (Newcombe and MacDonald 1991) was created using the following formula:

$$\text{Stress Index} = \ln[\text{Test Duration (Hours)} \times \text{Acute Toxicity (mg/L)}]$$

The data used to calculate the acute (mortality) and chronic (reduced growth) TSS stress indices are presented in Table 2-15 and Table 2-16. The resulting stress indices protecting different percentages of exposed species are presented in Table 2-17. Similarly, sedimentation rates that are protective of the stated percentages of benthic species are presented in Table 2-18.

The effect of scouring (loss of sediment by erosion) was determined to be the loss of species associated with specific layers of sediment (see Issue Paper No. 5, Appendix C). For example, an increase of velocity resulting from a CSO discharge that removed one centimeter of sediment would remove all the animals associated with that one centimeter layer. This would represent a loss to the benthic community of the roles provided by these animals. To identify which species these could be, we evaluated the depth of sediment exploited by commonly encountered Duwamish River benthic species. A review of the benthic community assessment conducted at the Duwamish/Diagonal CSO/storm drain and Kellogg Island (Section 7) indicated that the species identified in Table 2-19 could be at risk from scouring.

Table 2-15. Acute Data Used to Calculate Effects Threshold for TSS, Sorted by Stress Index

Rank	Stress Index	Test Duration (hours)	Acute Toxicity (mg/L)	Species Tested
1	8.3	4	1,000	Algae (*Chlorella* sp.)
2	8.3	4	1,000	Algae (*Monochrysis lutheri*.)
3	9.2	96	100	American shad (*Alosa sapidissima*)
4	9.5	24	570	Atlantic silverside (*Menidia menidia*)
5	9.6	20	750	White perch (*Morone americana*)
6	9.9	24	800	Menhaden (*Brevoortia tyrannus*)
7	9.9	24	800	Bluefish (*Pomatomus saltatrix*)
8	10.4	48	670	Spot (*Leiostomus xanthurus*)
9	10.5	72	500	Striped bass (*Morone saxatilis*)
10	10.8	96	488	chinook salmon (*Oncorhynchus tshawytscha*)
11	10.8	96	500	Yellow perch (*Perca flavescens*)
12	10.8	96	509	coho salmon (*Oncorhynchus kisutch*)
13	10.9	24	2,310	Bay anchovy (*Anchoa mitchilli*)
14	11.1	6	11,100	Pacific oyster (*Crassostrea gigas*)
15	11.5	96	1,047	Chum salmon (*Oncorhynchus keta*)
16	12.0	24	6,800	Bluefish (*Pomatomus saltatrix*)
17	12.5	24	11,400	Croaker (*Micropogon undulatus*)
18	13.3	24	23,770	Striped killifish (*Fundulus majalis*)
19	13.3	24	24,470	Mummichog (*Fundulus heteroclitus*)

Table 2-16. Chronic Data Used to Calculate Effects Threshold for TSS, Sorted by Stress Index

Rank	Stress Index	Test Duration	Chronic Toxicity NOEC or LOEC (mg/L)	Species Tested
1	8.2	15	250	Sea urchin (*Anthocidaris crassispina)*
2	11.1	240	270	Rainbow trout (*Oncorhynchus mykiss)*
3	11.9	504	300	Pacific salmon (*Oncorhynchus sp.)*
4	11.9	144	1,000	Rainbow trout (*Oncorhynchus mykiss*)
5	12.2	240	850	Rainbow trout (*Oncorhynchus mykiss*)
6	13.2	2,688	200	Rainbow trout (*Oncorhynchus mykiss*)

Table 2-17. Acute and Chronic Stress Indices Used to Evaluate TSS Effects

Stress Index[a]	Percent of Species Protected
Acute Effects	
8.1	95%
8.7	90%
9.1	85%
9.6	80%
Chronic Effects	
7.7	95%
8.7	90%
9.3	85%
10.1	80%

[a] The Stress Index is the natural log of the exposure duration (in hours) multiplied by TSS concentration in mg/L.

Table 2-18. Chronic Effect Threshold for Sedimentation Rates

Percent of Species Protected	Chronic Effect Threshold (mm/month)
95%	21
90%	37
85%	47
75%	60

Table 2-19. Scouring Effect Thresholds and Species Associated With Each Sediment Layer

Taxon	Range of Sed. Depth (cm) that Species are Found		Food Habitats
	Upper	Lower	
Chironomidae	0	0	Filters water column/sediment surface feeder
Epitonium sp.	0	0	Predator
Cumella vulgaris	0	0	Surface detrital feeder
Eudorella pacifica	0	0	Surface detrital feeder
Euphilomedes carcharodonta	0	0	Surface detrital feeder
Euchone sp.	0	0.5	Filters water column
Manayunkia aestuarina	0	0.5	Filters water column/sediment surface feeder
Pseudeopolydora kempi	0	0.5	Filters water column/sediment surface feeder
Pygospio elegans	0	0.5	Filters water column/sediment surface feeder
Corophium salmonis	0	0.5	Surface detrital feeder
Corophium spinicorne	0	0.5	Surface detrital feeder
Hobsonia florida	0	0.5	Surface detrital feeder
Oligochaeta	0	1	Sediment feeder
Eogammarus confervicolus	0	1	Surface detrital feeder
Capitella capitata	1	2	Sub-surface sediment feeder
Axinopsida serricata	2	2	Surface detrital feeder
Grandidierella japonica	0	2	Surface detrital feeder
Psephedia lordi	0	2	Surface detrital feeder
Aphelochaeta sp.	0	3	Surface detrital feeder
Macoma carlottensis	1	4	Surface detrital feeder
Cossura pygodactylata	0	5	Sub-surface deposit feeder
Scoletoma luti	1	5	Sub-surface sediment feeder
Neanthes sp.	0	5	Surface detrital feeder
Parvilucina tenuisculpta	0	5	Surface detrital feeder
Clinocardium sp.	0	6	Filters water column/sediment surface feeder
Heteromastus sp.	0	15	Sub-surface sediment feeder

2.3.2 Salinity Effects Thresholds

A salinity effects threshold was established at five ppt for stenohaline organisms (see Issue Paper No. 5, Appendix C for details). This threshold was selected because most freshwater fishes are not found at salinities above three to five ppt (Moyle and Cech 1988). Also, many species of marine organisms cannot tolerate estuarine situations or low salinities. This threshold was used to evaluate the percent of time that each model cell was below this threshold during a year, as well as the maximum number of contiguous days below this criterion.

2.3.3 Dissolved Oxygen Effects Thresholds

Exposure to low DO concentrations can result in adverse effects (mortality and reduced growth) to aquatic life. Thus, DO effect thresholds are minimums not maximums. The State of Washington has established a series of DO criteria based on the classification of surface water bodies (WSDOE 1995b). The Duwamish River has been designated Class B – Good. The DO criteria for the Duwamish River (Freshwater Class B) and Elliott Bay (marine waters) are:

- Freshwater Class B — DO shall exceed 6.5 mg/L
- Marine Water — DO shall exceed 5.0 mg/L. When natural conditions, such as upwelling, occur, the DO can be degraded by up to 0.2 mg/L by human caused activities.

These criteria were used to evaluate field-sampling data to determine the risk to aquatic life from reductions in DO.

2.3.4 Water Column Acidity (pH) Effects Thresholds

pH has both maximum and minimum criteria for the protection of aquatic life. The State of Washington has established a series of pH criteria based on the classification of surface water bodies (WSDOE 1995b). The pH criteria for the Duwamish River (Freshwater Class B) and Elliott Bay (marine waters) are:

- Freshwater Class B — pH shall be within the range of 6.5 to 8.5
- Marine Water — pH shall be within the range of 7.5 to 8.5

These criteria were used to evaluate field-sampling data to determine the risk to aquatic life from changes in pH.

2.3.5 Water Column Temperature Effects Thresholds

The State of Washington has established a series of temperature criteria based on the classification of surface water bodies (WSDOE 1995b). The temperature criteria associated with the Duwamish River (freshwater Class B) and Elliott Bay (marine waters) are:

- Freshwater Class B — Temperature shall not exceed 21 ^{o}C
- Marine Water — Temperature shall not exceed 19 ^{o}C

These criteria were used to evaluate the field-sampling data to determine the risk to aquatic life from changes in temperature.

2.3.6 Water Velocity/Displacement Effects Thresholds

Sustainable swimming speeds of 0.2 to 0.7 m/s were established from literature studies of coho salmon (Table 2-20). Increases in water velocity resulting from a CSO discharge that exceed these speeds could result in the displacement of fish to areas where acute (lethal) and chronic (sub-lethal) effects from other stressors (see Appendix C – *Issue Papers* for further discussion). This range of velocities were used to establish an effects threshold of 1.0 m/s to evaluate the estimated centerline plume velocity during a CSO discharge.

Table 2-20. Reported Sustainable Swimming Speeds for Coho Salmon *Oncorhynchus kisutch* Smolts

Life Stage/Fish Type	m/s[a]	Reference
Smolt (freshwater)	0.4	Flagg and Smith (1982)
Smolt (saltwater)	0.2	Flagg and Smith (1982)
Smolt (freshwater)	0.2	Smith (1982)
Wild fish (freshwater)	0.7	Brauner et al. (1994)
Hatchery fish (freshwater)	0.7	Brauner et al. (1994)
Wild fish (saltwater)	0.6	Brauner et al. (1994)
Hatchery fish (saltwater)	0.5	Brauner et al. (1994)
Smolts	0.6	Glova and McInerney (1977)

[a] Meters per second

2.4 Effects Characterization Uncertainty

Many factors in the effects characterization limit the influence uncertainties have in the development of the TRVs. For example, the water column TRVs used in the aquatic effects characterization are based on data that were screened against U.S. EPA guidelines for test acceptability (Stephan et al. 1985), as we described above in the effects characterization methodology. The TRVs are based on data searches of U.S. EPA ambient water quality criteria documents, EPA's AQUIRE database, and the scientific literature, so they are comprehensive. The WERF (1996) methodology used in the aquatic ecological effects characterization is peer-reviewed and generally accepted by aquatic ecological risk assessors. Even with this degree of confidence in the effects characterization, some uncertainties remain worthy of mentioning, including:

- Some stressors evaluated in the water quality assessment have not been tested for toxicity.
- Not all WQA receptor species have been tested with every stressor of interest.
- Only a limited range of concentrations/doses and exposure durations have been tested for some stressor evaluated in the WQA.
- A limited range of effects/endpoints has been evaluated for some of the stressors evaluated in the WQA.
- Extrapolation from laboratory to field conditions.

These sources of uncertainty have been discussed in detail in Issue Paper #6 - *Aquatic Life and Wildlife Toxicology* (Appendix C).

Another uncertainty in the aquatic life chemical effects characterization for the water column has to do with endpoints not evaluated in the Water Quality Assessment due to insufficient data. This type of uncertainty was recently highlighted by Arkoosh et al. (1998) who show immunosuppression in salmon smolt from the Duwamish Estuary (i.e., in our study area) relative to Nisqually estuary smolt. However, the cause of the observed immunosuppression has not been determined, so it cannot at this time be causally linked to any particular stressor or stressors, nor has it been linked to a population-level effect.

The sediment effects characterization uses Washington State sediment management standards as TRVs. The sediment management standards generally are not based on established cause and effect relationships, but instead on Apparent Effects Thresholds

(AETs) [14]. This reflects the fact that sediments are a complex environment in which it is difficult to establish causal relationships for toxic effects (Adams et al. 1992, Allen 1995, Ingersoll et al. 1997, National Research Council 1997). Therefore, it often is necessary to use observed correlations between stressors and effects to estimate TRVs. Uncertainty about sediment TRVs is higher than uncertainty about water column TRVs, which are based on controlled experiments that meet EPA guidelines for test acceptability. Sediment TRVs are conservative, because they essentially assume a stressor is causing risk if it is present where risks are observed or predicted. In the first volume of this report (Overview and Interpretation), we evaluated the reliability of the sediment TRVs for chemicals with sediment concentrations above the TRVs anywhere in the Duwamish Estuary.

Some sediment standards are more reliable (less uncertain) than others, because multiple lines of evidence give similar TRV estimates. PAHs and mercury are two chemical stressors for which we have reliable TRVs. Sediment management standards for PAHs are based on the oyster larval[15] and Microtox[16] AETs. The AETs generally are similar to other toxicity threshold values (Effects Range – Low (ER-L), Effects Range – Median (ER-M) (Long et al. 1995) and equilibrium partitioning-derived values (Di Toro et al. 1991)), so we consider them to be reasonably reliable TRVs. The sediment management standard for mercury falls between the ER-L and ER-M and is within a factor of three of the Ecotox EqP threshold. Because all the AETs for mercury are within a factor of three, we consider the sediment management standard for mercury to be a reliable TRV.

Examples of less reliable (more uncertain) sediment TRVs include those for 1,4-dichlorobenzene and bis(2-ethylhexyl)phthalate. The sediment management standard for

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14 The AET approach was developed specifically to assess and manage the quality of sediments in Puget Sound. It uses empirical data (field and laboratory) to identify concentrations of chemicals above which biological effects are always expected. AET values are derived using a comparison of biological effects and chemical data in paired data sets from field-collected samples. In a given data set, the AET for a particular chemical is the sediment chemical concentration above which biologically adverse effects are always observed (based on statistical significance, $p < 0.05$) relative to an appropriate reference sediment (Adams et al. 1992).

15 The test sediment has a mean survivorship of normal larvae that is less (statistically significant, *t*-test, $p<0.05$) than the mean normal survivorship in the reference sediment and the test sediment mean normal survivorship is less than eighty-five percent of the mean normal survivorship in the reference sediment (i.e., the test sediment has a mean combined abnormality and mortality that is greater than fifteen percent relative to time-final in the reference sediment) (Ch 173-204 WAC, page 17).

16 The mean light output of the highest concentration of the test sediment is less than 80 percent of the mean light output of the reference sediment, and the two means are statistically different (*t*-test, $p<0.05$) from each other (Ch 173-204 WAC, page 17).

1,4-dichlorobenzene is based on the AET for benthic invertebrate abundance.[17] AET does not establish a causal relationship between a stressor and an effect (e.g., between 1,4-dichlorobenzene and reduced benthic abundance); it establishes a correlation based on field observations (Spies 1989). The AET method cannot separate the effects of individual stressors when multiple stressors are present (Adams et al. 1992). For example, one would expect sediment-bound chemicals from CSOs to be correlated with physical changes in sediment particles, which could be the cause of an apparent effect like reduced benthic abundance. Studies to date of 1,4-dichlorobenzene generally have been observational and correlational (Chapman et al. 1996), and direct experimental evidence demonstrating that 1,4-dichlorobenzene in sediments causes risks is lacking. The water toxicity database for 1,4-dichlorobenzene is limited as well; U.S. EPA's criterion document contains only two data points: an acute LC_{50} for sheepshead minnow, and an acute LC_{50} for mysids. Searches of the aquatic toxicology literature revealed no additional 1,4-dichlorobenzene aquatic toxicity data.

The bis(2-ethylhexyl)phthalate sediment management standard is based on the Microtox bacterial luminescence bioassay[18] (47 mg/kg[19]), although the benthic abundance AET is only slightly higher at 60 mg/kg. In the case of bis(2-ethylhexyl) phthalate, the TRV estimated from the proposed U.S. EPA water quality criterion by equilibrium partitioning theory is approximately 700 times the sediment management standard, suggesting the sediment management standard may under estimate the toxic effects threshold (and therefore over estimate risk) for bis(2-ethylhexyl)phthalate.

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17 Test sediment has less than 50 percent of the reference sediment mean abundance of any one of the following major taxa: Class Crustacea, Phylum Mollusca or Class Polychatea, and the test medium abundance is statistically different (*t*-test, $p < 0.05$) from the reference sediment abundance (Ch 173-204 WAC, page 17).

18 The mean light output of the highest concentration of the test sediment is less than 80 percent of the mean light output of the reference sediment, and the two means are statistically different (*t*-test, $p<0.05$) from each other (Ch 173-204 WAC, page 17).

19 Normalized to organic carbon.

3. Methods and Assumptions Used in Characterizing Exposure

3.1 Aquatic Life Exposure to Chemicals

This section summarizes the methods used to determine the aquatic life estimated exposure concentrations (EECs) for COPCs in the Duwamish River and Elliott Bay. The Duwamish River and Elliott Bay water quality model divides the study area into 512 grid cells. For aquatic life, the 512 grid cells were separated into two patches: those above the downstream end of Harbor Island (the Duwamish River patch, consisting of 129 cells), and those below it (the Elliott Bay patch, consisting of 214 cells). The 169 model cells not included in either the Duwamish River or Elliott Bay patch occur either upstream of the Turning Basin, or west of Duwamish Head. The cells west of Duwamish Head were included in the model as a buffer against Puget Sound boundary effects. The cells above the Turning Basin were included in a separate screening-level assessment of risks from possible future peak flow discharges of treated effluent from the East Treatment Plant at Renton, Washington (Simmonds et al. 1998). These upstream cells were not evaluated in this risk assessment because they are upstream of the most upriver King County CSO on the Duwamish River (the Norfolk CSO).

Each model grid cell was further divided into 10 surface water layers and one sediment layer, resulting in 1,290 water column and 129 sediment “grid elements” in the Duwamish River patch, and 2,140 water column and 214 sediment grid elements in the Elliott Bay patch. The model predicted concentrations for each COPC in each of these grid elements every 15 minutes for a year. The simulated year was a composite of measured flows from October 1996 to June 1997, and simulated discharges for July 1997 to September 1997 generated from July 1981 to September 1981 rainfall data, as described in the Duwamish River and Elliott Bay modeling report. The modeling report is presented in Appendix B-1.

The significant amount of output generated by the EFDC model (reporting 5,120 data points every hour for 365 days) required that this information be further summarized—referred to as post-processing—prior to use in the risk assessment. For the water column, the post-processing program computed peak concentrations, averaged over acute and chronic exposure durations, for each month of the simulation. For the sediment layer, the post-processing program also computed peak concentrations for each month of the simulation, but only for chronic exposure durations, because sediment concentrations did not vary over time periods shorter than the chronic exposure duration.

The exposure durations used for assessing acute and chronic risks were one hour and four days, respectively, following the U.S. Environmental Protections Agency’s *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic*

Organisms and Their Uses (Stephan et al. 1985). The reasons for using a one-hour averaging period for acute exposure are as follows:

- Some stressors, for example ammonia and low DO, are fast acting toxicants. For these stressors exposure to acutely toxic concentrations for about one to three hours is sufficient to cause death.
- Even for substances that are not fast acting toxicants, organisms may suffer delayed effects from one-hour exposure to acutely toxic concentrations. Therefore, a one-hour averaging period provides an accurate-to-conservative (depending on the stressor) exposure duration for comparing to acute TRVs.

The reasons for using the maximum four-day averaging period in each month for chronic exposure are as follows:

- The averaging period should be shorter than the duration of chronic toxicity tests (20 to 30 days) because substantial fluctuations within the test period result in increased adverse effects.
- The results of chronic toxicity tests are, at least in some cases, determined by a sensitive life stage occurring during the chronic test. It is reasoned that a four-day averaging period is probably sufficiently short to prevent increased adverse effects on sensitive life stages.

The decision to focus on monthly peaks was based on the following considerations:

- The ecosystem doesn't have a chance to recover between events that occur close together in time; so we assumed that the maximum impact to aquatic life during a month would apply to the entire month. The time required for the Duwamish River and Elliott Bay to recover from peak exposures (assuming the peak exposures are high enough to cause risks) is unknown. Some recovery probably would occur within a small number of tide cycles, full recovery only after several to many seasons. These recovery issues were discussed in Issue Paper No. 9 – *Risk Predictions and Aquatic Community Responses*.
- Assessing EECs on a monthly basis allowed us consideration of potential seasonal changes in the ecosystem, for example, whether or not migratory species would be in the Duwamish River or Elliott Bay when EECs were high enough to pose risks. These seasonal issues were discussed in Issue Paper No. 2 - *Aquatic Life and Wildlife Site Use.*
- Assessing EECs on a monthly basis gave us a representative sampling of the seasonally varying rainfall and flow conditions in the Duwamish River and Elliott Bay watershed.

During the wet season, the Duwamish Estuary stratifies into a freshwater lens on the surface, with a saline layer below. Marine and estuarine organisms tend to avoid the freshwater lens, for example by closing their shells, swimming away, or sinking to more saline waters below the freshwater surface layer. By avoiding the freshwater, organisms will avoid exposure to any COPCs found therein. Conversely, for organisms that fail to avoid the freshwater, risk from osmoregulatory failure could override risk from exposure to COPCs. Therefore, we only analyzed EECs that occurred when salinity was greater than five parts per thousand, which in this situation represents a threshold between freshwater and saline conditions.

The EFDC model predicted water column concentrations in the 1-year model simulation for baseline conditions as well as for without CSO contributions. Acute and chronic chemical concentrations in the Duwamish River and Elliott Bay water column are summarized in Table 3-1 through Table 3-28 for baseline conditions. Chemical concentrations under without-CSO conditions are not presented because few substantial differences were observed between the baseline and without CSO conditions. For each chemical, the average, standard deviation, minimum, median, and maximum acute and chronic concentrations across all cells in the Duwamish River and Elliott Bay patches are reported for each month. For the metals evaluated, both total and dissolved concentrations are presented. Total metal concentrations are conservatively used in the Tier 1 evaluation, while dissolved metal concentrations are used in the Tier 3 evaluation. Dissolved metal concentrations are generally believed to be more strongly related to toxicity than total metal concentrations (Prothro 1993).

The EFDC model also predicted sediment concentrations in the 1-year model simulation for baseline conditions as well as for without CSO conditions. Chronic chemical concentrations in sediments throughout the study area at the end of the 1-year model simulation under baseline conditions are summarized in Table 3-29 for each COPC evaluated. However, in contrast to water column concentrations, changes in sediment chemicals over time are much less dynamic and, therefore, harder to detect over a 1-year time period. Consequently, a 10-year model simulation of seven chemicals[20] was conducted to determine if differences in baseline and without CSO sediment concentrations increased over a longer time period. These chemicals were selected based on the level of risk they posed to aquatic life in study area sediments. As stated above, sediment concentrations were predicted in a single layer in each cell of the study area. These concentrations were represented by the highest four-day running average in each

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[20] The seven chemicals were 1,4-dichlorobenzene, bis(2-ethylhexyl)phthalate, chrysene, copper, lead, mercury, and total PCBs. Chrysene was selected as a representative PAH value based on its high correlation with other PAH concentrations ($r^2 \geq 0.8$). The number of chemicals was limited to seven to reduce the time required to complete the ten year simulation. Even so, the ten-year simulation required 60 days to run.

month for each study area cell. These values were selected because only the chronic exposure of aquatic life to sediment was assessed (sediments being long-term integrators of chemical exposure).

3.2 Juvenile Salmon Dietary Exposure

To determine dietary risks to juvenile salmon, composited samples of gammarid amphipods (*Eogammarus* and *Corophium*) were collected in late July 1998 on the west side of Kellogg Island and from the beach adjacent to Kellogg Island east of West Marginal Way in the Duwamish River. A parallel collection of amphipods was conducted in early August 1998 at the mouth of the Nisqually River. At each location, amphipods were collected by hand or by screening surface sediments from the lower intertidal zone (-1 to +3 feet MLLW). The Kellogg Island composited samples consisted of approximately 87 percent *Eogammarus* and 13 percent *Corophium* while the Nisqually River composited sample consisted of approximately 80 percent *Eogammarus* and 20 percent *Corophium*.

Kellogg Island amphipods represent prey items that would be consumed by juvenile salmon outmigrating from the Green/Duwamish watershed (Leon 1980; Meyer et al. 1980; Parametrix 1990). Nisqually Delta amphipods would represent similar prey items from a relatively unimpacted watershed. These amphipods were identified by Kevin Li of the King County Environmental Laboratory and analyzed chemically for those Tier 1 chemicals that potentially posed risks to aquatic life in the water column. Amphipod chemical concentrations and associated method detection limits are presented in Table 3-30 for those chemicals with dietary TRVs available (see Section 2.2).

Table 3-30. Concentrations (µg/kg) of the Chemicals of Potential Concern[a] in Gammarid Amphipods from the Study Area and a Reference Site

Parameters	Kellogg Island		Nisqually Delta	
	Sample #1	Sample #2	Sample #1	Sample #2
Copper, Total (mg/kg)	9,770	11,900	4,160	4,130
Lead, Total (mg/kg)	1,310	952	164	175
Aroclor 1254 (µg/kg)	36.3	48.7	4[b]	4[b]
Aroclor 1260 (µg/kg)	43.1	50.8	4[b]	4[b]
Zinc, Total (mg/kg)	7,860	9,300	4,790	4,790

[a] Only chemicals with available dietary TRVs are reported here.

[b] Chemical not detected. Concentration equal to one-half of the MDL reported.

3.3 Estimation of Sediment PAH Exposure to Predict English Sole Liver Lesions

Liver lesions in English sole are a biomarker of exposure to PAHs in sediments (Johnson et al. 1998). Specifically, elevated occurrences of liver lesions have been associated with exposure to PAHs in enclosed embayments in the Puget Sound area as well as in other areas of the coastal waters of the United States (Myers et al. 1994; Johnson et al., 1998). Research conducted by the Northwest Fisheries Science Center has established a predictive relationship between bulk sediment PAH concentrations and the prevalence of a number of different types of liver lesions (Horness et al. 1998). This relationship was used to predict the prevalence of liver lesions in English sole populations that would be exposed to study area sediments, as described in Table 3-31. The prevalence of liver lesions was calculated using the monthly average concentration of PAHs for both the Duwamish River and Elliott Bay. Individual PAH concentrations were combined using the rules presented below:

1. Convert mg-PAH per kg-dry weight sediment concentrations to nanogram PAH per gram dry weight sediment by multiplying each value by 1,000.
2. Calculate total PAH concentration as the sum of the benzo(k)fluoranthene, phenanthrene, chrysene, benzo(b)fluoranthene, benzo(a)anthracene, benzo(a)pyrene benzo(g,h,i)perylene, dibenzo(a,h)anthracene, indeno(1,2,3-cd)pyrene concentrations (nanogram PAH per gram dry weight sediment). Adjust for missing PAHs[21] by adding 50 to resulting value, and then dividing by 0.9.
3. Calculate the $\log_{10}$(total PAH dry weight concentrations).
4. Calculate incidence of liver lesions for each lesion type listed in Table 3-31 using the following:
    - If $\log_{10}$ (total PAH dry weight concentrations) are less than or equal to the threshold given in Table 3-31, then prevalence of liver lesions equals reference levels.
    - If $\log_{10}$ (total PAH dry weight concentrations) is greater than the threshold, then incidence of liver lesions (ILL) = slope x $\log_{10}$(PAH) + y-intercept.

[21] By missing PAHs, we mean that the King County WQA list of PAHs was shorter than the list used by Horness et al. (1998) to develop this PAH-liver lesion relationship. The specific adjustment factors reported in Step 2 were calculated for us by Beth Horness, Northwest Fisheries Science Center, NOAA.

The slopes and intercepts for the different lesion types are provided in Table 3-31.

Table 3-31. Regression Equation Values Between Sediment PAH Concentrations and Incidence of Liver Lesions Used to Calculate the Prevalence of Liver Lesions in English Sole

Lesion Type	Threshold	Reference	Slope	Y-Intercept
Neoplasms	3.45	0.004	0.1	-0.341
Foci of Cellular Alteration	1.74	0.008	0.04	-0.062
Specific Degenerative/Necrotic Lesions	2.97	0.013	0.37	-1.086
Megalocytic Hepatosis	2.97	0.002	0.21	-0.622
Nuclear Pleomorphism	2.97	0.001	0.3	-0.890
Proliferative Lesions	2.37	0.024	0.09	-0.189
Any Lesion = Neo or FCA or SDN	2.79	0.024	0.31	-0.841

Source: Horness, et al. (1998)

The predicted prevalence of liver lesions, under baseline conditions, without CSOs and the naturally occurring rates of liver lesion formation present in populations not exposed to sediment PAHs (Horness et al. 1998) are presented in Table 3-32. Elevated liver lesions are predicted for the Duwamish River and Elliott Bay, both baseline and without CSOs. It is interesting to note that we did see a difference between baseline and without CSO biomarkers based on the one-year model simulation, indicating that CSOs are a source of PAHs to the study area. We further evaluated PAHs in sediment by including chrysene in the ten-year simulation[22]. These results show that the magnitude of the difference in chrysene concentrations between baseline and without CSO conditions was smaller[23] after the ten-year simulation than after the one-year simulation[24]. Therefore, the

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22 We chose chrysene for the ten-year comparison because the SPMD data showed the highest average concentration in the water column, and sediment data showed good spatial correlation ($r^2 > 0.8$) of chrysene with all the other measured PAHs (phenanthrene, pyrene, benzo(k)fluoranthene, chrysene and benzo(b)fluoranthene).

23 Statistically significant using a two-sided Wilcoxon signed rank test, $P = 0.05$.

differences between baseline and without CSO incidences of liver lesions shown in Table 3-32 would decrease as well.

Table 3-32 Predicted Incidence of English Sole Liver Lesions[a]

Specific Liver Lesion Types Formed by English Sole	Baseline, Annual Average[b]	Without CSO, Annual Average[b]	Change in Prevalence with CSO Removal	Percentage of Unexposed Populations with Liver Lesions[c]
Neoplasms	10%	9%	1%	0.40%
Foci of Cellular Alteration	11%	11%	0%	0.80%
Specific Degenerative/ Necrotic Lesions	53%	50%	3%	1.30%
Megalocytic Hepatosis	30%	28%	2%	0.20%
Nuclear Polymorphism	42%	40%	2%	0.10%
Proliferative Lesions	21%	20%	1%	2.40%
Risk of Forming Any Lesion	52%	49%	3%	2.40%

a Each column is the percent of the population predicted to develop a specific type of liver lesion.

b These data are predicted by the model developed by Horness et al. 1998 using sediment data from the EFDC model.

c Data directly taken from Horness et al. 1998.

These predictions, based on the model of Horness et al. (1998), are somewhat higher than those reported by Johnson et al. (1998) in Elliott Bay English sole, applying the same Horness et al. (1998) model. Predicted neoplasms were approximately 10 percent, whereas the observed incidence reported by Johnson et al. (1998) for Elliott Bay English sole was 3 percent. Predicted specific degenerative/ necrotic (SDN) lesions were approximately 53 percent, whereas the observed incidence (Johnson et al. 1998) was 22 percent. The incidence of liver lesions has not been correlated with any population-level effects on English sole, but it is a biomarker of English sole exposure to PAHs in Duwamish River and Elliott Bay sediments.

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24 Again, the purpose of the ten-year simulation was to determine whether baseline versus without CSO differences were increasing or decreasing over time.

3.4 Physical Stressors

Evaluating the risks to aquatic life from physical stressors presents a unique set of issues. This is because physical stressors have rarely been evaluated in risk assessments, which have traditionally focussed on chemicals released to the environment by human activities. To identify risks to aquatic life from physical effects, exposure was estimated either from project team's knowledge of the river (qualitative), from the WQA field sampling program, or from the EFDC hydrodynamic model. Exposure to physical stressors are discussed below.

