PORTLAND HARBOR RI/FS

REMEDIAL INVESTIGATION REPORT

DRAFT FINAL

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Prepared for
The Lower Willamette Group

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Figure 7.3-44. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized 4,4’-DDE

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Figure 7.3-54. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Total DDx

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Figure 7.3-64. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Total Chlordanes

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Figure 7.3-66. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Aldrin

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Figure 7.3-70. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Endrin

Figure 7.3-71. Background-Upriver Bedded Sediment Outlier Analysis, Endrin ketone Dry Weight

Figure 7.3-72. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Endrin ketone

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Figure 7.3-74. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Heptachlor

Figure 7.3-75. Background-Upriver Bedded Sediment Outlier Analysis, Heptachlor Epoxide Dry Weight

Figure 7.3-76. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Heptachlor Epoxide

Figure 7.3-77. Background-Upriver Bedded Sediment Outlier Analysis, MCPP Dry Weight

Figure 7.3-78. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized MCPP

Figure 7.3-79. Background-Upriver Bedded Sediment Outlier Analysis, Total PAHs Dry Weight

Figure 7.3-80. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Total PAHs

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Figure 7.3-82. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Total cPAH BaPEq

Figure 7.3-83. Background-Upriver Bedded Sediment Outlier Analysis, Total LPAHs Dry Weight

Figure 7.3-84. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Total LPAHs

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Figure 7.3-86. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Benzo(a)anthracene

Figure 7.3-87. Background-Upriver Bedded Sediment Outlier Analysis, Benzo(a)pyrene Dry Weight

Figure 7.3-88. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Benzo(a)pyrene

Figure 7.3-89. Background-Upriver Bedded Sediment Outlier Analysis, Benzo(b)fluoranthene Dry Weight

Figure 7.3-90. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Benzo(b)fluoranthene

Figure 7.3-91. Background-Upriver Bedded Sediment Outlier Analysis, Benzo(k)fluoranthene Dry Weight

Figure 7.3-92. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Benzo(k)fluoranthene

Figure 7.3-93. Background-Upriver Bedded Sediment Outlier Analysis, Chrysene Dry Weight

Figure 7.3-94. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Chrysene

Figure 7.3-95. Background-Upriver Bedded Sediment Outlier Analysis, Dibenzo(a,h)anthracene Dry Weight

Figure 7.3-96. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Dibenzo(a,h)anthracene

Figure 7.3-97. Background-Upriver Bedded Sediment Outlier Analysis, Indeno(1,2,3-cd)pyrene Dry Weight

Figure 7.3-98. Background-Upriver Bedded Sediment Outlier Analysis, OC-normalized Indeno(1,2,3-cd)pyrene

Figure 7.3-99. Background-Upriver Bedded Sediment Outlier Analysis, BEHP Dry Weight

Figure 7.3-100. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized BEHP

Figure 7.3-101. Background-Upriver Bedded Sediment Outlier Analysis, Diethyl phthalate Dry Weight

Figure 7.3-102. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Diethyl phthalate

Figure 7.3-103. Background-Upriver Bedded Sediment Outlier Analysis, Benzyl Alcohol Dry Weight

Figure 7.3-104. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Benzyl Alcohol

Figure 7.3-105. Background-Upriver Bedded Sediment Outlier Analysis, Carbazole Dry Weight

Figure 7.3-106. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Carbazole

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Figure 7.3-108. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Hexachlorobenzene

Figure 7.3-109. Background-Upriver Bedded Sediment Outlier Analysis, Pentachlorophenol Dry Weight

Figure 7.3-110. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Pentachlorophenol

Figure 7.3-111. Background-Upriver Bedded Sediment Outlier Analysis, 4-Methylphenol Dry Weight

Figure 7.3-112. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized 4-Methylphenol

Figure 7.3-113. Background-Upriver Bedded Sediment Outlier Analysis, Phenol Dry Weight

Figure 7.3-114. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Phenol

Figure 7.3-115. Background-Upriver Bedded Sediment Outlier Analysis, Aluminum Dry Weight

Figure 7.3-116. Background-Upriver Bedded Sediment Outlier Analysis, Antimony Dry Weight

Figure 7.3-117. Background-Upriver Bedded Sediment Outlier Analysis, Arsenic Dry Weight

Figure 7.3-118. Background-Upriver Bedded Sediment Outlier Analysis, Cadmium Dry Weight

Figure 7.3-119. Background-Upriver Bedded Sediment Outlier Analysis, Chromium Dry Weight

Figure 7.3-120. Background-Upriver Bedded Sediment Outlier Analysis, Copper Dry Weight

Figure 7.3-121. Background-Upriver Bedded Sediment Outlier Analysis, Lead Dry Weight

Figure 7.3-122. Background-Upriver Bedded Sediment Outlier Analysis, Mercury Dry Weight

Figure 7.3-123. Background-Upriver Bedded Sediment Outlier Analysis, Nickel Dry Weight

Figure 7.3-124. Background-Upriver Bedded Sediment Outlier Analysis, Selenium Dry Weight

Figure 7.3-125. Background-Upriver Bedded Sediment Outlier Analysis, Silver Dry Weight

Figure 7.3-126. Background-Upriver Bedded Sediment Outlier Analysis, Zinc Dry Weight

Figure 7.3-127. Background-Upriver Bedded Sediment Outlier Analysis, Tributyltin Ion Dry Weight

Figure 7.3-128. Background-Upriver Bedded Sediment Outlier Analysis, OC-Normalized Tributyltin Ion

Figure 7.4-1. Background-Upriver Surface Water Analysis, PCBs: PCB 077, PCB 081, PCB 105

Figure 7.4-2. Background-Upriver Surface Water Analysis, PCBs: PCB 118, PCB 126, PCB 156 & 157

Figure 7.4-3. Background-Upriver Surface Water Analysis, PCBs: PCB 169, Total PCB Congeners, PCB TEQ (ND=0)

Figure 7.4-4. Background-Upriver Surface Water Analysis, PCDD/Fs: 1,2,3,4,7,8-HCDF; 1,2,3,6,7,8 HCDD; and 1,2,3,7,8 PCDD

Figure 7.4-5. Background-Upriver Surface Water Analysis, PCDD/Fs: 2,3,4,7,8 PCDF; 2,3,7,8 TCDF; 2,3,7,8 TCDD

Figure 7.4-6. Background-Upriver Surface Water Analysis, PCDD/Fs: TCDD TEQ (ND=0), Total TEQ

Figure 7.4-7. Background-Upriver Surface Water Analysis, Pesticides: 4,4'-DDD; 4,4'-DDE, 4,4’-DDT

Figure 7.4-8. Background-Upriver Surface Water Analysis, Pesticides: Sum DDD; Sum DDE; Sum DDT

Figure 7.4-9. Background-Upriver Surface Water Analysis, Pesticides: Total DDx; δ-Hexachlorocyclohexane; γ-Hexachlorocyclohexane

Figure 7.4-10. Background-Upriver Surface Water Analysis, Pesticides: Total Chlordanes, Aldrin, Dieldrin

Figure 7.4-11. Background-Upriver Surface Water Analysis, Pesticides: Endrin, Endrin Ketone, Heptachlor

Figure 7.4-12. Background-Upriver Surface Water Analysis, Pesticides: Heptachlor Epoxide; Herbicides: MCPP

Figure 7.4-13. Background-Upriver Surface Water Analysis, PAHs: Total PAHs; CPAH BaPEq; Total LPAHs

Figure 7.4-14. Background-Upriver Surface Water-Analysis, PAHs: Benzo(a)anthracene; Benzo(a)pyrene; Benzo(b)fluoranthene

Figure 7.4-15. Background-Upriver Surface Water-Analysis, PAHs: Benzo(j+k)fluoranthene, Benzo(k)fluoranthene, Chrysene

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Panel 10.2-6C. Total Chlordanes – Biota

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Panel 10.2-7B. Aldrin – Subsurface

Panel 10.2-7C. Aldrin – Biota

Panel 10.2-8A. Dieldrin – Surface Sediment and Abiotic Media

Panel 10.2-8B. Dieldrin – Subsurface

Panel 10.2-8C. Dieldrin – Biota

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Panel 10.2-9B. Arsenic – Subsurface

Panel 10.2-9C. Arsenic – Biota

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Panel 10.2-13B. Tributyltin Ion – Subsurface

Panel 10.2-13C. Tributyltin Ion – Biota

LIST OF ACRONYMS

2-D	two dimensional
3-D	three dimensional
95 UCL	95th percentile upper confidence limit
95 UPL	95th percentile upper prediction limit
ADCP	acoustic doppler current profiler
AFSO	Amended Final Stipulated Order
AOC	Administrative Order on Consent
AOPC	area of potential concern
ARAR	Applicable or Relevant and Appropriate Requirement
ASFO	Amended Stipulation and Final Order
ATSDR	Agency for Toxic Substances and Disease Registry
BAP	benzo(a)pyrene
BaPEq	benzo(a)pyrene equivalent
BEHP	bis-2(ethylhexyl) phthalate
BERA	baseline ecological risk assessment
BES	Bureau of Environmental Services
BFPPA	Bona Fide Prospective Purchaser Agreement
bgs	below ground surface
BHHRA	baseline human health risk assessment
bml	below mudline
BMP	best management practice
BNSF	Burlington Northern Santa Fe Railway Company
BOD	biological oxygen demand
BSAF	biota-sediment accumulation factor
BSAR	biota-sediment accumulation regression
BTEX	benzene, toluene, ethylbenzene, and total xylene
BTV	background threshold values
CBWTP	Columbia Boulevard Wastewater Treatment Plant
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act
cfs	cubic feet per second
CGF	coarse-grained flood deposits
COC	contaminant of concern[1]
COI	contaminant of interest[2]
COPC	contaminant of potential concern[3]
cPAH	carcinogenic polycyclic aromatic hydrocarbon

[1] Prior deliverables and some of the tables and figures attached to this document may use the term "chemical of concern," which has the same meaning as "contaminant of concern" and refers to "contaminants" as defined in 42 USC 9601(33).

[2] Prior deliverables and some of the tables and figures attached to this document may use the term "chemical of interest," which has the same meaning as "contaminant of interest" and refers to "contaminants" as defined in 42 USC 9601(33).

[3] Prior deliverables and some of the tables and figures attached to this document may use the term "chemical of potential concern," which has the same meaning as "contaminant of potential concern" and refers to "contaminants" as defined in 42 USC 9601(33).

CPD	Commission of Public Docks
CRAG	Columbia Region Association of Governments
CRBG	Columbia River Basalt Group
CRD	Columbia River datum
CRITFC	Columbia River Inter-Tribal Fish Commission
C_s	concentration of the chemical associated with solids
CSM	conceptual site model
CSO	combined sewer overflow
CT	central tendency
C_w	aqueous concentration of the chemical
CWA	Clean Water Act
DCA	dichloroethane
DCB	dichlorobenzene
DCE	dichloroethene
DDD	dichloro-diphenyl-dichloroethane
DDE	dichloro-diphenyl-dichloroethene
DDT	dichloro-diphenyl-trichloroethane
DDx	2,4′ and 4,4′-DDD, -DDE, -DDT
DEA	David Evans and Associates, Inc.
decaCB	decachlorobiphenyl
DEHP	diethylhexyl phthalate
DEQ	Oregon Department of Environmental Quality
diCB	dichlorobiphenyl
DMR	Discharge Monitoring Report
DO	dissolved oxygen
DOC	dissolved organic carbon
DRH	diesel-range hydrocarbons
DSL	Oregon Department of State Lands
ECSI	Environmental Cleanup Site Information
EDB	ethylene dibromide
EDC	ethylene dichloride
EDI	equal discharge increment
EFC	Emergency Fleet Corporation
EFDC	Environment Fluid Dynamics Code
EOSM	Evraz Oregon Steel Mills
EPA	U.S. Environmental Protection Agency
EPC	exposure point concentration
ERA	ecological risk assessment
ERIS	Emergency Response Information System
FFA	fill, fine-grained facies of flood deposits, and recent alluvium
f_{oc}	fraction of organic carbon
FOIA	Freedom of Information Act
FS	feasibility study
FSP	field sampling plan
FSR	field sampling report

GIS	geographic information systems
GLISP	Guild's Lake Industrial Sanctuary Plan
GPS	global positioning system
GRH	gasoline-range hydrocarbons
GWPA	groundwater pathway assessment
HCH	hexachlorocyclohexane
heptaCB	heptachlorobiphenyl
hexaCB	hexachlorobiphenyl
HI	hazard index
HPAH	high-molecular-weight polycyclic aromatic hydrocarbon
HpCDD	heptachlorodibenzo-p-dioxin
HpCDF	heptachlorodibenzofuran
HQ	hazard quotient
HRGC	high resolution gas chromatography
HRMS	high resolution mass spectrometry
HST	hydrodynamic and sediment transport
HxCDD	hexachlorodibenzo-p-dioxin
HxCDF	hexachlorodibenzofuran
IC	indicator chemical
IRAF	infant risk adjustment factor
ISA	initial study area
JSCS	joint source control strategy
K_d	solid/water partitioning coefficient
K_{oc}	organic carbon partitioning coefficient
K_{ow}	octanol-water partitioning coefficient
LISST	laser in situ scattering and transmissometer
LOAEL	lowest-observed-adverse-effect level
LOE	line of evidence
LPAH	low-molecular-weight polycyclic aromatic hydrocarbon
LRMS	low resolution mass spectrometry
LWG	Lower Willamette Group
LWR	lower Willamette River
M&B	McCormick & Baxter Creosote Co.
MCB	monochlorobenzene
MDL	method detection limit
MFA	Maul Foster & Alongi, Inc.
MGP	manufactured gas production
mgy	million gallons per year
MHWM	mean high water mark
MNR	monitored natural recovery
MOA	memorandum of agreement
monoCB	monochlorobiphenyl
MOU	memorandum of understanding
MRL	method reporting limit
MS4	municipal separated storm sewer system

MSL	mean sea level
MT	metric ton
MTBE	methyl tert-butyl ether
mya	million years ago
N/m^2	Newton per square meter
NAPL	non-aqueous phase liquid
NAVD88	North American Vertical Datum of 1988
NB/NS	near bottom/near surface
NCP	National Contingency Plan
NGVD29/47	National Geodetic Vertical Datum of 1929 through the Pacific Northwest Supplemental Adjustment of 1947
NJADN	New Jersey Atmospheric Deposition Network
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no-observed-adverse-effect level
nonaCB	nonachlorobiphenyl
NPDES	National Pollutant Discharge Elimination System
NPL	National Priorities List
NRC	National Response Center
NRIU	natural resource inventory update
OC normalized	organic carbon normalized
OCDD	octachlorodibenzo-p-dioxin
OCDF	octachlorodibenzofuran
octaCB	octachlorinated biphenyl
ODFW	Oregon Department of Fish and Wildlife
ODHS	Oregon Department of Human Services
ODOT	Oregon Department of Transportation
OHW	ordinary high water
OHWM	ordinary high water mark
OLW	ordinary low water
OSSA	Oregon State Sanitary Service Authority
OSWER	EPA Office of Solid Waste and Emergency Response
PAH	polycyclic aromatic hydrocarbon
PBDE	polybrominated diphenyl ether
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-p-dioxin
PCDD/F	polychlorinated dibenzo-p-dioxin/furan
PCDF	polychlorinated dibenzofuran
PCP	pentachlorophenol
PeCDD	pentachlorodibenzo-p-dioxin
PeCDF	pentachlorodibenzofuran
PEF	potency equivalency factor
pentaCB	pentachlorobiphenyl
PGE	Portland General Electric
POC	particulate organic carbon
ppm	parts per million

PRD	Portland River Datum
PRE	preliminary risk evaluation
PRG	Preliminary Remediation Goal
PRP	potentially responsible party
PTRR	Portland Terminal Railroad
PVC	polyvinyl chloride
Q-Q	quantile-quantile
QA/QC	quality assurance and quality control
QAPP	quality assurance project plan
RA	risk assessment
RAL	remedial action level
RfD	reference dose
RG	remediation goal
RI	remedial investigation
RI/FS	remedial investigation and feasibility study
RM	river mile
RME	reasonable maximum exposure
ROD	Record of Decision
RRH	residual-range hydrocarbons
SAP	sampling and analysis plan
SCRA	site characterization and risk assessment
SCSR	site characterization summary report
SFO	Stipulation and Final Order
SHA	special habitat area
SIC	Standard Industrial Classifications
SLERA	screening-level ecological risk assessment
SLLI	StarLink Logistics
SMA	sediment management area
SOW	statement of work
SPI	Sediment Profile Imagery
SQG	sediment quality guideline
SQV	sediment quality value
SRM	Sandy River Mudstone
SSO	sanitary sewer overflow
STA	Sediment Trend Analysis®
SVOC	semivolatile organic compound
SWAC	surface weighted average concentrations
TBT	tributyltin ion
TCA	trichloroethane
TCDD	tetrachlorodibenzo-p-dioxin
TCDF	tetrachlorodibenzofurans
TCE	trichloroethene
T_{cr}	critical shear stresses
TEF	toxicity equivalency factor
TEQ	toxic equivalent concentration

tetraCB	tetrachlorobiphenyl
TMDL	total maximum daily load
TOC	total organic carbon
TPH	total petroleum hydrocarbons
triCB	trichlorobiphenyl
TRV	toxicity reference value
TSCA	Toxic Substance Control Act
TSS	total suspended solids
TZW	transition zone water
UCL	upper confidence limit
UPL	upper prediction limit
UPRR	Union Pacific Railroad
USACE	U.S. Army Corps of Engineers
USCG	U.S. Coast Guard
USGS	U.S. Geological Survey
UTL	upper tolerance limit
VI	vertically integrated
VI-EMW	vertically integrated: east-middle-west
VOC	volatile organic compound
WHO	World Health Organization
WISCO	Willamette Iron and Steel Company
WOE	weight of evidence
WPCL	Water Pollution Control Laboratory

1.0 INTRODUCTION

This remedial investigation report evaluates the environmental data collected and compiled by the Lower Willamette Group (LWG) since the inception of the Portland Harbor Remedial Investigation and Feasibility Study (RI/FS) in 2001. Portland Harbor, which encompasses the downstream portion of the Willamette River in Portland, Oregon, was designated as a Superfund site in 2000 under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA).

The LWG is performing the RI/FS for the Portland Harbor Superfund Site (Site) pursuant to a U.S. Environmental Protection Agency (EPA) Administrative Settlement Agreement and Order on Consent for Remedial Investigation/Feasibility Study (AOC; EPA 2001a, 2003b, 2006a). As provided in the Statement of Work (SOW) to the AOC, the objectives of the Portland Harbor RI/FS are as follows:

- Investigate the nature and extent of contamination for the Study Area[1]
- Identify sources of contamination that contribute, or have contributed, to unacceptable risk in Study Area[2]
- Assess potential risk to human health and the environment
- Develop and evaluate potential remedial alternatives to reduce risks to acceptable levels
- Recommend a preferred alternative for cleanup.

This RI Report addresses the first three objectives and includes the human health and ecological baseline risk assessments as appendices. The final two objectives will be addressed in the FS. The Final RI Report will capture any preliminary or incomplete information from the risk assessments that has changed pursuant to finalization of these documents.

Portland Harbor has been the focus of numerous environmental investigations completed by the LWG and various other governmental and private entities. Major LWG data collection efforts occurred during three sampling rounds to characterize the physical system of the lower Willamette River (LWR) and to assess the nature and extent of contamination in sediment, surface water, transition zone water (TZW)[3], stormwater, and biota. Media-specific investigations performed in Round 1 (summer 2002 to spring 2004) and Round 2 (fall 2004 to spring 2006) were previously documented in the *Comprehensive Round 2 Site Characterization Summary and Data*

[1] Defined as the in-water portion below or equal to +13 ft North American Vertical Datum (NAVD) from river mile 1.9 to 11.8.

[2] Upland source control efforts, including site-specific upland source control studies and implementation of source control measures, are performed under the oversight of the Oregon Department of Environmental Quality and are not within the scope of the AOC and SOW for the in-water portion of the Site.

[3] Transition zone water is the interval where both groundwater and surface water comprise some percentage of the water occupying pore space in the sediments.

Gaps Analysis Report (Round 2 Report; Integral et al. 2007), which was submitted to the EPA on February 21, 2007. Round 3 sampling activities took place during multiple field efforts in the winter of 2006, the summer/fall/winter of 2007, and February 2008. Data collected during these efforts supplemented the data collected in previous sampling rounds and closed data gaps identified by EPA and LWG.

The content and organization of this RI report adhere to CERCLA's *Guidance Document for Conducting Remedial Investigations and Feasibility Studies under CERCLA, Interim Final* (EPA 1988). The required content of this RI Report is specified in Section 7.8.2 of the SOW:

> This report shall summarize results of field activities to characterize the Site, sources of contamination, nature and extent of contamination, and the fate and transport of contaminants. Respondents will refer to the RI/FS Guidance for an outline of the report format and contents. Following comment by EPA, Respondents will prepare a final RI Report that satisfactorily addresses EPA comments.

In accordance with these requirements, this report assembles data collected by the LWG and others, characterizes the physical characteristics and nature and extent of contamination in the Study Area (described below in Section 1.2) based on those data, identifies sources of contaminants to the Study Area, provides an analysis of the fate and transport of contaminants, discusses background contaminant concentrations, presents the baseline human health and ecological risk assessments, and provides a revised conceptual site model (CSM). The identification of areas of potential concern (AOPCs) and preliminary remediation goals (PRGs) for those AOPCs will be addressed during the FS process, and are not discussed in this RI Report.

The revised CSM expands upon the preliminary CSM previously presented in the Programmatic Work Plan (Integral et al. 2004), and updated in the CSM Update (Integral and GSI 2005a,b,c) and the Round 2 Report. Unlike the AOPC-specific CSMs presented in Section 11 of the Round 2 Report, the scope of the revised CSM presented in this RI Report is site-wide and includes a cross-media conceptual understanding of sources, contaminant distribution, loading, fate and transport, and exposure pathways for human and ecological receptors for select contaminants of concern (COCs).[4]

1.1 RI/FS OVERVIEW AND OBJECTIVES

The Programmatic Work Plan presented a generalized road map of the RI/FS process for the Portland Harbor Superfund Site. The process is multifaceted and iterative, with specific sampling events, technical memoranda, and decision points made during various phases that inform the RI and direct future FS work. The RI and FS are conducted concurrently, and data collected during the RI phase are used to develop and

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evaluate remedial alternatives in the FS phase. The ultimate goal of the RI is to collect sufficient data to adequately characterize the Site so that EPA can select a remedy that is protective of human and ecological receptors. The ultimate goal of the FS is to ensure that appropriate remedial alternatives are developed and evaluated in accordance with CERCLA guidance to allow the selection of the most efficient remedies for cleanup that balance effectiveness, protectiveness, cost, compliance with Applicable or Relevant and Appropriate Requirements (ARARs), and public acceptance.

The RI was conducted consistent with EPA (1988) guidance (page 1-3, 2nd paragraph):

> The objective of the RI/FS process is not the unobtainable goal of removing *all* uncertainty, but rather to gather information sufficient to support an informed risk management decision regarding which remedy appears to be most appropriate for a given site. The appropriate level of analysis to meet this objective can only be reached through constant strategic thinking and careful planning concerning the essential data needed to reach a remedy selection decision. As hypotheses are tested and either rejected or confirmed, adjustments or choices as to the appropriate course for further investigations and analyses are required. These choices, like the remedy selection itself, involve the balancing of a wide variety of factors and the exercise of best professional judgment.