3.4.1 Suspended Solids – TSS/Scouring/Sedimentation

During CSO events, inorganic and organic particulate matter is discharged to the Duwamish Estuary. This particulate matter is composed of both settleable solids and TSS. Settleable solids are larger, heavier particles (e.g., sand) that quickly settle to the streambed (e.g., p. 2-57 of APHA 1995). Conversely, TSS are smaller, lighter particles (e.g., silt and clay) that remain suspended for a longer period (e.g., see p. 2-56 of APHA 1995).

Sedimentation (sediment deposition) is the settling of solids at the sediment-water interface and is measured as the depth of solids accumulating over time. Sediment deposition is a direct measure of how solids can cover and subsequently smother benthic organisms. Scouring is the removal of sediment from existing habitat by currents that resuspend and move sediment downstream. Sedimentation and scouring affect epibenthic and infaunal species. When sediment is deposited at high rates, sessile and slow moving species can be smothered. Alternatively, benthic organisms can be displaced and exposed to predation by scouring and loss of sediment habitat. Organisms may also leave the area where there is a high sediment deposition rate.

Evaluating risks to aquatic organisms in the Duwamish Estuary from TSS in CSO discharges as well as scouring and sedimentation rates required information on the magnitude of rate increases of TSS, scouring, and sedimentation. Data sources were measurements of CSO discharges and river concentrations for TSS, as well as predictions by the hydrodynamic model of TSS concentrations, and scouring/sedimentation rates in specific model cells.

These exposure concentrations were incorporated into summary statistics representing acute and chronic exposures (Table 3-33 and Table 3-34). Acute exposures were calculated as the 95^{th} percentile of the data set. These values represented an upper bound estimate of the exposure an organism could encounter from a CSO discharge. Chronic exposures were calculated as the 95^{th} upper confidence limit of the mean. These values represented upper bound estimates of the average concentration a population of aquatic organisms could encounter over a lifetime. The acute and chronic exposure concentrations were calculated separately for the Duwamish River and Elliott Bay.

Table 3-33. Monthly Sedimentation Rates (mm/day) for the Study Area

Summary Statistic	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug
Average	-0.003[a]	0.036	0.037	0.035	0.095	0.072	0.066	0.064	0.057	0.014	0.005	-0.001
Standard Deviation	0.035	0.136	0.307	0.148	0.634	0.382	0.793	0.409	0.266	0.072	0.028	0.021
Maximum	0.095	1.016	2.197	1.062	7.882	5.912	11.904	6.959	4.095	0.726	0.246	0.093
Median	0.000	0.000	0.000	0.000	0.004	0.001	0.007	0.001	0.001	0.001	0.000	0.000
Minimum	-0.199	-0.571	-1.545	-1.001	-2.008	-0.411	-3.242	-1.453	-0.652	-0.284	-0.110	-0.101

[a] Negative sediments means overall loss of sediment in that month.

Table 3-34. Monthly TSS Stress Indices for the Study Area

Summary Statistic	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug
Acute TSS Stress Indices												
Average	4.4	4.1	4.1	3.8	4.1	4.1	4.1	3.6	3.5	3.6	3.4	3.3
Standard Deviation	1.1	1.2	1.2	0.9	1.5	1.0	1.3	0.8	0.8	0.7	0.7	0.6
Maximum	6.4	9.3	8.8	8.6	8.5	7.8	7.9	7.5	7.5	7.6	7.0	6.6
Median	3.9	3.5	3.5	3.5	3.5	3.5	3.4	3.4	3.4	3.4	3.3	3.2
Minimum	3.3	3.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.1	2.2	2.6
Chronic TSS Stress Indices												
Average	8.7	8.1	7.4	7.3	7.3	7.3	7.4	7.0	7.0	7.2	7.3	7.5
Standard Deviation	0.8	0.5	2.2	2.3	2.6	2.5	2.6	2.6	2.2	2.0	1.6	0.4
Maximum	10.3	11.0	11.0	10.6	10.9	10.2	11.5	10.1	9.8	9.6	9.0	8.8
Median	8.2	7.9	7.9	7.8	7.8	7.9	7.9	7.9	7.7	7.8	7.6	7.5
Minimum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

3.4.2 Reduction in Salinity

During and following rainstorms, large amounts of freshwater enter the Duwamish River from CSO discharges. If these are of sufficient magnitude and duration, the salinity in the river's estuarine reaches and Elliott Bay can decline substantially, creating conditions lethal to stenohaline[25] species and stressful to some euryhaline[26] and immobile species. Differing regions of the U.S. have seen kills of organisms following freshwater inundation of these habitats (Boesch et al. 1976; Jarvis 1979). The following discussion explains how we identified the different salinity regimes occurring in the Duwamish Estuary.

To assess risks to aquatic organisms in the Duwamish Estuary from an influx of freshwater and decline in salinity requires an understanding of the magnitude and duration of declines. With this information, and knowledge of the salinity tolerance ranges of the selected receptors, risks to aquatic organisms can be estimated. Salinity declines in the Duwamish Estuary were estimated with the same model used for chemical stressors. We post-processed the salinity data to estimate the duration of salinity excursions below five parts per thousand (five ppt) in each grid element in the Duwamish River and Elliott Bay patches. The five ppt is a threshold level for most marine invertebrates, which if exceeded for several days or more, will result in stress and even mortality. Simulated salinity was evaluated within the each grid element to determine the percent of time that salinity at particular locations and depths fell below five ppt. As discussed earlier, the salinity risk characterization is qualitative in the sense that a numerical criterion was not established for the percent of time or duration of freshwater incursions that would constitute a risk.

3.4.3 Reduction in Dissolved Oxygen

Reductions in DO may result from the biological oxygen demand (BOD) of particulate matter in CSO discharges (Welch and Lindell 1992). The effects of CSO discharges on DO levels in receiving water may be mediated by the increased river flow of higher DO water, which can occur in conjunction with CSO discharge events (SPCC 1981; Welch and Lindell, 1992). Those waters near and in the sediment typically have the lowest DO concentrations, owing to the degradation of organic matter (sediment oxygen demand). In urban environments, sediment DO is often well below the minimum threshold of 3 mg/L (Davis 1975a,b) due to the cumulative effects of waste and nutrient loading (NOAA 1996; Rabalais and Harper 1992).

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25 Stenohaline organisms can tolerate only a narrow range of salinity. Here, they are mainly marine species colonizing Elliott Bay and the lower reaches of the Duwamish River during times of high stable salinity.

26 Euryhaline organisms can tolerate significant changes in salinity because they are physiologically capable of regulating their ionic balance.

Receptors evaluated in this pathway of the risk assessment exclude fish inhabiting the upper water column, but include benthic invertebrates, bottom-dwelling fish, plankton, and other relatively immobile aquatic organisms. These represent aquatic life that cannot readily avoid areas of low DO by emigrating.

The measures of DO exposure levels are presented in Table 3-35. No DO concentration data are available from the water quality model; the BOD component was not implemented due to programming needs and computing power. Consequently, sampling data in cells which receive CSO discharges were used directly to estimate exposures to reduced DO concentrations.

Table 3-35. Summary Statistics of Temperature, pH, and DO Measurements Made in Cells into which CSOs Discharge[a]

Summary Statistic	Temperature (^{o}C)	pH	DO (mg/L)
Average	9.2	7.7	9.1
Standard Deviation	1.9	0.3	0.9
Maximum	16.1	8.5	12.3
Median	9.1	7.7	8.9
Minimum	4.2	6.4	7.0

[a] Measurements were made adjacent to Brandon, Chelan, Connecticut, Duwamish/Diagonal, Denny Way, Hanford, Norfolk, South Michigan, and West Michigan CSOs. Measurements of pH and temperature were also made at the Tukwila Gauging Station adjacent to the East Division Reclamation Plant.

3.4.4 Change in pH

Changes in pH levels typically are not directly toxic to aquatic life, but instead can result in risk from the effect of pH on other toxicants (U.S. EPA 1986b). Our assessment endpoint for pH was the maintenance of sustainable populations of aquatic life. pH was not an output of the water quality model. Consequently, sampling data from cells which receive CSO discharges were used to directly estimate exposures to high or low pH (Table 3-35).

3.4.5 Change in Temperature

Temperature is a critical measure of the suitability of a particular environment for the presence of specific aquatic life species. Subsequently, temperature changes in water bodies can alter the existing aquatic life community (U.S. EPA 1986b). Both algal and fish communities will change as temperature increases. Sufficient increases in temperature can change a cold water fishery to a warm water fishery. The water quality model did not predict temperature, so field data were used to estimate temperature exposures. Consequently, sampling data from cells receiving CSO discharges were used to directly estimate exposures to high temperatures (Table 3-35).

3.4.6 Displacement

Influx of water from a CSO discharge can result in displacement of organisms from appropriate habitats (based on salinity, temperature and structure providing escape from predators) to habitats that are inappropriate. Centerline plume velocities in the vicinity of CSO discharge locations were predicted by the nearfield component of the water quality model. These velocities were compared to sustainable swimming speeds of salmonid fry and smolts to characterize the risk from increases in river flow.

3.5 Exposure Assessment Uncertainty

The uncertainty in the aquatic ecological exposure assessment is primarily uncertainty about the Duwamish Estuary model. Model uncertainty is discussed in detail in Volume 1 of this report in Section 5.3, *Precautions for Future Investigations*. There are uncertainties about the model – in particular about using the model to make predictions about conditions other than those for which it is calibrated. However, these model uncertainties have limited bearing on the exposure assessment because it is based on the conditions for which the model is calibrated. The EECs used in the risk assessment are monthly maximum acute and chronic values. While exposure estimates for individual locations and times contain uncertainties, the fact that we used monthly maximum acute and chronic concentrations suggests that it is unlikely that we substantially underestimated exposures to aquatic organisms.

Sediment EECs are more uncertain than water column EECs. Two primary uncertainties associated with sediment EECs are: (1) use of the average of all available data within each cell with more than one sediment sample available to estimate the initial conditions in the cell, and (2) use of a linear interpolation scheme to set initial sediment concentrations in model cells without sediment concentration data. Because we used the average concentrations within each cell, the resolution of the model is limited to the size of the cells. This implies that the sediment concentration at any location within the cell may be over- or under-estimated. An example of this may be observed at the footprints near the CSO and stormdrains, where for some chemicals the average cell concentration is lower than the peak concentration near the outfall, but higher than the background concentration at the edge of the footprint.

The use of a linear interpolation scheme could create an over estimation or under-estimation bias in initial sediment concentrations, depending on where sediments have been sampled, and the nature of the sources. The worst case scenario is a localized ("hot spot") contaminant that has only been sampled in the hot spot and at the boundaries of the study area. In this case, the linear interpolation scheme would reduce initial sediment concentration as a function of distance from the hot spot, whereas a more realistic approach would be to reduce initial sediment concentration as a function of the square of the distance from the hot spot. This would cause sediment EECs to be over estimated in the model cells for which sediment data were unavailable.

One way we dealt with the problems we encountered with the sediment initialization was to run the model for ten years, to allow sufficient time for initial sediments to become buried. This allowed us to better discern the difference between baseline and without CSO risks to organisms living on or in the sediment surface layer, because over time new sediments become an increasingly large fraction of the total surface sediment layer, and the importance of the initial conditions diminishes. While this approach was useful for comparing baseline and without CSO risks, a better initialization of the model, coupled with a more thorough hydrodynamic calibration, would clearly be preferred. Nonetheless, the model is useful for investigating the relative importance of CSOs in determining surface sediment concentrations, and it shows that at the level of resolution represented by the model's 512 grid cells, CSOs have little discernable impacts on sediment chemical concentrations.

4. AQUATIC LIFE RISK CHARACTERIZATION METHODS

Potential risks to aquatic life from chemical stressors in the Duwamish River and Elliott Bay surface water and sediment were estimated for both the baseline and without CSO scenarios for two exposure media-sediments and water. Additionally, risk estimates were calculated for reference sites in Puget Sound for use in evaluating study area risks. The risk characterization methods for each exposure medium (water, sediment) are described in detail below.

4.1 Surface Water

The aquatic life risk characterization for surface water consisted of two tiers of the WERF Methodology (WERF 1996). Tier 1 used a quotient approach, while in Tier 3 risks were assessed probabilistically. The purpose of Tier 1 was to eliminate those chemicals in the surface water that clearly did not pose a risk to aquatic receptors. Chemicals that could not be eliminated by this process were evaluated further in Tier 3. We went directly from Tier 1 to Tier 3, skipping WERF Tier 2. The methodology for Tier 2 and Tier 3 are the same, differing only in the use of site-specific data and validation studies. Use of the EFDC computer model provided us site-specific data, and the benthic assessment, and Brandon Street CSO effluent toxicity tests served to corroborate Tier 3 risk predictions. The quotient and probabilistic approaches used in Tier 1 and Tier 3, respectively, are described below.

4.1.1 Tier 1

In evaluating potential risks of surface water chemicals to aquatic organisms using the quotient approach, the surface water concentration for each chemical was divided by its respective TRV. The quotient is often termed the hazard quotient, or HQ:

$$\text{Hazard Quotient} = \frac{\text{Surface Water Concentration}}{\text{Toxicity Reference Value}} \qquad \text{Equation 4-1}$$

As defined earlier, surface water cell concentrations in the Duwamish River and Elliott Bay were represented by the peak monthly one-hour moving average to estimate the reasonable worst-case acute exposures. The peak monthly four-day moving average was used to estimate the reasonable worst-case chronic exposures. The monthly peak average was used to ensure that (1) key migratory species that use Elliott Bay and the Duwamish River only during certain months of the year could be evaluated and (2) to provide seasonal information on how varying rainfall and flow conditions affect aquatic life risk estimates. It was assumed that Duwamish River and Elliott Bay comprise two separate aquatic communities; therefore, HQs were calculated for the cells in each area separately.

A chemical with at least one HQ greater than 1.0 was considered to pose potential risk. Most Washington State and federal water quality standards/criteria are designed to protect 95 percent of aquatic species. Therefore, when a State standard or federal criterion is available, an HQ greater than 1.0 suggests that greater than 5 percent of the species may be at risk for an individual chemical. For chemicals with limited toxicity data available, it becomes less certain what an HQ greater than 1.0 implies (i.e., the risk potential may be over or under estimated). HQs are most useful as a screening tool because they are typically based on a single conservative concentration or a conservative estimate of an average concentration. However, HQs are not adequate for a detailed evaluation because they do not adequately reflect the range of chemical concentrations to which aquatic organisms may be exposed. Additionally, use of total recoverable metal concentrations can significantly over represent the bioavailability fraction in the Tier 1 assessment. For each chemical, acute and chronic HQs were therefore calculated in every cell in the river and bay for each month of the year.

4.1.2 Tier 3

Chemicals identified in Tier 1 with at least one HQ greater than 1.0 were evaluated probabilistically in Tier 3 when sufficient toxicity data were available. In Tier 3, risk was defined as the probability of affecting a given percentage of species. There are different viewpoints on what percentage of the species it is acceptable to affect without negatively influencing overall community function. For example, the Society for Environmental Toxicology and Chemistry recommends a 90 percent level of ecological protection, as do Solomon et al. (1996), while U.S. EPA recommends a 95 percent level (Stephan et al. 1985). If a commercially, economically, recreationally important or threatened or endangered species (e.g., salmon) is among the most sensitive species exposed to a chemical, it may be deemed unacceptable to affect any of the species in the community (Stephen et al. 1985).

In contrast with the point estimates of exposure and effect used in Tier 1, probability distributions were used in Tier 3. A probability distribution of risk was predicted for each aquatic community (Duwamish River and Elliott Bay) using all the modeled data for the site. Additionally, dissolved concentrations of metals estimated by the EFDC model were used in the Tier 3 assessment because they more closely represent the bioavailable fraction (Prothro 1993). The assessment assumed that aquatic receptors were equally likely to be exposed to chemicals in every modeled cell and layer. The comparison of the 1-hour and 4-day average chemical concentrations in each cell and layer with the chemical effects curves estimates the percent species expected to be affected. The estimated risks for each cell and cell layer can also be averaged separately for the Duwamish River and Elliott Bay to provide an estimate of the average risk to the aquatic community. Table 4-1 gives a hypothetical example of how risk was estimated using the modeled concentration data. Averaging risk over the Duwamish River and Elliott Bay was appropriate in determining the risk to populations of aquatic life in these areas.

Table 4-1. Example Probabilistic Risk Calculation

Cell	Max. 1-hour or 4-day Surface Water Conc. (µg/L)	Risk as Percent Species Affected
1	2	1%
2	4	3%
3	2	1%
4	3	2%
5	1	0%
6	5	5%
7	4	3%
8	7	8%
9	2	1%
10	1	0%
Average Risk =		2%

4.2 Salmon Chemical Exposures

The risk characterization for salmon exposure to water column and dietary prey concentrations was done similarly to Tier 1 of the risk characterization for surface water, (i.e., the HQ approach was used), where:

$$\text{Hazard Quotient} = \frac{\text{Water Column or Dietary Prey Concentrations}}{\text{Salmonid Toxicity Reference Value}} \qquad \text{Equation 4-2}$$

Both acute and chronic exposure concentrations were compared to appropriate TRVs. Water column HQs were first calculated using the Duwamish River and Elliott Bay aquatic life patch[27] data. Subsequently, surface water concentrations in cells containing

27 A patch is defined as those cells containing habitat used by an aquatic receptor. For aquatic life, the study area was divided into two patches — the Duwamish River and Elliott Bay.

habitat critical to juvenile salmon were also compared to TRVs. Average prey concentrations were compared to dietary TRVs when available. Salmonid species TRVs for water column exposures were identified from the U.S. EPA ambient water quality criteria documents (see Table 2-4). Dietary TRVs were taken from the scientific literature (see Table 2-5).

4.3 Sediment

The risk characterization for aquatic life in sediment was done similarly to Tier 1 of the risk characterization for surface water, (i.e., the HQ approach was used):

$$\text{Hazard Quotient} = \frac{\text{Sediment Concentration}}{\text{Toxicity Reference Value}} \qquad \text{Equation 4-3}$$

Because sediments are long term integrators of exposure, only chronic exposure concentrations were compared to TRVs (Table 2-7). Sediment HQs were calculated over the same cell patches used to assess risks to Duwamish River and Elliott Bay aquatic life.

Sediment TRVs were identified from several different sources. Sediment management standards for the State of Washington were the primary criterion used to determine if sediment risks existed (Table 2-7). If no State standard was available for a particular COPC, then sediment concentrations were compared to the other available standards (Table 2-7).

4.4 Physical Stressors

The risk characterization for the effect of physical stressors (DO, temperature, pH, sedimentation rate, displacement and TSS) on aquatic life was also conducted using the HQ approach. Salinity was evaluated qualitatively by examining the amount of time that the cells adjacent to each CSO location were below the five ppt threshold criterion as well the longest time period that this criterion was exceeded. Scouring was evaluated by examining a plot of sediment bed height in the cell adjacent to each CSO location to determine if sediment layers were removed during the year.

5. AQUATIC LIFE RESULTS

This section presents and discusses the results of the aquatic life risk characterization for chemical stressors in the Duwamish River and Elliott Bay. Potential risks to aquatic life were estimated assuming (1) baseline conditions and (2) without CSOs. The results of these two scenarios are compared to assess the possible effects of CSO removal. The results of the surface water risk characterization are provided and discussed first, followed by the sediment risk characterization.

5.1 Surface Water Risk Characterization

As described in Section 3 of this Appendix, an HQ approach was used in Tier 1 of the aquatic life risk assessment, while a probabilistic approach was used in Tier 3 for those chemicals retained from Tier 1. The risk characterization results for each tier are provided below.

5.1.1 Tier 1

Aquatic Community. Chemicals with HQs greater than 1.0 (i.e., where the peak monthly chemical concentration exceeded its Tier 1 TRV) are noted in Table 5-1 and Table 5-2 for the baseline and without CSO scenarios, respectively. Overall, HQs exceeded 1.0 for a greater number of chemicals in the Duwamish River than in Elliott Bay, and for a higher proportion of months. The elimination of CSOs from the model did not remove any COPCs, although the number of months where a COPC concentration exceeded its TRV decreased for some. All chemicals with HQs greater than 1.0 identified in Table 5-1 and Table 5-2 except fluoranthene, benzo(a)anthracene, benzo(g,h,i)perylene, and total PCBs were evaluated in Tier 3; for these chemicals, insufficient toxicity data were available to adequately estimate a probabilistic effects curve. Tier 1 results for the four chemicals with HQs greater than 1.0 that are not further evaluated in Tier 3 are presented in Table 5-3 through Table 5-6.

Salmonids. Potential risks to salmonid species, particularly chinook and coho salmon, from chemicals with Tier HQs greater than 1.0 identified above were evaluated by comparing monthly peak concentrations to toxicity values specific for salmonid species. Most of the toxicity data for salmonids were from freshwater studies conducted with pre-smolts (a freshwater lifestage). The freshwater toxicity data were assumed relevant to the estuarine conditions because dissolved estuarine surface water concentrations were compared to dissolved toxicity values from the literature, thereby minimizing the differences between saline water and freshwater that could influence the bioavailability of the COPCs. It was also assumed that the sensitivity of pre-smolt freshwater salmonids was similar to that for salmonid estuarine life stages.

Table 5-1. Aquatic Life Chemicals of Potential Concern in Surface Water—Baseline

COPC	September	October	November	December	January	February	March	April	May	June	July	August
Acute												
Duwamish River												
Arsenic		X	X									
Copper	X	X	X	X	X	X	X	X	X	X	X	X
Lead		X										
Nickel		X	X	X	X							
Zinc	X	X	X	X	X	X	X	X	X	X	X	
Benzo(a)anthracene					X							
Fluoranthene			X									
Elliott Bay												
Arsenic							X					
Copper	X	X		X	X		X	X	X	X	X	
Chronic												
Duwamish River												
Copper	X	X	X	X	X	X	X	X	X	X	X	X
Lead	X	X	X	X	X							
Nickel	X		X	X	X							
TBT	-[a]	-[a]	X		X							
Total PCBs	-[a]	-[a]	X	X	X	X	X					
Benzo(g,h,i)perylene	X		X									
Elliott Bay												
Copper	X											
Lead	X											
Nickel							X					
TBT	-[a]	-[a]										
Total PCBs	-[a]	-[a]	X	X	X		X					

[a] No data were available for these months due to model initial conditions.

Table 5-2. Aquatic Life Chemicals of Potential Concern—Without CSOs

COPC	September	October	November	December	January	February	March	April	May	June	July	August
Acute												
Duwamish River												
Arsenic		X	X									
Copper	X	X	X	X	X	X	X	X	X	X	X	X
Lead		X	X									
Nickel		X	X	X	X							
Zinc		X	X	X	X	X	X	X	X	X	X	X
Benzo(a)anthracene			X		X							
Fluoranthene			X									
Elliott Bay												
Arsenic							X					
Copper	X	X	X		X		X					
Chronic												
Duwamish River												
Copper	X	X	X	X	X	X	X	X	X	X	X	X
Lead	X	X	X	X	X		X					
Nickel	X	X	X	X	X		X					
TBT	-[a]	-[a]	X		X		X					
Total PCBs	-[a]	-[a]	X	X	X	X	X					
Benzo(g,h,i)perylene	X		X									
Elliott Bay												
Copper	X	X										
Lead	X											
Nickel							X					
TBT	-[a]	-[a]										
Total PCBs	-[a]	-[a]	X	X	X		X					

[a] No data were available for these months due to model initial conditions.

HQs for salmonid species were less than 1.0 for all COPCs evaluated (see Table 5-7 through Table 5-14). Accordingly, HQs for the modeled cells of the Duwamish River and Elliott Bay deemed to be critical salmon habitat were also less than 1.0 (see Table 5-15 through Table 5-22).

5.1.2 Salmon Dietary Risk Characterization

None of the HQs for any of the four chemicals analyzed were greater than one (Table 5-23). Consequently, juvenile salmon are not predicted to be at risk of increased mortality or reduced growth from consuming *Corophium* and *Gammarus* amphipods from the Kellogg Island area of the Duwamish River.

5.1.3 Tier 3

As explained in Section 3 of this volume, potential risks in Tier 3 were defined as expected percent species affected. The risk estimates, by month, are shown in Table 4-3 in Volume 1 and Table 5-24 to Table 5-29 of this Appendix. Estimated risks tend to be low (less than or equal to one) for most COPCs for most months. The COPCs expected to affect the greatest percent of species are TBT (chronic) and copper (acute, chronic). Maximum monthly TBT chronic risks are 2 percent in the Duwamish River and 1 percent in Elliott Bay. Maximum monthly chronic copper risks are 2 percent in the Duwamish River, but less than or equal to 1 percent in Elliott Bay. Acute copper risks in both the Duwamish River and Elliott Bay are less than or equal to 1 percent. Risk results were virtually the same for the baseline and without CSO scenarios, thereby demonstrating that CSOs do not significantly contribute to TBT and copper loading in the Duwamish River and Elliott Bay.

TBT risks in the Duwamish River were less than or equal to 1 percent for nine of the 10 months for which data were available[28]. The most sensitive species to TBT that could potentially be affected during certain months of the year include bivalves such as clams, mussels and oysters. Copper was estimated to affect up to 2 percent of the aquatic species during certain months of the year.

The derivation of the percent of species affected is shown in greater detail in Figure 5-1 through Figure 5-6. These figures present the distribution of Duwamish River EECs and the distribution of toxicity values on the same graph for each chemical evaluated for a representative month. As shown in these figures, the acute and chronic chemical concentrations in the Duwamish River are generally substantially lower than concentrations predicted to affect aquatic organisms.

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28 Due to initial conditions of TBT, data are only available for November through August.

Table 5-7. Acute Salmonid Hazard Quotients for the Duwamish River—Baseline

COPC	Species	Hazard Quotients											
		September	October	November	December	January	February	March	April	May	June	July	August
Arsenic	*Oncorhynchus mykiss* (rainbow trout)	2.04E-04	2.25E-04	1.56E-04	1.69E-04	1.56E-04	1.49E-04	1.47E-03	2.27E-04	1.60E-04	1.55E-04	1.66E-04	1.50E-04
	Salvelinus fontinalis (brook trout)	1.82E-04	2.01E-04	1.39E-04	1.51E-04	1.39E-04	1.33E-04	1.31E-03	2.02E-04	1.43E-04	1.38E-04	1.48E-04	1.34E-04
Copper	*Salvelinus fontinalis* (brook trout)	9.98E-03	1.10E-02	6.96E-03	1.03E-02	6.35E-03	7.63E-03	7.04E-03	6.57E-03	9.11E-03	1.38E-02	1.42E-02	1.39E-02
	Salmo salar (Atlantic salmon)	1.00E-02	1.10E-02	7.00E-03	1.03E-02	6.38E-03	7.66E-03	7.07E-03	6.60E-03	9.15E-03	1.38E-02	1.43E-02	1.40E-02
	Oncorhynchus mykiss (rainbow trout)	2.63E-02	2.89E-02	1.83E-02	2.70E-02	1.67E-02	2.01E-02	1.85E-02	1.73E-02	2.40E-02	3.63E-02	3.75E-02	3.65E-02
	Salmo clarkii (cutthroat trout)	1.66E-02	1.83E-02	1.16E-02	1.71E-02	1.06E-02	1.27E-02	1.17E-02	1.09E-02	1.52E-02	2.30E-02	2.37E-02	2.31E-02
	Oncorhynchus tshawytscha (chinook salmon)	2.62E-02	2.88E-02	1.82E-02	2.69E-02	1.66E-02	2.00E-02	1.84E-02	1.72E-02	2.39E-02	3.61E-02	3.73E-02	3.64E-02
	Oncorhynchus nerka (sockeye salmon)	4.72E-03	5.19E-03	3.29E-03	4.85E-03	3.00E-03	3.60E-03	3.32E-03	3.10E-03	4.30E-03	6.51E-03	6.73E-03	6.56E-03
	Oncorhynchus kisutch (coho salmon)	1.57E-02	1.73E-02	1.10E-02	1.62E-02	9.99E-03	1.20E-02	1.11E-02	1.03E-02	1.43E-02	2.17E-02	2.24E-02	2.19E-02
Lead	*Salvelinus fontinalis* (brook trout)	3.04E-06	4.24E-06	3.58E-06	6.11E-06	4.15E-06	2.96E-06	3.28E-06	2.90E-06	5.44E-06	7.60E-06	8.03E-06	7.71E-06
	Oncorhynchus mykiss (rainbow trout)	5.99E-06	8.36E-06	7.06E-06	1.20E-05	8.19E-06	5.83E-06	6.47E-06	5.71E-06	1.07E-05	1.50E-05	1.58E-05	1.52E-05
Nickel	*Oncorhynchus mykiss* (rainbow trout)	5.37E-05	6.21E-05	6.10E-05	6.35E-05	5.25E-05	5.46E-05	2.97E-04	5.07E-05	6.40E-05	8.37E-05	8.94E-05	8.34E-05
Zinc	*Oncorhynchus kisutch* (coho salmon)	2.24E-03	3.61E-03	3.42E-03	4.17E-03	3.25E-03	2.15E-03	2.21E-03	2.00E-03	3.16E-03	5.77E-03	6.02E-03	5.71E-03
	Oncorhynchus nerka (sockeye salmon)	2.43E-03	3.91E-03	3.71E-03	4.52E-03	3.52E-03	2.33E-03	2.39E-03	2.17E-03	3.42E-03	6.25E-03	6.53E-03	6.19E-03
	Oncorhynchus tshawytscha (chinook salmon)	8.18E-03	1.32E-02	1.25E-02	1.52E-02	1.18E-02	7.84E-03	8.06E-03	7.31E-03	1.15E-02	2.10E-02	2.20E-02	2.08E-02
	Oncorhynchus mykiss (rainbow trout)	5.29E-03	8.52E-03	8.08E-03	9.84E-03	7.67E-03	5.07E-03	5.22E-03	4.74E-03	7.46E-03	1.36E-02	1.42E-02	1.35E-02
	Salmo salar (Atlantic salmon)	1.68E-03	2.70E-03	2.56E-03	3.12E-03	2.43E-03	1.61E-03	1.65E-03	1.50E-03	2.36E-03	4.31E-03	4.51E-03	4.27E-03
	Salvelinus fontinalis (brook trout)	1.74E-03	2.80E-03	2.65E-03	3.23E-03	2.52E-03	1.67E-03	1.71E-03	1.55E-03	2.45E-03	4.47E-03	4.67E-03	4.42E-03

Table 5-8. Chronic Salmonid Hazard Quotients for the Duwamish River—Baseline

COPC	Species	Hazard Quotients											
		September	October	November	December	January	February	March	April	May	June	July	August
Copper	*Oncorhynchus mykiss* (rainbow trout)	1.52E-02	2.31E-02	1.42E-02	1.62E-02	1.33E-02	1.49E-02	1.52E-02	1.29E-02	1.47E-02	1.98E-02	1.77E-02	2.40E-02
	Salmo trutta (brown trout)	9.38E-03	1.42E-02	8.78E-03	9.98E-03	8.19E-03	9.17E-03	9.40E-03	7.93E-03	9.04E-03	1.22E-02	1.09E-02	1.48E-02
	Salvelinus fontinalis (brook trout)	1.78E-02	2.71E-02	1.67E-02	1.90E-02	1.56E-02	1.74E-02	1.79E-02	1.51E-02	1.72E-02	2.33E-02	2.08E-02	2.81E-02
	Salvelinus namaycush (lake trout)	9.48E-03	1.44E-02	8.87E-03	1.01E-02	8.27E-03	9.27E-03	9.50E-03	8.01E-03	9.14E-03	1.24E-02	1.11E-02	1.50E-02
	Salmo salar (Atlantic salmon)	7.14E-03	1.08E-02	6.69E-03	7.60E-03	6.23E-03	6.98E-03	7.16E-03	6.04E-03	6.88E-03	9.32E-03	8.33E-03	1.13E-02
	Salmo clarkii (cutthroat trout)	1.18E-02	1.80E-02	1.11E-02	1.26E-02	1.03E-02	1.16E-02	1.19E-02	1.00E-02	1.14E-02	1.55E-02	1.38E-02	1.87E-02
	Oncorhynchus tshawytscha (chinook salmon)	1.86E-02	2.83E-02	1.74E-02	1.98E-02	1.63E-02	1.82E-02	1.87E-02	1.57E-02	1.80E-02	2.43E-02	2.17E-02	2.94E-02
	Oncorhynchus nerka (sockeye salmon)	3.36E-03	5.10E-03	3.14E-03	3.57E-03	2.93E-03	3.28E-03	3.36E-03	2.84E-03	3.24E-03	4.38E-03	3.92E-03	5.30E-03
	Oncorhynchus kisutch (coho salmon)	1.12E-02	1.70E-02	1.05E-02	1.19E-02	9.77E-03	1.09E-02	1.12E-02	9.46E-03	1.08E-02	1.46E-02	1.31E-02	1.77E-02
Lead	*Oncorhynchus mykiss* (rainbow trout)	4.78E-05	6.88E-05	4.89E-05	7.12E-05	5.73E-05	4.70E-05	5.13E-05	4.36E-05	4.42E-05	7.40E-05	8.20E-05	7.78E-05
	Salvelinus fontinalis (brook trout)	3.89E-05	5.60E-05	3.98E-05	5.79E-05	4.67E-05	3.83E-05	4.17E-05	3.55E-05	3.60E-05	6.02E-05	6.67E-05	6.34E-05
Nickel	*Oncorhynchus mykiss* (rainbow trout)	2.72E-03	2.40E-03	2.41E-03	2.78E-03	2.46E-03	2.54E-03	1.48E-02	2.00E-03	2.79E-03	3.27E-03	3.32E-03	2.83E-03
Zinc	*Oncorhynchus nerka* (sockeye salmon)	3.76E-02	7.35E-02	6.76E-02	4.19E-02	4.92E-02	2.31E-02	7.36E-02	1.54E-02	1.49E-02	1.79E-02	1.01E-02	7.82E-03
	Oncorhynchus tshawytscha (chinook salmon)	1.84E-02	3.61E-02	3.32E-02	2.06E-02	2.41E-02	1.13E-02	3.61E-02	7.56E-03	7.32E-03	8.79E-03	4.94E-03	3.83E-03
	Oncorhynchus mykiss (rainbow trout)	1.57E-02	3.08E-02	2.83E-02	1.75E-02	2.06E-02	9.68E-03	3.08E-02	6.45E-03	6.24E-03	7.50E-03	4.21E-03	3.27E-03
	Salvelinus fontinalis (brook trout)	1.34E-02	2.62E-02	2.41E-02	1.49E-02	1.75E-02	8.24E-03	2.62E-02	5.49E-03	5.32E-03	6.39E-03	3.59E-03	2.79E-03
	Oncorhynchus kisutch (coho salmon)	1.67E-02	3.27E-02	3.00E-02	1.86E-02	2.18E-02	1.03E-02	3.27E-02	6.85E-03	6.62E-03	7.96E-03	4.47E-03	3.47E-03
	Salmo salar (Atlantic salmon)	1.25E-02	2.44E-02	2.25E-02	1.39E-02	1.63E-02	7.69E-03	2.45E-02	5.12E-03	4.96E-03	5.96E-03	3.35E-03	2.60E-03
TBT	*Oncorhynchus tshawytscha* (chinook salmon)	6.33E-01	1.02E-01	1.10E-01	5.07E-02	1.04E-01	2.17E-02	5.98E-02	3.14E-03	9.77E-03	9.72E-03	7.36E-03	1.21E-02
	Oncorhynchus mykiss (rainbow trout)	2.02E-01	3.25E-02	3.52E-02	1.62E-02	3.33E-02	6.92E-03	1.91E-02	1.00E-03	3.12E-03	3.11E-03	2.35E-03	3.88E-03
	Salvelinus namaycush (lake trout)	7.26E-02	1.17E-02	1.26E-02	5.81E-03	1.20E-02	2.48E-03	6.86E-03	3.60E-04	1.12E-03	1.12E-03	8.45E-04	1.39E-03
Total PCBs	*Salvelinus fontinalis* (brook trout)	7.05E-01	5.23E-02	9.12E-02	3.26E-02	3.54E-02	8.20E-03	1.95E-02	1.60E-03	2.20E-03	9.00E-04	8.00E-04	2.80E-03

Table 5-9. Acute Salmonid Hazard Quotients for Elliott Bay—Baseline

COPC	Species	Hazard Quotients											
		September	October	November	December	January	February	March	April	May	June	July	August
Arsenic	*Oncorhynchus mykiss* (rainbow trout)	1.77E-04	1.73E-04	1.75E-04	1.63E-04	1.61E-04	1.75E-04	7.46E-03	1.90E-04	2.02E-04	1.96E-04	1.86E-04	1.70E-04
	Salvelinus fontinalis (brook trout)	1.58E-04	1.55E-04	1.56E-04	1.45E-04	1.44E-04	1.56E-04	6.65E-03	1.69E-04	1.80E-04	1.75E-04	1.66E-04	1.51E-04
Copper	*Salvelinus fontinalis* (brook trout)	4.78E-03	5.74E-03	4.44E-03	5.71E-03	5.49E-03	5.52E-03	5.76E-03	4.97E-03	6.36E-03	8.76E-03	6.73E-03	6.07E-03
	Salmo salar (Atlantic salmon)	4.80E-03	5.76E-03	4.46E-03	5.73E-03	5.52E-03	5.54E-03	5.79E-03	5.00E-03	6.39E-03	8.80E-03	6.77E-03	6.10E-03
	Oncorhynchus mykiss (rainbow trout)	1.26E-02	1.51E-02	1.17E-02	1.50E-02	1.44E-02	1.45E-02	1.52E-02	1.31E-02	1.67E-02	2.30E-02	1.77E-02	1.60E-02
	Salmo clarkii (cutthroat trout)	7.97E-03	9.56E-03	7.41E-03	9.51E-03	9.15E-03	9.20E-03	9.61E-03	8.29E-03	1.06E-02	1.46E-02	1.12E-02	1.01E-02
	Oncorhynchus tshawytscha (chinook salmon)	1.25E-02	1.50E-02	1.16E-02	1.50E-02	1.44E-02	1.45E-02	1.51E-02	1.30E-02	1.67E-02	2.30E-02	1.76E-02	1.59E-02
	Oncorhynchus nerka (sockeye salmon)	2.26E-03	2.71E-03	2.10E-03	2.70E-03	2.59E-03	2.61E-03	2.72E-03	2.35E-03	3.01E-03	4.14E-03	3.18E-03	2.87E-03
	Oncorhynchus kisutch (coho salmon)	7.53E-03	9.04E-03	7.00E-03	8.99E-03	8.65E-03	8.69E-03	9.08E-03	7.83E-03	1.00E-02	1.38E-02	1.06E-02	9.56E-03
Lead	*Salvelinus fontinalis* (brook trout)	1.71E-06	1.70E-06	1.25E-06	2.15E-06	2.26E-06	1.33E-06	1.95E-06	2.00E-06	2.47E-06	3.97E-06	3.49E-06	1.61E-06
	Oncorhynchus mykiss (rainbow trout)	3.37E-06	3.36E-06	2.46E-06	4.24E-06	4.46E-06	2.62E-06	3.85E-06	3.94E-06	4.87E-06	7.83E-06	6.88E-06	3.18E-06
Nickel	*Oncorhynchus mykiss* (rainbow trout)	3.08E-05	3.01E-05	4.09E-05	3.92E-05	3.64E-05	3.64E-05	9.63E-04	3.76E-05	4.51E-05	4.39E-05	4.41E-05	2.80E-05
Zinc	*Oncorhynchus kisutch* (coho salmon)	1.23E-03	1.31E-03	9.17E-04	2.00E-03	1.90E-03	1.10E-03	1.49E-03	1.49E-03	1.64E-03	2.54E-03	1.95E-03	1.38E-03
	Oncorhynchus nerka (sockeye salmon)	1.33E-03	1.42E-03	9.95E-04	2.17E-03	2.05E-03	1.20E-03	1.61E-03	1.62E-03	1.78E-03	2.75E-03	2.12E-03	1.50E-03
	Oncorhynchus tshawytscha (chinook salmon)	4.47E-03	4.79E-03	3.35E-03	7.30E-03	6.91E-03	4.03E-03	5.42E-03	5.44E-03	5.98E-03	9.27E-03	7.13E-03	5.03E-03
	Oncorhynchus mykiss (rainbow trout)	2.90E-03	3.10E-03	2.17E-03	4.73E-03	4.48E-03	2.61E-03	3.51E-03	3.52E-03	3.87E-03	6.00E-03	4.62E-03	3.26E-03
	Salmo salar (Atlantic salmon)	9.17E-04	9.83E-04	6.86E-04	1.50E-03	1.42E-03	8.26E-04	1.11E-03	1.12E-03	1.23E-03	1.90E-03	1.46E-03	1.03E-03
	Salvelinus fontinalis (brook trout)	9.50E-04	1.02E-03	7.11E-04	1.55E-03	1.47E-03	8.55E-04	1.15E-03	1.16E-03	1.27E-03	1.97E-03	1.52E-03	1.07E-03

Table 5-10. Chronic Salmonid Hazard Quotients for Elliott Bay—Baseline

COPC	Species	Hazard Quotients											
		September	October	November	December	January	February	March	April	May	June	July	August
Copper	*Oncorhynchus mykiss* (rainbow trout)	1.02E-02	1.28E-02	1.11E-02	1.25E-02	1.27E-02	1.37E-02	1.48E-02	1.14E-02	1.18E-02	1.57E-02	1.24E-02	1.26E-02
	Salmo trutta (brown trout)	6.32E-03	7.92E-03	6.85E-03	7.71E-03	7.82E-03	8.47E-03	9.10E-03	7.02E-03	7.27E-03	9.67E-03	7.63E-03	7.76E-03
	Salvelinus fontinalis (brook trout)	1.20E-02	1.50E-02	1.30E-02	1.46E-02	1.49E-02	1.61E-02	1.73E-02	1.33E-02	1.38E-02	1.84E-02	1.45E-02	1.47E-02
	Salvelinus namaycush (lake trout)	6.39E-03	8.00E-03	6.92E-03	7.79E-03	7.90E-03	8.56E-03	9.20E-03	7.09E-03	7.35E-03	9.77E-03	7.71E-03	7.84E-03
	Salmo salar (Atlantic salmon)	4.81E-03	6.03E-03	5.22E-03	5.87E-03	5.95E-03	6.45E-03	6.93E-03	5.34E-03	5.54E-03	7.36E-03	5.81E-03	5.91E-03
	Salmo clarkii (cutthroat trout)	7.98E-03	1.00E-02	8.66E-03	9.74E-03	9.88E-03	1.07E-02	1.15E-02	8.86E-03	9.19E-03	1.22E-02	9.64E-03	9.80E-03
	Oncorhynchus tshawytscha (chinook salmon)	1.25E-02	1.57E-02	1.36E-02	1.53E-02	1.55E-02	1.68E-02	1.81E-02	1.39E-02	1.44E-02	1.92E-02	1.52E-02	1.54E-02
	Oncorhynchus nerka (sockeye salmon)	2.26E-03	2.83E-03	2.45E-03	2.76E-03	2.80E-03	3.03E-03	3.26E-03	2.51E-03	2.60E-03	3.46E-03	2.73E-03	2.78E-03
	Oncorhynchus kisutch (coho salmon)	7.54E-03	9.45E-03	8.18E-03	9.20E-03	9.33E-03	1.01E-02	1.09E-02	8.37E-03	8.68E-03	1.15E-02	9.11E-03	9.26E-03
Lead	*Oncorhynchus mykiss* (rainbow trout)	3.49E-05	3.47E-05	2.60E-05	3.78E-05	4.41E-05	3.04E-05	3.75E-05	2.28E-05	2.52E-05	5.30E-05	3.58E-05	3.63E-05
	Salvelinus fontinalis (brook trout)	2.84E-05	2.82E-05	2.11E-05	3.07E-05	3.59E-05	2.48E-05	3.05E-05	1.86E-05	2.05E-05	4.31E-05	2.92E-05	2.95E-05
Nickel	*Oncorhynchus mykiss* (rainbow trout)	2.18E-03	2.12E-03	2.09E-03	2.17E-03	2.19E-03	1.99E-03	2.98E-02	1.87E-03	2.07E-03	2.35E-03	2.26E-03	1.74E-03
Zinc	*Oncorhynchus nerka* (sockeye salmon)	2.63E-02	7.84E-03	4.69E-03	6.19E-03	6.79E-03	5.76E-03	6.97E-03	3.33E-03	3.11E-03	5.62E-03	4.39E-03	4.64E-03
	Oncorhynchus tshawytscha (chinook salmon)	1.29E-02	3.84E-03	2.30E-03	3.03E-03	3.33E-03	2.82E-03	3.42E-03	1.63E-03	1.53E-03	2.76E-03	2.15E-03	2.28E-03
	Oncorhynchus mykiss (rainbow trout)	1.10E-02	3.28E-03	1.96E-03	2.59E-03	2.84E-03	2.41E-03	2.91E-03	1.39E-03	1.30E-03	2.35E-03	1.84E-03	1.94E-03
	Salvelinus fontinalis (brook trout)	9.36E-03	2.79E-03	1.67E-03	2.20E-03	2.42E-03	2.05E-03	2.48E-03	1.19E-03	1.11E-03	2.00E-03	1.56E-03	1.65E-03
	Oncorhynchus kisutch (coho salmon)	1.17E-02	3.48E-03	2.08E-03	2.75E-03	3.01E-03	2.56E-03	3.09E-03	1.48E-03	1.38E-03	2.50E-03	1.95E-03	2.06E-03
	Salmo salar (Atlantic salmon)	8.73E-03	2.60E-03	1.56E-03	2.06E-03	2.25E-03	1.91E-03	2.32E-03	1.11E-03	1.03E-03	1.87E-03	1.46E-03	1.54E-03
TBT	*Oncorhynchus tshawytscha* (chinook salmon)	4.49E-01	3.45E-02	2.55E-02	1.48E-02	3.38E-02	1.08E-02	3.18E-02	8.96E-04	1.74E-03	1.52E-03	1.29E-03	2.90E-03
	Oncorhynchus mykiss (rainbow trout)	1.43E-01	1.10E-02	8.15E-03	4.71E-03	1.08E-02	3.46E-03	1.02E-02	2.86E-04	5.56E-04	4.87E-04	4.12E-04	9.27E-04
	Salvelinus namaycush (lake trout)	5.15E-02	3.95E-03	2.93E-03	1.69E-03	3.88E-03	1.24E-03	3.65E-03	1.03E-04	2.00E-04	1.75E-04	1.48E-04	3.33E-04
Total PCBs	*Salvelinus fontinalis* (brook trout)	7.33E-01	2.00E-02	1.25E-02	6.90E-03	9.30E-03	2.50E-03	5.90E-03	3.00E-04	6.00E-04	3.00E-04	2.00E-04	6.00E-04

Table 5-11. Acute Salmonid Hazard Quotients for the Duwamish River—Without CSOs

COPC	Species	Hazard Quotients											
		January	February	March	April	May	June	July	August	September	October	November	December
Arsenic	*Oncorhynchus mykiss* (rainbow trout)	1.52E-04	1.52E-04	1.48E-03	2.15E-04	1.80E-04	1.56E-04	1.70E-04	1.52E-04	2.45E-04	2.28E-04	1.55E-04	1.69E-04
	Salvelinus fontinalis (brook trout)	1.36E-04	1.35E-04	1.32E-03	1.91E-04	1.60E-04	1.39E-04	1.52E-04	1.35E-04	2.18E-04	2.03E-04	1.39E-04	1.51E-04
Copper	*Salvelinus fontinalis* (brook trout)	6.17E-03	8.62E-03	7.19E-03	6.52E-03	1.06E-02	8.88E-03	9.95E-03	1.15E-02	1.15E-02	1.11E-02	7.01E-03	7.01E-03
	Salmo salar (Atlantic salmon)	6.20E-03	8.66E-03	7.23E-03	6.56E-03	1.06E-02	8.92E-03	9.99E-03	1.15E-02	1.16E-02	1.12E-02	7.04E-03	7.04E-03
	Oncorhynchus mykiss (rainbow trout)	1.62E-02	2.27E-02	1.89E-02	1.72E-02	2.78E-02	2.34E-02	2.62E-02	3.02E-02	3.04E-02	2.92E-02	1.84E-02	1.84E-02
	Salmo clarkii (cutthroat trout)	1.03E-02	1.44E-02	1.20E-02	1.09E-02	1.76E-02	1.48E-02	1.66E-02	1.92E-02	1.92E-02	1.85E-02	1.17E-02	1.17E-02
	Oncorhynchus tshawytscha (chinook salmon)	1.62E-02	2.26E-02	1.89E-02	1.71E-02	2.77E-02	2.33E-02	2.61E-02	3.01E-02	3.02E-02	2.91E-02	1.84E-02	1.84E-02
	Oncorhynchus nerka (sockeye salmon)	2.91E-03	4.07E-03	3.40E-03	3.08E-03	5.00E-03	4.20E-03	4.70E-03	5.43E-03	5.45E-03	5.25E-03	3.31E-03	3.31E-03
	Oncorhynchus kisutch (coho salmon)	9.71E-03	1.36E-02	1.13E-02	1.03E-02	1.67E-02	1.40E-02	1.57E-02	1.81E-02	1.82E-02	1.75E-02	1.10E-02	1.10E-02
Lead	*Salvelinus fontinalis* (brook trout)	3.93E-06	3.01E-06	3.19E-06	2.97E-06	2.94E-06	4.12E-06	5.53E-06	6.13E-06	2.89E-06	4.19E-06	3.59E-06	4.03E-06
	Oncorhynchus mykiss (rainbow trout)	7.76E-06	5.93E-06	6.28E-06	5.85E-06	5.79E-06	8.13E-06	1.09E-05	1.21E-05	5.70E-06	8.26E-06	7.08E-06	7.95E-06
Nickel	*Oncorhynchus mykiss* (rainbow trout)	5.26E-05	5.47E-05	2.89E-04	5.74E-05	6.89E-05	6.99E-05	8.07E-05	6.62E-05	5.58E-05	6.08E-05	6.12E-05	5.96E-05
Zinc	*Oncorhynchus kisutch* (coho salmon)	3.08E-03	2.13E-03	2.21E-03	2.08E-03	2.98E-03	2.14E-03	3.58E-03	2.52E-03	2.53E-03	3.57E-03	3.43E-03	3.09E-03
	Oncorhynchus nerka (sockeye salmon)	3.34E-03	2.31E-03	2.40E-03	2.25E-03	3.23E-03	2.32E-03	3.88E-03	2.73E-03	2.74E-03	3.87E-03	3.72E-03	3.35E-03
	Oncorhynchus tshawytscha (chinook salmon)	1.12E-02	7.76E-03	8.06E-03	7.58E-03	1.09E-02	7.79E-03	1.31E-02	9.20E-03	9.21E-03	1.30E-02	1.25E-02	1.13E-02
	Oncorhynchus mykiss (rainbow trout)	7.27E-03	5.03E-03	5.22E-03	4.91E-03	7.04E-03	5.04E-03	8.46E-03	5.96E-03	5.96E-03	8.43E-03	8.11E-03	7.29E-03
	Salmo salar (Atlantic salmon)	2.30E-03	1.59E-03	1.65E-03	1.56E-03	2.23E-03	1.60E-03	2.68E-03	1.89E-03	1.89E-03	2.67E-03	2.57E-03	2.31E-03
	Salvelinus fontinalis (brook trout)	2.39E-03	1.65E-03	1.71E-03	1.61E-03	2.31E-03	1.66E-03	2.78E-03	1.95E-03	1.96E-03	2.77E-03	2.66E-03	2.39E-03

Table 5-12. Chronic Salmonid Hazard Quotients for the Duwamish River—Without CSOs

COPC	Species	Hazard Quotients											
		January	February	March	April	May	June	July	August	September	October	November	December
Copper	*Oncorhynchus mykiss* (rainbow trout)	1.30E-02	1.39E-02	1.69E-02	1.28E-02	1.39E-02	1.70E-02	1.72E-02	2.19E-02	1.54E-02	2.25E-02	1.43E-02	1.62E-02
	Salmo trutta (brown trout)	8.03E-03	8.58E-03	1.04E-02	7.89E-03	8.60E-03	1.05E-02	1.06E-02	1.35E-02	9.51E-03	1.39E-02	8.79E-03	1.00E-02
	Salvelinus fontinalis (brook trout)	1.53E-02	1.63E-02	1.98E-02	1.50E-02	1.63E-02	1.99E-02	2.01E-02	2.57E-02	1.81E-02	2.64E-02	1.67E-02	1.90E-02
	Salvelinus namaycush (lake trout)	8.11E-03	8.67E-03	1.05E-02	7.97E-03	8.69E-03	1.06E-02	1.07E-02	1.37E-02	9.61E-03	1.40E-02	8.89E-03	1.01E-02
	Salmo salar (Atlantic salmon)	6.11E-03	6.53E-03	7.95E-03	6.01E-03	6.55E-03	7.96E-03	8.07E-03	1.03E-02	7.24E-03	1.06E-02	6.69E-03	7.62E-03
	Salmo clarkii (cutthroat trout)	1.01E-02	1.08E-02	1.32E-02	9.96E-03	1.09E-02	1.32E-02	1.34E-02	1.71E-02	1.20E-02	1.75E-02	1.11E-02	1.26E-02
	Oncorhynchus tshawytscha (chinook salmon)	1.59E-02	1.70E-02	2.07E-02	1.57E-02	1.71E-02	2.08E-02	2.11E-02	2.69E-02	1.89E-02	2.75E-02	1.75E-02	1.99E-02
	Oncorhynchus nerka (sockeye salmon)	2.87E-03	3.07E-03	3.74E-03	2.82E-03	3.08E-03	3.74E-03	3.80E-03	4.84E-03	3.40E-03	4.96E-03	3.15E-03	3.58E-03
	Oncorhynchus kisutch (coho salmon)	9.58E-03	1.02E-02	1.25E-02	9.41E-03	1.03E-02	1.25E-02	1.27E-02	1.61E-02	1.13E-02	1.65E-02	1.05E-02	1.19E-02
Lead	*Oncorhynchus mykiss* (rainbow trout)	5.54E-05	4.68E-05	6.20E-05	3.96E-05	3.88E-05	6.86E-05	7.95E-05	7.24E-05	4.80E-05	6.92E-05	4.86E-05	7.18E-05
	Salvelinus fontinalis (brook trout)	4.51E-05	3.81E-05	5.05E-05	3.22E-05	3.16E-05	5.58E-05	6.47E-05	5.89E-05	3.91E-05	5.63E-05	3.95E-05	5.84E-05
Nickel	*Oncorhynchus mykiss* (rainbow trout)	2.43E-03	2.54E-03	1.60E-02	2.00E-03	2.65E-03	3.18E-03	3.20E-03	2.71E-03	2.84E-03	2.31E-03	2.44E-03	2.71E-03
Zinc	*Oncorhynchus nerka* (sockeye salmon)	3.76E-02	7.35E-02	6.76E-02	4.19E-02	4.92E-02	2.31E-02	7.36E-02	1.54E-02	1.49E-02	1.79E-02	1.01E-02	7.82E-03
	Oncorhynchus tshawytscha (chinook salmon)	1.84E-02	3.61E-02	3.32E-02	2.06E-02	2.41E-02	1.13E-02	3.61E-02	7.56E-03	7.32E-03	8.79E-03	4.94E-03	3.83E-03
	Oncorhynchus mykiss (rainbow trout)	1.57E-02	3.08E-02	2.83E-02	1.75E-02	2.06E-02	9.68E-03	3.08E-02	6.45E-03	6.24E-03	7.50E-03	4.21E-03	3.27E-03
	Salvelinus fontinalis (brook trout)	1.34E-02	2.62E-02	2.41E-02	1.49E-02	1.75E-02	8.24E-03	2.62E-02	5.49E-03	5.32E-03	6.39E-03	3.59E-03	2.79E-03
	Oncorhynchus kisutch (coho salmon)	1.67E-02	3.27E-02	3.00E-02	1.86E-02	2.18E-02	1.03E-02	3.27E-02	6.85E-03	6.62E-03	7.96E-03	4.47E-03	3.47E-03
	Salmo salar (Atlantic salmon)	1.25E-02	2.44E-02	2.25E-02	1.39E-02	1.63E-02	7.69E-03	2.45E-02	5.12E-03	4.96E-03	5.96E-03	3.35E-03	2.60E-03
TBT	*Oncorhynchus tshawytscha* (chinook salmon)	1.07E-01	2.18E-02	1.15E-01	5.19E-03	4.29E-03	9.20E-03	6.47E-03	5.41E-03	6.38E-01	1.02E-01	1.17E-01	5.19E-02
	Oncorhynchus mykiss (rainbow trout)	3.41E-02	6.96E-03	3.69E-02	1.66E-03	1.37E-03	2.94E-03	2.07E-03	1.73E-03	2.04E-01	3.25E-02	3.72E-02	1.66E-02
	Salvelinus namaycush (lake trout)	1.22E-02	2.50E-03	1.32E-02	5.95E-04	4.92E-04	1.06E-03	7.42E-04	6.21E-04	7.31E-02	1.17E-02	1.34E-02	5.95E-03
PCB	*Salvelinus fontinalis* (brook trout)	3.63E-02	8.28E-03	2.40E-02	1.77E-03	1.80E-03	1.09E-03	6.58E-04	1.01E-03	7.04E-01	4.71E-02	9.69E-02	3.33E-02

Table 5-13. Acute Salmonid Hazard Quotients for Elliott Bay—Without CSOs

		Hazard Quotients											
COPC	**Species**	**January**	**February**	**March**	**April**	**May**	**June**	**July**	**August**	**September**	**October**	**November**	**December**
Arsenic	*Oncorhynchus mykiss* (rainbow trout)	1.78E-04	1.73E-04	7.46E-03	1.94E-04	2.02E-04	1.97E-04	1.85E-04	1.72E-04	1.77E-04	1.72E-04	1.75E-04	1.65E-04
	Salvelinus fontinalis (brook trout)	1.59E-04	1.55E-04	6.65E-03	1.73E-04	1.80E-04	1.76E-04	1.65E-04	1.53E-04	1.58E-04	1.53E-04	1.56E-04	1.47E-04
Copper	*Salvelinus fontinalis* (brook trout)	5.19E-03	5.40E-03	5.48E-03	4.09E-03	6.29E-03	5.99E-03	5.56E-03	5.05E-03	3.88E-03	5.59E-03	4.29E-03	5.06E-03
	Salmo salar (Atlantic salmon)	5.21E-03	5.42E-03	5.50E-03	4.11E-03	6.32E-03	6.02E-03	5.59E-03	5.08E-03	3.90E-03	5.62E-03	4.31E-03	5.08E-03
	Oncorhynchus mykiss (rainbow trout)	1.36E-02	1.42E-02	1.44E-02	1.08E-02	1.65E-02	1.58E-02	1.46E-02	1.33E-02	1.02E-02	1.47E-02	1.13E-02	1.33E-02
	Salmo clarkii (cutthroat trout)	8.64E-03	8.99E-03	9.13E-03	6.82E-03	1.05E-02	9.99E-03	9.27E-03	8.42E-03	6.47E-03	9.33E-03	7.15E-03	8.43E-03
	Oncorhynchus tshawytscha (chinook salmon)	1.36E-02	1.41E-02	1.44E-02	1.07E-02	1.65E-02	1.57E-02	1.46E-02	1.32E-02	1.02E-02	1.47E-02	1.12E-02	1.33E-02
	Oncorhynchus nerka (sockeye salmon)	2.45E-03	2.55E-03	2.59E-03	1.93E-03	2.97E-03	2.83E-03	2.63E-03	2.39E-03	1.83E-03	2.64E-03	2.03E-03	2.39E-03
	Oncorhynchus kisutch (coho salmon)	8.17E-03	8.50E-03	8.62E-03	6.45E-03	9.90E-03	9.44E-03	8.76E-03	7.96E-03	6.11E-03	8.81E-03	6.76E-03	7.97E-03
Lead	*Salvelinus fontinalis* (brook trout)	2.31E-06	1.34E-06	2.25E-06	1.24E-06	1.46E-06	1.52E-06	1.96E-06	1.58E-06	1.37E-06	1.96E-06	1.41E-06	2.15E-06
	Oncorhynchus mykiss (rainbow trout)	4.55E-06	2.65E-06	4.43E-06	2.44E-06	2.87E-06	3.00E-06	3.86E-06	3.12E-06	2.70E-06	3.85E-06	2.77E-06	4.24E-06
Nickel	*Oncorhynchus mykiss* (rainbow trout)	3.64E-05	3.56E-05	9.73E-04	3.67E-05	4.34E-05	4.32E-05	3.88E-05	2.79E-05	3.08E-05	3.02E-05	4.11E-05	3.91E-05
Zinc	*Oncorhynchus kisutch* (coho salmon)	1.98E-03	1.12E-03	1.61E-03	1.10E-03	1.37E-03	1.30E-03	1.66E-03	1.28E-03	1.07E-03	1.52E-03	1.07E-03	2.06E-03
	Oncorhynchus nerka (sockeye salmon)	2.14E-03	1.22E-03	1.74E-03	1.19E-03	1.48E-03	1.41E-03	1.80E-03	1.38E-03	1.16E-03	1.65E-03	1.16E-03	2.23E-03
	Oncorhynchus tshawytscha (chinook salmon)	7.21E-03	4.09E-03	5.87E-03	4.00E-03	4.99E-03	4.76E-03	6.05E-03	4.66E-03	3.92E-03	5.54E-03	3.89E-03	7.52E-03
	Oncorhynchus mykiss (rainbow trout)	4.67E-03	2.65E-03	3.80E-03	2.59E-03	3.23E-03	3.08E-03	3.92E-03	3.02E-03	2.54E-03	3.58E-03	2.52E-03	4.87E-03
	Salmo salar (Atlantic salmon)	1.48E-03	8.40E-04	1.20E-03	8.21E-04	1.02E-03	9.76E-04	1.24E-03	9.56E-04	8.04E-04	1.14E-03	7.98E-04	1.54E-03
	Salvelinus fontinalis (brook trout)	1.53E-03	8.70E-04	1.25E-03	8.50E-04	1.06E-03	1.01E-03	1.29E-03	9.90E-04	8.33E-04	1.18E-03	8.26E-04	1.60E-03