Specific objectives of the RI, which conform to the CERCLA RI/FS guidance document (EPA 1988) and the SOW, include the following:

- Investigate and describe the Site's physical setting, including land use, hydrology, hydrogeology, river and sediment dynamics, aquatic and wildlife habitat, and human use
- Identify major sources of historical and ongoing chemical contamination and the status of source control activities
- Describe the nature and extent of indicator chemicals (ICs), vertically and laterally, that may pose risk for both harbor-wide and localized areas of contamination[5]
- Evaluate the loading, fate, and transport of ICs using available empirical information and model results
- Define the concentrations and statistical characteristics of ICs in appropriate upstream reference locations (i.e., background) for use in risk characterization, PRG development, and remedial alternatives evaluation during the FS

[5] Although chemicals in addition to those on the IC list are present within the Study Area and may pose risk to human health and the environment, the IC list was generated, in consultation with EPA (as documented in Appendix A5, Attachments A4, A7, A9, and A11), based on the preliminary COCs that emerged from the Round 2 risk screening and preliminary risk evaluation process and consideration of the following non-risk-based factors: (1) detection frequency; (2) suitability for cross-media comparisons in this RI Report; (3) representativeness of a given IC for a suite of related chemicals; and (4) specific EPA requests to include certain additional chemicals as ICs. See Section 5.0 for additional discussion.

- Assess human and ecological risk under baseline conditions and identify COCs that may pose unacceptable risk
- Develop a revised site-wide CSM for select ICs that describes the cross-media understanding of contaminant distribution, sources, loading, fate, transport, and exposure pathways for human and ecological receptors at the Site.

Ultimately, the primary information needed to make risk management decisions includes the following:

1. Identification of the receptors, exposure scenarios, and contaminants that are associated with potentially unacceptable risk from exposure to in-water media
2. For the above, identification of the exposure pathways (including media) and, to the extent practicable, locations that contribute most to a finding of unacceptable risk
3. For the receptors/pathways/contaminants identified in items "1" and "2" above, determination of whether the most important sources of contaminants to the river are ongoing or historical (not current releases), so that risk management actions can be taken to reduce exposures.

For in-water sources, actions may include active cleanup, natural recovery, or other management options for contaminated sediments in the river. The RI/FS must also identify potential sources that are not in the river, or are in the river upstream of the Study Area, but that are important ongoing sources of contamination to sediments and surface water, and therefore, may require management to address unacceptable risks to humans and biota associated with the Study Area. Management of these sources will be overseen by the Oregon Department of Environmental Quality (DEQ).

1.2 AREA OF STUDY

In accordance with the AOC, the RI/FS initially focused on the stretch of the LWR from river mile (RM) 3.5 to 9.2 and adjacent areas logically associated with an evaluation of the in-water portion of this stretch of the river. The SOW and the Programmatic Work Plan refer to that initial study area as the "ISA." The boundaries of the ISA were based on the results of a site investigation undertaken by EPA and DEQ in 1997 (described further in Section 1.3).

During development of the Programmatic Work Plan, EPA required the LWG to broaden the investigation upstream and downstream to include areas of the river extending from approximately RM 1.9 to 11; this expanded area was termed the "Study Area." Based upon its review of the Round 2 Report, EPA further expanded the Study

Area to include RM 1.9 to 11.8[6] (Map 1.2-1). The ISA and the subsequent Study Areas do not define the Portland Harbor Superfund Site, the boundaries of which will be determined by EPA upon issuance of a Record of Decision (ROD).

1.3 SITE BACKGROUND

The Willamette River is the 12th largest river in the United States and is one of 14 American Heritage Rivers in the country. During its 99-mile course, which ends by emptying into the Columbia River, it drains 11.7 percent of the state of Oregon. Today, the Willamette River is noticeably different from the river used by the Native Americans in the 18th century. Historically, the Willamette was wider, had more sand bars and shoals, and fluctuated greatly in volume. In contrast, the main river now has been redirected and channelized, several lakes and wetlands in the lower floodplain have been filled, and agricultural lands converted to urban or industrial areas. The end result is a river that is deeper and narrower than it was historically with higher banks that prevent the river from expanding during high-flow events. Further, the installation of a series of dams moderated fluctuations of flow. Little, if any, original shoreline or river bottom exists that has not been modified by the above actions, or as a result of them. Some riverbank areas and adjacent parcels have been abandoned and allowed to revegetate, and beaches have formed along some modified shorelines due to relatively natural processes.

From as early as the 1840s, the development of Portland has been connected to the development of the Willamette. Portland Harbor is a heavily industrialized reach of the LWR, located immediately downstream of downtown Portland and extending almost to the confluence with the Columbia River. Since the late 1800s, the harbor has been extensively modified by wetland draining, channelization, and dredging for creation and maintenance of the navigation channel and ship berthing areas. The harbor has been the site of numerous manufacturing, shipbuilding, petroleum storage and distribution, metals salvaging, and electrical power generation activities.

In 1891, the Oregon State Legislature created the Port of Portland. By 1930, following a period of railroad and riverfront development, shipping tonnage for the Port increased to 4.1 million tons. The Port of Portland is now the largest wheat exporting port in the country (Port of Portland 2011). Cargo from more than 40 U.S. states passes through Portland as part of the approximately $15 billion in goods that travel the Columbia River system. As Oregon's major port and population center, the LWR sees a great variety of uses. For example, shipping, industrial, fishing, recreational, natural resource, and other interest groups all use the LWR.

[6] As part of the RI, the LWG also sampled contiguous areas adjacent to the Study Area downstream to RM 0.8, in the upstream portion of the Multnomah Channel, and upstream to RM 12.2. This sampling was conducted to support the Site boundary definition and assess potential contaminant migration in these adjacent areas.

In the early 1900s, rivers in the United States were generally used as open sewers, which was also true for the Willamette (Carter 2006). The growing city's untreated sewage, as well as process water from a variety of industries, including slaughterhouses, chemical plants, electroplaters, paper mills, and food processors, was discharged directly into the river. Early 1920s studies of the Willamette's water quality found the river to be extensively polluted and described the river as "intolerable" and "ugly and filthy."

Today, contamination enters the river (i.e., Study Area, which consists of surface water and bedded sediment) through several pathways, including stormwater, permitted industrial discharges, atmospheric deposition, riverbank erosion, groundwater, combined sewer overflows (CSOs), and incidental releases within the Study Area, and in surface water and sediment inflows from upstream.

Once in the surface water, most contaminants bind to fine-grained sediment particles. The fate and transport of these particles in the river (i.e., where particles settle out, and whether they are permanently deposited or are episodically resuspended and transported by natural or anthropogenic disturbance factors) determines the degree to which these contaminants affect ecological and human receptors and which sediment areas pose risk over time. Understanding these internal fate and transport processes is a major focus of this RI/FS.

Public and private outfalls are located on both shores of the river. These outfalls have historically discharged stormwater, municipal waste, and industrial wastewater to the harbor from numerous drainage basins that have a variety of land uses and facilities. In addition to areas adjacent to the harbor, agricultural, industrial, transportation, and residential land uses in the Willamette Basin upstream of the harbor historically and currently discharge municipal, agricultural, and industrial wastewater and stormwater directly to the Willamette River and indirectly discharge through overland, overwater, and groundwater pathways, thereby contributing to chemical contamination of sediments in the Study Area. Although private industries and municipalities within the river watershed began installing waste control systems beginning in the 1950s, the legacy of past waste management practices remains in the river bottom sediments.

In March 1997, DEQ and EPA initiated a joint study of shallow, nearshore river sediment contamination in the LWR from approximately RM 3.5 to 9.5. Sediments containing metals, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), chlorinated pesticides, and dioxins were found throughout the harbor area (Weston 1998). Based on the concentrations of these contaminants, EPA determined in December 2000 that Portland Harbor qualified for placement on the National Priorities List (NPL) under CERCLA. In the listing, EPA and DEQ are the lead agencies for the in-water and upland portions, respectively, of the Portland Harbor Superfund Site. Both agencies coordinate their efforts through a Memorandum of Understanding (MOU) that was signed in 2001 by government stakeholders, including six tribal governments (the Confederated Tribes and Bands of the Yakama Indian

Nation, the Confederated Tribes of Grande Ronde, the Confederated Tribes of Siletz Indians of Oregon, the Confederated Tribes of the Warm Springs Reservation of Oregon, the Confederated Tribes of the Umatilla Indian Reservation, and the Nez Perce tribes) and several state and federal natural resource trustees (National Oceanic and Atmospheric Administration [NOAA], Oregon Department of Fish and Wildlife [ODFW], U.S. Fish and Wildlife, and U.S. Department of Interior).

In September 2001, EPA and the LWG entered into the AOC to complete an RI/FS of the Portland Harbor Superfund Site. The LWG includes private property owners along the Willamette River, the Port of Portland, and the City of Portland and represents a small subset of the potentially responsible parties (PRPs) identified by EPA. The final Programmatic Work Plan was approved by EPA in June 2004. The Programmatic Work Plan provides an outline and schedule for the sampling activities performed to support the RI/FS. All field sampling activities are documented in a series of field sampling plans (FSPs), quality assurance project plans (QAPPs), health and safety plans, field sampling reports (FSRs or cruise reports), and site characterization summary reports (SCSRs). The Programmatic Work Plan also stipulated the development of technical memoranda to address certain elements of the RI, such as modeling efforts and the development of the baseline risk assessments. The entire RI/FS process has been dynamic and iterative, with the evaluation of new site data resulting in modifications to the original RI approach specified in the Work Plan.

Cleanup activities are currently underway in Portland Harbor. EPA evaluated 32 upland sites in 1982, several of which are current sources to the river, and deferred most of them to DEQ for cleanup. DEQ is presently working with upland property owners to identify and control upland sources of contamination that may be affecting river sediments through such pathways as overland runoff, bank erosion, stormwater discharge, or groundwater seepage. A total of 84 upland sites are in various phases of cleanup, ranging from agreement negotiation to source control evaluation and implementation (DEQ 2009b). Cleanup at two upland NPL sites within or adjacent to Portland Harbor (Gould and McCormick & Baxter Creosote Co. [M&B]) has been implemented, including some in-water work at M&B.

DEQ is also collaborating with the City of Portland's Bureau of Environmental Services (BES) to identify, investigate, and control contaminant discharges to shared City stormwater conveyance lines. As part of the City's 20-year combined-sewer overflows abatement program, to be completed by 2011, all or a portion of the stormwater discharging through 15 City of Portland outfalls is being diverted to the wastewater treatment plant.

Early in-water removal actions have been or are being performed by three upland property owners under separate EPA orders: Port of Portland at Terminal 4 (RM 4.5), NW Natural adjacent to the former Gasco site (RM 6.2), and Arkema adjacent to its former plant (RM 7.2). These actions will probably precede any remediation that

occurs as a result of the Portland Harbor RI/FS and will be coordinated with cleanup actions ordered by EPA in the Portland Harbor ROD(s).

EPA is also working with the University of Portland to conduct cleanup activities on the Triangle Park property (RM 7.4). EPA is overseeing the investigation and eventual upland cleanup.

1.4 REPORT ORGANIZATION

The remaining sections of this report include the following information:

- **Section 2. Environmental Data Sets.** This section focuses on data quality reviews that were performed to evaluate the quality of LWG and non-LWG data and to determine their usability for various purposes in the RI/FS, Baseline Human Health Risk Assessment (BHHRA), and Baseline Ecological Risk Assessment (BERA). It summarizes the LWG investigative activities that have occurred since the Portland Harbor RI/FS began in 2001, including sediment, surface water, TZW, sediment trap, stormwater, and biota sampling, as well as physical characterizations. Historical and concurrent investigations performed by others and used in the risk evaluations are also described.

  The data set introduced in Section 2 represents data collected or received between May 1, 1997 and June 2, 2008. Sections 2 through 11 of this report are based on this initial lockdown date. Between June 2, 2008 and July 19, 2010, a significant amount of sediment data was collected from the portion of the river that passes through downtown Portland. A second lockdown date of July 19, 2010 was established to capture this new data as well as other data collected from the Study Area. Data collected or received between June 8, 2008 and July 19, 2010 are presented in Appendix H. The following table summarizes the location of data associated with these two lockdown dates:

Report Section	**Data Lockdown Date**
Sections 2–11 and Associated Appendices	June 2, 2008
Appendix F: BHHRA	June 2, 2008 plus select studies: EPA fish and sediment polybrominated diphenyl ether (PBDE) data, Terminal 4 Abatement Phase I construction project data
Appendix G: BERA	June 2, 2008 plus select studies: EPA osprey egg data, EPA sediment PBDE data, Terminal 4 Abatement Phase I construction project data, RM 11E data

Report Section	**Data Lockdown Date**
Appendix H	July 19, 2010, which includes data collected prior to June 2, 2008

In addition, map base layers were subject to lockdown dates:

Base Layer/Report Section	**GIS Base Layer Lockdown Date**
Highlighted Upland ECSI Sites/Sections 2-3, 5-6, 11 and Associated Appendices	2008
Highlighted Upland ECSI Sites/Sections 4 and 10	2009
Bathymetric Hillshade and Depth Change/Section 3	Historical Depiction (2001), Current Depiction (2009)
Outfalls/All Sections	2009
Dredge & Cap Areas/Detailed Core Plot Maps (Appendix D1.2) and Appendix H	2011
Dredge & Cap Areas/All Other Sections	2008

- **Section 3. Environmental Setting.** This section reviews the physical and human characteristics of the Study Area, including meteorology, geology, hydrogeology, bathymetry, sediment physical properties, habitat, current and historical land and harbor use, history of the municipal and non-municipal conveyance systems, and public access. Results from the application of the hydrodynamic sedimentation model are also discussed.
- **Section 4. Identification of Sources.** This section describes general land use changes to give a historical context to the Study Area. The types of known and potential contaminant sources that affect the Study Area are identified. This section augments general information on sources provided in the Programmatic Work Plan and the Round 2 Report.
- **Section 5. In-river Distribution of Indicator Chemicals.** This section describes the nature and extent of ICs in surface and subsurface sediment, in-river sediment traps, surface water, TZW, and biota.
- **Section 6. Loading, Fate, and Transport for Select Indicator Chemicals.** This section presents an overview of the primary known sources of contaminants to the river; describes the processes affecting the release, transport, and fate of ICs within the Study Area; and presents estimates of current

pathway-specific mass-loading rates of select ICs. Historical loading to the Study Area is discussed qualitatively in this section.

- **Section 7. Determination of Background Concentrations for Indicator Chemicals.** This section evaluates the concentrations and statistical characteristics of ICs in surface water and sediment samples collected from upstream reference locations (i.e., background) for use in risk characterization, PRG development, and remedial alternatives evaluation during the FS. Sediment trap data, surface water suspended solids, and subsurface sediment data collected from dredge borrow pits near the upstream end of the Study Area are also discussed as supporting lines of evidence for background.
- **Section 8. Baseline Human Health Risk Assessment Summary.** This section provides a summary of the BHHRA included in Appendix F.
- **Section 9. Baseline Ecological Risk Assessment Summary.** This section provides a summary of the BERA included in Appendix G.
- **Section 10. RI Conceptual Site Model for Select Indicator Chemicals.** This section presents a site-wide overview of the physical setting; contaminant distribution in sediments; contamination sources; external loading and internal fate and transport mechanisms; human health risk drivers and potentially complete exposure pathways/scenarios; and ecological risk drivers and ecological receptors/exposure pathways. For selected ICs, this section also presents integrated, chemical-specific evaluations of nature and extent in abiotic and biotic media in the Study Area, and the relationships between the observed distribution in the system and known or likely historical and current sources of contamination.
- **Section 11. Summary and Conclusions.** This section provides the major findings of the RI and the next steps in the RI/FS process.
- **Section 12. References.** Citations noted in the RI are found in this section.
- **Section 13. Glossary.** This section contains definitions of technical terms found in the RI.

Nine appendices are included with this document:

- **Appendix A. Data Sources and Site Characterization/Risk Assessment Database.** This appendix briefly summarizes the studies from which data in this RI report were obtained and includes the complete database in Access® files on compact disc. Data rules for reducing the site characterization/risk assessment (SCRA) database into the RI data set are provided. This SCRA database in Appendix A reflects the June 2, 2008 lockdown date. Further, this appendix includes the process for calculating chemical concentrations from whole-body bass and carp samples. Finally, an administrative appendix that documents the decision-making process between EPA and the LWG for the RI is included.

- **Appendix B. DEQ September 2010 Milestone Report Table 1.** This appendix presents Table 1 from DEQ's Joint Source Control Strategy (JSCS) Milestone Report (DEQ 2010c).
- **Appendix C. Stormwater Statistics and Groundwater Characterization.** Summary statistics for stormwater collected by the LWG and other parties are included in this appendix. Details of LWG's groundwater pathway assessment work, including identification of potential upland groundwater source areas and TZW investigation results, are also provided.
- **Appendix D. Nature and Extent of Contaminants in Biotic and Abiotic Media.** Summary statistics of the chemical and physical data for all media are provided. The appendix includes constituent concentrations used in each summed analyte group for all media. Scatter plots, histograms, and maps of contaminant distribution of ICs not included in the main report are also included in this appendix. A technical memo, Comparison of PCB Aroclor and Congener Data, is included in this appendix. The results of screening the surface water and TZW data against human-health-based criteria are also presented in this appendix..
- **Appendix E. Loading, Fate, and Transport Supporting Information and Calculations.** This appendix provides the analyses used to develop loading estimates for upstream surface water, stormwater, permitted point source discharges, atmospheric deposition, groundwater plumes, and advection through sediments.
- **Appendix F. Baseline Human Health Risk Assessment.** This appendix provides the complete BHHRA.
- **Appendix G. Baseline Ecological Risk Assessment.** This appendix provides the complete BERA.
- **Appendix H. Updated RI Data Set: Distribution of Indicator Chemicals in the Upper Study Area and Upstream and Downstream Areas.** This appendix updates the presentation of the Downtown Reach and the area off RM 11 using an updated RI data set. Data sets collected or received between June 2, 2008 (the Draft RI Report data lockdown date) and July 19, 2010 (the Final RI Report data lockdown date) are included in the updated database. The updated RI data set is provided in Access® on compact disc in an attachment to this appendix. Other attachments to this appendix include the following:
  - Summary statistics of the chemical and physical data for sediment and sediment trap data
  - Summaries of the new studies provided in the updated RI data set
  - The complete data report, *Contaminant Concentrations in Osprey (Pandion haliaetus) Eggs from Portland Harbor and Surrounding Areas: Data*

Summary Report prepared by the U.S. Fish and Wildlife Service and Associates Environmental Consulting, LLC

– A technical memo, Monitored Natural Recovery Area Radioisotope Evaluation in the upper Study Area.

- **Appendix I. Interactive Map Application of Section 10 Panels.** This appendix provides an interactive map application of the three-section panel analyte-specific series included in Section 10 that present cross-media contaminant distributions and available source information for each of the 13 CSM contaminants. The panels show the entire Study Area, upland site property boundaries, outfall locations, historical industries, and river mile markers, and these layers can be turned on and off to display different combinations of information (e.g., subsurface sediment and biota PCB distributions).

2.0 ENVIRONMENTAL DATA SETS

Environmental data have been collected within the Portland Harbor Study Area during numerous LWG sampling events since the inception of the Portland Harbor RI/FS in 2001. These data, along with data from historical and concurrent studies by other parties in the LWR constitute the Portland Harbor SCRA database. The Portland Harbor SCRA database consists of over one million analytical results representing a variety of sample matrices dating back to 1969. The most recent data are from June 2008.

Environmental data sets in the SCRA database have undergone rigorous data quality review and meet the data quality objectives established for the project in the Programmatic Work Plan (Integral et al. 2004; see Section 2.1.1). The SCRA database is the basis for defining the nature and extent of the contaminants and provides foundational information from which decisions are made regarding human health and ecological risk within the Study Area, as well as the development of remedial alternatives for the FS.

The Portland Harbor SCRA database was transmitted to members of the LWG risk assessment teams on June 2, 2008.[1] The RI, BERA, and BHHRA database managers separately queried the Portland Harbor SCRA database to derive subsets of data to support their respective efforts. For the RI, the BERA, and the BHHRA, a date of May 1, 1997 was used to define the initiation of the sediment data set to follow the last major flood of the Willamette River in the winter of 1996. Additional selections and data reduction steps were applied to the RI, BHHRA, and BERA data sets to satisfy specific data evaluation and presentation needs. Each data set is briefly described below:

- **RI Data Set** – The RI data set includes data from those matrices relevant to discussions of contaminant distributions, contaminant loading, background, and potential sources, including sediment/sediment trap, riparian sediment and soil, surface water, TZW, seep, tissue, and stormwater data. Selected data collected by other parties are presented in Table 2.0-1 and are described in Section 2.1.4 and Appendix A1. Exclusions from the RI data set are presented in Appendix A2. A flat file of the RI data set is provided in Appendix A3.
- **BHHRA Data Set** – The BHHRA data set includes only those matrices relevant for direct human health exposure pathways, including surface sediment, clam and crayfish tissue, fish tissue, surface water, seep water, and shallow TZW. Data sources and selection criteria for the BHHRA data set are summarized in Section 2.1.4 and are described in detail in Appendix F.
- **BERA Data Set** – The BERA data set includes matrices relevant to ecological exposure pathways, including surface sediment, benthic invertebrate and fish

[1] Additional sediment and tissue data collected subsequent to the June 2008 transmittal is included in an updated SCRA database, which is presented in Appendix H of this RI report.

tissue, surface water, and shallow TZW. Data sources and selection criteria for the BERA data set are summarized in Section 2.1.4 and are described in detail in Appendix G.

The remainder of Section 2 presents the data management and reduction process used in the RI and risk assessments (RAs; Section 2.1) and the environmental data collected by media (Section 2.2).

2.1 DATA EVALUATION PROCESS

This section focuses on data quality reviews that were performed to evaluate the quality of LWG and non-LWG data and determine their usability for various purposes in the RI/FS, BHHRA, and BERA. The principal issues related to the usability of historical and current data include data quality, sediment stability over time, and the intended use of the data. All of these factors must be acceptable for data to be considered usable.

2.1.1 Data Quality

Methods for performing data quality reviews for data generated by the LWG are described in the project-specific QAPP. In addition, a detailed review of the quality of each non-LWG chemical and biological data set was performed prior to entering those data sets into the project database. Methods for reviewing non-LWG data are described in the Programmatic Work Plan (Section 4 and Appendix F; see Integral et al. 2004).

Two overall data quality categories were established in the Programmatic Work Plan, as follows:

- **Category 1.** Category 1 data are of known quality and are considered acceptable for use in decision making for the Site. There is sufficient information on these data sets to confidently verify that the data, along with associated data qualifiers, accurately represent chemical concentrations present at the time of sampling.
- **Category 2.** Category 2 data are of generally unknown or suspect quality. The quality assurance and quality control (QA/QC) information shows that data quality is poor or suspect, or essential QA/QC data (e.g., surrogate recoveries, matrix spike/matrix spike duplicates) are either incomplete or lacking.

The evaluation of data quality was conducted at the finest level of detail available for each data set. This evaluation focused on individual analyte groups within each survey when possible, and so any given survey may contain all Category 1 data, all Category 2 data, or a combination of Category 1 and 2 data. In addition, data that received a QA1 or QA2 level of validation were flagged as such, providing a combined data quality category (e.g., Category 1 QA2). For chemistry data, Category 1 and 2 designations were entered into the project database for each sample and analyte. Sample counts of Category 1 and Category 2 data are summarized in Table 2.1-1.