Table 5-14. Chronic Salmonid Hazard Quotients for Elliott Bay—Without CSOs

		Hazard Quotients											
COPC	**Species**	**January**	**February**	**March**	**April**	**May**	**June**	**July**	**August**	**September**	**October**	**November**	**December**
Copper	*Oncorhynchus mykiss* (rainbow trout)	1.23E-02	1.32E-02	1.43E-02	1.09E-02	1.16E-02	1.26E-02	1.24E-02	1.23E-02	9.89E-03	1.28E-02	1.11E-02	1.24E-02
	Salmo trutta (brown trout)	7.59E-03	8.14E-03	8.79E-03	6.73E-03	7.13E-03	7.80E-03	7.66E-03	7.61E-03	6.10E-03	7.88E-03	6.85E-03	7.67E-03
	Salvelinus fontinalis (brook trout)	1.44E-02	1.55E-02	1.67E-02	1.28E-02	1.36E-02	1.48E-02	1.45E-02	1.45E-02	1.16E-02	1.50E-02	1.30E-02	1.46E-02
	Salvelinus namaycush (lake trout)	7.67E-03	8.22E-03	8.88E-03	6.80E-03	7.21E-03	7.88E-03	7.74E-03	7.69E-03	6.16E-03	7.96E-03	6.92E-03	7.75E-03
	Salmo salar (Atlantic salmon)	5.78E-03	6.20E-03	6.69E-03	5.12E-03	5.43E-03	5.94E-03	5.83E-03	5.80E-03	4.64E-03	6.00E-03	5.22E-03	5.84E-03
	Salmo clarkii (cutthroat trout)	9.59E-03	1.03E-02	1.11E-02	8.50E-03	9.01E-03	9.85E-03	9.67E-03	9.62E-03	7.70E-03	9.95E-03	8.65E-03	9.68E-03
	Oncorhynchus tshawytscha (chinook salmon)	1.51E-02	1.62E-02	1.75E-02	1.34E-02	1.42E-02	1.55E-02	1.52E-02	1.51E-02	1.21E-02	1.56E-02	1.36E-02	1.52E-02
	Oncorhynchus nerka (sockeye salmon)	2.72E-03	2.91E-03	3.15E-03	2.41E-03	2.55E-03	2.79E-03	2.74E-03	2.73E-03	2.18E-03	2.82E-03	2.45E-03	2.74E-03
	Oncorhynchus kisutch (coho salmon)	9.06E-03	9.71E-03	1.05E-02	8.03E-03	8.51E-03	9.31E-03	9.14E-03	9.09E-03	7.28E-03	9.40E-03	8.18E-03	9.15E-03
Lead	*Oncorhynchus mykiss* (rainbow trout)	4.55E-05	3.39E-05	4.31E-05	2.63E-05	2.42E-05	3.10E-05	3.52E-05	3.25E-05	3.29E-05	3.89E-05	3.00E-05	3.98E-05
	Salvelinus fontinalis (brook trout)	3.71E-05	2.76E-05	3.51E-05	2.14E-05	1.97E-05	2.52E-05	2.86E-05	2.65E-05	2.68E-05	3.17E-05	2.44E-05	3.24E-05
Nickel	*Oncorhynchus mykiss* (rainbow trout)	2.20E-03	2.08E-03	2.86E-02	1.94E-03	2.07E-03	2.30E-03	2.13E-03	1.78E-03	2.19E-03	2.14E-03	2.13E-03	2.18E-03
Zinc	*Oncorhynchus nerka* (sockeye salmon)	2.63E-02	7.84E-03	4.69E-03	6.19E-03	6.79E-03	5.76E-03	6.97E-03	3.33E-03	3.11E-03	5.62E-03	4.39E-03	4.64E-03
	Oncorhynchus tshawytscha (chinook salmon)	1.29E-02	3.84E-03	2.30E-03	3.03E-03	3.33E-03	2.82E-03	3.42E-03	1.63E-03	1.53E-03	2.76E-03	2.15E-03	2.28E-03
	Oncorhynchus mykiss (rainbow trout)	1.10E-02	3.28E-03	1.96E-03	2.59E-03	2.84E-03	2.41E-03	2.91E-03	1.39E-03	1.30E-03	2.35E-03	1.84E-03	1.94E-03
	Salvelinus fontinalis (brook trout)	9.36E-03	2.79E-03	1.67E-03	2.20E-03	2.42E-03	2.05E-03	2.48E-03	1.19E-03	1.11E-03	2.00E-03	1.56E-03	1.65E-03
	Oncorhynchus kisutch (coho salmon)	1.17E-02	3.48E-03	2.08E-03	2.75E-03	3.01E-03	2.56E-03	3.09E-03	1.48E-03	1.38E-03	2.50E-03	1.95E-03	2.06E-03
	Salmo salar (Atlantic salmon)	8.73E-03	2.60E-03	1.56E-03	2.06E-03	2.25E-03	1.91E-03	2.32E-03	1.11E-03	1.03E-03	1.87E-03	1.46E-03	1.54E-03
TBT	*Oncorhynchus tshawytscha* (chinook salmon)	3.40E-02	1.10E-02	6.88E-02	3.63E-03	1.77E-03	1.36E-03	1.10E-03	1.38E-03	4.49E-01	3.49E-02	2.54E-02	1.48E-02
	Oncorhynchus mykiss (rainbow trout)	1.09E-02	3.52E-03	2.20E-02	1.16E-03	5.65E-04	4.34E-04	3.50E-04	4.41E-04	1.44E-01	1.11E-02	8.13E-03	4.72E-03
	Salvelinus namaycush (lake trout)	3.90E-03	1.26E-03	7.90E-03	4.17E-04	2.03E-04	1.56E-04	1.26E-04	1.58E-04	5.15E-02	4.00E-03	2.92E-03	1.69E-03
PCB	*Salvelinus fontinalis* (brook trout)	9.30E-03	2.54E-03	1.13E-02	6.02E-04	6.19E-04	3.27E-04	1.41E-04	2.61E-04	7.33E-01	2.04E-02	1.26E-02	6.98E-03

Table 5-15. Acute Hazard Quotients for Salmonids in Critical Habitat Cells of the Duwamish River—Baseline

		Hazard Quotients by Cell														
COPC	**Species**	**3**	**44**	**45**	**78**	**112**	**113**	**118**	**119**	**120**	**124**	**125**	**126**	**130**	**131**	**132**
Arsenic	*Oncorhynchus mykiss* (rainbow trout)	<1.0E-07	<1.0E-07	<1.0E-07	9.34E-05	1.15E-04	1.14E-04	1.17E-04	1.17E-04	1.04E-04	1.18E-04	1.16E-04	9.77E-05	1.14E-04	1.02E-04	9.92E-05
	Salvelinus fontinalis (brook trout)	<1.0E-07	<1.0E-07	<1.0E-07	8.32E-05	1.03E-04	1.01E-04	1.05E-04	1.05E-04	9.26E-05	1.05E-04	1.03E-04	8.71E-05	1.02E-04	9.12E-05	8.85E-05
Copper	*Salvelinus fontinalis* (brook trout)	<1.0E-07	<1.0E-07	<1.0E-07	6.43E-03	6.40E-03	6.03E-03	6.71E-03	6.46E-03	5.71E-03	6.65E-03	6.75E-03	5.91E-03	6.79E-03	6.73E-03	5.79E-03
	Salmo salar (Atlantic salmon)	<1.0E-07	<1.0E-07	<1.0E-07	6.46E-03	6.43E-03	6.06E-03	6.75E-03	6.49E-03	5.73E-03	6.68E-03	6.78E-03	5.94E-03	6.82E-03	6.76E-03	5.82E-03
	Oncorhynchus mykiss (rainbow trout)	<1.0E-07	<1.0E-07	<1.0E-07	1.69E-02	1.68E-02	1.59E-02	1.77E-02	1.70E-02	1.50E-02	1.75E-02	1.78E-02	1.55E-02	1.79E-02	1.77E-02	1.52E-02
	Salmo clarkii (cutthroat trout)	<1.0E-07	<1.0E-07	<1.0E-07	1.07E-02	1.07E-02	1.00E-02	1.12E-02	1.08E-02	9.51E-03	1.11E-02	1.12E-02	9.85E-03	1.13E-02	1.12E-02	9.66E-03
	Oncorhynchus tshawytscha (chinook salmon)	<1.0E-07	<1.0E-07	<1.0E-07	1.69E-02	1.68E-02	1.58E-02	1.76E-02	1.69E-02	1.50E-02	1.74E-02	1.77E-02	1.55E-02	1.78E-02	1.76E-02	1.52E-02
	Oncorhynchus nerka (sockeye salmon)	<1.0E-07	<1.0E-07	<1.0E-07	3.04E-03	3.02E-03	2.85E-03	3.17E-03	3.05E-03	2.70E-03	3.14E-03	3.19E-03	2.79E-03	3.21E-03	3.18E-03	2.74E-03
	Oncorhynchus kisutch (coho salmon)	<1.0E-07	<1.0E-07	<1.0E-07	1.01E-02	1.01E-02	9.49E-03	1.06E-02	1.02E-02	8.99E-03	1.05E-02	1.06E-02	9.31E-03	1.07E-02	1.06E-02	9.12E-03
Lead	*Salvelinus fontinalis* (brook trout)	<1.0E-07	<1.0E-07	<1.0E-07	2.07E-06	2.65E-06	2.63E-06	3.08E-06	2.78E-06	2.23E-06	2.88E-06	2.85E-06	2.12E-06	3.28E-06	2.83E-06	2.35E-06
	Oncorhynchus mykiss (rainbow trout)	<1.0E-07	<1.0E-07	<1.0E-07	4.08E-06	5.23E-06	5.19E-06	6.07E-06	5.48E-06	4.39E-06	5.68E-06	5.62E-06	4.17E-06	6.47E-06	5.58E-06	4.63E-06
Nickel	*Oncorhynchus mykiss* (rainbow trout)	<1.0E-07	<1.0E-07	<1.0E-07	1.82E-05	6.11E-05	5.97E-05	5.45E-05	5.32E-05	5.53E-05	5.32E-05	5.13E-05	4.94E-05	5.26E-05	5.17E-05	4.90E-05
Zinc	*Oncorhynchus kisutch* (coho salmon)	<1.0E-07	<1.0E-07	<1.0E-07	1.55E-03	1.91E-03	1.78E-03	2.17E-03	1.93E-03	1.63E-03	2.04E-03	1.99E-03	1.54E-03	2.21E-03	1.96E-03	1.67E-03
	Oncorhynchus nerka (sockeye salmon)	<1.0E-07	<1.0E-07	<1.0E-07	1.68E-03	2.07E-03	1.93E-03	2.36E-03	2.09E-03	1.77E-03	2.22E-03	2.16E-03	1.67E-03	2.39E-03	2.13E-03	1.81E-03
	Oncorhynchus tshawytscha (chinook salmon)	<1.0E-07	<1.0E-07	<1.0E-07	5.65E-03	6.96E-03	6.50E-03	7.93E-03	7.03E-03	5.96E-03	7.45E-03	7.27E-03	5.60E-03	8.06E-03	7.16E-03	6.07E-03
	Oncorhynchus mykiss (rainbow trout)	<1.0E-07	<1.0E-07	<1.0E-07	3.66E-03	4.50E-03	4.21E-03	5.14E-03	4.55E-03	3.86E-03	4.83E-03	4.71E-03	3.63E-03	5.22E-03	4.64E-03	3.93E-03
	Salmo salar (Atlantic salmon)	<1.0E-07	<1.0E-07	<1.0E-07	1.16E-03	1.43E-03	1.33E-03	1.63E-03	1.44E-03	1.22E-03	1.53E-03	1.49E-03	1.15E-03	1.65E-03	1.47E-03	1.25E-03
	Salvelinus fontinalis (brook trout)	<1.0E-07	<1.0E-07	<1.0E-07	1.20E-03	1.48E-03	1.38E-03	1.69E-03	1.49E-03	1.27E-03	1.58E-03	1.54E-03	1.19E-03	1.71E-03	1.52E-03	1.29E-03

Table 5-16. Acute Hazard Quotients for Salmonids in Critical Habitat Cells of the Duwamish River—Without CSOs

		Hazard Quotients by Cell														
COPC	**Species**	**3**	**44**	**45**	**78**	**112**	**113**	**118**	**119**	**120**	**124**	**125**	**126**	**130**	**131**	**132**
Arsenic	*Oncorhynchus mykiss* (rainbow trout)	<1.0E-07	<1.0E-07	<1.0E-07	9.86E-05	9.70E-05	9.31E-05	9.70E-05	9.37E-05	9.10E-05	9.65E-05	9.70E-05	9.30E-05	<1.0E-07	<1.0E-07	<1.0E-07
	Salvelinus fontinalis (brook trout)	<1.0E-07	<1.0E-07	<1.0E-07	8.80E-05	8.65E-05	8.31E-05	8.65E-05	8.35E-05	8.11E-05	8.61E-05	8.65E-05	8.29E-05	<1.0E-07	<1.0E-07	<1.0E-07
Copper	*Salvelinus fontinalis* (brook trout)	<1.0E-07	<1.0E-07	<1.0E-07	6.34E-03	5.57E-03	5.65E-03	6.43E-03	5.88E-03	5.70E-03	6.21E-03	6.50E-03	5.92E-03	<1.0E-07	<1.0E-07	<1.0E-07
	Salmo salar (Atlantic salmon)	<1.0E-07	<1.0E-07	<1.0E-07	6.37E-03	5.60E-03	5.67E-03	6.46E-03	5.90E-03	5.73E-03	6.24E-03	6.53E-03	5.94E-03	<1.0E-07	<1.0E-07	<1.0E-07
	Oncorhynchus mykiss (rainbow trout)	<1.0E-07	<1.0E-07	<1.0E-07	1.67E-02	1.46E-02	1.48E-02	1.69E-02	1.55E-02	1.50E-02	1.63E-02	1.71E-02	1.56E-02	<1.0E-07	<1.0E-07	<1.0E-07
	Salmo clarkii (cutthroat trout)	<1.0E-07	<1.0E-07	<1.0E-07	1.06E-02	9.28E-03	9.41E-03	1.07E-02	9.79E-03	9.50E-03	1.04E-02	1.08E-02	9.86E-03	<1.0E-07	<1.0E-07	<1.0E-07
	Oncorhynchus tshawytscha (chinook salmon)	<1.0E-07	<1.0E-07	<1.0E-07	1.66E-02	1.46E-02	1.48E-02	1.68E-02	1.54E-02	1.49E-02	1.63E-02	1.70E-02	1.55E-02	<1.0E-07	<1.0E-07	<1.0E-07
	Oncorhynchus nerka (sockeye salmon)	<1.0E-07	<1.0E-07	<1.0E-07	3.00E-03	2.63E-03	2.67E-03	3.04E-03	2.78E-03	2.69E-03	2.94E-03	3.07E-03	2.80E-03	<1.0E-07	<1.0E-07	<1.0E-07
	Oncorhynchus kisutch (coho salmon)	<1.0E-07	<1.0E-07	<1.0E-07	9.98E-03	8.77E-03	8.89E-03	1.01E-02	9.25E-03	8.98E-03	9.78E-03	1.02E-02	9.32E-03	<1.0E-07	<1.0E-07	<1.0E-07
Lead	*Salvelinus fontinalis* (brook trout)	<1.0E-07	<1.0E-07	<1.0E-07	1.97E-06	2.22E-06	2.12E-06	3.06E-06	2.36E-06	2.23E-06	2.86E-06	2.76E-06	2.36E-06	<1.0E-07	<1.0E-07	<1.0E-07
	Oncorhynchus mykiss (rainbow trout)	<1.0E-07	<1.0E-07	<1.0E-07	3.88E-06	4.38E-06	4.19E-06	6.02E-06	4.65E-06	4.39E-06	5.63E-06	5.43E-06	4.65E-06	<1.0E-07	<1.0E-07	<1.0E-07
Nickel	*Oncorhynchus mykiss* (rainbow trout)	<1.0E-07	<1.0E-07	<1.0E-07	1.85E-05	6.97E-05	6.62E-05	6.23E-05	6.37E-05	6.14E-05	6.10E-05	6.01E-05	5.75E-05	<1.0E-07	<1.0E-07	<1.0E-07
Zinc	*Oncorhynchus kisutch* (coho salmon)	<1.0E-07	<1.0E-07	<1.0E-07	1.53E-03	1.52E-03	1.52E-03	2.21E-03	1.69E-03	1.54E-03	2.07E-03	1.84E-03	1.69E-03	<1.0E-07	<1.0E-07	<1.0E-07
	Oncorhynchus nerka (sockeye salmon)	<1.0E-07	<1.0E-07	<1.0E-07	1.66E-03	1.64E-03	1.65E-03	2.40E-03	1.83E-03	1.67E-03	2.24E-03	2.00E-03	1.84E-03	<1.0E-07	<1.0E-07	<1.0E-07
	Oncorhynchus tshawytscha (chinook salmon)	<1.0E-07	<1.0E-07	<1.0E-07	5.60E-03	5.53E-03	5.55E-03	8.06E-03	6.17E-03	5.62E-03	7.55E-03	6.72E-03	6.17E-03	<1.0E-07	<1.0E-07	<1.0E-07
	Oncorhynchus mykiss (rainbow trout)	<1.0E-07	<1.0E-07	<1.0E-07	3.62E-03	3.58E-03	3.59E-03	5.22E-03	4.00E-03	3.64E-03	4.89E-03	4.35E-03	4.00E-03	<1.0E-07	<1.0E-07	<1.0E-07
	Salmo salar (Atlantic salmon)	<1.0E-07	<1.0E-07	<1.0E-07	1.15E-03	1.13E-03	1.14E-03	1.65E-03	1.27E-03	1.15E-03	1.55E-03	1.38E-03	1.27E-03	<1.0E-07	<1.0E-07	<1.0E-07
	Salvelinus fontinalis (brook trout)	<1.0E-07	<1.0E-07	<1.0E-07	1.19E-03	1.17E-03	1.18E-03	1.71E-03	1.31E-03	1.20E-03	1.60E-03	1.43E-03	1.31E-03	<1.0E-07	<1.0E-07	<1.0E-07

Table 5-17. Chronic Hazard Quotients for Salmonids in Critical Habitat Cells of the Duwamish River—Baseline

		Hazard Quotients by Cell														
COPC	**Species**	**3**	**44**	**45**	**78**	**112**	**113**	**118**	**119**	**120**	**124**	**125**	**126**	**130**	**131**	**132**
Copper	*Oncorhynchus mykiss* (rainbow trout)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
	Salmo trutta (brown trout)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
	Salvelinus fontinalis (brook trout)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
	Salvelinus namaycush (lake trout)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
	Salmo salar (Atlantic salmon)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
	Salmo clarkii (cutthroat trout)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
	Oncorhynchus tshawytscha (chinook salmon)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
	Oncorhynchus nerka (sockeye salmon)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
	Oncorhynchus kisutch (coho salmon)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
Lead	*Oncorhynchus mykiss* (rainbow trout)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
	Salvelinus fontinalis (brook trout)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
Nickel	*Oncorhynchus mykiss* (rainbow trout)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
TBT	*Oncorhynchus tshawytscha* (chinook salmon)	<1.0E-07	<1.0E-07	<1.0E-07	3.02E-02	1.89E-02	1.84E-02	2.10E-02	2.29E-02	1.80E-02	2.56E-02	2.52E-02	1.09E-02	1.70E-02	1.21E-02	9.10E-03
	Oncorhynchus mykiss (rainbow trout)	<1.0E-07	<1.0E-07	<1.0E-07	9.65E-03	6.05E-03	5.86E-03	6.72E-03	7.32E-03	5.74E-03	8.17E-03	8.04E-03	3.49E-03	5.42E-03	3.85E-03	2.91E-03
	Salvelinus namaycush (lake trout)	<1.0E-07	<1.0E-07	<1.0E-07	3.46E-03	2.17E-03	2.11E-03	2.41E-03	2.63E-03	2.06E-03	2.93E-03	2.89E-03	1.25E-03	1.95E-03	1.38E-03	1.04E-03
PCB	*Salvelinus fontinalis* (brook trout)	<1.0E-07	<1.0E-07	<1.0E-07	7.68E-03	5.05E-03	5.10E-03	5.20E-03	5.53E-03	5.18E-03	5.63E-03	5.52E-03	3.27E-03	4.27E-03	3.25E-03	2.56E-03

Table 5-18. Chronic Hazard Quotients for Salmonids in Critical Habitat Cells of the Duwamish River—Without CSOs

COPC	Species	Hazard Quotients by Cell														
		3	**44**	**45**	**78**	**112**	**113**	**118**	**119**	**120**	**124**	**125**	**126**	**130**	**131**	**132**
Copper	*Oncorhynchus mykiss* (rainbow trout)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	1.60E-02	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
	Salmo trutta (brown trout)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	9.89E-03	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
	Salvelinus fontinalis (brook trout)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	1.88E-02	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
	Salvelinus namaycush (lake trout)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	9.99E-03	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
	Salmo salar (Atlantic salmon)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	7.53E-03	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
	Salmo clarkii (cutthroat trout)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	1.25E-02	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
	Oncorhynchus tshawytscha (chinook salmon)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	1.96E-02	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
	Oncorhynchus nerka (sockeye salmon)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	3.54E-03	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
	Oncorhynchus kisutch (coho salmon)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	1.18E-02	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
Lead	*Oncorhynchus mykiss* (rainbow trout)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	5.42E-05	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
	Salvelinus fontinalis (brook trout)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	4.41E-05	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
Nickel	*Oncorhynchus mykiss* (rainbow trout)	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07	2.38E-03	<1.0E-07	<1.0E-07	<1.0E-07	<1.0E-07
TBT	*Oncorhynchus tshawytscha* (chinook salmon)	<1.0E-07	<1.0E-07	<1.0E-07	3.74E-02	8.99E-02	8.63E-02	9.36E-02	8.68E-02	7.45E-02	8.80E-02	9.17E-02	5.48E-02	<1.0E-07	<1.0E-07	<1.0E-07
	Oncorhynchus mykiss (rainbow trout)	<1.0E-07	<1.0E-07	<1.0E-07	1.19E-02	2.87E-02	2.76E-02	2.99E-02	2.77E-02	2.38E-02	2.81E-02	2.93E-02	1.75E-02	<1.0E-07	<1.0E-07	<1.0E-07
	Salvelinus namaycush (lake trout)	<1.0E-07	<1.0E-07	<1.0E-07	4.29E-03	1.03E-02	9.90E-03	1.07E-02	9.95E-03	8.55E-03	1.01E-02	1.05E-02	6.29E-03	<1.0E-07	<1.0E-07	<1.0E-07
PCB	*Salvelinus fontinalis* (brook trout)	<1.0E-07	<1.0E-07	<1.0E-07	7.53E-03	1.63E-02	1.56E-02	1.65E-02	1.49E-02	1.42E-02	1.49E-02	1.55E-02	1.18E-02	<1.0E-07	<1.0E-07	<1.0E-07

Table 5-19. Acute Hazard Quotients for Salmonids in Critical Habitat Cells of Elliott Bay—Baseline

COPC	Species	Hazard Quotients by Cell										
		220	**254**	**270**	**285**	**286**	**299**	**312**	**313**	**326**	**340**	**355**
Arsenic	*Oncorhynchus mykiss* (rainbow trout)	9.17E-04	8.62E-04	9.62E-04	9.20E-04	2.00E-03	2.28E-03	1.67E-03	2.23E-03	1.56E-03	1.74E-03	1.51E-03
	Salvelinus fontinalis (brook trout)	8.18E-04	7.69E-04	8.58E-04	8.21E-04	1.79E-03	2.03E-03	1.49E-03	1.99E-03	1.39E-03	1.55E-03	1.35E-03
Copper	*Salvelinus fontinalis* (brook trout)	2.59E-03	2.61E-03	2.78E-03	2.45E-03	2.10E-03	1.97E-03	2.75E-03	1.97E-03	2.58E-03	2.59E-03	2.61E-03
	Salmo salar (Atlantic salmon)	2.60E-03	2.62E-03	2.80E-03	2.46E-03	2.11E-03	1.97E-03	2.76E-03	1.98E-03	2.60E-03	2.61E-03	2.63E-03
	Oncorhynchus mykiss (rainbow trout)	6.81E-03	6.86E-03	7.32E-03	6.45E-03	5.53E-03	5.17E-03	7.23E-03	5.19E-03	6.80E-03	6.82E-03	6.87E-03
	Salmo clarkii (cutthroat trout)	4.31E-03	4.35E-03	4.64E-03	4.09E-03	3.50E-03	3.28E-03	4.58E-03	3.29E-03	4.31E-03	4.32E-03	4.36E-03
	Oncorhynchus tshawytscha (chinook salmon)	6.78E-03	6.83E-03	7.29E-03	6.43E-03	5.51E-03	5.15E-03	7.21E-03	5.17E-03	6.77E-03	6.80E-03	6.85E-03
	Oncorhynchus nerka (sockeye salmon)	1.22E-03	1.23E-03	1.32E-03	1.16E-03	9.93E-04	9.29E-04	1.30E-03	9.32E-04	1.22E-03	1.23E-03	1.23E-03
	Oncorhynchus kisutch (coho salmon)	4.08E-03	4.11E-03	4.38E-03	3.86E-03	3.31E-03	3.10E-03	4.33E-03	3.11E-03	4.07E-03	4.08E-03	4.12E-03
Lead	*Salvelinus fontinalis* (brook trout)	6.71E-07	6.75E-07	8.22E-07	5.90E-07	4.43E-07	4.00E-07	6.88E-07	4.03E-07	6.13E-07	6.15E-07	6.32E-07
	Oncorhynchus mykiss (rainbow trout)	1.32E-06	1.33E-06	1.62E-06	1.16E-06	8.73E-07	7.88E-07	1.36E-06	7.94E-07	1.21E-06	1.21E-06	1.25E-06
Nickel	*Oncorhynchus mykiss* (rainbow trout)	2.18E-04	2.23E-04	2.64E-04	2.51E-04	3.63E-04	3.75E-04	2.36E-04	3.72E-04	2.32E-04	2.27E-04	2.00E-04
Zinc	*Oncorhynchus kisutch* (coho salmon)	5.65E-04	5.61E-04	7.14E-04	5.35E-04	4.11E-04	3.73E-04	5.36E-04	3.74E-04	4.95E-04	5.06E-04	5.12E-04
	Oncorhynchus nerka (sockeye salmon)	6.12E-04	6.09E-04	7.74E-04	5.81E-04	4.46E-04	4.04E-04	5.81E-04	4.06E-04	5.36E-04	5.49E-04	5.55E-04
	Oncorhynchus tshawytscha (chinook salmon)	2.06E-03	2.05E-03	2.60E-03	1.95E-03	1.50E-03	1.36E-03	1.96E-03	1.37E-03	1.80E-03	1.85E-03	1.87E-03
	Oncorhynchus mykiss (rainbow trout)	1.33E-03	1.33E-03	1.69E-03	1.26E-03	9.71E-04	8.80E-04	1.27E-03	8.84E-04	1.17E-03	1.20E-03	1.21E-03
	Salmo salar (Atlantic salmon)	4.22E-04	4.20E-04	5.34E-04	4.01E-04	3.07E-04	2.79E-04	4.01E-04	2.80E-04	3.70E-04	3.79E-04	3.83E-04
	Salvelinus fontinalis (brook trout)	4.38E-04	4.35E-04	5.53E-04	4.15E-04	3.19E-04	2.89E-04	4.16E-04	2.90E-04	3.84E-04	3.92E-04	3.97E-04

Table 5-20. Acute Hazard Quotients for Salmonids in Critical Habitat Cells of Elliott Bay—Without CSOs

COPC	Species	Hazard Quotients by Cell										
		220	254	270	285	286	299	312	313	326	340	355
Arsenic	*Oncorhynchus mykiss* (rainbow trout)	9.56E-04	9.17E-04	1.06E-03	1.11E-03	1.70E-03	1.86E-03	1.65E-03	1.86E-03	1.51E-03	1.72E-03	1.60E-03
	Salvelinus fontinalis (brook trout)	8.52E-04	8.18E-04	9.47E-04	9.85E-04	1.51E-03	1.66E-03	1.47E-03	1.66E-03	1.35E-03	1.53E-03	1.43E-03
Copper	*Salvelinus fontinalis* (brook trout)	2.56E-03	2.56E-03	2.67E-03	2.41E-03	2.01E-03	1.93E-03	2.03E-03	1.94E-03	1.95E-03	1.97E-03	1.98E-03
	Salmo salar (Atlantic salmon)	2.57E-03	2.57E-03	2.68E-03	2.42E-03	2.02E-03	1.94E-03	2.04E-03	1.95E-03	1.96E-03	1.98E-03	1.99E-03
	Oncorhynchus mykiss (rainbow trout)	6.72E-03	6.73E-03	7.03E-03	6.33E-03	5.29E-03	5.08E-03	5.34E-03	5.10E-03	5.13E-03	5.18E-03	5.20E-03
	Salmo clarkii (cutthroat trout)	4.26E-03	4.26E-03	4.45E-03	4.01E-03	3.35E-03	3.22E-03	3.38E-03	3.23E-03	3.25E-03	3.28E-03	3.30E-03
	Oncorhynchus tshawytscha (chinook salmon)	6.70E-03	6.70E-03	7.00E-03	6.31E-03	5.27E-03	5.06E-03	5.32E-03	5.09E-03	5.12E-03	5.16E-03	5.18E-03
	Oncorhynchus nerka (sockeye salmon)	1.21E-03	1.21E-03	1.26E-03	1.14E-03	9.51E-04	9.12E-04	9.59E-04	9.17E-04	9.22E-04	9.30E-04	9.34E-04
	Oncorhynchus kisutch (coho salmon)	4.02E-03	4.03E-03	4.21E-03	3.79E-03	3.17E-03	3.04E-03	3.20E-03	3.06E-03	3.07E-03	3.10E-03	3.11E-03
Lead	*Salvelinus fontinalis* (brook trout)	6.57E-07	6.69E-07	7.94E-07	5.67E-07	4.23E-07	3.86E-07	3.43E-07	3.91E-07	3.12E-07	3.20E-07	3.22E-07
	Oncorhynchus mykiss (rainbow trout)	1.30E-06	1.32E-06	1.56E-06	1.12E-06	8.34E-07	7.60E-07	6.77E-07	7.70E-07	6.15E-07	6.30E-07	6.34E-07
Nickel	*Oncorhynchus mykiss* (rainbow trout)	2.64E-04	2.67E-04	2.76E-04	2.46E-04	3.38E-04	3.57E-04	2.14E-04	3.54E-04	2.01E-04	2.00E-04	1.81E-04
Zinc	*Oncorhynchus kisutch* (coho salmon)	5.54E-04	5.62E-04	6.90E-04	5.14E-04	3.93E-04	3.60E-04	3.67E-04	3.63E-04	3.25E-04	3.40E-04	3.42E-04
	Oncorhynchus nerka (sockeye salmon)	6.01E-04	6.10E-04	7.48E-04	5.57E-04	4.26E-04	3.91E-04	3.98E-04	3.93E-04	3.52E-04	3.69E-04	3.70E-04
	Oncorhynchus tshawytscha (chinook salmon)	2.02E-03	2.05E-03	2.52E-03	1.88E-03	1.43E-03	1.31E-03	1.34E-03	1.32E-03	1.18E-03	1.24E-03	1.25E-03
	Oncorhynchus mykiss (rainbow trout)	1.31E-03	1.33E-03	1.63E-03	1.21E-03	9.28E-04	8.51E-04	8.68E-04	8.57E-04	7.67E-04	8.04E-04	8.07E-04
	Salmo salar (Atlantic salmon)	4.15E-04	4.21E-04	5.16E-04	3.85E-04	2.94E-04	2.70E-04	2.75E-04	2.72E-04	2.43E-04	2.55E-04	2.56E-04
	Salvelinus fontinalis (brook trout)	4.30E-04	4.36E-04	5.35E-04	3.99E-04	3.05E-04	2.79E-04	2.85E-04	2.81E-04	2.52E-04	2.64E-04	2.65E-04