Project decisions will be based on analyses using Category 1 data. Category 1 data that have had an EPA-approved level of data validation, comparable to Washington State Department of Ecology's "QA2" evaluation, are designated as "Category 1 QA2" data sets. All data generated by the LWG hold the Category 1 QA2 designation. Some data generated by other parties are also designated Category 1 QA2. Non-LWG Category 1 data that received an abbreviated level of review are termed "Category 1 QA1." Only Category 1, QA2 data are used in the BHHRA, the BERA, and the determination of background chemical concentrations (Section 7). Both Category 1, QA1 and QA2 data are used to describe the nature and extent of contamination (Section 5) and to evaluate contaminant loading, fate, and transport (Section 6). Category 2 data were generally used for project scoping. For example, Category 2 tissue data were used to help identify contaminants of interest (COIs)[2], and Category 2 sediment data were used in the initial assessment of trends in contaminant concentrations, which was useful for developing sampling programs. No Category 2 data for environmental media other than sediment are included in the RI data set provided in Appendix A3.

2.1.1.1 Chemical Data Review Criteria

Criteria for placing data sets into categories were developed during the compilation of existing information to identify basic data qualities and not to limit data to specific program uses. Chemical data quality was assessed by evaluating the following factors:

- Traceability
- Comparability
- Sample integrity
- Potential measurement bias
  - Accuracy
  - Precision.

All of these factors were known or supported by existing QA/QC information (analytical methods, chain-of-custody, sample holding time, method blanks, matrix spike/matrix spike duplicates, laboratory control samples, replicates, surrogates) for Category 1 data. If supporting documentation for each factor was not available or was not reinforced by the availability of other high-quality QA/QC information, data were assigned a Category 2 designation. If the acceptance criteria for any of the above factors were not satisfied for either the entire data set or a specific analyte group, data for that data set or group were generally qualified and were determined to have limited usefulness. The chemical data were reviewed by analyte group (e.g., metals, semivolatile organic compounds [SVOCs]). As a result, a data set may contain all Category 1 data, all Category 2 data, or both Category 1 and Category 2.

[2] Prior deliverables and some of the tables and figures attached to this document may use the term "chemical of interest," which has the same meaning as "contaminant of interest" and refers to "contaminants" as defined in 42 USC 9601(33).

2.1.1.2 Biological Data Review Criteria

Bioassay data quality was evaluated based on validation guidelines and performance criteria from the Puget Sound Estuary Program (PTI 1989). Bioassay validation guidelines include checks of completeness, holding conditions, standard reporting methods, and QA/QC results for negative control, reference sediment, positive control (reference toxicant), and measured water quality parameters according to standard testing methods and established performance criteria.

2.1.2 Sediment Stability and Temporal Integrity

The observed and modeled hydrodynamic and sediment transport patterns in the LWR, particularly in the Study Area, are detailed in Section 3. Early RI investigations—Sediment Trend Analysis® (STA), Sediment Profile Imagery (SPI), and time-series bathymetry studies—have been described previously (Integral et al. 2004, 2007), and these initial lines of evidence suggested the potential for widespread, small-scale (≤30 cm) surface sediment disturbance or movement during winter (rainy season) flow regimes. The measured, maximum net bathymetric change over the 25-month period between the January 2002 and February 2004 surveys was less than 30 cm (1 ft) over 90 percent of the ISA (see Section 3.1.4.2). While river bed elevations during flooding events were not measured, the 30-cm depth appears to represent an interval below which sediment erodibility decreases markedly (or conversely sediment column stability increases). While this characteristic is heterogeneous across the site (e.g., erodibility of sandy sediments being different than mud), a 30-cm surface sediment layer definition for the RI was designed to capture that portion of the sediment column that has the potential to be disturbed or transported under typical annual conditions. This 30-cm surface interval is significant from a physical perspective; however, the depth of the biologically-active layer, where sediment biota exposures occur, is typically only 10 to 15 cm deep.

Empirical surface sediment erodibility measurements (Sedflume) associated with the hydrodynamic and sediment transport (HST) modeling, as well as measured surface-to-subsurface gradients in chemical contamination, also indicate that significant river-wide sediment bed movement or resuspension does not occur under typical flow conditions in the Study Area. There is potential, however, for significant sediment bed movement in the high-energy portions of the LWR during rare high-flow events. Modeling simulations of extreme hydrological events (e.g., the 100-year flood) on the LWR indicate the potential for significant sediment bed erosion up to one meter (m) or more in the more dynamic portions of the Study Area (see Section 3.1.4.2). The February 1996 flood is the most recent hydrologic event on the LWR that approaches a 100-year flood level. The following winter (1997) had the second highest flow levels observed in the past 30+ years.

In consideration of these factors, the RI data set only includes data that were collected after the winter of 1996/1997 and that meet the other usability criteria described above; the earliest data included in the database were collected in May 1997. The assumption is that while near-surface changes in chemical concentrations due to sediment scour or

accretion certainly have occurred in places, no natural large-scale erosion events or re-exposure of buried deep sediments has occurred since that time.

2.1.3 Data Usability

A final component of the data evaluation process is to consider the intended use of the data throughout the complete RI/FS process. This is an important component because, although the same database may be used, various treatments and interpretations of the data may be necessary depending on the objectives of the specific evaluations. This section describes some of the more commonly encountered, and perhaps most consequential, data usability issues for the Portland Harbor RI/FS—elevated detection limits for undetected chemical concentration data and N-qualified chemical concentration data.

2.1.3.1 Elevated Detection Limits

The historical database for Portland Harbor contains some samples with undetected concentrations of constituents at high detection limits. From a purely analytical perspective, EPA categorizes all data meeting proper QA/QC procedures, regardless of detection limit level, as Category 1 (i.e., data of known quality and considered acceptable for use). However, the acceptability of these data is dependent on their specific use. For example, in the absence of other data, elevated detections limits may provide insight on the need for additional analyses for which lower detection limits are achievable. From a data-needs standpoint, however, these same data may not be useful because if inappropriately compared to some concentration benchmark, they could unnecessarily result in the perceived need for additional sampling and analysis (despite their attendant uncertainty in actual concentration). From a predictive risk assessment perspective, these data are recommended by EPA to be excluded from formal risk quantification because of their uncertainty in concentration (EPA 1989). From an applied engineering and feasibility standpoint, elevated detections limits are also not useful because they are not capable of defining with precision actual chemical concentration data that can be used to set boundaries for remedy considerations.

2.1.3.2 N-qualifiers

N-qualified data present another situation that requires evaluation in the context of data use. N-qualified data in the RI data set are identified and used in the RI and RAs with recognition of the potential limitations associated with using this data noted below.[3] The N-qualifier denotes that the identity of the analyte is presumptive and not definitive, generally as a result of the presence in the sample of an analytical interference, such as hydrocarbons or, in the case of pesticides, PCBs. Data that are N-qualified meet the primary identification criteria of the method; however, the confirmation criteria are not met and the identification is potentially a false positive. In addition to uncertainty regarding chemical identification, N-qualified data also indicate some uncertainty in the reported concentration level (EPA 1989). The degree of

[3] Consistent with Risk Assessment Guidance for Superfund Part A (EPA 1989), N-qualified data were included in the data set used for use in risk assessment, as documented here.

attendant uncertainty in both identification and concentration is commonly assessed on a sample-by-sample basis.

Given the uncertainty associated with N-qualified data, as well as the varying extent of this uncertainty, users must carefully weigh the impacts of its use throughout the RI/FS process. This careful attention is fully consistent with EPA guidance and other data-use guidance documents recommending the use of N-qualified data only on a case-by-case basis. The rationale for this is that data are evaluated and used in different ways throughout the RI/FS process. Provided below are some examples of how intended use of N-qualified data could vary:

- **Nature and Extent.** In nature and extent determinations, evaluations of data are generally predicated on individual, point-by-point analytical results (e.g., determining the individual chemical concentration results at various sample locations as opposed to calculating a mean across those samples). In instances when N-qualified data are spatially accompanied by data of more certain chemical identification and concentration, the role of N-qualified data is likely limited. This is often the case for Portland Harbor, where there exists an abundance and wide distribution of data for sediment, surface water, and biota that are not N-qualified. The number and percentage of N-qualified data for the nature and extent ICs (see Section 5.0) for all media are summarized in Table 2.1-2 and details are provided in data reports and SCSRs for the various sampling tasks.

- **Risk Assessment.** N-qualified data are present in the database, and EPA requested that these data be included in the risk assessments (EPA 2006b). EPA (1989) recognizes that while uncertainty in both chemical identity and chemical concentration exists for N-qualified data, their use in risk assessment is judged on a case-by-case basis. N qualification indicates "the presence of an analyte that has been 'tentatively identified,' and the associated numerical value represents its approximate concentration" (EPA 1999). The qualification indicates that the analyst believed that the result was the result of analytical interference from a chemical other than the target analyte. All N-qualified results are therefore biased high for organochlorine pesticides and may result in an overestimation of risk.

- **Feasibility Study.** As the RI process proceeds towards conclusion, the LWG will continue to engage EPA on the need and importance of utilizing N-qualified data. Because the FS stage relies on both elements of nature and extent and risk assessment, the LWG will continue to examine carefully to what degree, if any, N-qualified data impact the FS work and its findings.

2.1.4 SCRA Database

The LWG project database contains all of the data reported by the analytical laboratories. This includes field and lab replicates, lab dilutions, results for the same

analyte from multiple analytical methods (SW8270 and SW8270-SIM, for example), and laboratory QA samples such as matrix spikes, surrogates, and method blanks. The data handling rules described in *Guidelines for Data Averaging and Treatment of Non-detected Values for the Round 1 Database* (Kennedy/Jenks et al. 2004) were used to create a simplified data set for the SCRA data users. This involved creating a SCRA database that excludes laboratory QA results, contains only the most appropriate dilution result and analytical method for each analyte, and averages laboratory replicates and splits. The resulting data set contains only one result per analyte per sample.

2.1.4.1 Quality Assurance/Quality Control

QA/QC checks were made throughout the data management process, but those specific to SCRA data reduction steps focused on the following checks:

- **Completeness** – Chain-of-custody forms submitted to the laboratory listing samples, methods, and analyses were compared to samples, methods, and analytes loaded into the SCRA for each sampling event. These checks were made for all samples. For data sets prepared by other parties, the source document was typically used to check sample/method/analyte completeness in the SCRA.
- **Averaging** – Laboratory duplicates and field splits were averaged. Because averaging required significant data manipulation, a series of additional checks was performed on the SCRA database before distribution. Data were divided into subgroups, and approximately 40 percent of the data in each subgroup were verified. If any problems were found with the averaging, then 100 percent of the data in the subgroup were verified and problems were corrected.
- **Database Codes** – Project database codes were checked for sample type, matrix type, basis, and units. Unusually high and low values for a given method were checked to confirm potential unit or analytical errors.
- **Qualifiers** – Checks were made for unusual data qualifier codes. Qualifier codes for calculated averages followed guidelines provided in Kennedy/Jenks et al. (2004).

In addition to the standard checks above, Integral's senior chemists reviewed all final SCRA files prior to distribution.

2.1.4.2 Calculated Totals

This section presents the summation rules for the RI data set and baseline RA data sets and highlights where they deviate. Data management rules for all three data sets are summarized in Table 2.1-3.

2.1.4.2.1 General Summation Rules

RI data set summation rules are as follows:

- Calculated totals are the sum of all detected concentrations; non-detected concentrations are treated as zero
- If all analytes for a total are not detected, then the highest detection limit is used for the summation.

Baseline RAs and the background data set summation rules are as follows:

- Calculated totals are the sum of all detected concentrations, and non-detected results for analytes detected at least once in the RA data set within the Study Area for a given medium are included in the summation at one-half the detection limit
- If none of the analytes are detected for a given sample, but are determined to be present within the Study Area, then the highest detection limit is used for the summation
- Non-detects for analytes never detected within a data set for a given medium are excluded (i.e., treated as zero).

The determination of medium-specific data sets differs between the BHHRA and the BERA based on relevant exposure scenarios. Medium-specific data sets are described in Appendix F (BHHRA) and Appendix G (BERA).

2.1.4.2.2 Individual Analytes in Calculated Totals

Data sets for the RI and baseline RAs included calculated totals for the chemical groups listed in Table 2.1-4.

Individual analytes included in totals are as follows:

- **Total PCBs**—Sum of PCB Aroclors or PCB congeners. Total PCB Aroclors represent the sum of all reported Aroclors. Total PCB congeners represent the sum of all reported (up to 209) individual congeners. For the RI and BHHRA data sets, total PCB congeners were selected to represent total PCBs when available. If not available, total PCB Aroclors were selected. Total PCB selection in the BERA varied depending on the medium:
  - For all BERA surface sediment samples, the total PCB concentration is represented by total PCB Aroclors.
  - For the BERA tissue data set, the total PCB concentration is represented by total PCB Aroclors for Round 1 samples and total PCB congeners for Round 2 and Round 3 samples. Aroclors were selected over congeners for Round 1 because PCB congener analysis was performed on only a limited number of samples.
  - For the BERA surface water data set, the total PCB concentration is represented by total PCB congeners for all XAD samples and by total PCB Aroclors for locations where only peristaltic samples were collected.

- **Total PCDD/Fs**[4]**—**Total polychlorinated dibenzo-p-dioxin/furan (PCDD/Fs) reported in the RI are the sum of tetra and higher polychlorinated dioxin and furan homologs: tetrachlorodibenzo-p-dioxins (TCDDs), pentachlorodibenzo-p-dioxins (PeCDDs), hexachlorodibenzo-p-dioxins (HxCDDs), heptachlorodibenzo-p-dioxins (HpCDDs), octachlorodibenzo-p-dioxin (OCDD), tetrachlorodibenzofurans (TCDFs), pentachlorodibenzofurans (PeCDFs), hexachlorodibenzofurans (HxCDFs), heptachlorodibenzofurans (HpCDFs), and octachlorodibenzofuran (OCDF). Total PCDD/Fs for the BERA were calculated from the sum of individual PCDD/F compounds. The BHHRA relies solely on the 2,3,7,8-TCDD TEQ.
- **PCB and Dioxin TEQs**—Toxicity equivalency factors (TEFs) were used to calculate PCB and dioxin toxic equivalent concentrations (TEQs). Concentrations of congeners are multiplied by their TEFs to estimate toxicity of the congeners relative to 2,3,7,8-TCDD. Resulting concentrations are summed. TEFs are published by the World Health Organization (WHO) for fish and birds (Van den Berg et al. 1998) and for mammals (Van den Berg et al. 2006).
- **Total DDx**—Total DDx was calculated from the six DDx compounds: 2,4′-dichloro-diphenyl-dichloroethane (DDD); 4,4′-DDD; 2,4′-dichloro-diphenyl-dichloroethene (DDE); 4,4′-DDE; 2,4′-dichloro-diphenyl-trichloroethane (DDT); and 4,4′-DDT. Total DDD were calculated with 2,4′-DDD and 4,4′-DDD; total DDE were calculated with 2,4′-DDE and 4,4′-DDE; and total DDT were calculated with 2,4′-DDT and 4,4′-DDT.
- **Total LPAHs**—Total low molecular weight PAHs (LPAHs) are the sum of 2-methylnaphthalene, acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene, and phenanthrene.
- **Total HPAHs**—Total high molecular weight PAHs (HPAHs) are the sum of fluoranthene, pyrene, benzo(a)anthracene, chrysene, benzofluoranthene, benzo(a)pyrene, indeno(1,2,3,-c,d)pyrene, dibenzo(a,h)anthracene, and benzo(g,h,i)perylene.
- **Total PAHs**—Sum of the individual LPAHs and HPAHs.
- **Total cPAHs**—A benzo(a)pyrene (BAP) equivalent (BaPEq) concentration was calculated by multiplying the carcinogenic PAHs (cPAHs) by their respective potency equivalency factors (PEFs), and summing the resulting concentrations. PAHs classified as carcinogenic are benzo(a)anthracene, chrysene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, indeno(1,2,3,-c,d)pyrene, and dibenzo(a,h)anthracene. PEFs were assigned according to EPA (1993) and are shown in Appendix A3.
- **Total Chlordanes—**Sum of cis-chlordane, trans-chlordane, oxychlordane, cis-nonachlor, and trans-nonachlor.

[4] The term “PCDD/Fs” is equal in meaning to “dioxins/furans” in the RI.

- **Total Endosulfan**—Sum of alpha-endosulfan, beta-endosulfan, and endosulfan sulfate.
- **Total Xylene**—Sum of m,p-xylene, o-xylene, and xylene.
- **BTEX**—Sum of benzene, toluene, ethylbenzene, and total xylene.
- **Total Fines**—Sum of all silt and clay grain-size fractions passing U.S. standard sieve #230 (0.0625-mm openings).
- **TPH**—Total petroleum hydrocarbons (TPH) are the sum of diesel-range hydrocarbons, residual-range hydrocarbons, gasoline-range hydrocarbons, lube oil, and motor oil.

For both the RI and baseline RA data sets, a minimum number of individual analytes for a given sample was required to be analyzed in order to complete the totals. These rules are provided in Table 2.1-5. Totals with less than the expected number of analytes but above the minimum number of analytes were qualified with an "A." For PCB and dioxin TEQs, all analytes with TEFs were required in order to calculate a total.

2.1.4.3 Field Replicates

For the RI, the BHHRA, and the BERA data sets, field replicates were generally retained as individual sample results. For spatial analyses requiring the calculation of spatially weighted average concentrations, only one sample result was used for those results with identical sampling coordinates. In those cases, data associated with the first sample were used in the analysis. Field replicates in the background data set were averaged to avoid bias by overweighing a single sample location. The potential for bias is greater in the background data set due to the small number of samples. Otherwise, data presentations and analyses included field replicates as discrete samples.

2.1.4.4 Organic Carbon Normalization

Organic chemical results were organic carbon normalized (OC normalized) for subsequent evaluation in the BERA and background data sets, following the criteria described in Table 2.1-3. Dry-weight concentrations in mg/kg were divided by the decimal percent total organic carbon (TOC) value.

No upper TOC limit was set that would exclude normalization; however, for higher TOC values (>4.0%), each individual sample was evaluated for possible anthropogenic contributions to organic carbon (e.g., wood waste, petroleum, non-aqueous phase liquids [NAPLs], or sewage) that may have confounded partitioning assumptions. For TOC <0.2%, or high values with contribution from anthropogenic wastes, no OC-normalized value was calculated. In these few cases, sample data were evaluated on a dry-weight basis only. For samples without TOC data, the value was estimated using a regression equation based on site-specific TOC and grain size (as percent fines) from the upriver reach (RM >15.3).

2.1.5 RI Data Set

The data set used for each RI data type is summarized in Table 2.1-6. The first column of the table lists the various data types. The second column lists the data sources and the general data quality selection criteria (e.g., Category 1 versus Category 2 and QA1 versus QA2). Additional data inclusion or exclusion criteria used for subsequent data analyses and presentations in the RI Report are listed in the third column. Further data evaluation steps, such as the outlier analysis, that are specific to a particular data set, are not included here but are discussed in the appropriate RI report sections. A complete list of data exclusions for each data type is provided in Appendix A2. The RI data set is provided electronically in Appendix A3.

2.1.6 BHHRA Data Set

The data set used for each BHHRA data type is summarized in Table 2.1-7. The first column of the table lists the various data types. The second column lists the data sources and the general data quality selection criteria (e.g., Category 1 vs. Category 2 and QA1 vs. QA2). Additional data inclusion or exclusion criteria used to develop the BHHRA data set are listed in the third column. Specific data management procedures and rules and additional data reduction steps for the BHHRA are provided in Appendix F.

2.1.7 BERA Data Set

The data set used for each BERA data type is summarized in Table 2.1-8. Taken together, the inclusion and exclusion criteria listed below were used to develop the BERA data set. Specific data management procedures and rules and additional data reduction steps for the BERA are provided in Appendix G.

2.1.8 Query Manager™ Database

EPA and its government partners, as well as members of the general public, use the Query Manager™ database-mapping application developed by NOAA's National Ocean Service Office of Response and Restoration.[5] Sediment and biota chemistry data contained in the SCRA database were translated into Query Manager-compatible format files and uploaded to NOAA's Portland Harbor Watershed Database. Currently, water data are not stored in Query Manager. NOAA has integrated the Portland Harbor Watershed Database with data query software (MARPLOT®) and ArcView® GIS on a web-based portal (http://mapping2.orr.noaa.gov/website/portal/portland/). Users may analyze and display the data contained in Query Manager along with spatial information, such as aerial photos, bathymetry, shoreline types, and outfalls.

Summing methods used in Query Manager for the various compound groups discussed above depart slightly from the LWG's summing methods. In all cases, non-detects for individual substances in a compound group are treated as zero values, and if the sum of detected results for individual substances in a sample is less than the maximum

[5] The LWG relies on the Portland Harbor SCRA database for decision-making purposes and reporting.

undetected result, then the sum is reported at the higher detection limit with a U-qualifier. Specific summing rules are provided below:

- **Total PCBs**—Aroclor and congener data are summed separately (PCB SUM A [total Aroclors] and PCB SUM P [total congeners]). In Query Manager, the preferred PCB sum is reported for the Aroclor data since those data are reported for the majority of studies. In the LWG database, the preferred PCB sum is reported for the congener data.
- **Total DDx**—Calculated using six isomers where available. If three or fewer isomers are reported, the sums are not derived (routine assumes that only p,p'-isomers were reported). Also, the sum of isomer pairs of DDT and its derivatives were calculated. The following pairs were summed when both isomers were provided for samples in the data set: 2,4'-DDT and 4,4'-DDT; 2,4'-DDD and 4,4'-DDD; 2,4'-DDE and 4,4'-DDE.
- **Total LPAHs**—Sum of acenaphthene, anthracene, biphenyl, 2,6-dimethylnaphthalene, fluorene, 1-methylnaphthene, 2-methylnapthene, 1-methylphenanthrene, phenanthrene, naphthalene, where two or more are measured.
- **Total HPAHs**—Sum of benzo(a)anthracene, benzo(a)pyrene, benzo(e)pyrene, chrysene, dibenzo(a,h)anthracene, fluoranthene, perylene, and pyrene, where two or more are measured.
- **Total PAHs**—Calculated as the sum of the LPAH and HPAH chemicals. Only those samples with more than one chemical in the group (LPAH and HPAH) are summed.
- **Total Chlordanes**—Sum of alpha-chlordane, gamma-chlordane, beta-chlordane, cis- and trans-chlordane, oxychlordane, cis-nonachlor, and trans-nonachlor.

All field duplicates and splits are retained in the Query Manager database and are recorded as separate samples reported from the same location and similar species/tissue type (if applicable) or depths. No data within the Query Manager database is an average. For some queries run with the Query Manager interface, if there are two results with the same StationID/SampleID (lab replicates), the result that has been designated as the preferred result (the “normal” sample) is reported in the queries. “Multi-chem” queries show both the main sample results and lab replicate results.

To illustrate the outcome of a query first performed on the RI data set and then performed in Query Manager, the following example is provided. Table column headings below are those provided in the respective data sets and offer corresponding information. In this example, total LPAHs were queried for samples collected from a surface sediment location (DG-11) collected in Round 3. Field splits (DG11-2 and DG11-3) were also collected from this location.

From Query Manager:

Stationid	exsampid	chemcode	conc	qualcode	units
DG11-1	LW3-DG11	LPAH	30.1	CALC	PPB
DG11-2	LW3-DG11-2	LPAH	142.9	CALC	PPB
DG11-2	LW3-DG11-3	LPAH	155.7	CALC	PPB

From RI Data Set:

LocationName	SampleID	cas_rn	ValueNum	Qualifiers	Units
DG11-1	LW3-DG11	LPAH	32	JT	ug/kg
DG11-2	LW3-DG11-2	LPAH	160	JT	ug/kg

Because no field splits are averaged in Query Manager, three separate total LPAH results were reported. Further, because total LPAHs are calculated using an analyte list that differs from the RI data set, and because splits are averaged in the RI data set, the calculated totals are different. In this case, the results reported in the RI data set are slightly higher than the results reported in Query Manager.