Table 5-21. Chronic Hazard Quotients for Salmonids in Critical Habitat Cells of Elliott Bay—Baseline

COPC	Species	Hazard Quotients by Cells										
		220	254	270	285	286	299	312	313	326	340	355
Copper	*Oncorhynchus mykiss* (rainbow trout)	7.20E-03	7.16E-03	6.38E-03	6.64E-03	5.42E-03	5.31E-03	6.21E-03	5.34E-03	5.97E-03	6.03E-03	6.15E-03
	Salmo trutta (brown trout)	4.44E-03	4.42E-03	3.94E-03	4.09E-03	3.34E-03	3.28E-03	3.83E-03	3.29E-03	3.68E-03	3.72E-03	3.79E-03
	Salvelinus fontinalis (brook trout)	8.43E-03	8.39E-03	7.48E-03	7.78E-03	6.35E-03	6.23E-03	7.27E-03	6.26E-03	7.00E-03	7.07E-03	7.21E-03
	Salvelinus namaycush (lake trout)	4.48E-03	4.46E-03	3.98E-03	4.14E-03	3.38E-03	3.31E-03	3.87E-03	3.33E-03	3.72E-03	3.76E-03	3.83E-03
	Salmo salar (Atlantic salmon)	3.38E-03	3.36E-03	3.00E-03	3.12E-03	2.54E-03	2.49E-03	2.91E-03	2.51E-03	2.80E-03	2.83E-03	2.89E-03
	Salmo clarkii (cutthroat trout)	5.60E-03	5.58E-03	4.97E-03	5.17E-03	4.22E-03	4.14E-03	4.83E-03	4.16E-03	4.65E-03	4.70E-03	4.79E-03
	Oncorhynchus tshawytscha (chinook salmon)	8.81E-03	8.77E-03	7.82E-03	8.13E-03	6.64E-03	6.50E-03	7.60E-03	6.54E-03	7.31E-03	7.39E-03	7.53E-03
	Oncorhynchus nerka (sockeye salmon)	1.59E-03	1.58E-03	1.41E-03	1.47E-03	1.20E-03	1.17E-03	1.37E-03	1.18E-03	1.32E-03	1.33E-03	1.36E-03
	Oncorhynchus kisutch (coho salmon)	5.29E-03	5.27E-03	4.70E-03	4.88E-03	3.99E-03	3.91E-03	4.57E-03	3.93E-03	4.39E-03	4.44E-03	4.53E-03
Lead	*Oncorhynchus mykiss* (rainbow trout)	1.67E-05	1.67E-05	1.52E-05	1.48E-05	9.63E-06	8.73E-06	1.11E-05	8.85E-06	1.03E-05	1.05E-05	1.09E-05
	Salvelinus fontinalis (brook trout)	1.36E-05	1.36E-05	1.24E-05	1.20E-05	7.84E-06	7.11E-06	9.05E-06	7.20E-06	8.40E-06	8.55E-06	8.83E-06
Nickel	*Oncorhynchus mykiss* (rainbow trout)	1.13E-02	1.16E-02	1.24E-02	1.27E-02	1.46E-02	1.49E-02	1.06E-02	1.46E-02	1.10E-02	1.07E-02	1.06E-02
TBT	*Oncorhynchus tshawytscha* (chinook salmon)	1.07E-03	9.48E-04	9.80E-04	9.36E-04	3.54E-04	2.49E-04	1.09E-03	2.07E-04	1.03E-03	8.38E-04	4.09E-04
	Oncorhynchus mykiss (rainbow trout)	3.41E-04	3.03E-04	3.13E-04	2.99E-04	1.13E-04	7.95E-05	3.50E-04	6.61E-05	3.28E-04	2.68E-04	1.31E-04
	Salvelinus namaycush (lake trout)	1.23E-04	1.09E-04	1.12E-04	1.07E-04	4.06E-05	2.85E-05	1.26E-04	2.37E-05	1.18E-04	9.61E-05	4.69E-05
PCB	*Salvelinus fontinalis* (brook trout)	2.07E-04	1.83E-04	1.90E-04	1.82E-04	6.78E-05	4.78E-05	2.18E-04	3.95E-05	2.03E-04	1.64E-04	7.89E-05

Table 5-22. Chronic Hazard Quotients for Salmonids in Critical Habitat Cells of Elliott Bay—Without CSOs

		Hazard Quotients by Cell										
COPC	**Species**	**220**	**254**	**270**	**285**	**286**	**299**	**312**	**313**	**326**	**340**	**355**
Copper	*Oncorhynchus mykiss* (rainbow trout)	7.02E-03	7.01E-03	6.31E-03	6.58E-03	5.34E-03	5.25E-03	5.48E-03	5.28E-03	5.28E-03	5.39E-03	5.48E-03
	Salmo trutta (brown trout)	4.33E-03	4.32E-03	3.89E-03	4.06E-03	3.29E-03	3.24E-03	3.38E-03	3.25E-03	3.26E-03	3.33E-03	3.38E-03
	Salvelinus fontinalis (brook trout)	8.22E-03	8.21E-03	7.40E-03	7.71E-03	6.26E-03	6.15E-03	6.42E-03	6.19E-03	6.19E-03	6.32E-03	6.43E-03
	Salvelinus namaycush (lake trout)	4.37E-03	4.36E-03	3.93E-03	4.10E-03	3.33E-03	3.27E-03	3.41E-03	3.29E-03	3.29E-03	3.36E-03	3.42E-03
	Salmo salar (Atlantic salmon)	3.29E-03	3.29E-03	2.96E-03	3.09E-03	2.51E-03	2.46E-03	2.57E-03	2.48E-03	2.48E-03	2.53E-03	2.57E-03
	Salmo clarkii (cutthroat trout)	5.46E-03	5.46E-03	4.92E-03	5.12E-03	4.16E-03	4.09E-03	4.27E-03	4.11E-03	4.11E-03	4.20E-03	4.27E-03
	Oncorhynchus tshawytscha (chinook salmon)	8.59E-03	8.58E-03	7.73E-03	8.05E-03	6.54E-03	6.43E-03	6.71E-03	6.46E-03	6.47E-03	6.60E-03	6.71E-03
	Oncorhynchus nerka (sockeye salmon)	1.55E-03	1.55E-03	1.39E-03	1.45E-03	1.18E-03	1.16E-03	1.21E-03	1.17E-03	1.17E-03	1.19E-03	1.21E-03
	Oncorhynchus kisutch (coho salmon)	5.16E-03	5.15E-03	4.65E-03	4.84E-03	3.93E-03	3.86E-03	4.03E-03	3.88E-03	3.89E-03	3.97E-03	4.03E-03
Lead	*Oncorhynchus mykiss* (rainbow trout)	1.61E-05	1.62E-05	1.49E-05	1.46E-05	9.68E-06	8.78E-06	8.23E-06	8.91E-06	7.92E-06	8.03E-06	8.16E-06
	Salvelinus fontinalis (brook trout)	1.31E-05	1.32E-05	1.22E-05	1.18E-05	7.88E-06	7.15E-06	6.70E-06	7.25E-06	6.45E-06	6.53E-06	6.64E-06
Nickel	*Oncorhynchus mykiss* (rainbow trout)	1.25E-02	1.26E-02	1.33E-02	1.34E-02	1.52E-02	1.55E-02	1.18E-02	1.52E-02	1.15E-02	1.17E-02	1.19E-02
TBT	*Oncorhynchus tshawytscha* (chinook salmon)	1.00E-03	9.53E-04	9.51E-04	8.48E-04	7.94E-04	7.48E-04	2.93E-03	6.70E-04	2.93E-03	2.71E-03	1.55E-03
	Oncorhynchus mykiss (rainbow trout)	3.20E-04	3.05E-04	3.04E-04	2.71E-04	2.53E-04	2.39E-04	9.34E-04	2.14E-04	9.37E-04	8.65E-04	4.96E-04
	Salvelinus namaycush (lake trout)	1.15E-04	1.09E-04	1.09E-04	9.73E-05	9.10E-05	8.58E-05	3.36E-04	7.69E-05	3.36E-04	3.11E-04	1.78E-04
PCB	*Salvelinus fontinalis* (brook trout)	1.67E-04	2.45E-04	1.60E-04	1.41E-04	1.34E-04	1.26E-04	5.37E-04	1.13E-04	5.33E-04	4.88E-04	2.66E-04

Table 5-23. Salmonid Hazard Quotients for Chemicals in Prey Items for Each Dietary TRV Found in the Literature

Chemical Name	Test Species	Study Authors	Hazard Quotients
Aroclor 1254	*Salvelinus namaycush*	Mac and Seelye (1981)	0.059
Aroclor 1254	*Salvelinus namaycush*	Mac and Seelye (1981)	0.059
Aroclor 1254	*Salvelinus namaycush*	Mac and Seelye (1981)	0.059
Aroclor 1254	*Oncorhynchus mykiss*	Mayer et al. (1977)	<0.001
Aroclor 1260	*Oncorhynchus mykiss*	Mayer et al. (1977)	<0.001
Copper	*Oncorhynchus mykiss*	Lanno et al. (1985)	0.293
Copper	*Oncorhynchus mykiss*	Lanno et al. (1985)	0.131
Copper	*Oncorhynchus mykiss*	Lanno et al. (1985)	0.082
Copper	*Oncorhynchus mykiss*	Lanno et al. (1985)	0.063
Copper	*Oncorhynchus mykiss*	Lanno et al. (1985)	0.042
Copper	*Oncorhynchus mykiss*	Lanno et al. (1985)	0.038
Copper	*Oncorhynchus mykiss*	Lanno et al. (1985)	0.027
Copper	*Oncorhynchus mykiss*	Lanno et al. (1985)	0.021
Copper	*Oncorhynchus mykiss*	Lanno et al. (1985)	0.021
Copper	*Oncorhynchus mykiss*	Lanno et al. (1985)	0.016
Copper	*Oncorhynchus mykiss*	Lanno et al. (1985)	0.016
Copper	*Oncorhynchus mykiss*	Lanno et al. (1985)	0.015
Copper	*Oncorhynchus mykiss*	Lanno et al. (1985)	0.014
Copper	*Oncorhynchus mykiss*	Lanno et al. (1985)	0.014
Copper	*Oncorhynchus mykiss*	Lanno et al. (1985)	0.007
Copper	*Oncorhynchus mykiss*	Lanno et al. (1985)	0.004
Copper	*Oncorhynchus mykiss*	Miller et al. (1993)	0.833
Copper	*Oncorhynchus mykiss*	Miller et al. (1993)	0.02
Lead	*Oncorhynchus mykiss*	Goettl and Davies (1976)	0.0002

Table 5-24. Tier 3: Average Percent of Aquatic Species at Risk from Dissolved Arsenic

Month	Acute			
	Duwamish River		Elliott Bay	
	Baseline	Without CSOs	Baseline	Without CSOs
January	0%	0%	0%	0%
February	0%	0%	0%	0%
March	0%	0%	1%	1%
April	0%	0%	0%	0%
May	0%	0%	0%	0%
June	0%	0%	0%	0%
July	0%	0%	0%	0%
August	0%	0%	0%	0%
September	0%	0%	0%	0%
October	0%	0%	0%	0%
November	0%	0%	0%	0%
December	0%	0%	0%	0%

Table 5-25. Tier 3: Average Percent of Aquatic Species at Risk from Dissolved Copper

Month	Acute				Chronic			
	Duwamish River		Elliott Bay		Duwamish River		Elliott Bay	
	Baseline	Without CSOs	Baseline	Without CSOs	Baseline	Without CSOs	Baseline	Without CSOs
January	1%	1%	0%	0%	1%	1%	0%	0%
February	1%	1%	0%	0%	1%	1%	1%	1%
March	1%	1%	1%	0%	1%	1%	1%	1%
April	1%	1%	0%	0%	1%	1%	1%	1%
May	1%	1%	1%	1%	1%	1%	1%	1%
June	1%	1%	0%	0%	1%	1%	1%	1%
July	1%	1%	0%	0%	1%	1%	1%	1%
August	1%	1%	0%	0%	1%	1%	1%	1%
September	1%	1%	1%	1%	1%	1%	1%	1%
October	1%	1%	0%	0%	2%	2%	1%	1%
November	1%	1%	0%	0%	1%	1%	1%	1%
December	1%	1%	0%	0%	1%	1%	0%	0%

Table 5-26. Tier 3: Average Percent of Aquatic Species at Risk from Dissolved Lead

Month	Acute				Chronic			
	Duwamish River		Elliott Bay		Duwamish River		Elliott Bay	
	Baseline	Without CSOs	Baseline	Without CSOs	Baseline	Without CSOs	Baseline	Without CSOs
January	0%	0%	0%	0%	0%	0%	0%	0%
February	0%	0%	0%	0%	0%	0%	0%	0%
March	0%	0%	0%	0%	0%	0%	0%	0%
April	0%	0%	0%	0%	0%	0%	0%	0%
May	0%	0%	0%	0%	0%	0%	0%	0%
June	0%	0%	0%	0%	0%	0%	0%	0%
July	0%	0%	0%	0%	0%	0%	0%	0%
August	0%	0%	0%	0%	0%	0%	0%	0%
September	0%	0%	0%	0%	0%	0%	0%	0%
October	0%	0%	0%	0%	0%	0%	0%	0%
November	0%	0%	0%	0%	0%	0%	0%	0%
December	0%	0%	0%	0%	0%	0%	0%	0%

Table 5-27. Tier 3: Average Percent of Aquatic Species at Risk from Dissolved Nickel

Month	Acute				Chronic			
	Duwamish River		Elliott Bay		Duwamish River		Elliott Bay	
	Baseline	Without CSOs	Baseline	Without CSOs	Baseline	Without CSOs	Baseline	Without CSOs
January	0%	0%	0%	0%	0%	0%	0%	0%
February	0%	0%	0%	0%	0%	0%	0%	0%
March	0%	0%	1%	1%	1%	1%	3%	3%
April	0%	0%	0%	0%	0%	0%	0%	0%
May	0%	0%	0%	0%	0%	0%	0%	0%
June	0%	0%	0%	0%	0%	0%	0%	0%
July	0%	0%	0%	0%	0%	0%	0%	0%
August	0%	0%	0%	0%	0%	0%	0%	0%
September	0%	0%	0%	0%	1%	1%	0%	0%
October	0%	0%	0%	0%	1%	1%	0%	0%
November	0%	0%	0%	0%	0%	0%	0%	0%
December	0%	0%	0%	0%	0%	0%	0%	0%

Table 5-28. Tier 3: Average Percent of Aquatic Species at Risk from Dissolved Zinc

	Acute			
	Duwamish River		Elliott Bay	
Month	Baseline	Without CSOs	Baseline	Without CSOs
January	0%	0%	0%	0%
February	0%	0%	0%	0%
March	0%	0%	1%	1%
April	0%	0%	0%	0%
May	0%	0%	0%	0%
June	0%	0%	0%	0%
July	0%	0%	0%	0%
August	0%	0%	0%	0%
September	0%	0%	0%	0%
October	0%	0%	0%	0%
November	0%	0%	0%	0%
December	0%	0%	0%	0%

Table 5-29. Tier 3: Average Percent of Aquatic Species at Risk from Tributyltin

	Chronic			
	Duwamish River		Elliott Bay	
Month[a]	Baseline	Without CSOs	Baseline	Without CSOs
January	1%	1%	0%	0%
February	0%	0%	0%	0%
March	1%	2%	0%	1%
April	0%	0%	0%	0%
May	0%	0%	0%	0%
June	0%	0%	0%	0%
July	0%	0%	0%	0%
August	0%	0%	0%	0%
November	2%	2%	0%	0%
December	1%	1%	0%	0%

[a] September and October were not evaluated due to model initial conditions.

Figure 5-1. Acute EEC and Marine Toxicity Distributions for Dissolved Arsenic

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Figure 5-2. Acute and Chronic EEC and Marine Toxicity Distributions for Dissolved Copper

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Figure 5-3. Acute and Chronic EEC and Marine Toxicity Distributions for Dissolved Lead

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Figure 5-4. Acute and Chronic EEC and Marine Toxicity Distributions for Dissolved Nickel

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Figure 5-5. Acute EEC and Marine Toxicity Distributions for Dissolved Zinc

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Figure 5-6. Chronic EEC and Marine Toxicity Distributions for Dissolved TBT

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5.2 Sediment Risk Characterization

The evaluation of risks to benthos was based on a comparison of measured nearfield and model-predicted farfield sediment COPC concentrations to Tier 1 sediment TRVs. As described in Section 2.2, most of the Tier 1 sediment TRVs were based on Washington State sediment management standards. The model-predicted sediment concentrations were for the top ten centimeter layer, at the end of the one year baseline and without CSO simulations. We also reran the one-year simulation for ten sequential years, to discern whether differences between baseline and without CSO concentrations in the top ten centimeters increased from the first simulated year to the tenth. The methods and results of a benthic survey comparing a nearfield site at the Duwamish/Diagonal CSO and storm drain to a farfield site at Kellogg Island are presented in Section 7.

5.2.1 Farfield Sediment Risks

The comparison of sediment chemical concentrations to the Tier 1 sediment TRVs indicates that there are potential risks to benthic organisms in the sediments of the Duwamish River and Elliott Bay (Table 5-30). These risks are fairly widespread. Chemicals contributing to these risks include mercury, the organometalloid TBT, and several organic compounds (PAHs, PCBs, bis(2-ethylhexyl)phthalate, and 1,4-dichlorobenzene). The sediment concentrations of a few other chemicals exceeded sediment management standards occasionally. These included arsenic (1 percent of cells) cadmium (4 percent of cells), copper (2 percent of cells), lead (less than 1/10 of 1 percent of cells). Nickel slightly exceeded it's sediment management standard (maximum HQ = 2.3) over a large portion of the study area (82 percent of cells), but its maximum concentration was three times higher in reference sediments than in the study area.

5.2.2 Results of 10-Year Simulation of Sediment Concentrations

Our determination of risks to sediment-dwelling organisms was based on the results of the one-year model simulation. However, we were concerned that the period of one-year could be insufficient to detect changes in sediment concentrations following the elimination of CSO discharges to the Duwamish River and Elliott Bay. Consequently, an additional 10-year modeling simulation was conducted for concentrations of 1,4-dichlorobenzene, bis(2-ethylhexyl)phthalate, chrysene, copper, lead, mercury, and total PCBs in sediments to answer the following question:

- Does the difference between baseline and without CSO risks significantly change after 10 years of model simulation relative to the difference after one year of model simulation, assuming all other sources remain at baseline levels?

Table 5-30. Summary of Study Area and Reference Site Sediment Hazard Quotients

Chemicals	Study Area Baseline Condition			Reference Sediments	
	Maximum	Average	% Cells with HQs > 1	Maximum	Minimum
Arsenic	1.3	0.2	1%	0.4	<0.1
Benzo(a)anthracene	6.7	0.4	9%	<0.1	<0.1
Benzo(a)pyrene	0.8	0.1	0%	<0.1	<0.1
Benzo(b)fluoranthene	2.2	0.2	2%	<0.1	<0.1
Benzo(g,h,i)perylene	0.8	0.1	0%	0.1	<0.1
Benzo(k)fluoranthene	1.8	0.1	1%	<0.1	<0.1
Bis(2-ethylhexyl)phthalate	10.8	1.2	34%	NAV	NAV
Cadmium	1.5	0.3	4%	0.6	<0.1
Chrysene	7	0.4	9%	<0.1	<0.1
Copper	2.1	0.2	2%	0.1	<0.1
Dibenzo(a,h)anthracene	0.1	<0.11	0%	0.2	<0.1
1,4-Dichlorobenzene[a]	3.3	0.5	14%	NAV	NAV
Fluoranthene	10.3	0.4	9%	<0.1	<0.1
Indeno(1,2,3-c,d)pyrene	0.3	<0.13	0%	0.1	<0.1
Lead	2.1	0.1	0%	<0.1	<0.1
Mercury	8.3	0.8	23%	NAV	NAV
4-Methylphenol	4.9	0.2	4%	NAV	NAV
Nickel	2.3	1.3	82%	6.7	0.4
Phenanthrene	4.5	0.3	4%	0.2	<0.1
Pyrene	1.5	0.1	1%	<0.1	<0.1
TBT [a,] (In-House Criterion)	4,777[b]	NAV	NAV	NAV	NAV
TBT[a] (Roy F. Weston Criterion)	8,440[b]	NAV	NAV	NAV	NAV
Total PCBs[a]	27.5	2	63%	NAV	NAV
Zinc	1.4	0.3	1%	0.2	<0.1

[a] The HQs for these four chemicals are the initial conditions rather than the result of the one-year simulation. Initial conditions for these chemicals were regenerated with new data after the model simulations had been completed.

[b] The maximum HQ represented is based on an actual measurement of TBT in sediments located just north of Harbor Island.

N/AV = Not available

To address this question, differences between baseline and without CSO sediment concentrations for these seven chemicals were calculated for the 1-year[29] and 10-year model simulations. These differences between baseline and without CSO sediment concentrations were compared to determine if the magnitudes of the differences were statistically[30] different between the 1-year and 10-year model simulations. No differences in magnitude between the 1-year and 10-year model simulation could be detected for 1,4-dichlorobenzene, lead, mercury, or total PCBs. Differences in magnitude could be detected for bis(2-ethylhexyl)phthalate, chrysene, and copper. For copper, the difference between baseline and without CSO sediment concentrations after the 10-year model simulation was significantly larger than after the 1-year model simulation. This indicates that CSOs are contributing a higher copper concentration to sediments than are other sources. The analysis found a maximum copper concentration of twice the sediment management standard. The difference between baseline and without CSO sediment concentrations for bis(2-ethylhexyl)phthalate and chrysene decreased over this time period. As chrysene was used as the surrogate for all the PAHs in the 10-year simulation, this indicates that the difference in the predicted liver lesions in English sole observed for baseline and without CSOs would also decrease after 10 years of no CSO discharges.

5.2.3 Nearfield Sediment Risks

Risks to benthic organisms in the sediments near CSO discharges (nearfield[31]) were assessed using available chemistry, bioassay (as a measure of toxicity) and benthic survey data. Sediment chemistry data near CSOs in the study area were available for: South Magnolia, Denny Way, King Street, Connecticut Street, Lander, Hanford, Chelan, Duwamish/Diagonal, Brandon Street Michigan Street, West Michigan Street, Eighth Avenue and Norfolk. Sediment bioassay data were available for Denny Way, Duwamish/Diagonal, Connecticut, Chelan, and Hanford. Benthic community analyses

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29 Due to difficulties in initializing the model for 1,4-dichlorobenzene, bis(2-ethylhexyl)phthalate, and chrysene, differences were calculated for the 2-year simulation results.

30 The differences between baseline and without CSO sediment concentrations for the 1-year and 10-year simulations were compared using a Wilcoxon Sign Rank Test (Zar 1984). This test assigns ranks to the absolute value of the differences for each simulation run, then applies the sign (either positive or negative) to these ranks. A positive difference means that the baseline condition had a higher concentration than the without CSO condition at each time period. The signed ranks are summed and then compared to determine if a statistically detectable difference exists between the 1-year and 10-year simulation differences.

31 The nearfield is defined here as the environment directly adjacent to the CSO discharge. The size of the nearfield varies in relation to the volume of the discharge and in most cases is smaller than the farfield model cell. A critical difference between nearfield and farfield is that the farfield model predicts a single concentration for all sediments within a cell while nearfield measurements assessed in this section reflect the observed variability of chemicals in study area sediments near CSOs.

have been conducted at Duwamish/Diagonal (see Section 7 of this appendix) and Denny Way. Risk results from each data type are discussed in the sections that follow.

5.2.3.1 *Chemical Assessments*

Chemical concentrations in nearfield sediments were compared to WSDOE's (1995a) Sediment Quality Standards (SQS) and Cleanup Screening Levels (CSL). The SQS were used as the Tier 1 TRVs for assessing farfield sediment risks and are set at levels believed to result in no adverse effects on biological resources (WSDOE 1995a). The CSLs establish minor adverse effects as the level above which station clusters of potential concern are identified (WSDOE 1995a).

Concentrations of chemicals in sediment samples collected at South Magnolia were all below WSDOE's Sediment Management Standards (EBDRP, 1994). At Denny Way, samples collected from around the sediment cap exceeded the WSDOE's Cleanup Screening Levels (CSLs) for mercury, benzyl alcohol, bis(2-ethylhexyl)phthalate, a few individual PAHs, and silver. The WSDOE's Sediment Quality Standards (SQS) were exceeded for total LPAHs, total HPAHs, 1,4-dichlorobenzene, total PCBs, and butyl benzyl phthalate. (King County, 1996a).

At King Street a wide variety of metals, PAHs and PCB exceeded the CSL (EBDRP, 1994). Additionally, Hart-Crowser (1994) showed that mercury, silver and bis(2-ethylhexyl)phthalate exceeded the CSL and zinc and total PAHs exceeded the SQS. At Connecticut Street three PAHs, bis(2-ethylhexyl)phthalate, and butyl benzyl phthalate exceeded the SQS (King County unpublished data, 1995). At Lander, mercury exceeded the CSL, and a PAH, a phthalate and PCBs exceeded the SQS (EBDRP, 1994). At Hanford total PCBs exceeded the CSL and 1,4-dichlorobenzene, bis(2-ethylhexyl) phthalate, and mercury exceeded the SQS (King County unpublished data, 1995). At Chelan 1,4-dichlorobenzene, phenol, and total PCBs exceeded the SQS (King County unpublished data, 1995).

At Duwamish/Diagonal, mercury, total PCBs, bis(2-ethylhexyl)phthalate, and 4-methylphenol exceeded the CSL; and zinc, benzyl butyl phthalate exceeded the SQS (King County, 1997). Additionally, sediment samples collected near the Duwamish/Diagonal CSO/storm drain as part of the benthic assessment at Duwamish/Diagonal (included as Subappendix D) showed that bis(2-ethylhexyl) phthalate and 2,4-dimethylphenol exceeded the CSL and mercury, benzyl butyl phthalate, and 1,4-dichlorobenzene exceeded the SQS.

At Brandon, mercury, PAHs, PCBs, and phthalates exceeded the CSL (EBDRP, 1994). At Michigan Street PCBs and phthalates exceeded the CSL and PAHs exceeded the SQS (EBDRP, 1994). At West Michigan Street individual PAHs and phthalates exceeded the CSL and some individual PAHs exceeded the SQS (EBDRP, 1994). At Eighth Avenue no chemicals exceeded the SQS (EBDRP, 1994). At Norfolk mercury, total PCBs, 1,4-dichlorobenzene, bis(2ethylhexyl)phthalate, and benzoic acid exceeded the CSL and benzyl butyl phthalate and individual PAHs exceeded the SQS (King County, 1996b).

However, sediment remediation near the Norfolk CSO is expected to occur during the winter of 1999.

The results of sediment chemistry surveys suggest that benthic organisms near CSOs can be exposed to concentrations of chemicals exceeding the Washington Sediment Management standards. Because both the SQS and CSL are exceeded, adverse effects on the biological community are predicted.

5.2.3.2 *Toxicity Assessments*

Sediment toxicity samples were collected at Chelan, Connecticut, Hanford, and Denny Way (King County 1996c, 1998). Six sediment stations were sampled for toxicity at each of the Chelan, Connecticut, and Hanford CSOs. A total of two stations were sampled at the Denny Way CSO. Bioassays were conducted for three test species (Amphipod, Echinoderm, and Polychaete) at all stations.

At the Hanford CSO, two stations failed the amphipod bioassay and the echinoderm bioassay, and a third station failed the echinoderm bioassay. All other sediment bioassays at the Hanford CSO passed.

At the Chelan CSO, one station failed the echinoderm bioassay and another station failed the polychaete bioassay. All other sediment bioassays at the Chelan CSO passed.

At the Connecticut CSO, three stations failed all three bioassays. All sediment bioassays passed at the other Connecticut CSO stations.

At the Denny Way CSO, one station failed the echinoderm bioassay. All other sediment bioassays passed at the Denny Way CSO.

The results of laboratory toxicity tests confirm than some of the sediments near CSO discharges are toxic to benthic organisms and pose a risk to the benthic community.

5.2.3.3 *Benthic Community Assessments*

The benthic community assessment of the Duwamish/Diagonal CSO and storm drain is reported in its entirety in Section 7 of this appendix. Briefly, the assessment confirmed a clear pattern of effects (decreased abundances and species richness) to the benthic community close to outfall, which decreased with distance away from the outfall. As described in Section 7, the effects to the benthic community were correlated with organic enrichment and chemical contamination.

In an earlier study, a series of benthic community assessments were conducted near the Denny Way CSO to assess how benthic invertebrates recolonized the area that was capped just offshore of the Denny Way CSO. Benthic assessments were conducted in 1990, 1991, 1992, 1994, and 1996. Two stations, located on the cap approximately 300 feet offshore and 45 feet deep, were analyzed each year. The assessments showed that the cap was quickly recolonized but that biomass was low in the first years. The community then changed slowly through successional stages and in response to changing grain size and varying concentrations of total organic carbon. The most recent benthic community assessment showed a diverse benthic community where no affects from the CSO discharges could be observed (King County 1994, 1996a, 1998).

The results of these benthic surveys confirm that discharges from CSOs have adversely affected the nearfield benthic communities. These surveys also show that if impacted sediments are capped or otherwise remediated, the nearfield sediments have the potential to be recolonized by a more diverse benthic fauna that shows little or no influence of the CSO.