2.2 SUMMARY OF ENVIRONMENTAL DATA BY MEDIA

Between 2002 and 2008, the LWG collected the following data in the LWR:

- HST processes data
- Physical sediment characteristics
- Surface and subsurface sediment and tissue chemistry
- Sediment toxicity data
- Sediment trap chemistry
- Surface water chemistry
- TZW chemistry
- Stormwater chemistry
- Habitat type and distribution
- Species occurrence
- Biota tissue chemistry
- Cultural resources information
- Other regional and local information that informed and guided the RI studies.

The RI characterization process was multifaceted and iterative, involving three rounds of sampling for different data needs, often timed around varying river stages, river flows, and storm events. Table 2.2-1 provides an overview of the LWG sampling dates

for each major round of sampling, along with references for the data documents submitted to EPA. Each sampling event followed an approved FSP and QAPP. Field results were documented in an FSR. Analytical results were documented in a data report and/or a SCSR. Studies conducted by the LWG to characterize the LWR physical system are summarized in Table 2.2-2. Chemical and biological data from other parties were obtained primarily from individual LWG members, EPA, DEQ, the U.S. Geological Survey (USGS), and the U.S. Army Corps of Engineers (USACE). These investigations are summarized in Table 2.0-1. Data types included sediment, TZW, surface water, tissue, and seep chemistry and were compiled from both hard copy and digital sources. Appendix A1 provides additional information on the project objectives, sampling dates, sampling methods, sample types, and analyses for data collected by other parties listed in Table 2.0-1. As noted above, additional sediment data was collected subsequent to the transmittal of the Portland Harbor SCRA database. These data include, among others, USACE information related to recent Portland Harbor dredging projects, which was requested via the Freedom of Information Act (FOIA) in spring 2008 but had not yet been received prior to the June 2, 2008 lockdown date. These data are included in an updated Portland Harbor SCRA database, which is presented in Appendix H of the RI.

Environmental data collected during LWG Round 1 and 2 investigations, as well as studies by other parties during that same period, were summarized in the Round 2 Report (Integral et al. 2007). The Round 2 Report evaluated the physical, chemical, and biological information collected through the Round 2 (2006) sampling effort in order to focus the Round 3 data collection effort and, to the extent practicable, determine the final data needs for the RI/FS. Table 2.2-3 summarizes, by media, all data that are included in the RI data set. A subset of these data for matrices relevant to ecological and human health exposure are included in the BERA and BHHRA data set, and are discussed further in the sections below. Numbers of samples and analyses performed on each sample are summarized in Tables 2.2-4 (sediment), 2.2-5 (sediment trap), 2.2-6 (surface water), 2.2-7 (stormwater), and 2.2-8 (TZW), and Tables 2.2-9, 2.2-10, and 2.2-11 (biota).

2.2.1 Physical System

Many RI investigative activities were designed to develop a greater understanding of the Portland Harbor physical system. These included time-series bathymetric surveys, nearshore bank elevation change monitoring, and river flow measurements during high- and low-flow events. Additional time-critical data collection activities included a SPI field study, baseline bathymetric survey from RM 0 to Ross Island, and STA. Other field efforts involved the collection of water current direction and velocity profiles at several transects across the Study Area.

Along with existing information about the LWR, the physical data from the above studies were used to develop the physical CSM and to scope, develop, and calibrate a numerical HST model for the site (WEST 2004), which is described further in Section

3.1. The following physical system studies were conducted, and the following data types were collected for the RI:

- STA survey to document major trends in sediment transport (GeoSea Consulting 2001; SEA 2002a)
- SPI field study to provide reconnaissance information on physical and biological features of surface sediments in the LWR from Ross Island to the Columbia River (SEA 2002b)
- Five precision multibeam bathymetric surveys to document riverbed elevation changes over time (DEA 2002a, 2003b; SEA and DEA 2003; Integral and DEA 2004; Integral 2007e)
- Time-series sediment stake measurements to document nearshore bank elevation changes (Anchor 2004; Integral 2004b)
- Three acoustic doppler current profiler (ADCP) surveys to provide flow measurements during specific hydrological conditions, including a high-flow event and across tidal cycles (DEA 2002b, 2003b, 2004; Integral 2004b)
- Physical system data to calibrate the HST model, including total suspended sediment (TSS) concentrations, cohesive suspended sediment settling velocities, erosion rates, and critical erosion velocities (Integral 2006b,f; Integral et al. 2007).

The above studies are summarized in Table 2.2-2.

Data from a sixth bathymetric survey, conducted in 2009 on behalf of NOAA, was obtained for inclusion in the bathymetric change analyses discussed in Section 3 (DEA 2009).

Additional physical data meant to support the LWR FS were collected during RI sampling rounds (e.g., radioisotope analysis to determine mixing layer depth, a sidescan sonar survey to characterize Study Area riverbed conditions). Physical system data collected to support the RI are presented and described in detail in Section 3.0 of this report.

2.2.2 Sediment

Sediment chemistry data in the RI data set include LWG data collected from Rounds 1, 2, and 3 and data collected by other parties. The data documents are referenced in Tables 2.2-1 and 2.0-1. The selection criteria for non-LWG data are presented in Table 2.1-6. Only Category 1 QA2 surface sediment data that were not subsequently dredged or capped were used in the BHHRA and BERA.

The LWG data set is composed of samples collected from shorebird and human-use beaches (surface transect composites), riverbed samples (surface and subsurface), samples from biota sampling locations (collocated surface sediment), sediment toxicity

samples (surface), samples from TZW sampling locations (collocated surface sediment), and physical sediment characteristic samples (surface and subsurface). Data collected by other parties consist primarily of surface and subsurface riverbed samples. The majority of LWG surface and subsurface riverbed sediment samples were collected during Rounds 2 and 3 (some collocated surface sediment was collected in Round 1 from benthic invertebrate stations in the ISA). Surface and subsurface sediment data were collected from the Study Area (RM 1.9–11.8), Multnomah Channel, downstream (RM 0–1.9), downtown Portland (RM 11.8–15.3), and upriver (RM 15.3–28.4). Surface and subsurface sediment sampling locations for all three LWG rounds, as well as studies conducted by other parties, are shown in Maps 2.2-1a–y and 2.2-2a–t, respectively. LWG samples are identified by task/survey IDs "LWG01" (Round 1, surface samples only), "LWG02" (Round 2, surface and subsurface samples), and LWG03 (Round 3, surface and subsurface samples). Non-LWG sample survey IDs are cross-referenced to investigation summaries presented in Table 2.0-1 and Appendix A1. Numbers of samples and analyses performed on each sample are summarized in Table 2.2-4. Results for Rounds 1 and 2 were presented in detail in the Round 2 Report.

2.2.3 In-River Sediment Traps

The RI data set includes in-river sediment trap data collected by the LWG during Round 3. Data collected by the Port of Portland at Terminal 4 were excluded from the RI data set. Sediment trap data were not used in the BHHRA or BERA.

The LWG traps were deployed and maintained for one year at 12 locations within the Study Area, one location just downstream of the Study Area at RM 1.8, two upstream locations near RM 16, and at one location in Multnomah Channel (Map 2.2-3). Samples were collected quarterly, and the results of the in-river sediment trap sampling program were presented in a data report (Anchor and Integral 2008b). The number of sediment traps and the frequency of recovery and redeployment were designed to capture anticipated spatial and temporal variability of suspended sediment mass and to investigate the potential contributions of chemicals via waterborne sediment for various regions of the Study Area. The LWG sediment trap sampling program was not designed to support estimation of chemical mass loading within or throughout the system (Anchor 2006c). Table 2.2-5 lists the sample counts and analyses performed on each sample.

2.2.4 Bank Sediment and Soil

The RI data set includes bank (also referred to as the riparian zone; see EPA 2005b) sediment and soil data largely collected by other parties as part of bank and upland investigations. The LWG did not conduct bank/riparian zone investigations because the upper Study Area boundary is drawn at +13 ft NAVD88 elevation.[6] This elevation is based on a DEQ memorandum dated July 9, 2003 to EPA regarding the upland/in-water

[6] LWG Round 1 beach samples were composite samples from multiple grabs collected based on random-sampling scheme from the water line to the vegetation line, some individual grab subsamples from this sampling program were collected in the riparian zone (above +13 NAVD88, see Figure 2.2-1).

boundary for the Superfund Site (DEQ 2003b). Figure 2.2-1 depicts the shoreline boundary graphically. As noted in Table 2.1-6, surface sediment/soil data of any quality collected between +13 ft North American Vertical Datum of 1988 (NAVD88) and +20 ft NAVD88 are included in the RI data set. The bank sediment data are presented and discussed only in Sections 6 and 10 of the RI Report as part of the evaluation of potential sources of contamination to the Study Area. Bank sediment and soil data were not used in the BHHRA or BERA.

2.2.5 Surface Water

The RI data set includes LWG-collected data and data collected by other parties (Tables 2.2-1 and 2.0-1). The characterization of surface water in the following sections of the RI Report includes the LWG-collected data, TSS data collected by the City of Portland, and TSS and TOC data collected by NW Natural at the Gasco site. In addition, naphthalene data from samples collected off the Siltronic Corporation (Siltronic) facility is included in a discussion of groundwater pathways presented in Appendix C2. All other surface water data collected by other parties were excluded from the presentation of surface water data. Only LWG data were included in the BHHRA and BERA.

Surface water chemistry and conventional water quality parameter (i.e., temperature, pH, dissolved oxygen, and conductivity) data in the RI data set include samples collected during three surface water sampling events that took place during Round 2A and four events during Round 3A. Sampling events were executed at certain times of the year to determine if various river stages, river flows, and storm events have a measurable effect on the nature or concentration of surface water chemical constituents. Sampling stations included both river-wide transects and single-point sampling stations at specific locations. River-wide transect sampling was designed to estimate integrated water concentration through a cross section of the river, or fraction of a cross section, at a point in time. Round 2A transect samples involved a single vertically and horizontally integrated sample composite collected from multiple lateral substations across the width of the river channel. Round 3A transects at RM 2 and RM 11 were subdivided into three segments: east shoreline to navigational channel, navigational channel, and navigational channel to west shoreline. At each segment mid-point location, a vertically integrated (VI) water column sample was collected. For the other four transects (i.e., RM 4, 6.3, 16, and Multnomah Channel), separate near-bottom and near-surface (NB/NS) laterally integrated transect samples were collected. Single-point samples were stationary samples or sample pairs located adjacent to amphibian habitats to support the BERA, in generally quiescent areas adjacent to beaches that are used by swimmers to support the BHHRA, and near known or suspected sources.

Round 2A data were collected at three transect stations (RM 4, 6.3, and 11) and at 20 single-point stations. Round 3A surface water samples were collected at six transect stations (RM 2, 2.9 [Multnomah Channel], 4, 6.3, 11, and 16) and 12 single-point stations. At Gasco, single-point surface water samples were collected from 20 locations from three depths: near surface, mid-depth, and near-bottom for each of three tidal periods for a total of 180 samples. Near-bottom water samples were also collected at

three locations at slack points in the tidal cycle, for a total of 12 samples (Anchor 2006b). At Siltronic, surface water was collected from 17 near-bottom locations collocated with groundwater sampling locations. Five surface water samples were also collected upstream and downstream of the site (MFA 2005a). Map 2.2-4 shows the surface water sampling locations, and Table 2.2-6 lists the sample counts and analyses performed on each sample.

2.2.6 Stormwater

The RI data set includes LWG-collected data and stormwater grab, sediment trap, and catch basin solids sample data collected by other parties (Tables 2.2-1 and 2.0-1). The LWG-collected data are used to generate estimated stormwater loads to the Study Area for the purposes of fate and transport modeling and recontamination analysis (see Section 6). Non-LWG stormwater data were provided by DEQ in early 2008 for sites collecting data under the JSCS program; no stormwater discharge permit data are included in the RI data set because chemical monitoring requirements for these permits are typically limited. Of the non-LWG data, Category 1 data collected since June 1, 2004, are presented in Section 4 of the RI Report for reference purposes only, but, as directed by EPA, are not used in estimating stormwater loads. Stormwater data were not used in the BHHRA or BERA. The LWG performed stormwater sampling during Round 3, which consisted of flow-weighted composite samples, grab stormwater samples, continuous stormwater monitoring, and sediment trap deployment at selected stormwater outfalls within the Study Area. Samples were collected from a subset of drainage basins/outfalls within each of five land use categories in the Study Area. These locations were sampled by LWG and Port of Portland (Terminal 4) during two sampling efforts in the spring/summer of 2007 (Round 3A) and the fall/winter of 2007–2008 (Round 3B). One additional site (GE Decommissioning) was sampled by GE during the same time frame. Results from the GE investigation are also included in the overall LWG stormwater data set. In early 2008, the City of Portland collected three additional samples to supplement the residential data set, and these samples are included as well. This sample set is used for calculating stormwater loading in Section 6.

In addition to the LWG stormwater data, at LWG's request, DEQ provided stormwater data in early 2008 for sites that had thus far collected data under the JSCS program. Additional information on these non-LWG data is provided in Section 4.4. These data are not used in Section 6 loading calculations but are used as a qualitative comparison to the LWG-collected data. Samples were collected during eight separate storm events during the spring of 2007. An additional mobilization consisting of stormwater composite and sediment trap samples was conducted in the fall of 2007.

The results of the LWG stormwater sampling program were presented in a data report (Anchor and Integral 2008a). In a separate field effort, the LWG collaborated with the City to update, rectify, and verify information regarding private outfalls discharging to the Study Area (Integral 2008a). Sampling locations for both the LWG and non-LWG data are shown on Map 2.2-5, and sample counts and analyses performed on each sample are summarized in Table 2.2-7.

2.2.7 Groundwater/Transition Zone Water

The RI data set includes all TZW chemistry data collected by the LWG during Round 2, as well as groundwater, seep, and TZW data collected by other parties. The data documents are referenced in Tables 2.2-1 and 2.0-1. The transition zone is defined as the interval where both groundwater and surface water comprise some percentage of the water occupying pore space in the sediments (EPA 2008a). The primary focus of the transition zone analyses presented in Section 5.4 and Appendix C2 of this report is the surface sediment layer, which is considered to be the upper 30 cm of the sediment. For RI sampling purposes, the surface or shallow TZW samples were collected at depths up to 38 cm below mudline (bml). Deeper (90 to 150 cm bml) TZW samples are also discussed in some cases to lend insight into observed chemical distribution patterns. TZW data were evaluated in the BHHRA and BERA. Seep data collected from Outfall 22B were also evaluated in the BHHRA. Seeps are defined as locations where water discharges from the ground either above or below the river surface (GSI 2003b). Additional upland and baseline groundwater data, which are not included in the Portland Harbor SCRA database, were reviewed for use in the Groundwater Pathway Assessment Appendix for selected sites (Appendix C2). These data are described in detail in Appendix C2.

TZW data were collected by the LWG at nine sites located within the Study Area (see Section 4), selected in agreement with EPA as sites with a confirmed or reasonable likelihood for discharge of upland groundwater COIs to Portland Harbor. These sites are adjacent to the Kinder Morgan Linnton Terminal, ARCO Terminal 22T, ExxonMobil Oil Terminal, Gasco, Siltronic, Rhone Poulenc, Arkema, Willbridge Terminal, and Gunderson (Map 2.2-6). Additional stratigraphic characterization of a riverbed area offshore of the Gunderson site was conducted during Round 3, but it was determined that sampling of TZW at this site would not be necessary because the stratigraphic data did not provide physical evidence of a potentially complete flow pathway.

The LWG TZW samples were collected with either a Trident push probe at depths ranging from 30 to 150 cm or with a small-volume peeper deployed at sediment depths of 0 to 38 cm. The TZW data set also includes samples collected by other parties offshore of the Gasco and Siltronic sites with Geoprobe©. Table 2.2-8 lists the numbers of samples and analyses performed on each sample.

2.2.8 Biota

The RI data set includes LWG-collected biota tissue data and adult Chinook, adult lamprey, and adult sturgeon fish tissue data from the Oregon Department of Human Services (ODHS)/EPA/Agency for Toxic Substances and Disease Registry (ATSDR) Fish Contaminant Study (ODHS et al. 2003). The data documents are referenced in Tables 2.2-1 and 2.0-1. Biota tissue types included in the BHHRA or BERA are provided in Appendix F and Appendix G, respectively. Data collected as part of the

ODHS study were the only non-LWG fish tissue data of acceptable quality utilized for BHHRA evaluation.

Fish and biota collection was often preceded by surveys to identify species being harvested for human consumption (Kennedy/Jenks 2003), to determine the presence of targeted species, and to identify the proper techniques for collecting biota samples and behavior information (e.g., Ellis Ecological Services 2002; SEA and Windward 2003; Windward and Integral 2005). Fish and invertebrate tissue chemistry data were collected from the Study Area by the LWG and other parties to estimate exposure concentrations (as tissue residues or diet) for appropriate species or groups of ecological receptors (i.e., benthic invertebrates and fish). Biota tissue data were also collected upriver of the Study Area. Results of the LWG's laboratory bioaccumulation bioassays were also included in the RI data set. The bioassays were performed using commercially supplied clams (*Corbicula fluminea*) and laboratory-cultured worms (*Lumbriculus variegatus*) exposed to surface sediments collected at the same locations where field clams and worms were collected within the Study Area (Map 2.2-7).

Sampling locations for field-collected biota during all three sampling rounds are shown on Maps 2.2-8 through 2.2-12. Sampling locations specific to small-home-range species of fish and invertebrates are shown on Maps 2.2-8 and 2.2-9; large-home-range fish species are shown on Maps 2.2-10 through 2.2-12. Table 2.2-9 summarizes the biota samples and analyses. Table 2.2-10 lists the LWG and non-LWG sample counts and analyses performed on each sample. Table 2.2-11 provides the number of fish and invertebrates in each sample composite.

2.2.8.1 Bioassay

In Rounds 2 and 3, 293 surface sediment samples from the Study Area and upriver were submitted to a bioassay testing laboratory for toxicity testing (Windward 2005; Integral 2008c). Toxicity testing was performed to support the development of one or more predictive models characterizing the relationship between sediment chemistry and benthic invertebrate toxicity in the Study Area. Two whole-sediment toxicity testing protocols were employed. The 10-day *Chironomus tentans* and the 28-day *Hyalella azteca* sediment toxicity tests measuring survival and growth were conducted. Bioassay reference stations were also collected upriver of the Study Area. Sediment bioassay sampling locations are shown on Map 2.2-1a–y. Bioassay data are included in the BERA data set only (Appendix G).

2.2.8.2 Invertebrates

Invertebrate tissue in the RI data set included LWG field-collected tissue samples for crayfish (*Pacifasticus leniusculus)*, clam (*Corbicula fluminea*), mussels (tentatively identified as *Margaritifera falcata* and *Anodonta nuttalliana*), which were incidental by-catch during clam sampling, and epibenthic invertebrates and zooplankton collected with multiplate samplers. Invertebrate sampling locations for these small-home-range species are shown on Map 2.2-8. For clams, mussels, and crayfish, the map locations are shown as centroids of the specific sampling areas for each species (i.e., crayfish

sampled in an area of 100-ft shoreline contour by 100-ft extension into the river channel, and for clams and mussels sampled in variable benthic sledge tow areas). Table 2.2-9 provides the total number and type of invertebrate tissue data and the analyses performed on each sample. Invertebrate samples were analyzed for the same suite of chemicals as fish. Collocated surface sediment samples were also collected at clam and crayfish tissue sampling locations (or as close as possible) and analyzed for a similar suite of chemicals (Map 2.2-8). The data documents for this data set are referenced in Table 2.2-1.

2.2.8.3 Fish

The following fish species were selected as ecological receptors for the various feeding guilds in the LWR:

- **Omnivorous and herbivorous fish**—Largescale sucker (*Catostomus macrocheilus*), carp (*Cyprinus carpio carpio*), and pre-breeding white sturgeon (*Acipenser transmontanus*)
- **Invertivorous fish**—Sculpin (*Cottus asper*, *C. perplexus*, and *C.* spp.), peamouth (*Mylocheilus caurinus*), and juvenile Chinook salmon (*Oncorhynchus tshawytscha*)
- **Piscivorous fish**—Smallmouth bass (*Micropterus dolomieui*) and northern pikeminnow (*Ptychocheilus oregonensis*)
- **Detritivorous fish**—Larval stages of (ammocoetes and macropthalmia) Pacific lamprey (*Lampetra* sp.).

LWG-collected fish tissue data are included in the RI, BHHRA, and BERA data sets. In addition, data for adult Chinook salmon, adult sturgeon, and adult lamprey collected by other parties were included in the RI and BHHRA data sets. The data documents are referenced in Tables 2.2-1 and 2.0-1.

Table 2.2-11 provides the number of fish and invertebrates in each sample composite. Fish species composites were based on individual fish collected over various reaches of the river. Sculpin were composited from areas similar to where crayfish were collected. The map locations are shown as centroids of the sampled area of 100-ft shoreline contour by 100-ft extension into the river channel (Map 2.2-8). Largescale sucker, peamouth, and northern pikeminnow were composited over 1-mile stretches (Map 2.2-9); smallmouth bass were composited over 1-mile reaches for Round 1 and composited from either side of the river over 1-mile reaches for Round 3 (Maps 2.2-10a–d); and black crappie, brown bullhead, and carp were composited over 3-mile reaches (Maps 2.2-11 and 2.2-12a–c). Map 2.2-13 shows sturgeon and juvenile Chinook samples collected within discrete set line areas (for sturgeon) or beach seine areas (for juvenile Chinook). Juvenile sturgeon samples were not composited. Three juvenile sturgeon were collected and individually analyzed for each of five reaches that ranged from 1 to 2 miles long. The 15 points on Map 2.2-13 show the individual location of all sturgeon collected (three at each reach). Pacific lamprey ammocoetes

and micropthalmia were collected wherever suitable habitat was encountered (Map 2.2-14). For lamprey ammocoetes and macropthalmia samples, composites were made up of samples collected at several different areas within the Study Area and the map locations are shown as each successful sampling site of the sampling areas for each composite. Three ammocoetes collected during Round 1 were not analyzed. Note that collected lamprey ammocoetes and micropthalmia specimens were not positively identified to species because as larvae they are difficult to distinguish from other lampreys.

Whole-body and fillet tissue types were composited separately for carp, black crappie, smallmouth bass, and brown bullhead. During Round 1, fillets were collected from different fish than were used for whole-body samples, including black crappie, brown bullhead, carp, and smallmouth bass. During Round 3B, however, fillet and whole-body data were obtained using the same fish. Fillets were removed from Round 3B carp and smallmouth bass, and fillets and bodies without fillets (i.e., remaining bodies) were composited and analyzed separately. Methods for calculating whole-body concentration for smallmouth bass and carp are provided in Appendix A4.

Stomach contents were also examined, and prey species were enumerated for juvenile Chinook salmon and juvenile sturgeon; stomach contents were analyzed for the same select chemicals relative to fish dietary risks. Collocated surface sediment samples were also collected at sculpin tissue sampling locations (or as close as possible) and analyzed for a similar suite of chemicals (Map 2.2-8).

2.2.9 Cultural Resources Survey

According to CERCLA and its implementing regulations, EPA is required to comply with federal statutes that provide protection of archaeological and historical resources, including Native American burials and places of traditional religious and cultural significance. In 2001, EPA and DEQ signed an MOU with six tribal governments and three federal and state agencies that identified cultural resources as an area of special concern to the signatory tribes. The AOC requires a cultural resources survey as part of the RI/FS. The survey included the LWR from the confluence of the Willamette and Columbia rivers to Willamette Falls, including upland areas adjacent to this stretch of the river. Results of the survey are documented in *Cultural Resource Analysis Report for the Portland Harbor Superfund Site, Portland, Oregon* (AINW 2005).

2.2.10 Other Information Reviewed

Over the course of the RI, numerous documents were reviewed that informed the LWG's understanding of Portland Harbor and the LWR, contributed to the development of the CSM, BHHRA, and BERA, and guided the sampling programs. Information obtained from these outside sources includes the following:

- Regional setting information from geologic maps, government and scientific literature regarding structural geology and hydrogeology, information on

regional datums and meteorology, and historical and recent bathymetric studies conducted by NOAA

- Regional development and current human use area information from aerial photographs, zoning and fire insurance maps, City of Portland Bureau of Planning documents, and the City of Portland Willamette River Atlas

- Extensive information on LWR upland site characteristics and contaminant pathways that guided RI sampling programs and supported CSM development—including historical development and land uses, industrial operations, and COIs—provided by DEQ's Environmental Cleanup Site Information (ECSI) database, individual upland site investigation reports, and government publications (e.g., EPA 2006b)

- Extensive information on private and municipal outfalls and upland drainage systems that contributed to the CSM, provided by the City of Portland, DEQ's Joint Source Control Investigations, the Columbia Regional Association of Governments (CRAG) studies, and individual upland site reports

- Dredging and capping histories from individual site reports and USACE permits

- Site habitat information and riverbank type designations supporting the BHHRA and BERA, provided by ODFW documents, the City of Portland Natural Resources Inventory Update (City of Portland 2008d), and the scientific literature

- Site fish and wildlife resident populations, behavior, and consumption information that support the BHHRA and BERA (e.g., ATSDR documents, Columbia River Inter-Tribal Fish Commission [CRITFC] documents, ODHS et al. 2003)

- Upland discharge information supporting fate and transport loading estimates, provided by government planning documents, DEQ, and National Pollutant Discharge Elimination System (NPDES) discharge permit information

- Willamette River stage and flow data from the USGS Morrison Bridge station (#14211720) used to support field work planning and calculate CSM loading estimates

- Chemical use and toxicity information to support the BHHRA, BERA, and CSM from government documents (e.g., ATSDR toxicological profiles, etc.) and the scientific literature (e.g., Batt 2004; Friberg et al. 1986)

- Studies on contaminant hazards posed to fish and wildlife (e.g., Eisler 1986, 1987, 1988)

- Contaminant fate and transport process information provided by other site investigations (e.g., Steuer 1995) and the scientific literature (e.g., Erickson 1997).

The above list and examples given are not meant to be inclusive of all the information sources utilized in the LWR RI. Additional references are provided in the following sections of this report and in other site documents previously submitted to EPA.

NOTE TO REVIEWERS:
At the request of EPA, Section 3 text has been reorganized since the Draft RI submittal. Section 3.1 Physical Environment (previously Section 3.2 Hydrogeology) has been moved to the front of the section. Section 3.2 Human Environment (previously Section 3.1 Current Land and Harbor Use) has been moved to the middle of the section. In addition, sections previously found in Section 4 that cover historical land use and development of the municipal sewer system have been moved and incorporated into Section 3.2. Text that was moved shows as strike out in its original location but is not shown as redline in its new location for ease of review.

3.0 ENVIRONMENTAL SETTING

This section describes the natural and human-altered environmental setting of the Portland Harbor Superfund Site. Physical characteristics of the site include meteorology, regional geology and hydrogeology, surface water hydrology, the in-water physical system (which includes bathymetry, physical sediment characteristics, and hydrodynamics and sediment transport), and habitat. Human characteristics of the site that are discussed here include historical and current land and harbor use, the municipal sewer system, and human access and use. In addition to providing context to the RI sampling and analysis, the factors presented in this section are considered in the refinement of the Study Area-wide CSM, which is presented in Section 10.

Section 3.1 focuses primarily on the physical setting of the Study Area (RM 1.9 to 11.8). However, the LWR and physical features of the Willamette River and basin from Willamette Falls (RM 26) to the Columbia River (RM 0), as well as the upstream portion of Multnomah Channel, are discussed as needed to place the Study Area's physical characteristics into a regional context.

The Willamette River basin has a drainage area of 11,500 square miles and is bordered by foothills and mountains of the Cascade and Coast ranges up to 10,000 feet high to the south, east, and west (Trimble 1963). The main channel of the Willamette forms in the southern portion of the valley near Eugene, at the convergence of the Middle and Coast forks. It flows through the broad and fertile Willamette Valley region and at Oregon City flows over the Willamette Falls and passes through Portland before joining the Columbia River (Integral et al. 2004) (Map 3.1-1).

The portion of the river from Willamette Falls to the Columbia River is considered the LWR (see Map 1.2-1). Multnomah Channel is a distributary channel of the LWR that begins at RM 3.1 and flows approximately 21 miles to its confluence with the Columbia River.

The upstream reaches of the Willamette River above Willamette Falls constitute a meandering and, in some cases, braided river channel. Upstream flooding is largely controlled by 13 major tributary reservoirs (Uhrich and Wentz 1999). In the LWR, especially in the vicinity of Portland Harbor, the channel banks have been stabilized by

the placement of riprap, and construction of seawalls, bulkheads, etc. These measures have created a much more stable channel in the LWR.

The portion of the river where the federal navigation channel is maintained at –40 ft Columbia River Datum (CRD) defines Portland Harbor and extends upstream from the Columbia River to RM 11.7 (Broadway Bridge; see Map 1.2-1 and Section 3.1.3.1). From 1973 through 2007, average annual mean flow in the Willamette River was approximately 33,000 cubic feet per second (cfs) at the Morrison Bridge (near RM 12.8) in Portland.[1]

3.1 PHYSICAL ENVIRONMENT

3.1.1 Meteorology

Located about 65 miles inland from the Pacific Ocean, the city of Portland and Portland Harbor are situated near the confluence of the Willamette and Columbia rivers. This area lies approximately 20 ft above sea level and is about midway between the Coast Range to the west and the Cascades Range to the east. The climate of Portland is usually described as temperate or Oceanic, with mild, damp winters and relatively dry, warm summers. The Coast Range provides limited protection from Pacific Ocean storms while the steep slope of the Cascades Range impedes moisture-laden westerly winds, resulting in moderate rainfall in the area, especially during the winter months (NOAA 2010).

Precipitation falls primarily as rain, with nearly 90 percent occurring between mid-October and mid-May. Rainfall varies across the metropolitan area with the West Hills receiving nearly 60 inches of rain per year while the Portland International Airport only receives about 36 inches. Measurable snow accumulations are rarely more than 2 inches, occurring most frequently at elevations over 500 ft (including the West Hills) or along Portland's eastern boundary near the Columbia River Gorge at Troutdale (NOAA 2010). The city has experienced some major snow and ice storms caused by cold air outflow from the gorge. A storm in 1893 resulted in approximately 60 inches of snow accumulation (NOAA 2010).

Winds are from the north and west during the late spring and summer dry season and from the east and south during the fall and winter rainy season. Annual monthly wind speeds average 8.0 mph at the Portland airport (NOAA 2011). Average temperatures range from a low of 45°F in the winter months to a high of the middle 90s in the late summer (NOAA 2000). The lowest temperature ever recorded in Portland was −3°F (−19°C), which occurred on February 2, 1950. The highest temperature ever recorded was 107°F (42°C), on July 30, 1965 and again on August 8 and 10, 1981 (NOAA 2011).

[1] Data obtained from the USGS Water Resources web site (http://waterdata.usgs.gov/or/nwis/sw).

3.1.2 Hydrogeology

The current understanding of the generalized hydrogeology of the Study Area is presented in this section. The detailed hydrogeology of the upland areas on both sides of the river varies by location. This generalized discussion is intended to describe the important basic hydrogeologic units and their properties and groundwater flow within the Study Area and is not representative of any one particular location. An upland groundwater data review that summarizes hydrogeologic information and groundwater quality data from specific upland sites in the vicinity of the Study Area has been completed by LWG (GSI 2003a).

3.1.2.1 Geologic Setting

The Study Area is located along the southwestern edge of a large geologic structure known as the Portland Basin. The Portland Basin is a bowl-like structure bounded by folded and faulted uplands. These northwest-trending structural zones are interpreted as dextral wrench faults that delineate the Portland pull-apart basin (Beeson et al. 1985; Yelen and Patton 1991).

The basin has been filled with up to 1,400 ft of alluvial and glacio-fluvial flood deposits since the middle Miocene (approximately 12 million years ago). These sediments overlie older (Eocene and Miocene) rocks including the Columbia River Basalt Group (CRBG), Waverly Heights basalt, and older marine sediments. The older rocks are exposed where uplifting has occurred on the margins of the basin, including adjacent to the Study Area.

Because the Study Area is located at the edge of the basin, both the older rocks and overlying sediments are present near the surface and play a significant role in defining interactions between groundwater and the river. The geologic units found in the vicinity of the Study Area are illustrated in Figure 3.1-1 and briefly described below, from youngest to oldest (Beeson et al. 1991; Swanson et al. 1993):

- **Recent Anthropomorphic Fill.** Anthropomorphic fill blankets much of the lowland area next to the river and is predominantly dredged river sediment, including fine sand and silty sand. Hydraulic dredge fill was used to fill portions of the flood plain, such as Doane Lake, Guild's Lake, Kittridge Lake, Mocks Bottom, Rivergate, and a number of sloughs and low-lying areas. The fill also was used to connect Swan Island to the east shore of the Willamette River, and to elevate or extend the bank along significant lengths of both sides of the riverfront by filling behind artificial and natural silt and clay flood levee dike structures. Rocks, gravel, sand, and silt also were used to fill low-lying upland and bank areas. The thickness of this unit ranges from 0 to 20 or more feet. The permeability of this unit, where composed of clean dredge fill sand, is higher than the natural fine-grained alluvium. The presence of silt fill or a silty matrix in the sand fill generally reduces the permeability of the unit significantly.

- **Fine-grained Pleistocene Flood Deposits and Recent Alluvium (Undifferentiated).** This unit includes fine-grained facies of the Pleistocene Flood Deposits, as well as recent alluvium deposited by the present Willamette River. This unit generally consists of silt, clay, silty sand, and fine-to-medium sand that borders and underlies the present floodplain of the river (Beeson et al. 1991). The lower portions of this unit and where it forms the large bluffs bordering the east side of the river likely consist of the fine-grained facies of the flood deposits, whereas the upper portions near the river are likely more recent alluvium. The upper fine-grained portion of the unit has likely been reworked and deposited by the present Willamette River. The sands of this unit may be indistinguishable from overlying dredge fill in some places (Landau 2002a). The thickness of this unit ranges from 20 to over 100 ft. The permeability of the clay, silt, and silty sand of this unit is generally relatively low, whereas the portions of the unit consisting of clean sands may have a relatively higher permeability. This unit forms part of the Unconsolidated Sedimentary Aquifer regional hydrostratigraphic unit proposed by Swanson et al. (1993).
- **Coarse-grained Pleistocene Flood Deposits (Gravels).** The gravels include fluvial deposits from the Pleistocene Missoula floods. The deposits fill deep channels that were incised into the Troutdale Formation and CRBG during the floods. The unit consists of uncemented sand, gravel, and cobbles with boulders in places. This unit is generally between 10 and 200 ft thick in the vicinity of the Study Area and underlies fine-grained flood deposits and recent alluvium under much of the Study Area. The Willamette River subsequently incised the flood deposits in places. The rise in sea level from the end of the Pleistocene to the present resulted in the filling of the incised channel by finer-grained flood and recent alluvial facies to form the current floodplain channel of the river.
- **Upper Troutdale Formation.** The upper Troutdale Formation in the vicinity of the LWR includes cemented and uncemented alluvial sand, gravel, and cobbles deposited by the ancestral Willamette and Columbia rivers. The Troutdale Formation comprises the Troutdale Gravel Aquifer hydrostratigraphic unit. This unit is present in some places on the west side of the Study Area to thicknesses of 100 ft and is present along the entire length of the east side of the Study Area at thicknesses of up to 200 ft (Swanson et al. 1993).
- **Lower Troutdale Formation/Sandy River Mudstone.** The Sandy River Mudstone (SRM) is a fine-grained equivalent of the lower Troutdale Formation that overlies the CRBG in the center of the basin and at the margins of the basin away from the axis of the Columbia River. The lower Troutdale Formation/SRM is present in places under the LWR (Swanson et al. 1993) and borders the Portland Hills, but is not considered a significant hydrogeologic unit within the Study Area. The lower Troutdale Formation/SRM consists mostly of silt and clay with lenses of sand and gravel and tends toward fine-grained (low permeability) textures at the basin margins (Swanson et al. 1993).

- **Columbia River Basalt Group.** The CRBG consists of a thick sequence of Miocene basalt flows dating from between 17 and 6 million years ago (mya), but the CRBG flows that underlie much of the Portland Basin entered the area between 16.5 mya and 12 mya. Basalt flows of the CRBG were folded and faulted during the uplift of the Tualatin Mountains, concurrent with eruption and emplacement of younger flows present in the Portland Basin (Beeson et al. 1991). The CRBG is present at the surface or at relatively shallow depths along the west side of the Study Area and may be in direct contact with the river in places. The top of the unit drops off below ground surface (bgs) over a relatively short distance and is 400 or more feet bgs on the east side of the Study Area. The thickness of the CRBG in the vicinity of the Study Area is estimated to be approximately 600 ft (Beeson et al. 1991).

3.1.2.2 Hydrogeologic Units

The geologic units described above can be grouped into Study Area-wide hydrogeologic units on the basis of having generally similar hydrogeologic characteristics. Important hydrogeologic characteristics include the position of the groundwater surface relative to each hydrogeologic unit, the physical relationship between each hydrogeologic unit and the river, and physical characteristics of each hydrogeologic unit, such as permeability, heterogeneity, and anisotropy.

These hydrogeological units are described from uppermost to lowermost in the following sections.

3.1.2.2.1 Fill, Fine-grained Facies of Flood Deposits, and Recent Alluvium (FFA)

The FFA unit is composed of the fill, the combined fine-grained facies of the Pleistocene flood deposits, and the recent alluvium geologic units described by Beeson et al. (1991) and in Section 3.1.2.1. These units were grouped together on the basis of each unit's shared textures and intrinsic heterogeneity, proximity to the river and to each other, and importance with regard to the occurrence of upland groundwater and interactions with the river.

This unit, which encompasses a broad range of soil textures and hydraulic characteristics, blankets much of the lowland area next to the river and includes much of the material abutting the river. The unit consists of the fine sand and silty sand dredge fill overlying recent and Pleistocene silt and clay overbank sediments, which are interbedded with lenses and layers of fine to coarse sand. As discussed in Section 3.1.2.1, the dredge fill was placed behind low-permeability, artificial and natural flood levee dike structures in some locations. The thickness of this unit can be up to 150 ft, but it typically ranges between 30 and 100 ft.

The FFA hydrogeologic unit is the primary unit of importance in defining the interactions between upland groundwater and the river because of the following characteristics of the unit:

- The unit forms most of the river channel within the Study Area as well as the surrounding upland areas and, therefore, controls groundwater interactions with the river
- Most groundwater chemical plumes present in the upland areas occur within strata of this unit.

The distribution of textures and thus groundwater flow properties of the unit vary both vertically and horizontally by location along the Study Area. Silt, clay, and silty sand are present adjacent to the river at a majority of locations where the unit is observed near low river stage levels. Boring logs at sites north of RM 4 on the east side of the river indicate that a greater portion of the unit north of RM 4 and at depths below low river stage levels consists of sand layers. Comparison of hydraulic conductivity values for different textures within the FFA unit listed below illustrates the importance of the channel sand lenses and layers in focusing groundwater fluxes to the river at any particular location where present within this unit:

- Silt/clay: 0.005 to 2 ft per day
- Silty sand: 0.1 to 2 ft per day
- Sand: 0.5 to 30 ft per day.

The typical measured hydraulic conductivities in the silt/clay facies of the FFA indicate that groundwater fluxes from these sediments within the Study Area are generally low. Identification of only a few seeps present in silt/clay during the seep reconnaissance survey (GSI 2003b) is consistent with this conclusion. Conversely, groundwater fluxes from the uplands to the river within the FFA are expected to be greater in those areas where more permeable sand zones are present.

3.1.2.2.2 Coarse-grained Flood Deposits and Upper Troutdale Formation (CGF)

The CGF unit combines the unconsolidated coarse-facies flood deposits, including sands, gravels and cobbles, with the underlying uncemented and cemented gravels and cobbles of the upper Troutdale Formation. The flood gravels that compose the upper portion of this unit typically occupy scour channel surfaces on older units (e.g., the CRBG). Anthropomorphic fill; silt, clay, and sand of the flood deposits; and alluvium mostly cover the CGF, except in places on the highland bluffs on the east side of the river where the unit may be exposed.

The CGF unit is adjacent to and underlies much of the Study Area to thicknesses exceeding 200 ft. The overall thickness of the unit is more typically in the range of 100 ft. However, the unit is missing in places, including on the west side of the river towards the south end of the Study Area and directly under the river at RM 7. The top of the CGF unit is present at elevations of 0 ft to over –100 ft mean sea level (MSL). The unit is present at relatively shallow depths adjacent to the west side of the river in the vicinity of the Doane Lake area and may be in contact with river sediments. The

hydraulic conductivity of this unit measured in the vicinity of the Doane Lake area ranges from 3 ft per day to greater than 40 ft per day (AMEC 2001).

Because this unit has a relatively higher hydraulic conductivity than the overlying FFA unit, groundwater may flow more readily through this unit to deeper units where downward gradients are present and where the unit is present adjacent to the river, allowing deeper groundwater to more readily discharge to the river. Higher fluxes to the river within the CGF unit may increase downward gradients and thus increase groundwater and contaminant plume movement in the FFA unit. The effect of the CGF unit on groundwater flow in the FFA is a factor in the selection of characterization methods. Locations where the CGF unit may exert a stronger influence on deeper groundwater flow to the river, and thus vertical gradients in the FFA, include the Doane Lake area, the southern edge of the Study Area, and on the east side of the river in the vicinity of the International Terminal.

3.1.2.2.3 Lower Troutdale Formation/Sandy River Mudstone

This hydrogeologic unit is present in some places under the west side of the Study Area and is present under the entire length of the east side of the Study Area. The unit is predominantly silt and clay where explored in the vicinity of the Study Area, and thus the permeability of the unit is low. Where present, the unit overlies the CRBG below depths of –100 to –150 ft MSL and tends to pinch out on the west side and towards the southern end of the Study Area where the CRBG is present at shallower depths. The unit typically is separated from the river by at least 100 to 200 ft of alluvium and deposits of the upper Troutdale Formation. Based on the hydrogeologic characteristics of this unit and the depth relative to the river, it is not considered to contribute significantly to surface water/groundwater interactions within the Study Area.

3.1.2.2.4 Columbia River Basalt Group

The CRBG consists of a concordant sequence of basalt lava flows. Groundwater flow in the CRBG is focused along the higher permeability interflow zones and in some areas of fracture-enhanced permeability (e.g., faults). Hydraulic conductivities measured in individual basalt interflow zones in the vicinity of the Study Area range from 1.5 to 10.9 ft per day (AMEC 2001). Hydraulic conductivities measured in CRBG basalt flow interiors at Hanford, Washington, range from 1×10^{-4} to 1×10^{-7} ft per day (Strait and Mercer 1986), illustrating that the basalt interflow zones (flow top and bottom collectively) are the primary groundwater flow pathways in the CRBG.

The CRBG is present at relatively shallow depths along portions of the west side of the Study Area and may be in direct contact with the river in places. The top of the unit is irregular on the west side of the Study Area with channels from scouring by flood events and the ancestral Willamette River. The top of the unit on the west side of the Study Area is between elevation 0 ft and –50 ft MSL north of RM 9, except for an ancestral channel in the vicinity of Doane Lake (Figure 3.1-1). The top of the CRBG slopes down to an elevation of –250 ft MSL or more across the river on the east side of the Study Area. The relief of the unit across the Study Area appears to be due to

structural downwarping towards the center of the basin, and may be accentuated by normal faulting postulated along both sides of the Study Area (Beeson et al. 1991; Beeson 2003, pers. comm.). The overall significance of the CRBG with regard to groundwater/surface water interactions within the Study Area is not well characterized; however, the CRBG is considered to be most relevant to groundwater interactions with the river on the west side of the river downstream of about RM 9 because of its proximity to the river.

3.1.2.3 Groundwater Flow

Up to three general groundwater flow systems of interest are recognized along the Study Area: a shallow (shallow FFA), an intermediate (deep FFA), and a deep (CGF and CRBG) system. A deeper, regional flow system also is present, which includes the CRBG, where it is deep below the river (on the east side of the river), and lower Troutdale Formation/SRM. This deeper, regional flow system is not considered to be important in understanding the interactions between upland groundwater and the river that are relevant to this RI/FS.

At a local level, these divisions between flow systems are likely indistinct in places along the Study Area. Additionally, some investigations have identified further flow system refinements or divisions based on the local hydrogeology. However, the general flow systems described above appear to apply for the majority of the Study Area and provide a general model from which variations can be evaluated on a local scale. Figure 3.1-2 presents the generalized conceptual picture of groundwater flow through these flow systems. This figure supports the following discussions of groundwater flow systems.

3.1.2.3.1 Shallow Flow System

The shallow, unconfined, groundwater flow system along the margins of the Study Area consists mostly of fill and alluvial silt and clay deposits and some medium- to coarse-grained channel sand of the shallow FFA that blankets the lowlands next to the river, as shown in the generalized conceptual image on Figure 3.1-2. At many locations, the shallow flow system is hosted within the lower portion of fine dredge-fill sand and underlying silty sand and silt. The shallow system is recharged by direct precipitation and infiltration, infiltration from the hills on the west side of the Study Area, and exchange with several surface water bodies along the Study Area (e.g., Doane Lake). Groundwater in this system is unconfined. Groundwater level data in the upland areas indicate that there is a downward gradient toward deeper units from the shallow system. Groundwater levels and fluxes in the shallow system are affected by seasonal river stage changes, as well as by diurnal tidal influences. The degree of tidal influence decreases with increasing distance from the river and shallower groundwater depths. Groundwater gradients within the shallow system are generally steep immediately adjacent to the river and flatten out away from the river bank. The shallow flow system discharges to the river either above the river surface as surface seeps, or below the river surface as subsurface discharge, generally in nearshore areas. Because of tidal and

seasonal river stage fluctuations, a given groundwater discharge may express above or below the river surface at different times.

The presence of low-permeability features, such as silt and clay dikes constructed to retain hydraulically emplaced dredge fill, cutoff walls, and retaining walls, may act to impede groundwater flow locally in the shallow system, resulting in higher groundwater levels and steep shallow groundwater gradients near the shore. The presence of preferential pathways (human-made and natural) in the shallow FFA can be a significant, albeit localized, influence on the discharge of groundwater to the river.

3.1.2.3.2 Intermediate Flow System

The intermediate flow system occurs within thicker sequences of the fine-grained alluvial sediments of the FFA. Groundwater in the intermediate system generally discharges to the Willamette River below the river surface to deeper portions of the river (Figure 3.1-2), with discharge focused at the locations where more permeable strata (typically sand) may intersect the river. Horizontal hydraulic gradients within the intermediate flow system tend to be flatter near the river than observed in the shallow system, and thus high river stages and tidal changes may exert a greater influence on fluxes from the intermediate system to the river by further flattening or perhaps reversing the gradient locally.