5.3 Physical Stressors

Salinity, pH, temperature, DO, TSS, sedimentation rate, scouring, and displacement were evaluated for their effect on aquatic life in the Duwamish River and Elliott Bay.

5.3.1 Salinity

To estimate risk to estuarine aquatic organisms (fish, invertebrates), we evaluated the simulated salinity within the study area to determine the percentage of time that salinity at particular locations and depths fell below five parts per thousand (ppt). Five ppt is a threshold level for most marine invertebrates. If salinity is less than five ppt for several days or more, stress and eventually mortality will result (Table 5-31).

The most upstream penetration of the salt wedge, or toe of the salt wedge, occurred not far from the outfall of the Norfolk CSO (model cell #4). In the baseline simulation, this area of the river fell below five ppt salinity all the way to the bottom 42 percent of the year. The surface layer at this location fell below the criterion 84 percent of the time. We know that the salt wedge penetrates to this location in the river and above during the dry season. We also know that during the wet season with the attendant increase in runoff, the extent of the salt wedge penetration is greatly diminished.

The percent of time that salinity fell below five ppt decreased down river. Salinities less than five ppt were limited to the surface layers. For example, at the 8th Avenue CSO (cell # 76), salinity fell below the criterion in the surface layer and at mid-depth 68 and 6 percent of the time, respectively. Below mid-depth, salinity did not fall below the criterion. At the Brandon Street CSO (cell number 111), the salinity fell below the criterion in each of the top two layers of the water column 38 and 2 percent of the time, respectively. In cell number 148, which receives the Chelan Street CSO discharge, salinity fell below the minimum in the top two layers for only 11 and 0.2 percent of the time, respectively. Finally in cell numbers 153 and 161, which receive the Hanford Avenue and Lander Street CSO outfalls, respectively, salinity at the surface fell below the criterion only 1 percent of the time.

Table 5-31. Percent Time and Maximum Duration Below the Minimum Salinity Criterion (Five ppt) at the Model Cell into Which Each CSO Discharges

Layer	Percent of Time Below Minimum	Maximum Duration Below Minimum (days)	Percent of Time Below Minimum	Maximum Duration Below Minimum (days)	Percent of Time Below Minimum	Maximum Duration Below Minimum (days)
	8th Avenue		**Brandon**		**Chelan**	
10 (Top)	68.46%	40	38.09%	14	11.81%	6
9	57.85%	21	2.28%	1	0.23%	0
8	32.9%	8	0%	0	0%	0
7	15.69%	5	0%	0	0%	0
6	6.41%	2	0%	0	0%	0
5	2.88%	1	0%	0	0%	0
4	1.35%	0	0%	0	0%	0
3	0.46%	0	0%	0	0%	0
2	0.31%	0	0%	0	0%	0
1 (Bottom)	0.19%	0	0%	0	0%	0
	Connecticut		**Denny Way**		**Hanford**	
10 (Top)	0%	0	0%	0	1.37%	1
9	0%	0	0%	0	0%	0
8	0%	0	0%	0	0%	0
7	0%	0	0%	0	0%	0
6	0%	0	0%	0	0%	0
5	0%	0	0%	0	0%	0
4	0%	0	0%	0	0%	0
3	0%	0	0%	0	0%	0
2	0%	0	0%	0	0%	0
1 (Bottom)	0%	0	0%	0	0%	0

Table 5-31. Percent Time and Maximum Duration Below the Minimum Salinity Criterion (Five ppt) at the Model Cell into Which Each CSO Discharges (continued)

Layer	Percent of Time Below Minimum	Maximum Duration Below Minimum (days)	Percent of Time Below Minimum	Maximum Duration Below Minimum (days)	Percent of Time Below Minimum	Maximum Duration Below Minimum (days)
	Hanford/Rainier		**Harbor**		**King**	
10 (Top)	30.46%	12	0%	0	0%	0
9	2.17%	2	0%	0	0%	0
8	0%	0	0%	0	0%	0
7	0%	0	0%	0	0%	0
6	0%	0	0%	0	0%	0
5	0%	0	0%	0	0%	0
4	0%	0	0%	0	0%	0
3	0%	0	0%	0	0%	0
2	0%	0	0%	0	0%	0
1 (Bottom)	0%	0	0%	0	0%	0
	Lander		**Norfolk**		**South Magnolia**	
10 (Top)	0.79%	0	83.83%	241	0%	0
9	0%	0	78.12%	152	0%	0
8	0%	0	73.41%	44	0%	0
7	0%	0	66.74%	39	0%	0
6	0%	0	56.94%	36	0%	0
5	0%	0	50.46%	19	0%	0
4	0%	0	46.46%	19	0%	0
3	0%	0	44.41%	18	0%	0
2	0%	0	42.85%	18	0%	0
1 (Bottom)	0%	0	41.19%	18	0%	0

Table 5-31. Percent Time and Maximum Duration Below the Minimum Salinity Criterion (Five ppt) at the Model Cell into Which Each CSO Discharges (continued)

Layers	Percent of Time Below Minimum	Maximum Duration Below Minimum (days)	Percent of Time Below Minimum	Maximum Duration Below Minimum (days)
	South Michigan		West Michigan	
10 (Top)	52.36%	16	54.94%	16
9	8.94%	5	9.3%	5
8	0.18%	0	0.36%	0
7	0%	0	0%	0
6	0%	0	0%	0
5	0%	0	0%	0
4	0%	0	0%	0
3	0%	0	0%	0
2	0%	0	0%	0
1 (Bottom)	0%	0	0%	0

A comparison of the cell into which the CSO empties to those cells in the model adjacent to, above, or below, generally revealed little difference in salinity structure. Two possible exceptions are the Brandon Street and Hanford/Rainier CSO sites. The minimum criterion was exceeded at Brandon Street (cell number 111) in the top two layers 38 and 2 percent of the time, while in cell number 110 (cell above) they were exceeded in the top three layers for 46, 8, and 0.3 percent of the time. In cell number 112 (cell below), salinity fell below the minimum in each of the top four layers, and for 32, 11, 1, and .02 percent of the time. In cell number 113 (cell further below), salinity fell below the minimum again in the four surface cells, for 34, 14, 2, and 0.2 percent of the time.

The Hanford/Rainier CSO discharges into cell number 129 where we observed that only the two surface layers fell below the minimum criterion, and then for 30 and 2 percent of the time, respectively. Cell number 130 might show an influence of the nearby discharge as the top three layers fell below the minimum criterion. They were affected for 28, 20, 5, 1, and 0.03 percent of the time. In cell number 131, only the top three layers fell below the minimum salinity and then for only 29, 11, and 1 percent of the time, respectively. Cell number 128 fell below the criterion in the top three layers of the water column at 40, six, and 0.7 percent of the time, respectively. The Hanford/Rainier CSO discharges through the Duwamish/Diagonal Way outfall that also receives a significant discharge of separated storm water.

In summary, the model simulation suggests that there is a minimal influence of CSOs on the salinity in surface waters of the study area, which occurs only adjacent to the Brandon Street and Hanford/Rainier CSOs. There could be, then, a slight adverse effect to some aquatic life inhabiting these areas. Those species most vulnerable are immobile and can only tolerate a narrow range of salinities. These would be marine species near the most upriver extent of the range.

5.3.2 pH

The State of Washington (WAC 173-200) has established minimum and maximum criteria for both fresh and marine waters. These values are required to be within the range 6.5 to 8.5 for freshwater and 7.5 to 8.5 for marine water, with human-caused variation to be <0.5 pH units. The pH database we reviewed consisted of measurements at different depths of the Duwamish River and Elliott Bay from various CSO locations (Table 5-32). The pH database also contained measurements from Elliott Bay at two depths from two CSO locations. Our evaluation focused on exceedances of the marine criterion because most CSOs discharge into the marine environment.

The number of times where and when pH exceeded (fell below) the marine criterion generally decreased down river. Most exceedances were also associated with surface samples. Fewest exceedances were encountered in Elliott Bay. Generally, locations where pH exceeded the marine criteria were also the locations where salinity fell below five ppt, which likely only reflected the lower pH of freshwater entering the river during the wet season. For example, at the Norfolk CSO, 55 of 62 samples collected over the

Table 5-32. Number of Observations and Number of Exceedances of Freshwater and Marine pH Criteria at Selected CSO Locations

CSO Location	River Location[a]	Depth[b]	Number of Observations	Number of Freshwater Standards Exceedances	Number of Marine Standards Exceedances
Brandon	Center	Top	32	0	12
Brandon	Center	Bottom	32	0	0
Brandon	East	Top	32	0	20
Brandon	East	Bottom	32	0	1
Brandon	West	Top	32	0	14
Brandon	West	Bottom	32	0	0
Chelan	Center	Top	32	0	13
Chelan	Center	Bottom	32	0	0
Chelan	East	Top	31	0	15
Chelan	East	Bottom	30	0	1
Chelan	West	Top	32	0	9
Chelan	West	Bottom	32	0	0
Connecticut	Center	Top	32	0	0
Connecticut	Center	Bottom	32	0	0
Connecticut	East	Top	32	0	0
Connecticut	East	Bottom	32	0	0
Connecticut	West	Top	32	0	0
Connecticut	West	Bottom	32	0	0
Duwamish Head	-[c]	Top	29	0	0
Duwamish Head	-[c]	Bottom	29	0	0
Denny Way	Cap	Top	29	0	0
Denny Way	Cap	Bottom	29	0	1
Denny Way	Outfall	Top	28	0	3

Table 5-32. Number of Observations and Number of Exceedances of Freshwater and Marine pH Criteria at Selected CSO Locations (continued)

CSO Location	River Location[a]	Depth[b]	Number of Observations	Number of Freshwater Standards Exceedances	Number of Marine Standards Exceedances
Hanford	Center	Top	32	0	2
Hanford	Center	Bottom	32	0	0
Hanford	East	Top	32	0	1
Hanford	East	Bottom	32	0	1
Hanford	West	Top	32	0	0
Hanford	West	Bottom	32	0	0
Norfolk	East	Top	31	1	27
Norfolk	West	Top	31	0	28
South & West Michigan	Center	Top	32	0	11
South & West Michigan	Center	Bottom	32	0	0
South & West Michigan	East	Top	32	0	18
South & West Michigan	East	Bottom	32	0	1
South & West Michigan	West	Top	32	0	15
South & West Michigan	West	Bottom	32	0	4
Tukwila Gauging Station	-[c]	Top	32	1	29

a Indicates where in the river/bay the sample was collected relevant to the CSO discharge location.

b Top measurements were made 1 meter below the surface and bottom measurements were made one meter above the sediment.

c Samples were collected directly adjacent to CSO discharge location.

seven month period exceeded (fell below) the marine pH criterion. At the South/West Michigan and Brandon Street CSOs, 49 and 46 out of 192 samples over the same time frame, respectively, were in exceedance. At the Hanford CSO, only 4 out of 192 samples exceeded (fell below) the marine criterion.

At each location, evaluation of pH values from both sides of the river, and from the centerline when available, revealed few differences, which suggested that CSOs have little or no effect on water column. One might expect that the side of the river receiving the CSO discharge would show the greater number of exceedances. For example, at the Norfolk CSO, 27 samples from the east side and 28 samples from the west side of the river, of a total of 55 samples, exceeded the marine criterion. The CSO enters the river on the east side. Of the 46 samples from the Brandon Street CSO that exceeded the marine criterion, 20 samples were from the east side, 12 samples were from the centerline, and 14 samples were from the west side. The discharge occurs on the east side of the river. Of the 49 samples from West Michigan that exceeded the marine criterion, 19 were from the east side of the river, 11 from the centerline, and 19 from the wets side. The discharge is located on the west side of the river.

On no occasion did samples collected at the Connecticut Street CSO exceed the marine pH criterion. Similarly, pH did not fall below the criteria in any samples collected at Duwamish Head. On the Denny Way Cap, pH fell below the criterion in only one sample over the seven-month collection period. At the Denny Way Outfall, there were three samples (one each in November, December, and January) when the pH fell below the marine criterion.

These data would suggest that there is little or no influence of CSOs on the pH of surface waters in either the Duwamish River or Elliott Bay, and hence there can be little or no effect of pH on the health of aquatic life inhabiting the study area. While pH exceeds (falls below) the State of Washington marine criterion in a number of areas within the study area, including areas influenced by CSOs and areas removed from CSO influence, the pH shifts appear to be associated with shifts in salinity associated with general runoff.

5.3.3 Temperature

Temperature is a critical measure of the suitability of the environment for the presence of aquatic life. Each aquatic species seeks and maintains itself within a preferred range of temperatures. Departures from this range will affect diet, activity and general health. Temperatures outside the preferred range of the community of aquatic organisms will also change community structure (numbers of species and numbers of individuals present).

State of Washington temperature criteria (WAC 173-200) were the standards against which we compared temperature data collected from the Duwamish River and Elliott Bay over a seven-month period in 1996-1997. Water samples were collected at three points (east, west, and centerline) and at two depths at each of five CSO locations in the Duwamish River. Water samples were also collected at two depths at three CSO locations in Elliott Bay.

Review of all data (on the order of 192 measurements from each location) indicated that there were no temperatures at any location, depth, or sampling interval that exceeded the State of Washington temperature criteria, either for freshwater or marine water.

5.3.4 Dissolved Oxygen

Reductions of DO may result from the biological oxygen demand of particulate matter in CSO discharges (Welch and Lindell 1992), although the potential harmful effects of CSO discharges on receiving waters can be mediated by the increased flows which occur in conjunction with CSO discharge events (SPCC 1981, Welch and Lindell 1992). Reductions in DO can affect the behavior, metabolism, growth, reproduction, and survival of aquatic organisms. Typically, early life stages of aquatic organisms, except embryos, are sensitive to DO reductions, with juvenile life stages being the most sensitive (U.S. EPA 1986b). Those waters near or in the sediments generally have the lowest DO concentrations, owing to deposition and degradation of organic matter.

The State of Washington DO criteria for Class B (good) marine waters (WAC 173-200) was the standard against which we compared DO data collected in the Duwamish River and Elliott Bay over seven months in 1996-1997. Water samples for DO analysis were collected from three points (east, west, and centerline) and at two depths at five CSO locations at in the Duwamish River. Water samples were also collected from three CSO locations in Elliott Bay.

Review of all data (approximately 192 measurements from each location) indicated that at no location, depth, or time interval, did the DO fall below the State of Washington marine or freshwater criteria.

5.3.5 TSS

Acute HQs from TSS were less than 1.0 for the cells into which the Norfolk, 8th Avenue, South and West Michigan, Brandon Street, Hanford/Rainier, Harbor, Chelan and Hanford, Lander, and Denny Way CSOs discharge (Table 5-33). The exceptions were the Norfolk and 8th Avenue CSOs and these were only associated with the bottom layer in each case.

Chronic HQs greater than 1.0 resulted at all CSO locations, both in the Duwamish River and in Elliott Bay. Most chronic HQs, however, did not exceed 1.0; a relatively few ranged up to 1.37. Exceedances generally occurred in every month except the summer months (June, July, and August) although greatest exceedances occurred in winter months (January, February, March) and were associated with bottom layers.

Generally, the areas (cells of the model) of the river adjacent to, above, and below each CSO, behaved similarly (the acute and chronic HQs were of the same magnitude), suggesting minimal influence from CSOs, or that the influence of CSOs was not measurable at this level of model resolution. When the model was rerun without TSS loading from the CSOs, chronic HQs were reduced over a range of 2 to 5 percent. Greatest reductions occurred at CSO locations furthest upriver. For example at the Norfolk CSO, the reduction was 12 percent and was associated with the bottom layer. At the 8th Avenue CSO, the reduction was 6 percent, and again was associated in the bottom layer. In the two cases where acute HQs greater than 1.0 occurred (Norfolk and 8th

Table 5-33. TSS Hazard Quotients in Cells Receiving CSO Discharges

CSO Discharging to Cell	Exposure Conditions		Layer	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug
8th Avenue	Acute	Baseline	1	0.72	0.76	0.91	0.80	1.05	0.85	0.96	0.73	0.76	0.64	0.48	0.59
8th Avenue	Acute	Without CSO	1	0.85	0.63	0.85	0.76	1.03	0.85	0.97	0.74	0.78	0.64	0.51	0.62
8th Avenue	Acute	Baseline	10	0.74	0.66	0.51	0.37	N/AP	0.41	0.28	0.30	0.28	0.32	0.35	0.32
8th Avenue	Acute	Without CSO	10	0.76	0.64	0.37	0.37	N/AP	0.32	0.27	0.29	0.28	0.32	0.36	0.35
8th Avenue	Chronic	Baseline	1	1.26	1.18	1.30	1.28	1.40	1.30	1.33	1.21	1.18	1.08	1.01	1.09
8th Avenue	Chronic	Without CSO	1	1.33	1.11	1.27	1.24	1.40	1.30	1.33	1.20	1.18	1.07	1.00	1.10
8th Avenue	Chronic	Baseline	10	1.30	1.13	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	0.90
8th Avenue	Chronic	Without CSO	10	1.29	1.03	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	0.89
Brandon	Acute	Baseline	1	0.56	0.49	0.54	0.47	0.68	0.69	0.77	0.45	0.43	0.40	0.42	0.49
Brandon	Acute	Without CSO	1	0.65	0.49	0.51	0.52	0.68	0.54	0.74	0.44	0.43	0.42	0.41	0.50
Brandon	Acute	Baseline	10	0.74	0.62	0.57	0.41	0.39	0.43	0.35	0.37	0.35	0.36	0.35	0.35
Brandon	Acute	Without CSO	10	0.75	0.59	0.46	0.42	0.36	0.34	0.32	0.37	0.37	0.32	0.32	0.33
Brandon	Chronic	Baseline	1	1.15	1.06	1.09	1.09	1.22	1.17	1.29	1.05	1.02	1.01	0.99	1.00
Brandon	Chronic	Without CSO	1	1.20	1.06	1.07	1.07	1.22	1.12	1.24	1.04	1.02	1.02	0.99	1.00
Brandon	Chronic	Baseline	10	1.30	1.14	0.90	0.90	N/AP	N/AP	N/AP	N/AP	N/AP	0.89	0.90	0.93
Brandon	Chronic	Without CSO	10	1.31	1.09	0.89	0.89	N/AP	N/AP	N/AP	N/AP	N/AP	0.89	0.90	0.91
Chelan	Acute	Baseline	1	0.53	0.43	0.51	0.45	0.62	0.88	0.63	0.44	0.39	0.40	0.40	0.38
Chelan	Acute	Without CSO	1	0.59	0.45	0.49	0.52	0.62	0.54	0.63	0.43	0.39	0.41	0.40	0.38
Chelan	Acute	Baseline	10	0.73	0.59	0.54	0.39	0.49	0.45	0.38	0.37	0.38	0.33	0.32	0.33

Table 5-33. TSS Hazard Quotients in Cells Receiving CSO Discharges (continued)

CSO Discharging to Cell	Exposure Conditions		Layer	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug
Chelan	Acute	Without CSO	10	0.77	0.56	0.46	0.39	0.48	0.47	0.38	0.39	0.37	0.33	0.33	0.33
Chelan	Chronic	Baseline	1	1.09	1.01	1.03	1.02	1.11	1.13	1.16	1.03	0.99	1.01	0.99	0.98
Chelan	Chronic	Without CSO	1	1.08	1.01	1.02	1.03	1.10	1.06	1.12	1.03	0.99	1.01	0.99	0.98
Chelan	Chronic	Baseline	10	1.30	1.12	1.00	0.94	0.92	1.01	0.91	0.92	0.89	0.91	0.91	0.93
Chelan	Chronic	Without CSO	10	1.32	1.08	0.94	0.94	0.92	0.92	0.91	0.92	0.90	0.91	0.92	0.93
Connecticut	Acute	Baseline	1	0.44	0.40	0.41	0.40	0.40	0.67	0.40	0.42	0.40	0.41	0.39	0.38
Connecticut	Acute	Without CSO	1	0.41	0.40	0.40	0.41	0.41	0.40	0.40	0.42	0.40	0.41	0.39	0.37
Connecticut	Acute	Baseline	10	0.66	0.49	0.47	0.38	0.53	0.44	0.59	0.38	0.36	0.36	0.35	0.35
Connecticut	Acute	Without CSO	10	0.71	0.46	0.44	0.39	0.52	0.44	0.55	0.37	0.35	0.36	0.35	0.35
Connecticut	Chronic	Baseline	1	1.05	1.00	1.01	1.01	1.01	1.04	1.01	1.03	0.99	1.02	0.99	0.98
Connecticut	Chronic	Without CSO	1	1.02	1.00	1.01	1.01	1.01	1.01	1.02	1.03	0.99	1.02	0.99	0.98
Connecticut	Chronic	Baseline	10	1.23	1.06	0.99	1.01	1.08	1.03	1.14	0.97	0.96	0.95	0.95	0.95
Connecticut	Chronic	Without CSO	10	1.27	1.04	0.98	1.00	1.07	1.02	1.09	0.97	0.95	0.95	0.95	0.95
Denny Way	Acute	Baseline	1	0.44	0.39	0.41	0.41	0.41	0.42	0.40	0.42	0.40	0.41	0.39	0.37
Denny Way	Acute	Without CSO	1	0.41	0.40	0.41	0.41	0.42	0.42	0.40	0.42	0.39	0.41	0.38	0.37
Denny Way	Acute	Baseline	10	0.53	0.41	0.39	0.39	0.41	0.40	0.44	0.39	0.37	0.38	0.38	0.36
Denny Way	Acute	Without CSO	10	0.51	0.40	0.39	0.39	0.42	0.40	0.42	0.39	0.37	0.38	0.37	0.36
Denny Way	Chronic	Baseline	1	1.04	1.00	1.02	1.01	1.01	1.02	1.01	1.03	1.00	1.01	0.99	0.98
Denny Way	Chronic	Without CSO	1	1.01	1.00	1.02	1.01	1.01	1.02	1.01	1.03	0.99	1.01	0.98	0.98
Denny Way	Chronic	Baseline	10	1.13	1.01	0.99	0.99	1.01	1.01	1.01	1.00	0.97	0.99	0.97	0.97

Table 5-33. TSS Hazard Quotients in Cells Receiving CSO Discharges (continued)

CSO Discharging to Cell	Exposure Conditions		Layer	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug
Denny Way	Chronic	Without CSO	10	1.10	1.01	1.00	0.99	1.01	1.00	1.00	1.00	0.98	0.99	0.97	0.97
Hanford	Acute	Baseline	1	0.47	0.40	0.44	0.40	0.57	0.60	0.61	0.42	0.39	0.41	0.38	0.38
Hanford	Acute	Without CSO	1	0.55	0.40	0.42	0.42	0.56	0.48	0.58	0.42	0.39	0.41	0.38	0.38
Hanford	Acute	Baseline	10	0.72	0.58	0.54	0.40	0.62	0.51	0.68	0.38	0.36	0.34	0.33	0.34
Hanford	Acute	Without CSO	10	0.75	0.55	0.49	0.40	0.62	0.52	0.61	0.38	0.36	0.33	0.32	0.33
Hanford	Chronic	Baseline	1	1.06	1.01	1.02	1.01	1.10	1.10	1.15	1.03	0.99	1.01	0.99	0.98
Hanford	Chronic	Without CSO	1	1.09	1.00	1.01	1.01	1.10	1.05	1.12	1.03	1.00	1.02	0.99	0.99
Hanford	Chronic	Baseline	10	1.29	1.12	1.02	1.04	1.10	1.05	1.03	0.97	0.96	0.92	0.91	0.93
Hanford	Chronic	Without CSO	10	1.33	1.08	1.00	1.04	1.09	1.05	1.02	0.96	0.95	0.92	0.92	0.93
Hanford/Rainier	Acute	Baseline	1	0.55	0.49	0.56	0.49	0.68	0.83	0.72	0.46	0.43	0.41	0.43	0.43
Hanford/Rainier	Acute	Without CSO	1	0.63	0.53	0.53	0.55	0.67	0.56	0.69	0.45	0.43	0.44	0.43	0.42
Hanford/Rainier	Acute	Baseline	10	0.74	0.61	0.56	0.41	0.45	0.43	0.35	0.36	0.36	0.36	0.35	0.35
Hanford/Rainier	Acute	Without CSO	10	0.76	0.58	0.44	0.41	0.44	0.37	0.32	0.36	0.36	0.32	0.32	0.33
Hanford/Rainier	Chronic	Baseline	1	1.13	1.06	1.08	1.07	1.20	1.16	1.26	1.04	1.01	1.01	1.00	0.99
Hanford/Rainier	Chronic	Without CSO	1	1.18	1.07	1.06	1.07	1.20	1.11	1.22	1.03	1.01	1.02	1.00	0.99
Hanford/Rainier	Chronic	Baseline	10	1.30	1.14	0.90	0.90	N/AP	0.94	N/AP	N/AP	0.87	0.90	0.90	0.93
Hanford/Rainier	Chronic	Without CSO	10	1.31	1.09	0.90	0.90	N/AP	0.89	N/AP	N/AP	0.88	0.89	0.90	0.91
Harbor	Acute	Baseline	1	0.46	0.41	0.43	0.40	0.53	0.59	0.55	0.42	0.39	0.40	0.39	0.37
Harbor	Acute	Without CSO	1	0.43	0.40	0.41	0.41	0.53	0.46	0.54	0.42	0.39	0.40	0.39	0.36
Harbor	Acute	Baseline	10	0.71	0.54	0.50	0.38	0.61	0.49	0.68	0.38	0.37	0.34	0.33	0.33

Table 5-33. TSS Hazard Quotients in Cells Receiving CSO Discharges (continued)

CSO Discharging to Cell	Exposure Conditions		Layer	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug
Harbor	Acute	Without CSO	10	0.73	0.51	0.44	0.38	0.61	0.48	0.63	0.38	0.36	0.34	0.34	0.34
Harbor	Chronic	Baseline	1	1.07	1.01	1.02	1.01	1.07	1.06	1.11	1.03	0.99	1.01	0.99	0.98
Harbor	Chronic	Without CSO	1	1.02	1.01	1.02	1.01	1.06	1.04	1.08	1.03	0.99	1.01	0.99	0.98
Harbor	Chronic	Baseline	10	1.28	1.10	1.01	1.03	1.12	1.04	1.20	0.97	0.96	0.93	0.93	0.94
Harbor	Chronic	Without CSO	10	1.31	1.07	1.00	1.03	1.12	1.04	1.15	0.97	0.96	0.93	0.93	0.94
King	Acute	Baseline	1	0.43	0.39	0.40	0.41	0.41	0.41	0.39	0.42	0.40	0.41	0.38	0.37
King	Acute	Without CSO	1	0.40	0.39	0.40	0.41	0.41	0.41	0.40	0.42	0.40	0.41	0.38	0.37
King	Acute	Baseline	10	0.59	0.43	0.39	0.37	0.44	0.41	0.49	0.38	0.36	0.37	0.36	0.36
King	Acute	Without CSO	10	0.60	0.41	0.39	0.37	0.44	0.40	0.46	0.38	0.35	0.37	0.35	0.36
King	Chronic	Baseline	1	1.04	1.00	1.01	1.01	1.02	1.02	0.99	1.03	0.99	1.01	0.99	0.98
King	Chronic	Without CSO	1	1.01	1.00	1.01	1.01	1.02	1.02	1.01	1.03	1.00	1.02	0.99	0.99
King	Chronic	Baseline	10	1.18	1.02	0.99	0.98	1.02	1.01	1.06	0.99	0.97	0.97	0.96	0.96
King	Chronic	Without CSO	10	1.20	1.01	0.98	0.98	1.02	1.00	1.03	0.99	0.96	0.97	0.96	0.96
Lander	Acute	Baseline	1	0.45	0.40	0.43	0.40	0.54	0.69	0.60	0.41	0.39	0.40	0.38	0.37
Lander	Acute	Without CSO	1	0.45	0.40	0.42	0.40	0.53	0.45	0.56	0.41	0.39	0.40	0.38	0.38
Lander	Acute	Baseline	10	0.71	0.58	0.54	0.39	0.60	0.49	0.66	0.37	0.36	0.34	0.33	0.33
Lander	Acute	Without CSO	10	0.75	0.55	0.48	0.39	0.60	0.49	0.60	0.37	0.36	0.33	0.32	0.33
Lander	Chronic	Baseline	1	1.05	1.01	1.01	1.01	1.08	1.09	1.13	1.02	0.99	1.01	0.99	0.98
Lander	Chronic	Without CSO	1	1.04	1.01	1.01	1.01	1.09	1.04	1.10	1.02	0.99	1.01	0.99	0.99
Lander	Chronic	Baseline	10	1.29	1.12	1.00	1.03	1.12	1.04	1.15	0.96	0.95	0.92	0.91	0.93

Table 5-33. TSS Hazard Quotients in Cells Receiving CSO Discharges (continued)

CSO Discharging to Cell	Exposure Conditions		Layer	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug
Lander	Chronic	Without CSO	10	1.32	1.08	0.99	1.03	1.11	1.04	1.11	0.95	0.95	0.92	0.92	0.93
Norfolk	Acute	Baseline	1	0.86	1.00	0.91	0.88	0.76	0.72	0.82	0.78	0.71	0.86	0.94	0.76
Norfolk	Acute	Without CSO	1	0.92	0.92	0.90	0.91	0.78	0.76	0.84	0.65	0.74	0.78	0.79	0.73
Norfolk	Acute	Baseline	10	0.74	0.70	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	0.26	0.29	0.52
Norfolk	Acute	Without CSO	10	0.84	0.62	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	0.23	0.27	0.53
Norfolk	Chronic	Baseline	1	1.34	1.29	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	1.04	1.10	1.05
Norfolk	Chronic	Without CSO	1	1.34	1.28	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	1.05	1.06	1.03
Norfolk	Chronic	Baseline	10	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP
Norfolk	Chronic	Without CSO	10	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP
South Magnolia	Acute	Baseline	1	0.43	0.43	0.44	0.44	0.44	0.45	0.42	0.43	0.43	0.44	0.44	0.42
South Magnolia	Acute	Without CSO	1	0.43	0.43	0.43	0.44	0.43	0.43	0.42	0.43	0.43	0.44	0.44	0.42
South Magnolia	Acute	Baseline	10	0.52	0.41	0.39	0.39	0.40	0.40	0.41	0.39	0.37	0.38	0.37	0.36
South Magnolia	Acute	Without CSO	10	0.50	0.40	0.39	0.39	0.41	0.40	0.40	0.39	0.37	0.38	0.37	0.36
South Magnolia	Chronic	Baseline	1	1.04	1.03	1.03	1.03	1.03	1.03	1.03	1.03	1.02	1.03	1.03	1.02
South Magnolia	Chronic	Without CSO	1	1.01	1.03	1.03	1.03	1.03	1.03	1.03	1.03	1.02	1.03	1.02	1.02
South Magnolia	Chronic	Baseline	10	1.14	1.01	1.00	0.99	1.00	1.01	1.00	1.00	0.97	0.99	0.97	0.97
South Magnolia	Chronic	Without CSO	10	1.09	1.01	1.00	0.99	1.00	1.00	0.99	1.00	0.98	0.99	0.97	0.97
South Michigan	Acute	Baseline	1	0.64	0.56	0.64	0.49	0.76	0.91	0.77	0.49	0.47	0.41	0.38	0.43
South Michigan	Acute	Without CSO	1	0.72	0.48	0.58	0.48	0.75	0.62	0.73	0.48	0.47	0.41	0.37	0.43
South Michigan	Acute	Baseline	10	0.74	0.63	0.59	0.42	0.42	0.43	0.36	0.33	0.36	0.39	0.35	0.36

Table 5-33. TSS Hazard Quotients in Cells Receiving CSO Discharges (continued)

CSO Discharging to Cell	Exposure Conditions		Layer	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug
South Michigan	Acute	Without CSO	10	0.76	0.60	0.49	0.42	0.35	0.33	0.32	0.32	0.38	0.40	0.35	0.33
South Michigan	Chronic	Baseline	1	1.21	1.09	1.14	1.13	1.25	1.21	1.32	1.08	1.04	1.01	0.99	1.00
South Michigan	Chronic	Without CSO	1	1.26	1.06	1.11	1.10	1.25	1.15	1.26	1.07	1.04	1.00	0.98	1.00
South Michigan	Chronic	Baseline	10	1.30	1.15	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	0.90	0.91	0.93
South Michigan	Chronic	Without CSO	10	1.30	1.10	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	0.88	0.90	0.92
West Michigan	Acute	Baseline	1	0.63	0.55	0.65	0.49	0.75	0.89	0.80	0.52	0.49	0.41	0.38	0.44
West Michigan	Acute	Without CSO	1	0.75	0.47	0.60	0.48	0.75	0.59	0.76	0.51	0.50	0.41	0.38	0.44
West Michigan	Acute	Baseline	10	0.74	0.64	0.63	0.40	0.41	0.42	0.34	0.31	0.33	0.33	0.31	0.33
West Michigan	Acute	Without CSO	10	0.76	0.61	0.46	0.40	0.34	0.32	0.29	0.31	0.41	0.38	0.33	0.32
West Michigan	Chronic	Baseline	1	1.20	1.08	1.14	1.13	1.26	1.20	1.32	1.08	1.04	1.01	0.99	1.00
West Michigan	Chronic	Without CSO	1	1.27	1.06	1.11	1.10	1.25	1.16	1.28	1.07	1.04	1.00	0.98	1.00
West Michigan	Chronic	Baseline	10	1.31	1.14	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	0.88	0.92
West Michigan	Chronic	Without CSO	10	1.31	1.10	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	N/AP	0.89	0.91

N/AP – Not applicable, salinity less than 5 ppt

Avenue), removal of the CSO source of TSS resulted in acute risk being reduced 3 to 4 percent.