3.1.2.3.3 Deep Flow System

The deep flow system occurs within the CGF and basalt interflow zones of the CRBG, where the CRBG is present near the surface on the west side of the river. Downstream of about RM 9 on the west side of the river, residual basalt gravels immediately overlying the CBRG have been identified as important hydrogeologic features and potential conduits for groundwater contaminant transport. Groundwater in the deep system discharges to the Willamette River only in deeper portions of the river, with discharges focused at the locations where the gravels and/or basalt interflow zones are near or intersect the river sediments (Figure 3.1-2).

The CRBG does not play a role in the deep flow system on the east side of the river, because it occurs at substantially greater depth due to structural downwarping and associated normal faulting. Deep groundwater flow in the east side of the river occurs in the CGF, which is generally highly transmissive; however, gradients may be relatively low. Seasonal gradient reversals are known to occur during periods of high river stages. Where near the river, the connection, and thus response, to river stage changes is expected to be great.

3.1.2.4 Processes Governing Discharge of Groundwater to the Study Area

Generally, groundwater flow adjacent to the Study Area is toward the river. The Tualatin Mountains (Portland Hills) form a ridge that runs parallel to the Willamette River to the west, from the Multnomah Channel to the City of Portland. The mountains define the western edge of the Portland Basin; groundwater and creeks and channels

along the east face of the mountains flow downward to the Willamette River. On the east side of the river, starting upstream of RM 4, a broad terrace divides the floodplains of the Willamette and Columbia Rivers. Deep groundwater flows are influenced by the Columbia on the east side of the river, with effects increasing as distance from the Willamette River increases. Groundwater gradients are relatively flat in some areas along the east side of the river, due to both underlying geology and the influence of the Columbia River.

In the absence of preferential pathways, groundwater flow to the sediments and river will be diffuse along the length of the interface of each flow system with the river. However, permeability contrasts of several orders of magnitude can be expected in the FFA where alluvial processes create lenses and channels of sand within or surrounding finer-grained materials. The result of these permeability contrasts is that groundwater discharge will tend to be heavily influenced by the location and geometry of higher permeability layers (e.g., sands) in relation to the river.

Discharge from the shallow water-table groundwater system tends to be focused at or below the river/shore interface. Low river stages expose zones of focused discharge as seeps along the bank where the shallow groundwater surface intersects the ground surface. Preferential pathways, including coarse backfill (e.g., around utilities), historic stream channels, or sand/gravel layers focus groundwater flow, particularly where they occur in predominantly fine-grained sediment sequences in the shallow groundwater system. The groundwater flow regimes of all of the flow systems show seasonal patterns related to seasonal river stage and precipitation variations.

The gradient and the resultant flux from these systems vary with seasonal river stage changes. Diurnal tidal stage changes also result in temporary gradient and thus flow changes, particularly where the degree of connection between the river and adjacent aquifer is greater. Groundwater discharge through the river sediments to surface water is controlled by: 1) the permeability contrast between the sediments and underlying aquifer, and 2) the difference between the hydraulic head in groundwater at the aquifer/sediment interface and the river stage, which determines hydraulic gradient.

Direct measurements of groundwater seepage rates to the river were taken during the LWG Round 2 investigation and during the offshore investigation performed by Gasco (Anchor et al. 2007). Locations of these measurements are discussed in Appendix C2. Measurements were taken in nearshore areas as well as farther offshore, including several locations within the navigational channel. In all, 77 ultrasonic seepage meter measurements were taken (70 LWG measurements and 7 non-LWG measurements), primarily during the season of presumed maximum groundwater flux (high upland groundwater levels and low river stage). Daily average measurements ranged from –18.2 cm/day (recharge) to 14.2 cm/day (discharge to the river), with an average of 1.5 cm/day. These measurements were taken in areas of suspected higher groundwater flux, as part of the investigations of upland plume discharges. As such, these values are expected to be higher than the average flux rate for the entire channel. Measured

groundwater flux rates showed substantial variability between measurement sites; in general, the highest seepage rates were observed in sandy areas, and the lower values were observed in less conductive clay zones, as expected. Site-specific groundwater discharge estimates were used to calculate groundwater chemical loads to the river (Section 6 and Appendix E).

In addition to the empirical seepage measurements, Appendix E presents a calculated estimate of groundwater discharge to the river based on Darcy's Law (which describes flow rates through permeable media) and observed upland groundwater hydraulic conductivity values. This calculation estimated total groundwater discharge to the Study Area to be between 4.5 and 10.9 cfs, with an average of 7.3 cfs. This average corresponds to a seepage discharge rate of 0.1 ft/day (3.0 cm/day) across the entire channel surface of the Study Area, which is almost 10 miles long. The Darcy's Law groundwater discharge estimates fall within the range of groundwater fluxes (7 to 85 cfs) modeled for the lower Willamette River based on regional (Portland Basin) groundwater head measurements (Morgan and McFarland 1996).

3.1.2.5 Groundwater/Surface Water Transition Zone

The groundwater/surface water transition zone represents a region beneath the bottom of a surface water body, where conditions change from a groundwater-dominated to surface-water-dominated system within the substrate. It is a region that includes both the interface between groundwater and surface water, as well as the broader region in the substrate (and, on occasion, up into the surface-water body) where groundwater and surface water mix. The transition zone is the interval where both groundwater and surface water comprise some percentage of the water occupying pore space in the sediments. The physical and chemical properties of water within the transition zone reflect the effects of mixing between groundwater and surface water that occurs within the sediments, as well as biological and geochemical processes occurring within the sediment matrix and pore water. The transition zone is the location where chemical and biological transformation processes occur that affect the properties of chemicals that may be present in pore water and sediment, and it encompasses the sediment biologically active zone where benthic infaunal ecological receptors may reside.

The thickness of the transition zone is temporally and spatially variable due to changes in gradients between the surface water and groundwater, sediment texture and stratigraphy, and hydraulic conductivity. In general, areas with steeper groundwater gradients toward the river and/or lower sediment hydraulic conductivities are expected to exhibit thinner mixing zones (absent biological activity or other disturbances), with less diurnal response to tidal fluctuations.

3.1.3 Hydrology

River stage and currents in the LWR and Portland Harbor are influenced by hydrologic conditions in both the Willamette and Columbia rivers, and are further affected by the operations of federal and non-federal dams along these two rivers. River stage refers to

the height of the river measured relative to a specific elevation or "datum." A variety of vertical datums are used in the Portland Harbor region, and these are discussed below.

3.1.3.1 Regional Datums

Current or historical bathymetric and topographic data may be referenced to a variety of vertical datums in Portland Harbor. The bathymetric data collected as a part of this RI/FS are presented relative to NAVD88. This vertical datum is the national standard geodetic reference for heights and was selected for this project because it is a level datum and is easy to use with global positioning systems (GPS). NAVD88 is a fixed datum derived from local mean sea level observations at Father Point/Rimouski, Quebec, Canada. NAVD88 replaced NGVD29/47 as the national standard geodetic reference for heights.

The National Geodetic Vertical Datum of 1929 through the Pacific Northwest Supplemental Adjustment of 1947 (NGVD29/47) is a fixed datum adopted and adjusted in 1947 as a national standard geodetic reference for heights prior to June 24, 1993 and is now considered superseded by NAVD88. NGVD29 is sometimes referred to as Sea Level Datum of 1929 or as MSL on some early issues of USGS topographic quads. NGVD29 was originally derived from observations at 26 long-term tide stations in the U.S. and Canada. Data referencing MSL as the vertical datum in the Portland Harbor is technically based on NGVD29/47.

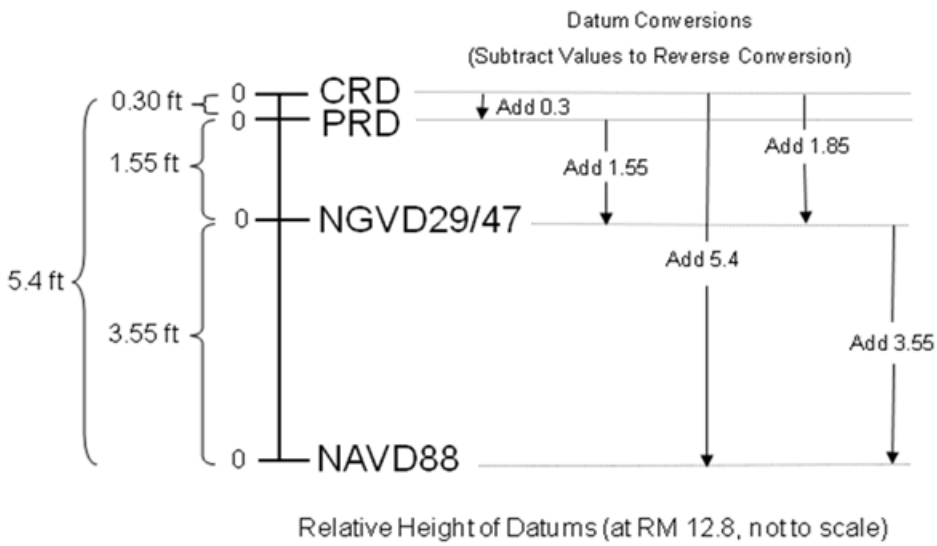
The CRD is used as the nautical chart datum for the LWR. CRD is a reference plane established by the USACE in 1912 by observing low water elevations at various points along the Columbia and Willamette rivers (USACE 1966). Consequently, the CRD is not a fixed/level datum but slopes upward as one moves upstream. The CRD is used upstream of RM 24 on the Columbia to the Bonneville Dam and on the Willamette River to Willamette Falls. Mariners can obtain the depth on a chart and apply tide or river-level gauge readings, relative to CRD, to compute actual water depth at the time of sailing. Low water values are used for navigation charting to provide conservative depth values in the event accurate tide data are not available to the mariner.

NAVD88, NGVD29/47, and CRD are the major datums used on maps and charts of Portland Harbor. In the lower Willamette, elevations reported relative to the CRD are approximately 5 ft less than NAVD88 elevations (e.g., the –15 ft NAVD88 contour on LWG bathymetry maps equates to a –20 ft CRD elevation).

Water level (river stage) data measured by the Morrison Bridge gauge (RM 12.8) are recorded as the Portland River Datum (PRD) and are 1.55 ft above NGVD29/47 (USACE 1991). The CRD is 1.85 ft above NGVD29/47 at the Morrison Bridge. On December 27, 2001, David Evans and Associates, Inc. (DEA) confirmed the relationship between this gauge and the CRD by running a differential leveling circuit from a nearby control monument used in the control network for the Willamette multibeam surveys. This survey confirmed that the Morrison Street staff gauge reports water levels 0.30 ft above CRD, as defined by the USACE (1991).

The river stages discussed below in Section 3.1.3.2.1 are the directly measured Morrison Bridge gauge levels and are therefore reported as PRD elevations in feet. To convert from PRD to CRD, subtract 0.3 ft from the reported river level.

The datum relationships discussed for Portland Harbor above are illustrated below:



3.1.3.2 Regional Surface Water Hydrology

The Columbia River drains a large segment of the northwestern United States and parts of western Canada. The Columbia basin is so large that isolated events such as localized rainstorms may have little or no effect on river flow. In its natural state, high flows on the Columbia River are most influenced by snow melt, which takes place during the spring months. This results in high water typically occurring in late May or early June, followed by receding water levels until the rains begin in late fall.

Lowest water on the Columbia River typically occurs in October or early November, reflecting a lack of precipitation and snowmelt in the basin during the summer months. With the onset of winter rains and snow, runoff will vary during the winter months, until the spring snowmelt leads to the high water period.

The Willamette River is a major tributary of the Columbia River and flows into the river at Columbia River Mile 103. Over the water years 1973 through 2007—a 35-year period of record—the Willamette River average daily mean discharge was 33,000 cfs, while that of the Columbia River above the confluence of the Willamette was 177,000 cfs.[2]

[2] The average daily mean values noted are based on data from the USGS gauge #14211720 (Willamette River at Portland, OR; see Section 3.1.3.2.1 below) and from USGS gauge #14105700 (Columbia River at The Dalles, OR), which is located downstream of the confluence of the Willamette but is the closest USGS gauge for which data are available for water years 1973–2007. These data are available at: http://waterdata.usgs.gov/nwis/sw.

Lowest water in the Willamette, as in the Columbia, typically occurs between September and early November prior to the initiation of the winter rains. Unlike the Columbia River, however, Willamette River flows generally increase in response to regional storms due to the comparatively small size of the basin. Record winter floods (e.g., 1964 and 1996) occurred when periods of heavy snowfall at lower elevations were followed by warming periods and heavy rains, resulting in rapid increases in runoff.

The effect of the multipurpose dams on the Columbia River and its tributaries has generally been to reduce the spring high water flows with retention and storage of the water through the system-wide management of reservoir pool levels at each dam. Beginning in late summer, stored water is released, which increase flows above the naturally occurring low-flow hydrograph. By winter, these reservoirs have been drawn down, and the storage capacity is used to take the peak off of winter flows and to optimize the generation of electricity.

There are 13 federal reservoirs on the Willamette River and its tributaries, having a combined storage capacity of over 1.6 million acre-ft. These reservoirs reduce the river flow during the winter snow and rain events by storing water (Table 3.1-1). With each major storm, water is retained and then released at the end of the storm to dampen hydrographic peaks and valleys. During persistent rainy periods and/or during exceptionally large precipitation events, the storage capacity may be exceeded, and additional flow entering the system leads to flooding, as occurred in 1964 and 1996. During these flood events, water flow in the river can be up to 50 times greater than the flow during low-water periods. Late in the winter, after the likelihood of a major flooding event has passed, the reservoirs are filled to capacity. These reservoirs are used for recreation during the summer and are drawn down in the fall to supplement natural low flows and to provide storage capacity in preparation for the flood season.

Water levels and currents in the LWR can be influenced by the Columbia River in several ways. The most apparent influence occurs during spring when high flows from the Columbia River act as a hydraulic dam to Willamette River, resulting in rises in the Willamette River stage. The Columbia River flow drops as the summer progresses, and this effect is diminished. During the winter, high seasonal flows on the Willamette River can be allowed to pass through to the Columbia River, which may have diminished flows due to retention at dams. This mechanism was used in the 1996 flood to lower the flood stage levels of the Willamette River in Portland.

Tidal action also compounds the hydrology and interplay of the two rivers, and affects the Willamette River upstream as far as Portland Harbor and beyond. Tides along the North American West Coast are mixed semidiurnal (two unequal high tides and two unequal low tides daily), with an average tidal range of approximately 8 ft in the Pacific Ocean. The high (i.e., flood) tide can influence Willamette River levels by up to 3 ft in Portland Harbor when the river is at a low stage. These tidal fluctuations can result in short-term flow reversals (i.e., upstream flow) in Portland Harbor during times of low river stage combined with large flood tides; this effect was measured in May 2003 as

part of the bathymetry survey effort using an acoustic Doppler current meter (DEA 2003b). As river stage rises, the tidal effect is gradually dampened and disappears at river levels around 10 ft CRD.

3.1.3.2.1 Willamette River Stages and Discharges

Figure 3.1-3 shows a plot of the mean daily river stage data from October 1, 1972 through March 31, 2008 at the Morrison Bridge in Portland near RM 12.8 (reported in feet PRD, USGS gauge #14211720).[3] Mean historical daily discharge (cfs) calculations from this gauge are shown in Figure 3.1-4, and Figure 3.1-5 presents the annual average discharges by water year[4] over the period of record. Flow data from October 1972 to September 1994 were computed by the USGS using an acoustic velocity meter (Lee 2002, pers. comm.). Most data after September 1994 are USGS estimates based on measurements from regional stations (Miller 2006, pers. comm.).

The seasonal cycle of water levels on the Willamette River is illustrated in Figure 3.1-3. Annual low water levels occur during the regional dry season from August to November. Winter (November to March) river stage is relatively high but variable due to short-term changes in precipitation levels in the Willamette Basin. Finally, a distinct and persistent period of relatively high water levels occurs from late May through June when Willamette River flow into the Columbia is slowed by high-water stage/flow in the Columbia River during the spring freshet in the much larger Columbia River Basin, as described above. The two highest peaks in the daily mean discharge record occurred in the winters of 1996 and 1997, when peak flows reached 420,000 cfs on February 9, 1996 and 293,000 cfs on January 2, 1997 (Figure 3.1-4).

For the water years 1973 through 2007—a 35-year period of record—the mean annual daily discharge was between 20,000 and 30,000 cfs during 14 years of this period (Figure 3.1-5). Annual mean daily flows were above 30,000 cfs during 19 years, with seven of those years above 40,000 cfs, and three in excess of 50,000 cfs. Only two water years (1977 and 2001) had average daily flows between 10,000 and 20,000 cfs.

Figure 3.1-6 presents the frequency (number of days per year) distribution of daily mean discharge values from the October 1, 1972 through March 31, 2008 data set. Flow on the Willamette River is most often between 10,000 and 30,000 cfs. Approximately 75 percent of the time flows are less than about 40,000 cfs, and exceed 90,000 cfs less than 10 percent of the time.

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[3] Data obtained from Regulation and Water Quality Section Web site (http://www.nwd-wc.usace.army.mil/perl/dataquery.pl?k=id:PRTO+record://PRTO/HG//1DAY/MEAN/) and the USGS National Water Information System Web site (http://waterdata.usgs.gov/or/nwis/uv?14211720). Where USGS data are available, they replaced USACE data for compiling the graphs shown in this section. The USACE site notes that these "data have not been verified and may contain bad and/or missing data and are only provisional and subject to revision and significant change." The data are used here only to illustrate long-term relative trends in the Willamette River stage at Portland. No data are available for 1991 and 1992.

[4] A water year extends from October 1 to September 30 (e.g., October 1, 1972 to September 30, 1973 comprises the 1973 water year).

Figures 3.1-7a–h show river stage data through each of the RI sample-collection years (i.e., 2001–March 31, 2008). For comparison, the graphs also include a plot of average annual river stage values based on the entire period of record (October 1972–March 2008), and plots of the values within one and two standard deviations from the average (representing approximately 68 percent and 95 percent of the recorded values, respectively). The LWR flood stage (18.3 ft PRD [18 ft CRD]) was not reached during the RI period.

Figures 3.1-8a–h present plots of river discharge data through each of the RI years (2001–March 31, 2008), with plots of the average daily discharge (October 1, 1972–March 31, 2008) and values within one and two standard deviations from the average shown for comparison. LWR discharge rates during the RI years followed a typical seasonal pattern and, as with river stage levels, have generally been within the range of typical discharges on record. Early 2001 and early 2005 were relatively low-flow winter/spring periods and early and late 2006 had relatively high flows compared with the long-term averages.

3.1.3.3 Hydrodynamics

Recent investigations of the hydrodynamics of the LWR, Study Area, and Multnomah Channel are summarized in this section. Both empirical information (flow measurements) and HST modeling have been conducted as part of this RI to support the understanding of the physical system and hydrodynamics. Additional work is being conducted as part of the FS. The primary objective of these efforts for the RI was to gain a sufficient understanding of the physical system to support the RI site characterization, the BHHRA, the BERA, and site-wide fate and transport modeling. A central issue for this large river system was the evaluation of sediment stability throughout the Study Area (i.e., where and to what depth in the sediment column are sediments stable and unstable over time under a range of hydrologic conditions, including rare flood events).

3.1.3.3.1 Flows in the Study Area

Flows were measured directly at multiple locations in the LWR using an ADCP during three of the four time-series bathymetric surveys which were conducted to measure riverbed elevation changes over time (see Section 3.1.4.2). The ADCP data provided snapshot observations of flows in the Study Area across a range of flow and tidal conditions (DEA 2002 b, 2003b, 2004). The empirical flow data also supported the development and calibration of a hydrodynamic model developed for this RI/FS (WEST and Integral 2005). The revised Phase 2 HST model (WEST and Tetra Tech 2009) was used here to develop vector plots of current velocities throughout the Study Area during both mid-ebb and mid-flood tides for both high- and low-river-flow periods (Figures 3.1-9a through 3.1-12c). Vector plots were also generated that show current velocities during maximum flood tide coupled with low river flow (Figures 3.1-13a–h) for the entire LWR (to assess the maximum extent of upstream flow reversals) and during high flows in both the Willamette and Columbia rivers (Figures 3.1-14a–h). Lower

Columbia and Willamette flow and stage data are included in footnotes on the vector plots.

The model output shows that currents generally flow downstream during four of the six flow-tide combinations. Reverse or upstream flows occur when river flow is low and the tide is in flood.

In general, flow in many of the relatively shallow nearshore embayments and slips is characterized by eddies and/or inshore flow, except on ebbing tides during low-flow periods, when downstream or offshore flow directions are dominant. As expected, higher current speeds occur in the deeper portions of the river channel, and lower speeds occur in the shallow nearshore areas, regardless of flow direction. Flow in Multnomah Channel is downstream under all flow/tide combinations modeled.

Based on this hydrodynamic model output, at the maximum flood tide during the low-flow period, reversed flows extend upstream to approximately RM 15, where upstream flow velocities are minimal, approximately 0.2 ft per second in the channel (Figures 3.1-13a–h), and are very low upstream of RM 15 to about RM 18.

During high flows on the Willamette and comparable flows on the Columbia (Figures 3.1-14a–h), flow is consistently downstream on the LWR, and the model predicts that there is an apparent eddy effect (reduced circular flows) where the Willamette River flows into the Columbia River.

3.1.3.3.2 Flows to Multnomah Channel

The flow data collected during the ADCP surveys in April 2002, May 2003, and January 2004, and summarized in the Round 1 Site Characterization Summary Report (Integral 2004b) suggested that LWR discharge through Multnomah Channel could be significant, ranging from 25 to 50 percent of the discharge volume of the Willamette during the “snap-shot” ADCP measurement periods. The percentage of Willamette River flow through Multnomah Channel is a function of the relative flow regimes in the Willamette and Columbia rivers, as well as tidal stage.

To investigate Multnomah Channel flows on a more continuous temporal basis, the CE-QUAL-W2 hydrodynamic model of the Columbia River/Willamette River System developed by Portland State University was used to model daily average flows in the system over a nearly four-year period from January 1999 through December 2002. The results of this study were discussed in the Round 3A Upstream & Downstream Sediment FSP (Integral 2006c) and are briefly summarized here and in Figure 3.1-15.

Figure 3.1-15 shows the flows (daily average cubic meters per second) for the Willamette and Columbia rivers and the modeled flows for the Multnomah Channel over the 1,400+ day (approximately 4-year) model run. The figure also shows the fraction of the total Willamette River flow through Multnomah Channel (black line). “Fraction” values greater than 1 indicate that flow down Multnomah Channel exceeds the Willamette River flow upstream of Multnomah Channel (i.e., at these times,

Multnomah Channel flows are a mixture of Willamette River water and inflow from the Columbia River).

The modeling effort identified three distinct river flow combinations and evaluated the proportion of discharge carried by Multnomah Channel:

1. **Low flows in both the Columbia River and Willamette River**—When flows are relatively low in both the Willamette and Columbia rivers, about 50 to 60 percent of the Willamette flow goes down Multnomah Channel.
2. **Low flow in the Columbia River and high flow in the Willamette River**—When relatively high flows in the Willamette River are concurrent with relatively low flows in the Columbia River, the proportion of Willamette River flow carried by Multnomah Channel decreases to about 25 to 30 percent of the total Willamette River flow.
3. **High flow in the Columbia River and low flow in the Willamette River—** When Columbia River flows are high and Willamette River flows are low, the increased river stage at the Columbia/Willamette confluence forces much of the Willamette River flow down Multnomah Channel. At certain low-flow Willamette periods (summer/early fall), all of the Willamette River flow, in terms of daily average volumes, plus some flow from the Columbia River, goes down Multnomah Channel. This last condition occurs about 25 percent of the time over the period modeled (January 1999 to December 2002).

No clear periods of concurrent high flows in both the Willamette River and Multnomah Channel were identified within the nearly four-year model simulation period. Averaged over the study period, flows in Multnomah Channel represent about 60 percent of the Willamette River flow upstream of Multnomah Channel. It should be kept in mind that some of the Multnomah Channel flow is Columbia River water, but the relative volumes of Willamette River versus Columbia River water flowing down Multnomah Channel cannot be determined from these modeling results.

3.1.4 Riverbed Characteristics and Dynamics

Several types of investigations have been conducted as part of the RI to characterize the physical nature of bedded sediments and their potential for movement within and through the LWR due to natural or anthropogenic forces. The results of the early physical system studies (STA, SPI, and time-series bathymetry surveys) conducted from 2000 to 2004 have been documented previously in individual technical reports and were summarized in the Programmatic Work Plan (Integral et al. 2004). These data were used to develop the initial physical CSM for the LWR presented in the Programmatic Work Plan, and this CSM helped direct the extensive sediment (and other media) chemical sampling efforts that were conducted throughout the harbor and the LWR from 2004 to 2008.

As noted above, in conjunction with the collection of empirical information on physical site characteristics, a numerical HST model was developed (Integral 2006b; Integral and WEST 2005; WEST 2004, 2005, 2006b; WEST and Integral 2005; WEST and Tetra Tech 2009). The primary RI objective of this modeling effort was to predict the potential impact of extreme (flood) events on site sediments, particularly the potential for buried contaminated sediments to be re-exposed. A secondary objective was to provide physical surface water and sediment flux data for the fate and transport modeling being conducted as part of the FS.

The sections that follow provide an overview of the major physical system site information, including sediment characteristics, bathymetric trends, site-specific data collected to support the HST model, a summary of the key HST model outputs, and finally, a description of the major sediment transport regimes based on this body of empirical and modeling information. This description represents a refinement of the physical CSM presented in the Programmatic Work Plan and Round 2 Report.

3.1.4.1 Physical Characteristics of Sediments

Physical sediment sample texture data (grain size, specific gravity, total solids) and TOC have been collected as part of all sediment sampling for the RI. Sediment texture data are also available from non-LWG sampling efforts conducted in the LWR.

As discussed in Section 2.1.2, the interval from 0–30 cm bml was used to define surface sediments within the LWR. This surface sediment interval definition was based on the empirical bathymetric change data, which indicate that most changes to the riverbed (e.g., erosion and deposition) occur within this interval under typical conditions within the Study Area (see Section 3.1.4.2.2 below). Below 30 cm, LWG-collected subsurface core samples were processed such that major discontinuities in sediment texture (e.g., sand/silt) were used to define subsurface sample breaks (see Integral, Anchor, and Windward 2004).

The grain-size data measured in surface sediment samples in the nature and extent database[5] were used to generate contour maps of surface sediment grain size (as percent fines; i.e., coarse silt [63 µ] and finer) and TOC (percent) in the Study Area (Maps 3.1-2 and 3.1-3). In the absence of anthropogenic activities that affect sediment textures, the physical characteristics of surface-bedded sediment are general indicators of the energy regime of the riverbed at that location. Typically, fine-grained sediments (silts, clays) dominate in relatively low-energy environments where current velocities are low enough to allow fine particles to settle out of the water column and remain deposited, whereas coarse sediments (sands, gravels) are indicative of higher-energy environments where fines are kept in suspension in the water column and/or winnowed out of previously deposited material and transported away during transitory high-energy events (e.g., floods or anthropogenic disturbances, such as prop wash, dredging, etc.).

[5] The majority of surface samples in the nature and extent database were collected from intervals within 30 cm of the mudline; however, a limited number of non-LWG samples were collected from 0 to 40 cm bml. These were included in the surface sample data set as these samples include the near-surface sediment texture.

Starting at the upstream end of the Study Area (RM 11.8), Map 3.1-2 shows that coarse-grained surface deposits (i.e., 0–20 percent fines) are predominant from upstream of the Study Area downstream to RM 11, especially along the western half of the river. The river gradually widens from RM 11 to 10, and this area is a mosaic of mostly sandy (21–40 percent fines) and mixed (41–60 percent fines) textures, and deeper holes and nearshore areas and embayments dominated by fines (61–100 percent fines).

The river widens markedly from RM 10 to 7, and surface sediments are dominated by fines, with the exception of some nearshore bank areas and some discontinuous areas along the western edge of the navigation channel. The finest texture sediments (81–100 percent fines) are widespread from RM 9 to 7, including locations within Willbridge Terminal, in the downstream lee of Swan Island (Portland Shipyard), and in Swan Island Lagoon.

From about RM 7 to 5, the river and navigation channel narrows again, and this reach is dominated by sands with relatively small subareas (e.g., within Willamette Cove and western nearshore around RM 6) that are dominated by fines characteristic of lower energy environments. Much of the remainder of the Study Area and beyond, to about RM 1.5, is dominated by fines, with a texture of 61–80 percent fines dominant upstream of Multnomah Channel (RM 3–5) and 81–100 percent fines widespread downstream of Multnomah Channel (RM 1.5–3). Conversely, the relatively shallow and narrow Multnomah Channel is dominated by sands, as is a portion of the Study Area upstream and immediately adjacent to the Multnomah Channel entrance extending to the east bank. This is the largest area in the LWR between RM 1.5 and 5 that is not dominated by fines.

As expected, the TOC content of the surface sediments (Map 3.1-3) generally mirrors the sediment grain-size distribution, with higher TOC content collocated with the finer-grained deposits. TOC levels generally range from 0.5 to approximately 3 percent, but a few isolated areas contain higher levels (6 to up to 27 percent); these are all downstream of RM 7 and include the head of Willamette Cove, an area west of the main channel from RM 6.2 to 6.4, a mid-channel area at RM 5.7, and a relatively large area east of the channel at RM 2.

Vertical gradients in grain size can be examined visually across the Study Area by comparing Map 3.1-2 (contoured surface sediment texture) with Map 3.1-4 (contoured sediment texture for the shallow subsurface horizon; i.e., subsurface intervals beginning at 30 cm). Overall, the surface and immediate subsurface sediment textures are consistent across the Study Area, suggesting that the energy regimes in the system are relatively stable over time. There is, however, a subtle but perceptible widespread shift from finer-grained surface sediments to a slightly coarser-grained subsurface layer (e.g., from 81–100 percent fines to 61–80 percent fines) across much of the site. This may reflect seasonal or inter-annual winnowing of the finer sediments from the sediment bed during higher flow periods and the subsequent long-term burial of the slightly coarser

residual sediments. Finally, there are three areas that show distinctly coarser surface sediments overlying finer material; these include the head of Swan Island Lagoon, the M&B/Willamette Cove area, and the area outside the entrance to Multnomah Channel, extending into the channel itself. Anthropogenic placement of fill and the recent sand cap appear to explain this pattern in Swan Island Lagoon and around M&B/Willamette Cove, respectively. The vertical shift to finer material at depth immediately adjacent to and within the mouth of the Multnomah Channel is not as apparent, but the "relict" muds may reflect the less dynamic sedimentary environment that existed in this portion of the river prior to the Portland Harbor navigation channel dredging and other land use modifications in the region (e.g., bank treatments).

3.1.4.2 Bathymetric Survey Data

Four major multibeam bathymetric surveys were conducted by the LWG to measure LWR riverbed elevation for the RI/FS: January 2002, July–September 2002, May 2003, and February 2004. A fifth, smaller survey focused in Multnomah Channel was conducted by the LWG in February 2007. The data were processed using a 1-m grid size to generate a digital terrain model, and the survey results were plotted in both three-dimensional (3-D) color-graded (i.e., "hillshade") and contour formats. Except for the Multnomah Channel survey, the results of each survey were compared to those from preceding surveys to analyze net bed elevation changes and change trends over time. The results of the four major surveys and the time-series change analyses have been reported previously (Integral et al. 2004; Integral 2004b). The Multnomah Channel survey data are reported in the Round 3A Upstream and Downstream Data Report (Integral 2007d).

In addition, a multibeam survey of the LWR was conducted by NOAA in January 2009, and this survey data was obtained by the LWG and compared with the 2002 baseline data to produce a 7-year bathymetric change data set.[6] This new data set is presented and discussed below.

3.1.4.2.1 Bathymetry

Map 3.1-5 shows the most recent LWR riverbed and upper Multnomah Channel bathymetry data. This bathymetric survey in the LWR was conducted in January 2009 by NOAA and the data obtained by the LWG. The Multnomah Channel bathymetric survey was conducted in February 2007 as part of the Round 3A sampling effort. The primary goal of this survey was to obtain high resolution riverbed elevation data in the channel.

Map 3.1-5 shows that most of the Study Area is from –30 to –50 ft CRD (–25 to –45 ft NAVD88) and is dominated by the authorized federal navigation channel, which runs from RM 0 (Columbia River) to RM 11.7 (Broadway Bridge) and extends nearly bank-to-bank from RM 4 to 6 and again from RM 8 to 11.7. Elevations in the federal

[6] This 2009 survey was conducted by DEA under contract to NOAA. DEA was the contractor that conducted the Portland Harbor bathymetric surveys for the LWG, and the survey equipment, set-up, and data processing techniques used in 2009 were very similar and fully compatible with methods used previously.

navigation channel are generally –40 to –50 ft CRD. Except along the western channel edge from RM 8 to 10 where extensive shoaling has occurred, these portions of the Study Area have very narrow and steeply sloped off-channel areas. Broader off-channel areas with shallow benches (–10 to –30 ft CRD) occur from RM 1 to 4 along the outside curve of the river, including across the head of Multnomah Channel, between RM 6 and 8, and at the head of Swan Island Lagoon. There are a number of off-channel areas, such as the outer portions of Swan Island Lagoon, Willbridge Terminal, Willamette Cove, and Terminal 4, that vary widely in depth as a function of their history and current land use (e.g., actively dredged berths). Finally, there are several deep areas in the harbor that extend from –60 to –80 ft CRD. These are historical borrow areas that were dredged to create the adjacent uplands; the two most extensive ones are in the eastern portion of the channel from RM 4.3 to 5 and RM 9.2 to 10. Map 3.1-6 shows the long-term bathymetric changes that occurred in the LWR between 1888 and 2001 and illustrates the large-scale deepened, diverted, and filled areas.

3.1.4.2.2 Riverbed Elevation Changes

Map 3.1-7 shows the net bathymetric change over the seven-year period between the first (January 2002) survey and the January 2009 survey in the LWR. The elevation change maps were created by overlaying the 1-m cells from each survey and subtracting the January 2009 data from the January 2002 data (the depth values are generally negative numbers, e.g., –15 ft NAVD88) to generate a direction and magnitude of change for each cell. The vertical resolution of the multibeam survey overlay was ±0.25 ft (approximately 7.6 cm), so cell comparisons that show positive or negative changes less than or equal to 0.25 ft represent no discernable change in riverbed elevation.[7] On Map 3.1-7, positive elevation changes (shallower in 2009 compared to 2002) indicate shoaling, and negative elevation changes (deeper in 2009 compared to 2002) indicate deepening. The no-change areas are shaded gray, while shoaling areas (positive change) are shown in yellow to orange shades, and areas that deepened (negative change) are shown in blue shades.

Previous evaluations of the time-series bathymetric change data between each survey period from 2002 to 2004 (i.e., 2002 to 2003, 2003 to 2004, and 2002 to 2004) showed generally consistent results and are discussed in the context of the LWR physical system in the Programmatic Work Plan (Integral et al. 2004; Integral 2006f). Key observations and conclusions from those evaluations are listed below:

- Over the 25-month period of observation, only 10 percent of the riverbed exhibited net bathymetric changes (erosion or accretion) greater than 1 ft (30 cm), but relatively small-scale scour or accretion from 0.25 ft (8 cm; the limit of resolution) to 1 ft (30 cm) in extent was widespread.

[7] The survey vertical accuracy specification of ≤0.5 ft was exceeded for both individual surveys (DEA 2002, DEA 2004). An analysis of bathymetric change data indicated that the vertical resolution of the survey overlay was ±0.25 ft for approximately 80 percent of the channel data. Therefore, this interval was used as the no-change category (Integral and DEA 2004).

- The reaches between RM 5 and 7 and RM 10 and 11.8, where the river is relatively narrow, are dominated by areas of small-scale net erosion, as is the western off-channel area from RM 0 to 3 (outside bend of the LWR as it turns toward the Columbia).
- Wide areas of deposition occur in the channel and along channel margins in the broader sections of the river (RM 1.5 to 3 [eastern margin], RM 4 to 5, and RM 7 to 10). The downstream and upstream areas mentioned in the previous sentence are known to be long-term sediment accumulation areas based on historical dredging records.
- Signs of in-filling are apparent in formerly dredged borrow areas (e.g., RM 5.2, RM 9 to 10, and RM 10.5 to 11.9).
- Many deepening areas are closely associated with berthing areas, slips, and pier structures (e.g., Terminal 4 riverfront dock, Portland Shipyard, Willbridge Terminal), likely the result of anthropogenic factors, such as prop wash from ships and dredging. Since 1997 dredging has occurred at Port of Portland Terminals 2, 4, and 5; the Willbridge Terminal; the CDL Pacific Grain Irving Elevator; the Glacier NW dock; the former Goldendale Aluminum dock; the International Terminals; the BP West Coast Products Terminal 22; the Vigor Industrial dock; the City Fire Bureau Station 6 dock; the Portland Cement Terminal; and the Ash Grove Cement Rivergate Lime Plant. The details of known dredging projects dating back to 1997 are presented in Section 3.2.5.2.
- Map 3.1-7 shows the net bathymetric change from 2002 to 2009 from the Columbia River (RM 0) to the upper end of Ross Island (RM 15.3). This time frame includes a winter (late 2005-2006) with a prolonged period of relatively high flows approaching 200,000 cfs. Spatial patterns noted above for the Study Area hold for the 2002 to 2009 time period also for this larger reach. Over this longer time frame, nearly three-quarters of the surveyed area (72 percent) shows positive or negative elevation changes of less than 1 ft (30 cm). Overall, shoaling is the predominant change observed. The extent of sediment accumulation in shoaling areas increased, with 22 percent of the riverbed surveyed showing accretion exceeding 1 ft (30 cm), whereas erosion exceeding 1 ft is noted in only 6 percent of the riverbed, including dredged areas. More importantly, and as noted above, the locations subject to scour, accretion, or no change within the Study Area remain consistent with those inferred from the two-year data set over this longer time frame.

3.1.4.2.3 Sediment Erodibility Measurements

Sediment erosion rates and critical erosion shear stress values for LWR sediment were measured directly as part of the HST model data collection effort conducted in the spring of 2006 (Integral 2006b). This study involved the collection of 17 cores from locations throughout the Study Area selected to represent a range of bottom conditions in terms of sediment texture and local hydrodynamic conditions. The primary use of these data was to support the sediment transport modeling effort, and these data and

their incorporation into the HST model are detailed by WEST and Tetra Tech (2009). These data are discussed here for their empirical value as a measure of riverbed erodibility of surface sediments across the Study Area in late March 2006.

The sediment cores were subjected to various flows using a Sedflume system to produce a range of shear stresses (a force applied parallel or tangentially to a surface; from 0.1 Newtons [N]/m^2 to 10 N/m^2) to the sediment surface. Resulting critical erosion flow velocities and erosion rates were measured at approximately 5-cm intervals to depths of approximately 25 cm. Physical properties of bulk density and grain-size distributions were also analyzed at approximately 5-cm intervals. Erosion rates per shear stress applied varied depending on sediment grain size, bulk density, and sediment depth. A summary of the number of applications per shear stress value and the range of observed erosion rates (in cm/s, depth of sediment eroded per unit time) on all Sedflume cores is presented in the following table.

Shear Stress (N/m^2)	Measurements Count	Erosion Rate (cm/s) Min	Erosion Rate (cm/s) Max
0.1	16	0	0.0003
0.2	23	0	0.0003
0.4	55	0	0.001
0.8	74	0	0.04
1.6	76	0.0002	0.1
3.2	76	0.0003	0.3
6.4	60	0.007	0.4
10	2	0.02	0.04

Critical erosion velocity shear stress values (defined in the Sedflume method as the shear stress at which erosion occurs at 10^{-4} cm/s; Sea Engineering 2006) were calculated at approximately 5-cm intervals. Median grain size (d50) values for the sediment intervals ranged from 9.7 µm (medium silt) to 401 µm (medium sand), and critical shear stresses (T_{cr}) were calculated to range from 0.06 N/m^2 to 1.28 N/m^2. These data from the SedFlume cores, summarized by core depth interval, are tabulated below.

Sample Depth Category	d50 (µm) Min	d50 (µm) Max	T_{cr} (N/m^2) Min	T_{cr} (N/m^2) Max
0–5 cm	9.7	401	0.06	0.64
5–10 cm	12	367	0.32	1.28
10–15 cm	10	378	0.22	1.28
15–20 cm	7.8	384	0.26	1.28
20–25 cm	10.9	357	0.24	1.28

The Phase 1 Hydrodynamic model (WEST and Integral 2005) was used to predict bed shear stresses that would occur in the LWR under typical low-flow (e.g., 40,000 cfs)

and relatively infrequent high-flow (e.g., 160,000 cfs) conditions.[8] Map 3.1-8 shows modeled bed shear under these low- and high-flow conditions. Under the low-flow conditions, bed shear values are predicted to remain below 0.4 N/m^2 throughout most of the channel and below 0.1 N/m^2 in the nearshore areas. Slightly higher shear stresses (up to 0.7 N/m^2) are predicted for the channel near RM 11 and for the head of Multnomah Channel. As a first-order approximation, these data indicate that significant sediment bed movement or resuspension due to natural hydrodynamic forces does not occur under the typical flow conditions that take place over much of the year (i.e., less than 50,000 cfs) in the LWR.

Under the relatively rare high-flow conditions, the predicted bed shear values remain low in most nearshore areas, slips, and embayments but are much higher, as well as more variable, in the channel. The predicted bed shear values in the main channel range from 0.614 N/m^2 between RM 2 and 2.3 to the maximum value of 19.7 N/m^2, which occurs in the channel at approximately RM 10.3. The highest values (>5.0 N/m^2) occur in both the nearshore and channel areas in the more constricted reaches (e.g., between RM 10 and 11, and again between RM 5 and 7; Map 3.1-8). The predicted high-flow bed shear values in the channel approach or exceed the highest critical shear stress calculated from the Sedflume study (1.28 N/m^2) throughout much of the Study Area, indicating that sediment transport is likely to occur throughout much of the channel during this flow condition.

3.1.4.3 Suspended Sediment

Suspended sediment loads and dynamics are an important component of the LWR physical system, and TSS data have been collected both as part of the surface water characterization and hydrodynamic model data collection efforts. Suspended solids data were collected in November 2004, March 2005, and July 2005 as part of the Round 2A surface water characterization effort, and during sampling conducted from November 2005 to April 2006 to directly support the hydrodynamic modeling effort (Table 3.1-2). The Round 2A surface water data, collected during the fall of 2004, winter of 2005, and summer of 2005, were reported previously in the Round 2A Surface Water Site Characterization Data Report (Integral 2006j). The results of conventional parameter analyses (TSS and TOC) from the November 2005 to April 2006 surface water sampling efforts (summarized in Table 3.1-3), and recently provided TSS data from the City of Portland, are presented in this section. Section 5.3.5 provides additional details on the RI suspended sediment sampling efforts and data compilation.

3.1.4.3.1 November 2005 to April 2006 LWG Sampling

Suspended solids concentrations were analyzed in a time series (November 2005 to April 2006) of vertically and horizontally integrated composite water samples from upriver of the Study Area, which were collected to verify the hydrodynamic model's

[8] Mean daily flows of approximately 160,000 cfs or more were recorded on 119 days (0.9 percent) over the 30-year period of record and on 14 days (0.5 percent) over the RI water years 2001 through March 31, 2008. Mean daily flows of 40,000 cfs or less were recorded on 9,374 days (74 percent) over the period of record and on 2,031 days (77 percent) over the RI water years 2001 through March 31, 2008.

suspended sediment/discharge rating curve, and from samples collected in April 2006 from points downstream of RM 11 and in Multnomah Channel, to support hydrodynamic model calibration (Integral 2006f).

The data from the sampling events are presented in Tables 3.1-3 and 3.1-4. TSS concentrations at the upriver stations ranged from 7 to 50 mg/L over the measurement period. Although the data set is relatively small, a plot of the upstream TSS data against the discharge hydrograph shows the expected pattern of higher concentrations on the rising limb of the hydrograph peak (i.e., 50 mg/L on December 22, 2005 at a discharge rate of 67,700 cfs) compared with the falling limb (i.e., 49 mg/L on January 19, 2006 at a discharge rate of 169,000 cfs; 39 mg/L on February 3, 2006 at 139,000 cfs; and 25 mg/L on February 7, 2006 at 108,000 cfs; Figure 3.1-16).

Available upriver TOC data (from March 3 and April 4, 2006 samples) show organic fractions of total solids in the water column remaining relatively consistent, ranging from 2.2 to 2.4 mg/L. In the same samples, TSS concentrations decrease from 21 to 22 mg/L at a flow of 41,500 cfs in March to 9 mg/L at a flow of 28,000 cfs in April, perhaps suggesting a higher inorganic suspended solids fraction with increased discharge. This would be expected; however, the particulate organic carbon (POC) was not directly measured.

Samples were collected twice per day in early April 2006 at stations in the Study Area and in Multnomah Channel, once on the rising tide and once on the falling tide. TSS concentrations in these samples ranged from 7 to 12 mg/L, and TOC concentrations ranged from 1.8 to 2.4 mg/L (Table 3.1-4). Because of the low variability of the results and limited number of data points, there is no clear pattern for the concentrations based on location or tidal phase; however, TOC values were slightly higher at the RM 11 and RM 6.3 stations than at the RM 2 and Multnomah Channel stations.

In situ suspended particle sizes were measured at HMV01 through HMV05 (RM 2, 6.3, 11, and 18) in early April 2006 using a laser *in situ* scattering and transmissometer (LISST) as part of the hydrodynamic model data needs collection (Integral 2006c). Particle size was measured in 0.5-m increments through the water column. The median grain-size measurements with depth at each station are plotted in Figure 3.1-17, and a summary of the grain-size ranges measured is tabulated below. As indicated by the data, particles primarily in the silt and fine-to-medium sand size ranges were in suspension, under a flow of approximately 30,000 cfs. The coarsest median grain sizes were found upstream of the harbor at station HMV05 (RM 18) where the river is relatively narrow.

		Maximum Grain Sizes per Station		
Location	**Station**	**d10(µm)**	**d50(µm)**	**d90(µm)**
RM 2 West Side	HMV01	4.28	25.76	204.36
RM 6.3 East Side	HMV02	4.1	40.66	274.53
RM 11 West Side	HMV03	3.32	35.28	242.34
RM 11 Mid-Channel	HMV04	2.98	32.11	242.91

		Maximum Grain Sizes per Station		
Location	**Station**	**d10(µm)**	**d50(µm)**	**d90(µm)**
RM 18 Mid-Channel	HMV05	3.78	79.3	383.11

Note:
d10, d50, d90 = diameter of the 10th, 50th, and 90th percentiles of the grain-size distribution.

3.1.4.3.2 Long-Term City Data

The City of Portland has provided TSS data collected from the Willamette River at locations between RM 1.1 and 20 from February 1992 to March 2008 (City of Portland 2008d). Data from the RI years (2001 to March 31, 2008) within the Study Area are available only from the City's RM 6.8 sampling location. These data are presented against the USGS discharge data for the water years 2001 through March 2008 in Figures 3.1-18a–h.