5.3.6 Sedimentation Rate

In general, risks to aquatic life from sedimentation were low in the study area except for the turning basin where high sedimentation levels were observed (maximum values in Table 5-34. Only one area (cell in the model) of the river receiving a CSO discharge indicated a HQ greater than 1.00. This was cell number 76 that receives the 8th Avenue CSO. The exceedance was small (1.66) and occurred in February. Areas (cells) on either side or above and below also showed HQs ranging from 1.23 to 2.30, again in the month of February. These model cells are numbers 75, 77, 78, 79, and 80. Cell number 76 is in relatively shallow water so it is possible that the CSO at this location could result in an increase in sedimentation rate. It is more likely; however, that the increased sedimentation rate in this cell and its adjacent cells reflect the greater sedimentation rates routinely encountered in the upper river, particularly during the wet season. All other cells showing increased sedimentation (HQs >1) in this data set, and there are 30, occurred in the upper river.

When the model was rerun without the input of CSOs, generally substantial reductions occurred over winter months in about 125 out of 153 cells stretching from the Norfolk CSO (above the head of navigation) down to the Hanford Avenue CSO located in the East Waterway. However, in the cells where reductions were observed between baseline and without CSO conditions, baseline HQs were less than 0.01, indicating minimal risk from sedimentation under baseline conditions. This implies that the decrease in sedimentation HQs under without CSO conditions do not have any bearing on the risk conditions.

5.3.7 Scouring

Scouring was assessed in King County's model by following changes in sediment bed height, in this case, decreases in sediment bed height (Figure 5-7). Risk to benthos was based on severity of scouring in the sediment column and whether or not the depth at which a particular benthic species was normally found was compromised by a scouring event. For example, most amphipods are surface detrital feeders that are found no deeper in the sediments than 1 cm. Their effects level or threshold, then, is 1 cm. The loss of the top centimeter of the sediment column would mean the displacement or loss of all amphipods from that habitat. We have established similar effects levels for 26 species of the benthos inhabiting the study area and most were found to inhabit the upper 1.0 cm of the sediment column. Some species exploit the sediment column down to a depth of 5 cm or more. We evaluated the model simulation for changes in bed height that exceeded these effects-levels.

Table 5-34. Summary of Monthly Sedimentation Hazard Quotients Across All Model Cells

Summary Statistic	**Sep**	**Oct**	**Nov**	**Dec**	**Jan**	**Feb**
Average	0.01	0.06	0.11	0.06	0.23	0.11
Standard Deviation	0.02	0.19	0.32	0.19	0.82	0.52
Maximum	0.14	1.5	3.1	1.6	11.6	8.2
Median	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Minimum	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Summary Statistic	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**
Average	0.20	0.10	0.09	0.02	0.01	0.01
Standard Deviation	1.0	0.52	0.39	0.10	0.04	0.02
Maximum	17.6	9.9	6.0	1.0	0.36	0.14
Median	0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Minimum	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01

Results simulating a period of seven months over the period October 1996 to June 1997 indicated few changes in bed height suggestive of scouring. Most CSOs (Chelan Street, Hanford Avenue, Lander Street, Harbor, Connecticut Street, King Street, Denny Way, and South Magnolia) were characterized by a change in sediment bed height of less than 1.0 cm over the seven month study period. A few CSOs (8th Avenue, Brandon Street, West and South Michigan Streets, and Hanford/Rainier) were characterized by a step-wise build-up of sediment bed height over the study period.

Only one site (the Norfolk CSO) was characterized by a sediment bed height that both increased and decreased over the study period. Actually the model simulation indicated that bed height increased and decreased by more than 0.5 cm 16 times over the study period. The increases and decreases exceeded 1.0 cm/month on several occasions and were associated with large storms that occurred in January and March 1997. Erosional events of at least 1.0 cm exceeded the threshold effects values for 14 of 26 species for which effects thresholds were generated, and clearly impacts were predicted at this location. While some disturbance of the benthos can be attributed to the CSO (channeling of the bottom occurs immediately below the outfall), perhaps the greater impact on the benthos can be attributed to the general bathymetry of the site. The CSO is located on a reach of the river that is narrower than the navigational channel into which it flows. Its slope is also relatively steep and its depth very shallow, all of which suggest that the site is located in a high-energy environment that may not be favorable to colonization by a diverse benthic community.

Figure 5-7. Change in Sediment Bed Height in Cells Adjacent to CSOs

Pg. 2 of Figure 5-7

5.3.8 Displacement

Risk of displacement was determined by comparing the centerline plume velocity resulting from a CSO discharge with the displacement threshold for juvenile salmon (1.0 m/s). The centerline plume velocity was predicted by near-field computer model that described plume size, direction, and velocity. This comparison revealed that displacement was a potential risk at all CSO locations for which velocity predictions were developed (Table 5-35). The number of days for which displacement occurred during the model year ranged from 1 for 8th Avenue to 32 days for Denny Way.

Table 5-35. The Number of Days on which the Plume Velocity Resulting from a CSO Discharge Exceeded 1.0 m/s at CSO Discharge Locations

CSO Location	Number of Days on Which Plume Velocity Exceeded 1.0 m/s
8th Avenue	1
Brandon	20
Chelan	20
Connecticut	4
Denny Way	32
Hanford	7
King	7
Lander	9
Norfolk	2
W. Michigan	6
Harbor	2

5.4 Uncertainty in the Risk Characterization Results

Uncertainties in the risk characterization results include those associated with the exposure and effects characterizations, along with those associated with interpretation of the risk results. We believe that while some of these uncertainties are worthy of discussion, none of the uncertainties are of sufficient magnitude, alone or in combination, to alter our results and conclusions. This belief is supported by the observation that the vast majority of the observed and predicted water column EECs fell below water quality criteria, below the estimated fifth percentile of the distribution of TRVs in the aquatic community, and below the TRVs for salmonids. These observations indicate that water column risks are low despite uncertainties in specific EECs and TRVs.

Our estimates of risks to aquatic life from physical stressors are uncertain due to uncertainty about physical stressor exposure levels and uncertainty about physical stressor effect thresholds, as described in the exposure and effects characterization uncertainty sections. The principle uncertainty of the risk characterization of physical stressors measured directly in the field (salinity, DO, temperature, and pH) will be associated with operation of the field instruments and the data used to set the State criteria. Field measurements can over or under estimate the actual value, while the State standards are constructed to conservatively protect aquatic like (i.e., over estimating risks). We combined field measurements with State criteria using HQs. Dividing the over or under estimation of field measurements by the over estimation of State criteria is likely to under estimate risks from these physical stressors.

Exposure levels of TSS, scouring, and sedimentation rates were all determined using the EFDC model. Consequently, uncertainties in the model will be reflected in predictions of the concentrations of physical stressors. Effects thresholds for TSS and sedimentation rates are likely over estimates of risk because non-estuary adapted species were used in the development of the TRVs for these stressors. Thus, the uncertainty of characterizing risks for TSS and sedimentation is likely to err on the side of over estimated risks. Uncertainty of scouring risks is linked with model uncertainty and any inefficiency associated with benthic collection and identification.

Our estimates of risks to benthic life are uncertain due to uncertainty about sediment EECs and uncertainty about sediment TRVs, as described in the exposure and effects characterization uncertainty sections. The sediment EECs may be either over- or under-estimated on a very fine scale (smaller than the model grid size) because we used average concentrations within each cell where sediment data were available. On the scale represented by the model cells, the sediment EECs are likely to tend to be over estimated because we used a linear interpolation scheme to estimate concentrations in cells with no sediment data available from cells with available data, which were largely collected to characterize hot spot contamination. The TRVs are likely to tend to be under estimated because they are based on AETs, which establish correlation between the presence of a stressor and an effect on the benthic community, but do not establish a cause and effect relationship between the stressor and the effect. We used the quotient method to estimate sediment risks, where the HQ = EEC/TRV. If EECs tend to be over estimated, and TRVs tend to be under estimated, sediment risks will tend to be over estimated. Therefore, while the sediment risk estimates presented in this report are uncertain, it is fairly certain that they do not, in general, under estimate risk, although there may be some instances where risks are under predicted in small-scale, localized areas of sediment contamination.

6. TOXICITY EVALUATION OF BRANDON STREET CSO EFFLUENT TO *CERIODAPHNIA DUBIA* AND *PIMEPHALES PROMELAS*

6.1 Introduction

This section summarizes the procedures and results of biological testing conducted on a Brandon Street CSO effluent sample collected by King County on 9 October 1997. This study was undertaken to evaluate risks to aquatic life present in the surface water receiving a CSO discharge. Effluent discharging from the Brandon Street CSO was collected during the course of the WQA and tested for chronic toxicity[32] to two common freshwater test species – *Ceriodaphnia dubia* (an invertebrate called a water flea) and a vertebrate fish species, *Pimephales promelas,* the fathead minnow. All testing was conducted by Parametrix's Environmental Toxicology Laboratory in Kirkland, Washington. Testing consisted of two chronic definitive bioassays using *Ceriodaphnia dubia* and *Pimephales promelas* as the test species. The median lethal concentration (LC_{50}), the lowest observed effect concentrations (LOEC), and the no observed effect concentrations (NOEC) are reported for each bioassay.

6.2 Test Methods and Conditions

6.2.1 Sample Collection

A composite sample from the Brandon Street CSO was collected by an ISCO company autosampler from the wet-well at the CSO outfall structure, located just prior to point of discharge to the river. A sample aliquot was collected every 10 minutes and deposited into a carboy over the course of the entire overflow event to form the composite sample tested.

6.2.2 Sample Handling

King County personnel collected a composite effluent sample on October 9 1997. The sample was shipped to Parametrix's Environmental Toxicology Laboratory, and refrigerated at 4°C until used for testing. Subsamples of the effluent were taken upon arrival for determination of temperature, pH, salinity, DO, conductivity, hardness, alkalinity, total residual chlorine, and ammonia.

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[32] The chronic toxicity test protocol developed by U.S. EPA measures both acute and chronic toxicity endpoints.

6.2.3 Source and Condition of Organisms

C. dubia were obtained from laboratory stock cultures and were ≤24 hours old at test initiation for the acute and chronic bioassays. Fathead minnows, *P. promelas,* were purchased from Aquatic Biosystems Inc., Fort Collins, Colorado and were 7 days old for the acute bioassay and ≤24 hours old at test initiation for the chronic bioassay.

A reference toxicant was used to assess the relative health of the test organisms and to ensure that their sensitivity fell within an expected concentration range. Sodium chloride was used as the reference toxicant for the *C. dubia* test. Potassium chloride was used as the reference toxicant for the *P. promelas* test.

6.2.4 Test Methods

The chronic tests were conducted according to WSDOE WAC Chapter 173-205, 1993; WSDOE Publication No. WQ-R-95-80; and *Short-term Methods for Estimating the Chronic Toxicity of Effluents and Receiving Waters to Freshwater Organisms,* U.S. EPA/600/4-91/002, July 1994. Summaries of test conditions for each test are presented in Table 6-1 and Table 6-2.

Table 6-1. Summary of Test Conditions for the Chronic Definitive *Ceriodaphnia dubia* Bioassay

Job Name:	King County Department of Natural Resources
Date:	9-16 October 1997
Test Protocol:	WSDOE, WAC Chapter 173-205, 1993; WSDOE Publication No. WQ-R-95-80; and *Short-term Methods for Estimating the Chronic Toxicity of Effluents and Receiving Waters to Freshwater Organisms*, U.S. EPA/600/4-91/002, July 1994.
Test Material:	Brandon Street CSO effluent
Test Organism/Age:	≤24 hrs old
Source:	In-house culture
Number/Test Chamber:	One
Volume/Test Chamber:	15 mL
Test Concentrations:	0, 6.25, 12.5, 25, 50, and 100% effluent
Replicates:	Ten
Reference Toxicant:	Sodium chloride
Test Duration:	7 days

Table 6-1. Summary of Test Conditions for the Chronic Definitive *Ceriodaphnia dubia* Bioassay (continued)

Control/Dilution Media:	Natural spring water (80-100 mg/L hardness as $CaCO_3$)
Test Chambers:	30 mL polypropylene cups
Lighting:	Fluorescent bulbs (50-100 foot candles)
Renewal:	Daily
Photoperiod:	16 hours light; 8 hours dark
Aeration:	None
Feeding:	Daily: 100 µL *Selenastrum* suspension; 100 µL yeast/Cerophyl/trout chow (YCT)
Temperature:	25 ± 1°C
Chemical Data:	pH and DO for each test concentration and the control (both initial and final solutions); temperature and specific conductivity at test initiation and every 24 hours; hardness, alkalinity, ammonia and total residual chlorine for each new sample
Effect Measured:	Mortality (defined as immobility) and reproduction
Test Acceptability:	Control mortality ≤ 20%; ≥ 60% of control organisms produce three broods, an average total of 15 or more offspring for the first three broods must be produced

Table 6-2. Summary of Test Conditions for the Chronic Definitive *Pimephales promelas* Bioassay

Job Name:	King County Department of Natural Resources
Date:	10-17 October 1997
Test Protocol:	WSDOE, WAC Chapter 173-205, 1993; WSDOE Publication No. WQ-R-95-80; and *Short-term Methods for Estimating the Chronic Toxicity of Effluents and Receiving Waters to Freshwater Organisms*, U.S. EPA/600/4-91/002, July 1994.
Test Material:	Brandon Street CSO effluent
Test Organisms/Age:	≤24 hours old
Source:	Aquatic Biosystems; Fort Collins, Colorado
Acclimation Period:	None

Table 6-2. Summary of Test Conditions for the Chronic Definitive *Pimephales promelas* Bioassay (continued)

Number/Test Chamber:	Ten
Volume/Test Chamber:	400 mL
Test Concentrations:	0, 6.25, 12.5, 25, 50, and 100% effluent
Replicates:	Four
Reference Toxicant:	Potassium chloride
Test Duration:	7 days
Control/Dilution Media:	Laboratory-prepared synthetic water (80-100 mg/L hardness as $CaCO_3$)
Test Chambers:	800 mL polyethylene beakers
Lighting:	Fluorescent bulbs (50-100 foot candles)
Photoperiod:	16 hours light; 8 hours dark
Aeration:	None
Feeding:	0.15 mL newly hatched brine shrimp nauplii twice daily
Renewal:	Daily
Temperature:	25 ± 1°C
Chemical Data:	DO and pH for each test concentration and the control (both initial and final solutions); temperature and specific conductivity at initiation and every 24 hours; hardness, alkalinity, ammonia and total residual chlorine at initiation for each new sample
Effect Measured:	Mortality and growth
Test Acceptability:	Control mortality ≤ 20%, average mean control weight ≥ 0.25 mg

6.3 Results

6.3.1 Initial Chemical and Physical Determinations

The results of initial chemical and physical determinations made for the 100 percent effluent samples are summarized in Table 6-3. Complete data are available in Subappendix A.

Table 6-3. Initial Chemical and Physical Determinations

Parameter Measured	Laboratory Measurement
Temperature (°C)	11
Salinity (ppt)	0
DO (mg/L)	9.2
pH	7.1
Conductivity (µS)	30
Total Hardness (mg/L $CaCO_3$)	18
Total Alkalinity (mg/L $CaCO_3$)	20
Total Residual Chlorine (mg/L)	<0.01
Ammonia (mg/L)	<1

6.3.2 Bioassay Results

Bioassay results are summarized below in Table 6-4. Control responses and reference toxicant results are within acceptable ranges. Complete information is available in Subappendix A. In summary, no toxicity was observed in either chronic bioassay, with NOECs of 100 percent effluent for both tests.

Table 6-4. Summary of Bioassay Results

	C. dubia		***P. promelas***	
Evaluation	**Survival (% effluent)**	**Reproduction (% effluent)**	**Survival (% effluent)**	**Growth (% effluent)**
NOEC	100	100	100	100
LOEC	>100	>100	>100	>100
LC_{50}	>100	N/A	>100	N/A
Reference Toxicant (LC_{50}) =	1.8 g/L NaCl		0.7 g/L KCl	

N/A = Not applicable

7. BENTHIC INFAUNAL COMMUNITY ANALYSIS

7.1 Introduction

In February 1997 Striplin Environmental Associates (SEA) contracted with King County through Parametrix, Inc. to conduct a benthic infauna community analysis. The objective of the benthic infauna community analysis was to determine whether or not near-field effects from a CSO or storm drain could be identified, and if possible, to determine the extent of the effect. To meet this objective the following analyses were undertaken:

- Compare benthic infauna data from a CSO or storm drain to benthic infauna data from an in-river reference site.
- Correlate benthic infauna data at each station with sediment chemistry and conventionals data.
- Compare benthic infauna data from both locations to the Puget Sound Reference Value data set (SEA 1996).
- Identify dominant taxa at each location and compare CSO/storm drain data with reference site data.
- Discuss and describe the ecosystem and ecological function of benthic infauna present and absent from each location.

Potential differences in the benthic infaunal communities found at the CSO or storm drain in the present study were undertaken in an attempt to validate predictions of risk to those communities based on model-derived sediment chemical concentrations.

7.1.1 Selection of the Study Area

Four CSOs and two reference sites were identified as candidate study areas. The candidate sites included:

1. Connecticut Street CSO
2. Brandon Street CSO
3. Denny Way CSO
4. Duwamish/Diagonal Way CSO/storm drain
5. Kellogg Island (reference site)
6. Snohomish River (reference site)

Several factors were considered in the selection of the study locations. These included the availability of sediment chemistry data, sediment conventional data (total organic carbon [TOC], grain size), and preferably some quantitative benthic community data.

Discussions among the study team indicated that the Duwamish/Diagonal Way CSO/storm drain was recently studied (it is in the initial stages of remediation). An extensive sediment chemistry database has been developed in the vicinity of the outfall and clear gradients of organic enrichment (based on TOC) and chemical contamination have been documented (King County 1997). For these reasons the Duwamish/Diagonal Way CSO/storm drain was selected as the study site for the benthic community assessment.

The Duwamish/Diagonal Way CSO/storm drain can discharge both combined sanitary and storm water as well as separated storm water. This pipe drains the Diagonal and Hanford drainage basins of Seattle. A 1987 separation project in the Hanford drainage basin eliminated more than 300 million gallons of CSO discharge per year at the Duwamish/Diagonal Way CSO/storm drain outfall. Winter 1996-1997, however, was unusually wet, resulting in twelve CSO overflows with an estimated volume of 186 million gallons (Zhong Ji, Wastewater Treatment Division, Seattle, Washington, personal communication). The March 18^{th} and 19^{th} storm alone resulted in a CSO discharge of approximately 35 million gallons. The annual discharge of separated storm water from the Duwamish/Diagonal Way storm drain is estimated to be 1,230 million gallons.

The selection of an appropriate reference site was more problematic because neither of the two potential reference sites had all of the preferred elements. Ideally the reference site should have all of the physical characteristics of the study site except the chemical contamination. For example, a study site in a river should have a reference site in a river. If physical disturbance (e.g., ship/boat traffic) could influence benthic communities at the study site, then the same physical disturbances should be present at the reference site.

The first candidate reference site in the Duwamish River, Kellogg Island, was extensively studied by Cordell et al. (1994, 1996). This research was established to set up long-term reference sites and to collect initial baseline data. Their research showed that the Kellogg Island site appeared to be the most promising reference site in the lower stretch of the river. They reported high numbers of species, high abundances and in general a higher diversity at the Kellogg Island site compared to other areas studied. King County, as part of the Duwamish River water quality assessment, collected and analyzed 12 sediment samples for chemical contaminants off of Kellogg Island from October 1996 through June 1997. The results indicated low concentrations of metals and organic compounds with no exceedances of the Washington State sediment quality standards (SQS, Chapter 173-204 WAC). The other potential reference site was either Steamboat or Ebey Sloughs in the Snohomish River. Cardwell (1997) qualitatively sampled several locations in each slough. They recommended using Steamboat Slough as a reference for this project because it is of a similar size to the Duwamish, primarily undeveloped, and contains relatively pristine habitats.

The Kellogg Island site was selected as the preferred reference site for a number of reasons. First, there were quantitative benthic community data that indicated that the community was healthy and diverse and thus most likely unaffected by chemical contamination. Second, sediment chemistry data indicated that no chemicals exceeded the SQSs. Third, Kellogg Island was located across and slightly upstream of the CSO, and so is influenced by generally the same physical factors as the CSO location. Finally, little or no quantitative sediment chemistry or benthic infauna data exist for the Snohomish River sloughs, and so limited the evaluation of this candidate reference site.

7.1.2 Section Organization

The remainder of this section describes the benthic community analysis. Section 7.2 describes the methods used to collect and process the benthic samples and the analytical techniques used to analyze and evaluate the benthic infaunal community. Section 7.3 presents the results of the analysis. Section 7.4 discusses the results and describes the ecological significance of the dominant species. Section 7.5 summarizes the results in relation to the project objective.

The field sampling forms are found in Subappendix B. The raw taxonomic data are found in Subappendix C, and sediment chemistry and conventional (i.e., TOC, and grain size) data obtained from King County (not analyzed in detail for this section) are included in Subappendix D.

Samples were also collected for sediment chemical analysis. The chemical results show that the stations in proximity to the CSO contained higher levels of chemicals relative to the reference stations.

7.2 Methods

7.2.1 Field Sampling

Nine stations were sampled for the benthic infaunal community analysis (Figure 7-1). Five stations were located off of the Duwamish Diagonal CSO/storm drain on a transect line oriented towards Kellogg Island. The remaining four stations were located roughly along the same line but were adjacent to Kellogg Island. The Kellogg Island stations were the project reference stations.

Benthic infaunal and sediment chemistry samples were collected between September 22 and September 25, 1997. Samples were collected from an aluminum 26-foot Almar, owned by Parametrix, Inc. The vessel was equipped with an A-frame and a hydraulic winch carrying 300 feet of ¼-inch stainless steel cable.

Station positioning was accomplished using a Trimbel Pathfinder Global Positioning System (GPS) with differential correction provided by an OMNI-STAR correction antenna. GPS corrected positions were then translated into Washington State Plane Coordinates (Table 7-1).

Need map, see figure 1, appendix m of the July 98 draft

Figure 7-1. Benthic Assessment Station Locations

Table 7-1. Station Coordinates

Station	Northing	Easting	Status
Duwamish Diagonal Stations			
DD-1	209120	1267153	Processed
DD-2	209059	1267092	Archived
DD-3	208929	1267040	Processed
DD-4	208785	1266933	Archived
DD-5	208606	1266844	Processed
Kellogg Island Stations			
KI-1	208552	1266651	Processed
KI-2	208274	1266665	Processed
KI-3	208216	1266675	Archived
KI-4	207755	1266615	Processed

Sediment samples were collected using a single 0.1 m^2-modified van Veen grab sampler. Sampling procedures followed the Puget Sound Estuary Program Methodology (PSEP 1986). Sampling equipment used to collect sediment for chemical analysis was decontaminated between stations using the following procedure:

1. Scrubbing with a nylon brush
2. On-board rinsing with sea water
3. Final rinsing by dunking the sampler several times prior to reaching the next sampling location.

The benthic infaunal samples were placed in a 1.0 mm mesh sieve screen box and gently rinsed of adhering sediment. Once the sieving was complete the remaining material was rinsed into thick plastic bags and preserved with a formaldehyde solution buffered with sodium borate.

7.2.2 Laboratory Analysis

Three of the five stations sampled near the CSO were processed and the remaining two were archived for later processing. At Kellogg Island, three of the four stations sampled were processed and the final one was archived.

Sediment chemical (metals, semivolatile organics, PCBs, TBT) and conventional variables (particle size, TOC) were analyzed by the King County Environmental laboratory following Puget South Estuary Program methodology (PSEP 1996a,b). Metals were analyzed using inductively coupled plasma (ICP) emission spectrometry with the exception of mercury. Mercury was analyzed by cold-vapor atomic absorption spectrophotometry. Semivolatile organics were extracted with an organic solvent and then analyzed by gas chromatography/mass spectrometry (GC/MS). PCBs were extracted with organic solvents and then analyzed using a gas chromatograph equipped with an electron capture detector (ECD). TBT was analyzed by GC/MS following the methods of Unger et al. (1986) and Krone et al. (1989).

Benthic infaunal samples were processed and analyzed according to Puget Sound Estuary Program Methodology (PSEP 1987) by Fukuyama-Hironaka Taxonomic and Environmental Services. In a deviation of the protocol, only three of the five benthic replicates from each station were sorted and identified. The taxonomic identifications of samples that were processed were completed to the lowest taxonomic level possible.

7.2.3 Data Analysis

The raw benthic data were imported into the SEA database system and summary statistics for 21 benthic endpoints were calculated by sample and station. These endpoints are presented in Table 7-2:

Table 7-2. Benthic Endpoints

ENDPOINT	
Total abundance (TOAB)	Oligochaete abundance (OLIGO)
Total number of taxa (TOTAX)	Miscellaneous taxa abundance (MISCAB)
Shannon-Wiener Diversity (H')	Polychaete taxa (POTAX)
Pielou's Evenness Index (J')	Mollusca taxa (MOTAX)
Infaunal Trophic Index (ITI)	Arthropod taxa (ARTAX)
Polychaete abundance (POAB)	Crustacean taxa (CRTAX)
Mollusca abundance (MOAB)	Amphipod taxa (AMPTAX)
Arthropod abundance (ARAB)	Echinoderm taxa (ECHTAX)
Crustacean abundance (CRAB)	Miscellaneous taxa (MISCTAX)
Amphipod abundance (AMPHAB)	Swartz's Dominance Index (SDI)
Echinoderm abundance (ECHAB)	

The raw species level data are presented in Subappendix B. Summary data from the CSO/storm drain stations were compared to summary data from the reference stations. Stations were paired for comparison based on distance from each respective shoreline, water depth and by grain size (represented by percent fines, which is the combined amount of silt and clay in the sample). The top ten numerically dominant taxa at each station were listed. Data from all stations were also compared to the 1996 reference value ranges developed by SEA for WSDOE (SEA 1996).

Procedures outlined in the Washington State Sediment Management Standards were used to determine whether a station was considered to be impacted. For this to occur the mean abundance of the major taxa groups (polychaetes, mollusks, and crustaceans) at the test station must be 50 percent reduced and statistically different from the mean abundance at the reference station. Statistical testing (t-test) was conducted using the software package SYSTAT for Windows, version 7.0. Prior to conducting statistical testing, histogram plots were prepared to examine the data for departures from normality and the abundance data were log transformed prior to testing.

7.3 Results

7.3.1 Chemical Results

In general terms the stations located on the CSO/storm drain side of the Duwamish River were organically enriched compared to stations on the Kellogg Island side of the river. All CSO/storm drain stations had moderate concentrations of low and high molecular weight polycyclic aromatic hydrocarbons (LPAH and HPAH respectively), bis(2-ethylhexyl) phthalate and coprostanol (Subappendix C). Coprostanol is a fecal steroid compound produced by the microbial breakdown of cholesterol in the digestive tracts of mammals which makes it useful as a tracer for sewage. All of these chemicals decreased in concentration with increasing distance from the CSO/storm drain (Subappendix C).

HPAH compounds were found at the two outermost Kellogg Island reference stations, and concentrations decreased from offshore to inshore. Coprostanol was undetected near Kellogg Island suggesting that these stations were not affected by discharges from the CSO/storm drain. These stations also showed a gradient of increasing organic carbon and percent fines moving away from the shoreline.

7.3.2 General Community Characteristics

A total of 28,428 benthic infaunal organisms representing 171 taxa were found in the 18 samples from the study area. Mean total abundance at each station ranged from 259.7/ $0.1m^2$ at Station DD-1 (closest to the CSO) to 5,444.7/$0.1m^2$ at Station KI-4 (closest to Kellogg Island) (Table 7-3). The mean total number of taxa ranged from 10.7 at Station DD-1 to 54.0 at Station KI-2. The Shannon-Wiener diversity values were less than 1.0 at all stations except at DD-5 (1.1) and KI-2 (1.2). With the exception of Station DD-1, the infaunal trophic index (ITI) ranged from 62.3 to 65.0, indicating communities dominated by surface detrital/surface deposit feeding organisms. The ITI at Station DD-1 was 1.0, indicating a community dominated by subsurface deposit feeding organisms.

Table 7-3. Summary of Benthic Endpoints

Sample	Total Abundance	Total Taxa	H’	J’	ITI	SDI	Abundance							
							Polychaete	Mollusk	Arthropod	Crustacea	Amphipod	Echinoderm	Oligochaete	Misc.
DD-1														
Rep 3	333	15	0.391	0.332	0	2	71	1	7	6	5	0	247	3
Rep 4	302	9	0.448	0.47	0	2	118	0	3	3	2	0	170	8
Rep 5	144	8	0.536	0.594	3	2	47	0	1	1	1	0	83	5
Fines = 14.6% TOC[a] = 2.7%														
Mean	259.7	10.7	0.5	0.5	1.0	2.0	78.7	0.3	3.7	3.3	2.7	0	166.7	5.3
STDS[b]	101.4	3.8	0.1	0.1	1.7	0	36.1	0.6	3.1	2.5	2.1	0	82.1	2.5
CV[c]	39.0	35.5	15.9	28.2	173.2	0	45.9	173.2	83.3	75.5	78.1	0	49.2	47.2
DD-3														
Rep 1	888	45	0.686	0.415	61	3	749	101	22	22	3	0	16	0
Rep 2	1,257	43	0.518	0.317	64	1	1,112	96	44	44	9	0	5	0
Rep 4	1,031	45	0.729	0.441	62	4	800	130	52	52	4	0	8	0
Fines = 81.2% TOC = 3.7%														
Mean	1,058.7	44.3	0.6	0.4	62.3	2.7	887.0	109.0	39.3	39.3	5.3	0	9.7	0
STDS	186.0	1.2	0.1	0.1	1.5	1.5	196.5	18.4	15.5	15.5	3.2	0	5.7	0
CV	17.6	2.6	17.3	16.7	2.5	57.3	22.2	16.8	39.5	39.5	60.3	0	0	0

[a] TOC – Total organic carbon

[b] STDS = Sample standard deviation

[c] CV = Coefficient of variation

Table 7-3. Summary of Benthic Endpoints (Continued)

Sample	Total Abundance	Total Taxa	H’	J’	ITI	SDI	Abundance							
							Polychaete	Mollusk	Arthropod	Crustacea	Amphipod	Echinoderm	Oligochaete	Misc.
DD-5														
Rep 3	815	43	0.997	0.61	63	5	392	388	29	29	0	2	1	4
Rep 4	960	51	1.17	0.685	64	7	371	372	110	110	5	2	1	3
Rep 5	626	47	1.118	0.669	63	6	198	289	138	138	8	1	0	0
Fines = 85.2% TOC = 1.9%														
Mean	800.3	47.0	1.1	0.7	63.3	6.0	320.3	349.7	92.3	92.3	4.3	1.7	0.5	2.3
STDS	167.5	4.0	0.1	0.0	0.6	1.0	106.5	53.1	56.6	56.6	4.0	0.6	0.7	2.1
CV	20.9	8.5	8.1	6.0	0.9	16.7	33.2	15.2	61.3	61.3	93.3	34.6	141.4	89.2
KI-1														
Rep 2	1,409	47	0.675	0.403	65	2	1,104	248	44	44	5	5	0	4
Rep 3	1,440	51	0.786	0.461	65	4	1,073	280	83	83	19	0	0	4
Rep 4	1,223	38	0.707	0.447	65	3	958	227	38	38	6	0	0	0
Fines = 90.6% TOC = 2.3%														
Mean	1,357.3	45.3	0.7	0.4	65.0	3.0	1,045.0	251.7	55.0	55.0	10.0	1.7	0	2.7
STDS	117.4	6.7	0.1	0.0	0.0	1.0	76.9	26.7	24.4	24.4	7.8	2.9	0	2.3
CV	8.6	14.7	7.9	6.9	0.0	33.3	7.4	10.6	44.4	44.4	78.1	173.2	0	86.6

Table 7-3. Summary of Benthic Endpoints (Continued)

Sample	Total Abundance	Total Taxa	H'	J'	ITI	SDI	Abundance							
							Polychaete	Mollusk	Arthropod	Crustacea	Amphipod	Echinoderm	Oligochaete	Misc.
KI-2														
Rep 1	526	62	1.257	0.701	65	9	231	220	64	64	3	5	1	3
Rep 2	509	46	1.163	0.699	61	8	224	190	54	54	1	0	36	5
Rep 5	631	54	1.279	0.738	63	9	277	256	80	79	3	1	3	11
Fines = 93.2% TOC = 2.0%														
Mean	555.3	54.0	1.2	0.7	63.0	8.7	244.0	222.0	66.0	65.7	2.3	2.0	13.3	6.3
STDS	66.1	8.0	0.1	0.0	2.0	0.6	28.8	33.0	13.1	12.6	1.2	2.6	19.7	4.2
CV	11.9	14.8	5.0	3.1	3.2	6.7	11.8	14.9	19.9	19.2	49.5	132.3	147.4	65.7
KI-4														
Rep 3	4,755	29	0.716	0.49	62	2	2,737	55	1,795	1,795	1,646	0	164	4
Rep 4	5,410	33	0.514	0.338	65	2	3,924	20	1,357	1,357	1,238	0	103	4
Rep 5	6,169	33	0.593	0.391	64	2	4,139	21	1,872	1,872	1,754	0	135	2
Fines = 24.7% TOC = 1.0%														
Mean	5,444.7	31.7	0.6	0.4	63.7	2	3,600.0	32.0	1,674.7	1,674.7	1,546.0	0	134.0	3.3
STDS	707.6	2.3	0.1	0.1	1.5	0	755.1	19.9	277.8	277.8	272.1	0	30.5	1.2
CV	13.0	7.3	16.8	19.0	2.4	0	21.0	62.3	16.6	16.6	17.6	0	22.8	34.6

Table 7-3. Summary of Benthic Endpoints (Continued)

Sample	Number of Taxa						
	Polychaete	**Amphipod**	**Mollusk**	**Echinoderm**	**Crustacea**	**Arthropod**	**Misc.**
DD-1							
Rep 3	4	4	1	0	5	6	2
Rep 4	3	2	0	0	3	3	1
Rep 5	3	1	0	0	1	1	2
Fines = 14.6%							
TOC = 2.7%							
Mean	3.3	2.3	0.3	0	3.0	3.3	1.7
STDS	0.6	1.5	0.6	0	2.0	2.5	0.6
CV	17.3	65.5	173.2	0	66.7	75.5	34.6
DD-3							
Rep 1	17	2	18	0	9	9	0
Rep 2	15	6	15	0	12	12	0
Rep 4	16	3	19	0	8	8	0
Fines = 81.2%							
TOC = 3.7%							
Mean	16.0	3.7	17.3	0	9.7	9.7	0
STDS	1.0	2.1	2.1	0	2.1	2.1	0
CV	6.3	56.8	12.0	0	21.5	21.5	0

Table 7-3. Summary of Benthic Endpoints (Continued)

Sample	Number of Taxa						
	Polychaete	Amphipod	Mollusk	Echinoderm	Crustacea	Arthropod	Misc.
DD-5							
Rep 3	21	0	11	2	6	6	3
Rep 4	20	2	19	2	5	5	3
Rep 5	17	5	17	1	12	12	0
Fines = 85.2%							
TOC = 1.9%							
Mean	19.3	2.3	15.7	1.7	7.7	7.7	2.0
STDS	2.1	2.5	4.2	0.6	3.8	3.8	1.7
CV	10.8	107.9	26.6	34.6	49.4	49.4	86.6
KI-1							
Rep 2	15	4	17	4	8	8	2
Rep 3	17	6	20	0	11	11	3
Rep 4	15	4	15	0	8	8	0
Fines = 90.6%							
TOC = 2.3%							
Mean	15.7	4.7	17.3	1.3	9.0	9.0	1.7
STDS	1.2	1.2	2.5	2.3	1.7	1.7	1.5
CV	7.4	24.7	14.5	173.2	19.2	19.2	91.7

Table 7-3. Summary of Benthic Endpoints (Continued)

Sample	Number of Taxa						
	Polychaete	**Amphipod**	**Mollusk**	**Echinoderm**	**Crustacea**	**Arthropod**	**Misc.**
KI-2							
Rep 1	33	2	13	2	6	6	3
Rep 2	20	1	15	0	6	6	3
Rep 5	21	1	16	1	7	8	6
Fines = 93.2%							
TOC = 2.0%							
Mean	24.7	1.3	14.7	1.0	6.3	6.7	4.0
STDS	7.2	0.6	1.5	1.0	0.6	1.2	1.7
CV	29.3	43.3	10.4	100.0	9.1	17.3	43.3
KI-4							
Rep 3	10	6	4	0	12	12	2
Rep 4	11	5	5	0	13	13	2
Rep 5	11	7	6	0	14	14	1
Fines = 24.7%							
TOC = 1.0%							
Mean	10.7	6.0	5.0	0.0	13.0	13.0	1.7
STDS	0.6	1.0	1.0	0.0	1.0	1.0	0.6
CV	5.4	16.7	20.0	0.0	7.7	7.7	34.6

The relative abundance of major taxa groups at each station are summarized in Table 7-4. Station DD-1 was numerically dominated by annelid worms with oligochaetes accounting for 64.2 percent of the abundance followed by polychaetes at 30.3 percent of the abundance. Station DD-3 was strongly dominated by polychaete worms, which accounted for 83.8 percent of the population, followed by mollusks at 10.3 percent. Station DD-5 was equally dominated by polychaetes and mollusks with relative abundances of 40 and 43.7 percent respectively. Arthropods at this station accounted for 11.5 percent of the population.

Table 7-4. Relative Abundance of the Major Taxa Groups

Station	Poly-chaeta	Mollusca	Arthro-poda	Echnio-dermata	Oligo-chaeta	Misc.
DD-1	30.3	0.1	1.4	0.0	64.2	2.1
DD-3	83.8	10.3	3.7	0.0	0.9	0.0
DD-5	40.0	43.7	11.5	0.2	0.0	0.3
KI-1	77.0	18.5	4.1	0.1	0.0	0.2
KI-2	43.9	40.0	11.9	0.4	2.4	1.1
KI-4	66.1	0.6	30.8	0.0	2.5	0.1

Off of Kellogg Island, Station KI-4 was dominated by polychaetes with 66.1 percent of the total abundance followed by arthropods at 30.8 percent. Station KI-2 was equally dominated by polychaetes (43.9 percent) and by mollusks (40.0 percent). Arthropods at this station accounted for 11.9 percent of the abundance. Station KI-1, which was closest to the center of the river on the Kellogg Island side, was dominated by polychaetes which accounted for 77 percent of the abundance followed by mollusks at 18.5 percent.

7.3.3 Comparison of CSO and Kellogg Island Stations

Stations for comparison were determined by distance from shore, water depth, and by the sediment grain size (percent fines). The distance from shore was included as a criterion because the river has been channeled by dredging for navigation purposes and ship traffic, by physically disturbing the river bed (i.e., propeller wake at shallow depths), may be impacting the community at the margins of the channel. The conventional parameters for each station pair are presented in Table 7-5.

The conventional parameters show that the percent fines were fairly similar between station pairs, however TOC was different. This is especially true for station pairs DD-1/KI-4 and DD-3/KI-2. The excess TOC at the DD stations is more than likely due to the discharge of the CSO/storm drain.

Table 7-5. Conventional Parameters for Sediment Sampled for Benthic Diversity

Station	Water Depth (ft)	Percent Fines	TOC
DD-1	9.4	14.6	2.7
KI-4	10.7	24.7	1.0
DD-3	24.9	81.2	3.7
KI-2	24.1	93.2	2.0
DD-5	39.5	85.2	1.9
KI-1	40.8	90.6	2.3

The Washington State Sediment Management Standards use a 50 percent reduction in the mean abundance of one of the major taxa groups (polychaetes, crustaceans, and mollusks) relative to the reference station and statistical significance ($p<0.05$) to differentiate between an impacted and an unimpacted station. Results of the t-tests are presented in Table 7-6. At Station DD-1, 11 of the 14 endpoints tested were significantly depressed compared to Station KI-4. All three SMS endpoints were statistically different and had abundances less that 50 percent of the reference station mean. Four endpoints at Station DD-3 were depressed compared to Station KI-2, one of which was mollusk abundance. In addition to being statistically different, mollusk abundance was 50 percent less than the reference station mean, indicating an impacted station according to the SMS. Two other endpoints, polychaete abundance and the total abundance, were significantly enhanced above the reference station mean. Three endpoints at Station DD-5 were statistically different from the reference mean and one of these, polychaete abundance, was 50 percent less than the reference station mean. Four other endpoints at this station were enhanced compared to Station KI-1.

7.3.4 Numerically Dominant Taxa

The ten most abundant species at each station are shown in Table 7-7. The values represent the total abundance from all three replicates. Station DD-1 was dominated by oligochaetes (marine earthworms) and the polychaetes *Capitella capitata* and *Neanthes* sp. There was similarity among the dominant taxa at Stations DD-3, DD-5, KI-1, and KI-2. These stations were located away from the shoreline. Taxa found in common among Stations DD-3, DD-5, KI-1, and KI-2 include the polychaete worms *Aphelochaeta* sp. and *Scoletoma luti*. The two bivalve mollusks *Axinopsida serricata* and *Psephedia lordi* were found in common among Stations DD-5 (most offshore CSO/storm drain) and Stations KI-1 and KI-2 (most offshore Kellogg Island Station). The ostracod *Euphilomedes carcharodonta* was dominant at the two offshore CSO stations (DD-3 and DD-5). With the exception of the oligochaetes and *Capitella capitata*, the dominant species at KI-4 were found at no other stations. Station KI-4 was strongly dominated by the polychaete *Pygospio elegans* and two species of amphipods *Corophium salmonis* and *C. spinicorne*.

Table 7-6. Summary of t-Test Results

Station Comparisons	DDS-1 Versus KI-4			DDS-3 Versus KI-2			DDS-5 Versus KI-1		
Benthic Endpoint	**DDS-1 Mean**	**KI-4 Mean**	**P**	**DDS-3 Mean**	**KI-2 Mean**	**P**	**DDS-5 Mean**	**KI-1 Mean**	**P**
Total Abundance	259.7	5,444.7	0.004	1,058.7	555.3	0.009	800.3	1,357.3	0.035
Total Taxa	10.7	31.7	0.003	44.3	54.0	0.169	47	45.3	0.733
Crustacea Abundance	**3.3**	1,674.4	0.005	39.3	65.7	0.154	92.3	55.0	0.531
Crustacea Taxa	3.0	13.0	0.005	9.7	6.3	0.100	7.7	9.0	0.620
Amphipod Abundance	2.7	1,546.0	0.003	5.3	2.3	0.169	4.3	10.0	0.357
Amphipod Taxa	2.3	6.0	0.032	3.7	1.3	0.185	2.3	4.7	0.246
Polychaete Abundance	**78.7**	3,600.0	0.001	887.0	244.0	0.002	**320.3**	1,045.0	0.027
polycheate Taxa	3.3	10.7	0.000	16.0	24.7	0.171	19.3	15.7	0.073
Mollusca Abundance	**0.3**	32.0	0.025	**109.0**	222.0	0.005	349.7	251.7	0.050
Mollusca Taxa	0.3	5.0	0.005	17.3	14.7	0.155	15.7	17.3	0.591
Shannon-Wiener Diversity (H')	0.500	0.600	0.115	0.600	1.200	0.004	1.100	0.700	0.006
Pielou's Eveness Index (J')	0.500	0.400	0.546	0.400	0.700	0.008	0.700	0.400	0.002
Infaunal Trophic Index (ITI)	1.0	63.7	0.000	62.3	63.0	0.672	63.3	65.0	0.037
Swartz's Dominance Index (SDI)	2.0	2.0	1.000	2.7	8.7	0.012	6.0	3.0	0.021

= Endpoints with significant depressions compared to the reference station and cells surrounded by a box represent endpoints with significant enhancements relative to the reference station.

= Darkly shaded cells with bold numbers are mean values less than 50 percent of the reference station mean and are statistically different.

Table 7-7. The Ten Most Abundant Species at Each Station

Taxon	Station					
	DD-1	DD-3	DD-5	KI-1	KI-2	KI-4
Oligochaeta	500	29	2	0	40	402
Capitella capitata	203	121	2	8	0	281
Neanthes sp.	25	0	0	0		16
Nematoda	15	41*	2	5	5	2
Chironomidae larvae	13	0	0	0	0	0
Eteone sp.	4	0	6	3	1	54
Corophium acherusicum	3	0	0	0	3	4
Harpacticus sp.	2	0	0	0	0	0
Diptera pupae	2					
Aphelochaeta sp.	0	2,284	295	2,503	117	0
Scoletoma luti	0	78	321	474	347	0
Euphilomedes carcharodonta	0	71	198	105	59	3
Clinocardium sp.	0	65	13	18	14	0
Axinopsida serricata	0	26	474	158	154	2
Psephedia lordi	0	55	239	176	272	1
Cossura pygodactylata	0	46	30	18	49	0
Macoma sp.	0	45	13	24	13	1
Eucone limnicola	0	34	53	29	14	0
Parvilucina tenuisculpta	0	31	32	98	41	0
Macoma carlottensis	0	17	190	128	88	1
Heteromastus filobranchus	0	23	139	45	63	0
Eudorella pacifica	0	1	46	8	102	0
Epitonium sp.	0	4	24	65	5	0
Pseudeopolydora kempi	0	0	0	0	42	28
Pygospio elegans	0	0	0	0	0	9,865
Corophium salmonis	0	0	0	0	0	3,507
Corophium spinicorne	0	0	0	0	0	582
Manayunkia aestuarina	4	0	0	0	0	262
Grandidierella japonica	2	0	0	0	0	271
Eogammarus confervicolus	0	0	0	0	0	260
Cumella vulgaris	0	0	0	0	0	200
Hobsonia florida	0	0	0	0	0	158

* Indicates that all 41 individuals were found in one sample.

= Indicates the ten taxa at each station.

7.3.5 Comparison to 1996 Reference Range Values

Reference ranges for benthic infauna communities in Puget Sound have been developed by WSDOE to help in the identification of reference areas and to use as a yardstick to compare against site specific reference stations (SEA 1996). Ranges for 14 benthic infauna endpoints were developed for four habitat categories (SEA 1996). These categories were based on ranges of sediment grain size (percent fines) and include: 0 to 20, 20 to 50, 50 to 80, and 80 to 100 percent fines.

The ranges, shown in Table 7-8, represent one standard deviation around the mean for each benthic endpoint. If a test station mean is outside of the range, it is considered to be statistically different from the mean for that endpoint. (For a detailed review of the reference ranges see SEA [1996].)

Table 7-8. Reference Value Ranges for Puget Sound Habitats[a]

Benthic Endpoint	Habitat Category <150 ft							
	N	0-20% Fines	N	20-50% Fines	N	50-80% Fines	N	80-100% Fines
Total Abundance	184	295-983	69	342-647	79	156-531	97	178-436
Total Taxa	183	47-90	66	50-78	81	38-66	99	24-42
Crustacean Abundance	180	43-198	68	40-167	77	0-104	98	4-148
Crustacean Taxa	181	8-17	66	6-16	80	4-10	103	3-7
Amphipod Abundance	186	8-47	63	0-27	83	1-29	95	0-44
Amphipod Taxa	185	4-10	66	2-7	78	1-5	92	1-3
Polychaete Abundance	178	72-322	67	126-322	82	78-215	97	31-145
Polychaete Taxa	193	21-47	68	28-51	81	21-36	99	9-22
Mollusk Abundance	178	26-150	65	27-192	78	0-232	98	24-104
Mollusk Taxa	185	12-21	66	9-17	82	8-18	100	6-13
Shannon-Wiener Diversity (H')	185	1.12-1.57	69	1.10-1.53	86	1.01-1.45	95	0.88-1.23
Pielou's Eveness Index (J')	182	0.65-0.83	69	0.63-0.82	86	0.59-0.85	99	0.6-0.82
Infaunal Trophic Index (ITI)	183	67.7-81.1	65	65.9-77.3	83	63.2-77.2	101	67.3-87.1
Swartz's Dominance Index (SDI)	186	6.8-21.6	68	8.3-19.2	84	5.5-16.5	98	4.2-9.6

[a] All Values are Presented in per $0.1m^2$

N = Number of samples

Stations from this project fall into the following three habitat categories: 0 to 20, 20 to 50, and 80 to 100 percent fines. Results of the comparisons to reference value ranges are tabulated in Table 7-9. Shaded values represent endpoints that were outside of the reference range for that endpoint.

Eleven of the 12 endpoints for Station DD-1 were depressed below the reference area range indicating a severely impacted station. Polychaete abundance (POAB) at five of the six stations, and mollusk abundance (MOAB) at four of the six stations were enhanced above the reference range. Because these two endpoints contribute to the total abundance (TOAB), this endpoint was also enhanced above the reference range at five of the six stations.

The total number of taxa at four of the six stations was enhanced above the reference range for that endpoint. This endpoint at Stations DD-1 (closest to the CSO/storm drain) and KI-4 (closest to Kellogg Island) was depressed below the range. The number of polychaete and molluscan taxa were within the range except at Stations DD-1 and KI-4. The ITI was below the reference range at all stations. Swartz's Dominance Index (SDI),

Shannon-Wiener Diversity Index (H'), and Pielou's Eveness Index (J') were depressed at all stations except at DD-5 and KI-2.

7.4 Discussion

The benthic infauna communities at the CSO/storm drain stations show the typical gradients associated with chemical contamination and organic enrichment (Pearson and Rosenberg 1978; Swartz et al. 1985; Tetra Tech and PTI 1988a,b; Tetra Tech 1985; Gray 1989). The cumulative effects of CSO and storm drain discharges have led to distinct infaunal communities grading from impacted at the CSO/storm drain station nearest the outfall to relatively unimpacted at the station furthest from shore.

Stations along the gradient show decreasing numbers of endpoints being statistically different from reference at increasing distance from the CSO/storm drain outfall. Station DD-1 had all three SMS endpoints indicating an impact, while Station DD-3 and DD-5 each had only one (mollusk and polychaete abundance, respectively) indicating an impact. Increases in the abundance of certain taxa groups with increasing distance from the point source discharge were also seen. This can result from sediments being outside of areas of heavy deposition (and potential contaminant effects) and being in an area where the generally increased organic material acts as a food source for opportunistic species. These differences in community structure and function can be seen at the project stations due in some part to the CSO/storm drain and to the natural deposition of river sediment.

Station DD-1 was very strongly dominated by oligochaetes, and the polychaetes *Capitella capitata* and *Neanthes* sp. These three taxa are often called "indicators" of marine pollution (Reish 1955, 1957). These opportunistic species have long been known to be found in great abundances at outfalls (Filice 1954 in Reish 1957; Grassle and Grassle 1974; Word 1978; Word and Mearns 1979). These two taxa groups were

Table 7-9. Results of the Comparisons to Reference Value Ranges

Station	Percent Fines	Abundance				Number of Taxa				Calculated Indices			
		Total	Polychaete	Molluscs	Crustacea	Total	Polychaete	Molluscs	Crustacea	H’	J’	ITI	SDI
DD-1	0 to 20	259.7	78.7	0.3	3.3	10.7	3.3	0.3	3.0	0.5	0.5	1.0	2.0
DD-3	80 to 100	1,058.7	887.0	109.0	39.3	44.3	16.0	17.3	9.7	0.6	0.4	62.3	2.7
DD-5	80 to 100	800.3	320.3	349.7	92.3	47.0	19.3	15.7	7.7	1.1	0.7	63.3	6.0
KI-1	80 to 100	1,357.3	1,045.0	251.7	55.0	45.3	15.7	17.3	9.0	0.7	0.4	65.0	3.0
KI-2	80 to 100	555.3	244.0	222.0	65.7	54.0	24.7	14.7	6.3	1.2	0.7	63.0	8.7
KI-4	20 to 50	5,444.7	3,600.0	32.0	1,674.7	31.7	10.7	5.0	13.0	0.6	0.4	63.7	2.0

= Less than the reference range.

= Greater than the reference range.

H' = Shannon-Weiner Diversity Index

J' = Pielou's Evenness Index

ITI = Infaunal Trophic Index

SDI = Swartz's Dominance Index

typically identified as being present at sites with severe organic enrichment and chemical contamination because they are able to tolerate low levels of DO, high levels of BOD and hydrogen sulfides, which other species cannot tolerate. They are motile to some extent, feed on subsurface deposit material and have relatively short generation times (Grassle and Grassle 1974, Pearson and Rosenberg 1978, Word 1990).

The next shift in community structure and function occurred at Station DD-3. There were fewer statistical differences among endpoints and enhancements outnumbered depressions compared to the reference values. This station was very strongly dominated by the polychaete *Aphelochaeta* sp. This particular species builds thin walled tubes and feeds by selecting detrital material from the sediment surface and in some cases from the water column if currents are not too strong (Word 1978). It typically cannot handle large quantities of organic material as it is not particularly mobile. It is also considered to be an “indicator” of marine pollution. Note that the second most dominant taxa, although far lower in abundance, is *Capitella capitata* which is also an indicator of pollution. The amount of TOC at Station DD-3 was actually greater than at DD-1, however, the deposition of new organic material can apparently be assimilated by the benthic community. Larger abundances and numbers of species of crustaceans and mollusks were present at DD-3 and nine of the eleven most dominant taxa at this station did not occur at Station DD-1.

A third shift in community structure and function occurs at Station DD-5. The infauna at this station appears to be more affected by the physical disturbance of the habitat than by chemical contaminants or excess organic carbon. The life histories of the dominant species are such that they are somewhat protected from the transport of sediment loads down the river. The abundance of *Aphelochaeta* sp. at station DD-5 decreases greatly compared to station DD-3, while the abundance of the other dominant taxa increase dramatically compared to Station DD-3. Many of these species inhabit the upper few centimeters of the sediment surface and construct short tubes through which they pump and filter water to obtain food (i.e., *Axinopsida serricata* and *Psephedia lordi*). Other species, like *Macoma carlottensis*, lie roughly 5 centimeters below the sediment surface and extend palps to the surface where they select recently deposited organic material for ingestion (Myers 1977, Pearson 1971, Wooden 1978). *Euphilomedes carcharodonta* is an ostracod crustacean which was dominant at both DD-3 and DD-5, but was found in greater abundance at Station DD-5, most likely due to the decreased amount of organic carbon.

Station KI-1 (the furthest offshore of the Kellogg Island stations) was, like DD-3, strongly dominated by *Aphelochaeta* sp. This was most likely a result of the greater amount of organic carbon in the sediment at Station KI-1 compared to DD-5. But unlike DD-3, the remaining dominant taxa at KI-1 were more similar to those at DD-5. This also may be a function of the station’s location in relation to the navigation channel in the river. The larger amount of organic material may sustain a greater population of *Aphelochaeta* sp., yet the physical disturbance associated with the navigation channel, sediment type and water currents were also supportive of a surface detrital/deposit feeding community.

Station KI-2 was dominated by almost the same suite of organisms as Station KI-1. However the abundance of *Aphelochaeta* sp. was substantially lower at Station KI-2 than at KI-1; most of the other taxa had slightly lower abundances.

Station KI-4 was considerably different from the remaining Kellogg Island stations. It was dominated by a few species with very high abundances. Most of the dominant species were found at no other station, with the exception of oligochaetes and *Capitella capitata*. The dominant organism at Station KI-4 was the polychaete *Pygospio elegans*. This worm lives in tubes at or near the sediment surface and uses its palps to feed on small particulates in the clay size fraction (Fauchald and Jumars 1979). The tubes form large mat-like congregates of organisms each living individually. However, the congregation of these individual tubes causes the settling of detrital material upon which they feed. These mats also provide habitat for a large number of other species, which feed on settled detrital material. These species include large numbers of the amphipods in the genera *Corophium*, *Manayunkia*, *Grandidierella*, and *Eogammarus*, among others. The results of this analysis are similar to with those reported by Cordell et al. (1994, 1996) at other locations in the intertidal areas around Kellogg Island.

The ecological significance of benthic infaunal species is that they primarily serve as one of the lower tiers on the marine food web. Those species found at the CSO/storm drain (DD) and the outer Kellogg Island (KI) stations are fed upon by foraging juvenile flatfish such as English sole, starry flounder, and sand sole. Subtidal benthic infaunal organisms typically do not serve as a food supply for juvenile salmonids which tend to feed in shallow water areas and among marine vegetation beds. The primary food for juvenile salmonids include harpacticoid copepods, amphipods, and some surface dwelling polychaetes such as those identified at Station KI-4.

7.5 Summary

The primary objective of this benthic assessment was to determine whether nearfield effects from a combined sewer overflow (CSO/storm drain) could be identified and, if possible, to determine the extent of the effect. To reach the objective a number of analyses and comparisons were conducted.

The sediment chemistry and benthic infauna evaluation showed that Station DD-1 was severely impacted from the CSO/storm drain. Station DD-3 was slightly affected by the CSO/storm drain primarily by the large amount of organic carbon. Station DD-5 was primarily affected by the physical characteristics of the river. Station DD-4, which was archived, would have been interesting to evaluate because it was located mid-way between DD-5 and DD-3 and may not have been influenced by the physical disturbances associated with an urban-channeled river.

The benthic data from stations off of the CSO/storm drain were compared to data from an in-river reference site at Kellogg Island. This was accomplished by pairing CSO/storm drain stations with Kellogg Island stations having similar grain sizes, water depths, and distances from shore. Using this approach the station closest to the CSO/storm drain was identified as being severely impacted with the majority of benthic infaunal endpoints being statistically different from the reference station. The center station in the transect (DD-3) was apparently affected by excess organic carbon because mollusks, which are sensitive to burial from excess organic material, were depressed. The outermost station on the transect (DD-5) appears to be more affected by the physical characteristics associated with the channeling of the Duwamish River than by the CSO/storm drain.

The results of the comparison to the Puget Sound Reference Ranges also identified Station DD-1 as being impacted. At Station DD-3, three of the four endpoints that were below the lower reference range were also identified by t-tests as being statistically different from its reference station. The two methods of analysis did not track each other at Station DD-5. The t-test analysis identified three endpoints as being statistically different from reference, while the reference range analysis identified only one of the three. The comparison of the three Kellogg Island stations to the reference ranges found some endpoints to be within the reference range as well as some endpoints to be enhanced or depressed. The depressions tended to be in the calculated indices.

The data clearly shows the presence of different benthic communities at the CSO/storm drain and Kellogg Island. It is likely that the benthic communities in the vicinity of the CSO/storm drain would be quite similar to that at Kellogg Island in the absence of the CSO/storm drain. The differences in physical characteristics between these two areas are likely due to the CSO/storm drain itself. Significant differences in benthic community structure and function on opposite sides of the Duwamish River are not otherwise anticipated given the hydrography of the river.

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SUBAPPENDIX A

Brandon Street Bioassay
Raw Data and Statistical Analyses

SUBAPPENDIX B

Benthic Community Survey Field Sampling Forms

SUBAPPENDIX C

Raw Benthic Infaunal Data

SUBAPPENDIX D

Sediment Chemistry Data for Samples Colocated with the Benthic Community Survey Sampling Stations