Flows on the Willamette River are typically 40,000 cfs or less (Figure 3.1-6). Figure 3.1-18i combines the long-term RM 6.8 TSS data set and plots suspended load versus discharge. This plot shows that there is a strong positive relationship between TSS and flow from discharges levels of 40,000 cfs and above ($r^2 = 0.75$); below that discharge level the measured TSS levels are not correlated with flow ($r^2 < 0.01$).

The lack of a trend in observed TSS values with flow during relatively low discharge periods is not unexpected. Based on the hydrodynamic model, significant sediment bed movement or resuspension due to natural hydrodynamic forces does not occur under flows of less than 50,000 cfs. In the absence of significant bed material resuspension, a variety of natural and anthropogenic factors could potentially play a dominant role in TSS concentrations during low-flow periods, such as precipitation events occurring locally or farther up the watershed, plankton blooms, prop wash, outfall discharges, etc. The influence of plankton growth on TSS concentrations could be expected to be seen in seasonal patterns shown by the data; however, no seasonal trend is apparent (Figure 3.1-18j). Plankton data collection was not part of the RI, and the relationship between TSS and plankton was not part of the scope of the surface water objectives.

3.1.4.4 Hydrodynamic and Sediment Transport Modeling

As noted previously, a numerical HST modeling effort was conducted as part of the Portland Harbor RI to complement the empirical observations and gain a further understanding of physical system dynamics. A primary objective of this modeling for the RI was to predict the potential impact of extreme (flood) events on site sediment stability, particularly the potential for buried contaminated sediments to be re-exposed. Other objectives include understanding the complex hydrodynamics (i.e., the movement of surface water) of the LWR system (e.g., see Section 3.1.3.3) and providing preliminary surface water and sediment flux data to the fate and transport modeling effort being conducted for the FS over a range of flow regimes. The initial fate and transport modeling effort is detailed in Anchor et al. (2007).

Development of the HST model began in 2003, and the model has been through several development phases with EPA coordination and input. The RI HST modeling work is detailed in a series of documents (Integral 2006b; Integral and WEST 2005; WEST 2004, 2005, 2006b; WEST and Integral 2005), and the final revised RI Phase 2 HST modeling report has been provided under separate cover (WEST and Tetra Tech 2009). Key aspects of the model, important developmental milestones, site-specific data collected to improve model performance, and major model sediment transport outputs are summarized in the sections that follow. The revised RI Phase 2 HST model was modified in 2009 by the LWG FS consultant Anchor QEA to address difficulties encountered in the sediment transport calibration and to improve the ability of the model to support FS objectives. The refined model will be presented in the FS.

The LWR HST model uses the Environmental Fluid Dynamics Code (EFDC). EFDC is a public domain, multifunctional, surface water modeling system, which can include hydrodynamic, sediment-transport, and eutrophication components. EFDC has been used for more than 80 modeling studies of rivers, lakes, estuaries, coastal regions, and wetlands in the United States and abroad.

The EFDC model's sediment-transport component is capable of simulating the transport of multiple size classes of cohesive and non-cohesive sediment (Tetra Tech 2002). A sediment processes function library allows the model user to choose from a wide range of currently accepted parameterizations for settling, deposition, resuspension, and bedload transport. The sediment bed is represented by multiple layers and includes a number of armoring representations for noncohesive sediment and a mixed bed material finite-strain consolidation formulation for dynamic prediction of bed-layer thickness, void ratio, and pore water advection. The sediment-transport component can operate in a morphological mode, with full coupling between the hydrodynamic components, to represent dynamic evolution of bed topography. Water column/bed exchange processes include particulate deposition and resuspension, pore water entrainment, and pore water advection and diffusion.

3.1.4.4.1 Phase 1 Modeling

Following EPA approval of the Modeling Approach Technical Memorandum (WEST 2004), Phase 1 of the RI modeling, including model setup, an analysis of model sensitivity, and initial model calibration and validation runs for both hydrodynamics and sediment transport were conducted (WEST and Integral 2005) and revised (WEST 2005). The Phase 1 revisions incorporated refinements identified in EPA's review of the initial Phase 1 results, as well as site-specific sediment data collected in Round 2 of the Portland Harbor RI/FS in the latter half of 2004. The primary objective of the Phase 1 modeling was to determine if a two-dimensional (2-D) model would be adequate for the site, in terms of addressing model objectives. Due to the relatively small tidal influence in the LWR and the general lack of a significant density structure (WEST 2004), Phase 1 concluded that a 2-D model was adequate. The secondary Phase 1 modeling objective was to gain an understanding of the site's physical processes and the impact of various model parameters on the model predictions. Based on the model

sensitivity and performance analyses, additional potential site-specific data needs were identified.

Overall, the Phase 1 model effectively simulated the hydrodynamics. However, bed elevation changes were not well captured by the model at the target accuracy levels. As a result, a number of site-specific data needs related to improving the sediment transport performance of the model were identified and collected in 2006. The data needs collection effort for the model is described below.

3.1.4.4.2 Hydrodynamic/Sediment Transport Model Data Collection

The Phase 1 HST modeling effort identified a suite of site-specific data types that would potentially improve the performance of the sediment transport portion of the model, and thus the ability of the model to achieve the specific modeling objectives (WEST and Integral 2005). In general, these data needs were associated with the behavior of cohesive sediments in the system (e.g., settling velocities and erodibility). The hydromodeling data needs FSP (Integral and WEST 2005) included the following field sampling and laboratory analysis tasks:

- Measure TSS at an upstream location just below the confluence of the Willamette and Clackamas rivers (~RM 24.7) across a range of flow conditions
- Measure TSS in the Study Area (at RM 11, 6.3, and 2) and in the Multnomah Channel under both relatively low- and high-flow conditions across a tidal cycle
- Determine fine sediment (less than 63 micron diameter) settling velocity under both relatively low- and high-flow conditions in the Study Area (between RM 2 and RM 11) and upstream of the Study Area (at RM 18) using a transmissometer
- Determine erosion rates and critical erosion shear stress with depth in the sediment column from sediment cores collected at 17 locations throughout the Study Area (Sedflume system)
- Collect duplicate cores from the Sedflume sample locations for the laboratory analysis of bulk sediment properties (grain size, bulk density, and TOC) with depth in the sediment column
- If the proper hydrologic conditions are present (i.e., flows on the Willamette approach or exceed 100,000 cfs), perform a short-term, time-series bathymetric survey; TSS sampling; and fine-grained settling velocity measurements in the Study Area.

The FSP was implemented in April 2006. All target data sets were collected in accordance with the FSP with the exception of the short-term, high-flow bathymetric changes. These data could not be collected because a qualifying hydrologic event did not occur in the spring of 2006. The HST data collection effort and data sets are detailed in Integral (2006b) and the Round 2 Report (Integral et al. 2007).

These site-specific data were incorporated in the Phase 2 modeling effort, which is summarized below. The detailed Phase 2 modeling efforts are provided in WEST and Tetra Tech (2009).

3.1.4.4.3 Phase 2 Modeling

In Phase 2, the HST model was revised and recalibrated using the site-specific modeling data collected in 2006 (described above). The revised model computation domain extends from the confluence with the Columbia River (RM 0) to the confluence with the Clackamas River (RM 24.1), and the Multnomah Channel to its confluence with the Columbia River near St. Helens, Oregon (WEST 2006b). The upstream boundary of the Phase 2 recalibration model was shifted to approximately 2.4 miles downstream of the Willamette Falls (RM 26.6), which was the upstream boundary in the Phase 1 model (WEST 2006b).

The recalibration focused on identifying a combination of the reference critical shear stress for deposition, reference resuspension rate, reference critical shear stress for resuspension, and reference void ratio to minimize the differences (both statistically and graphically) between the measured and simulated bed change over the calibration period.

Compared to the revised Phase 1 results, the Phase 2 model showed some improvement in the agreement between simulated and measured bed elevations by incorporating site-specific data. The model did a better job in the deeper portions of the river than the nearshore areas. This is expected as sediment transport in nearshore areas might also be affected by other factors (e.g., local flow features near structures and prop wash) that are not explicitly represented in the model. The revised Phase 2 calibration results are detailed in WEST and Tetra Tech (2009).

In 2009, the LWG fully modified the sediment transport portion of the HST model in order to meet FS objectives. The outputs of this fully calibrated HST model on erosion risk during a 100-yr flood event on the LWR are discussed below and have been incorporated into the physical CSM descriptions included in this RI Report.

3.1.4.4.4 FS HST Model Application – Flood Simulation

The primary HST model application for the RI is the evaluation of the risk of contaminated subsurface sediment re-exposure due to a major flood event in the LWR (i.e., the future risk scenario).

The 2009 revised, fully calibrated FS HST model was used to predict the bed elevation changes (i.e., the areas and magnitude of erosion and deposition in the Study Area) that would result from five different high-flow scenarios. A range of high-flow simulations were run because bed response can be a function of the long-term hydrographic conditions that exist leading up to the flood event. Figure 3.1-19 shows the simulated hydrograph for the flood event that produced the largest overall riverbed elevation changes; note that this hydrograph includes a simulation of the 1996 flood following

five years of high flow. Map 3.1-9a shows the net bed elevation changes, both erosion and accretion, following the simulated high-flow event. For comparison, Map 3.1-9b shows the maximum erosion levels predicted for each model cell during this simulated high-flow event; this map is a mosaic of maximum erosion per cell at any point during the simulation, and so shows the maximum extent of erosion for each cell regardless of backfilling that might occur on the falling limb of the hydrograph. The flood event maximum erosion map (Map 3.1-9b) shows that 38 percent of model cells in the Study Area undergo erosion at any point during the simulated flood; most of these cells (85 percent) undergo erosion of 30 cm (1 ft) or less. Overall, 6 percent of Study Area cells undergo maximum erosion greater than 30 cm (up to 192 cm) and, therefore, are predicted to exceed the 30-cm project-defined surface sediment layer during the modeled flood event. This erosion of deeper, "subsurface" sediments (greater than 30 cm to 192 cm) is localized in three regions of the Study Area (see the three darkest shades of blue cells in Map 3.1-9b):

- The navigation channel from RM 10 to 11.8, particularly upstream of RM 10.7
- The navigation channel from about RM 5.2 to 6.8 and adjacent eastern nearshore zone cells between RM 6.1 and 6.7
- An isolated cell in the eastern nearshore at RM 3.1.

These more deeply eroded areas correspond to areas that are predominantly sandy in texture and are also coherent with the areas that the empirical time-series bathymetry data show to be erosional (or non-depositional) under typical seasonal flows.

Beyond these areas, the flood simulation predicts that most areas of erosion will occur within the navigation channel. Much of the navigation channel will experience erosion on the order of 15 to 30 cm. Most nearshore or off-channel (e.g., Swan Island Lagoon) areas are not predicted to erode. Similarly, several portions of the navigation channel (i.e., RM 1.7 to 3.0), most of RM 4 to 5, and the western half of the channel from RM 7.3 to 9.2, also are not predicted to be erosional. It should be noted that any predicted erosion brings deeper sediment into the 30-cm surface sediment layer defined for this RI.

The degree and areal extent of backfilling of eroded areas predicted by the HST model during the high flow event can be estimated by comparing Map 3.1-9b (maximum erosion) with Map 3.1-9a (net change). The relatively deeply scoured areas from RM 11 to 12 and from RM 5.7 to 7 show minimal backfilling over the course of the event. In the other areas where erosion occurs, such as in the channel from RM 8 to 11, between RM 3 and 5, and downstream of the Study Area between RM 1 and 2, the final net change map indicates that backfilling occurs in these areas by the end of the simulation, with some cells showing net accumulations following the initial scour. Inspection of the cell-by-cell model outputs indicates that across all cells in the Study Area which experience some erosion during the high flow simulation, about one third (34 percent) show no or very little (<2 cm) backfilling from their maximum erosion levels, approximately half (50 percent) show less than 6 cm of backfilling, nearly

three-quarters (74 percent) experience less than 30 cm of backfilling, and about 7 percent of the cells show more than 100 cm of sediment accumulation following some initial erosion.

3.1.5 Sediment Transport Regimes (Physical Conceptual Site Model)

In the deeper, offshore areas of the harbor (i.e., the navigation channel and adjacent areas in the main stem of the LWR deeper than about –20 ft NAVD88, see Map 3.1-5), the movement of water and sediment appears to be controlled in large part by the physical shape of the river, both the cross-sectional area and anthropogenic alterations such as borrow pits, dredged areas, and structures (e.g., bridge footings). In the off-channel, nearshore areas, especially areas less than –20 ft (NAVD88) in depth, the sediment dynamics are complicated by local riverbank morphology, seasonal changes in water levels, bank treatments, and other anthropogenic factors such as prop wash. Map 3.1-10 shows several cross-sectional channel profiles from RM 1 to 13 and illustrates the variability of the river morphology through the Study Area. The cross-sectional profiles include both the 2002 (blue) and 2009 (red) bathymetry and show where relatively large-scale deposition (mostly) and erosion have occurred. Select sediment-profile images from the 2001 survey are included on Map 3.1-10 to show how river bed surface textures and sediment shear strength (as indicated by the depth of the SPI camera prism penetration; SEA 2002b) vary in accordance with the river's cross-sectional area and depositional setting. Finally, the plan view 2002 to 2009 bathymetric change data (Map 3.1-7) is included as the background layer on Map 3.1-10.

Map 3.1-11 shows predicted (EFDC hydrodynamic model) bottom shear forces in the LWR from the RM 24 (the upstream end of the revised HST model domain) to the Columbia under a relatively high flow regime (160,000 cfs); this was the flow condition observed in the LWR in late January 2004 when the Columbia River was relatively low. This map shows the predicted bottom shear forces of the Study Area in relation to the bottom shear forces in the upriver reaches. With the exception of the area from approximately RM 15 to 17, Map 3.1-11 shows that narrower upriver areas from RM 12 to 24 experience much higher near-bottom shear forces than RM 12 to the Columbia (Portland Harbor).

The physical system data collected throughout the Portland Harbor RI, summarized in the preceding sections, support and expand upon the physical conceptual site model described previously for the LWR (Integral et al. 2004; Integral 2007a). Table 3.1-5 summaries some of the key hydrodynamic/sediment transport characteristics of the LWR by major reaches with a focus on the distinct variations observed in subsections of the Study Area. The hydrodynamic character and sediment transport regimes of the LWR may be broadly described in terms of the ten reaches discussed in the following subsections.

3.1.5.1 Upriver Reaches

There are two reaches upstream of the Study Area that are summarized in Table 3.1-5 and described below:

- Upriver Reach (RM 15.3 to 26)[9]
- Downtown Reach (RM 11.8 to 15.3).

3.1.5.1.1 Upriver (RM 15.3 to 26)

The upriver segment includes the stretch of the river from Willamette Falls to the upstream end of Ross Island (approximately RM 26 to 15.3). Here the river is relatively narrow and flows through suburban areas under largely natural conditions, with the exception of the control structure (USACE Locks) at the Willamette Falls (approximately RM 26). Much of the river bottom consists of exposed basalt bedrock (GeoSea Consulting 2001). Bed shear stresses through this area are generally high (averaging 5.8 N/m^2), with the highest shear stresses occurring in the bend between RM 23 and 24 (>40 N/m^2), Map 3.1-11. Sustained current speeds in this reach appear to prevent all but the coarsest material from settling in the main stem of the river. Some low to moderate shear stresses occur in the smaller bifurcated channels, embayments, and sheltered nearshore areas. The most extensive relatively low-energy area occurs at the downstream end of this reach from approximately RM 15 to 17 and includes the river channel that runs behind (east of) Ross Island; predicted shear stresses here range from 0.4 to 4 N/m^2. This portion of the LWR (including the downtown reach) appears to most closely match the physical conditions observed in much of the Study Area under this flow regime (Map 3.1-11).

3.1.5.1.2 Downtown Reach (RM 11.8 to 15.3)

The downtown segment of the LWR extends from the upstream end of Ross Island (RM 15.3) to the upstream end of the Study Area at RM 11.8. Like the upriver reach, this is also a relatively high-energy segment of the river, where the main channel of the river is narrow (average cross-sectional area estimated at 34,000 ft^2) with steep channel margins that are largely constrained by upland bulkheads along both riverbanks. The deepest areas of the channel are found on the outer edges of bends in the river below Ross Island, and in the dominant bifurcation channel west of Ross Island. The –40 ft CRD authorized federal navigation channel does not extend into this reach (Map 3.1-5). Relatively high bed shear stresses (averaging 3.4 N/m^2) occur in the main portions of the channel, while lower shear stresses occur in the channel east of Ross Island and in shallower nearshore areas associated with some bends in the river (Map 3.1-11).

The high-energy environment of the main channel is evidenced by the observed bedded sediment texture, which consists primarily of gravels and sands (SEA 2002). Localized areas of exposed bedrock occur, particularly near bridges where scouring appears related to footing structures (GeoSea Consulting 2001). Fine-grained deposits are observed in some nearshore areas sheltered from the main flow of the river (SEA 2002b). The SPI image from RM 12.4 illustrates the high energy setting of this area,

[9] The upstream end of the upriver reach is defined hydrodynamically by the Willamette Falls at RM 26. In Section 7 of this RI, the upstream end of the upriver reach is extended to RM 28.4 in order to incorporate some surface sediment data collected upstream of the falls as part of EPA's 2007 Blue Heron & West Linn Paper Mill Site Investigations in the upriver/background data set.

showing an apparently small-scale transgressive, well-sorted fine to medium brown sand bedform overlying and advancing over a poorly sorted gray, silty, fine sand (Map 3.1-10; SEA 2002b). The 2002 to 2009 bathymetric change data show limited sediment accretion throughout this reach, particularly downstream of RM 14, where areas showing no change and small-scale deepening (≤1 ft) are dominant (Integral 2004b).

3.1.5.2 RI Study Area

The Study Area extends from RM 1.9 to 11.8 and the LWR -40 ft CRD authorized federal navigation channel nearly overlaps it, extending upstream from the Columbia River to RM 11.7 (Broadway Bridge). The varying physical/hydrodynamic conditions observed and measured within the Study Area during the RI support its classification into the following six separate segments that share characteristics (Table 3.1-5):

- RM 10 to 11.8
- RM 9.2 to 10
- RM 6.9 to 9.2
- RM 5 to 6.9
- RM 3 to 5
- RM 1.9 to 3.

Map 3.1-12 juxtaposes on a single panel the contoured surface grain-size patterns, the measured bathymetric change from 2002 to 2009, and the HST-predicted net riverbed elevation changes following a major flood event for the Study Area. The overlap of certain elements of these features across the Study Area helps support the discussions provided below.

3.1.5.2.1 RM 11.8 to RM 10

The cross-sectional area of the river begins to increase in this segment as the river broadens in a downstream direction, but the hydrodynamic energy in this segment of the Study Area remains relatively high (Maps 3.1-10 and 3.1-11) and comparable to the upriver reaches (e.g., see high-flow bed shear values in Table 3.1-5). This is evidenced by the high potential bed shear stresses, particularly in the eastern portion of the main channel where the channel bank is steep (Map 3.1-5), and by the observed bed sediment texture, which is dominated by sand (Map 3.1-12). The lower bed shear stresses predicted to occur along the eastern channel flank at RM 11.5 at the Goldendale Aluminum facility (Map 3.1-11) is supported by the historical dredging that has been required to maintain that facility's docking berth (CH2M Hill 2001).

The off-channel, nearshore areas of this reach are narrow, and show a nearly equal proportion of small-scale deepening, shoaling, and no-change areas (Integral 2004b). The channel through this reach has generally undergone minor net deepening over the study period (on the order of 30 cm [1 ft], or less), though small areas have deepened more substantially. Deposition on the order of several feet has occurred in the deep

areas of previously dredged holes (borrow pits) on the western side of the channel (Map 3.1-12). These are the farthest-upstream areas of significant net deposition in the LWR surveyed bathymetrically (i.e., from the Columbia River to the upper end of Ross Island) as part of the Portland Harbor RI/FS. Sand waves are evident migrating along the western portion of the channel between RM 11 and 11.7 (Map 3.1-7).

The flood scenario predicts areas of deep (>100 cm) erosion occurring in some central portions of the navigation channel between RM 10.7 and 11.6 in this reach (Map 3.1-9b), but deposition reduces the extent of net deepening, or even dominates, in other portions of the channel, and dominates in nearly all off-channel areas (Map 3.1-9a).

3.1.5.2.2 RM 10 to 9.2

The river becomes predominantly depositional as it widens significantly around RM 10. The increase in cross-sectional area reduces flow velocities, as reflected by the lower predicted bed shear stresses (Table 3.1-5), particularly along the broad western flank of the channel (Map 3.1-8), and the widespread sediment accumulation measured in this area (Map 3.1-9a). An extensive shoaling on the order of 60 to 150 cm (~2 to 5 ft) in extent is evident along the broad western flank of the channel here. Observed bed sediment textures reflect the cross-channel energy differences, with coarser-grained deposits dominating the eastern portion of the riverbed and finer-grained deposits occurring along the western portion (Map 3.1-12). The SPI image taken at RM 9.3 (station STA66F; Map 3.1-10) shows the riverbed to be composed of a thin silt layer overlying well-sorted medium sand, evidence that this eastern nearshore location undergoes alternating periods of sediment transport, when the fines are winnowed from the sands, followed by quiescent periods that allow deposition of the silt (SEA 2002b). The model predicts erosion to depths of approximately 30 cm in the navigation channel in this reach during the flood event (Map 3.1-9b), but net results show deposition dominating in the nearshore areas and reducing or eliminating the net erosion in some parts of the channel (Map 3.1-9a).

3.1.5.2.3 RM 6.9 to 9.2

This reach is the broadest segment of the Study Area with a relatively wide cross-sectional area (Map 3.1-10), estimated at an average of 63,000 ft^2, and moderate to low bottom shear stresses. The reach is dominated by fine-grained surface sediments (Maps 3.1-10 and 3.1-12). The depositional nature of the majority of this reach is seen in the areas of shoaling observed in the channel between RM 7.8 and 9.2 and along the eastern (directly downstream of Swan Island) and western channel-edge areas downstream to RM 6.9, between 2002 and 2009 (Map 3.1-7). Maintenance dredging has been required historically along the western shoreline of this reach. The large off-channel areas in this reach (e.g., Swan Island Lagoon, Willbridge Terminal) are characterized by very low bed shear but little or no sediment deposition (Map 3.1-12). Isolated areas of deepening observed in the lagoon are likely the result of anthropogenic factors such as prop wash. Dredging of sediments along the Willbridge Terminal piers occurred between winter 2002 and winter 2009 (Map 3.1-7).

3.1.5.2.4 RM 5 to 6.9

The river again narrows in this reach to an average cross-sectional area of approximately 57,000 ft^2 (Map 3.1-10). This stretch of river is a relatively high-energy sediment transport zone with high-flow bed shear rates (4.2 N/m^2) that approach the values predicted upstream of RM 10. Predicted maximum bed shear stresses are moderate to high (Map 3.1-11).

The high-energy nature of this segment of the river results in predominantly sandy surface sediments (Map 3.1-12). Examples of this are illustrated in the SPI photos in Map 3.1-10. The riverbed surface in the SPI snapshot from RM 6.9 (STA 47C) is composed of fine to medium, brown sand; the tan silt lenses within the sand matrix are evidence of active sediment transport (SEA 2002b). The riverbed seen at RM 5.5 (STA 36B) is composed of poorly to moderately sorted, fine to medium sand, and also appears to be undergoing sediment transport (SEA 2002b).

The 2002 to 2009 bathymetric change (Map 3.1-7) shows that the channel in this reach is a mosaic of no change, small areas of sediment accumulation (mostly associated with channel depressions), and some small-scale scour. Localized areas of exposed bedrock have been noted, particularly on the west side of the river near the St. Johns Bridge. Sand wave migration is evident along the central portion of the channel between RM 5 and 6. Outside the channel, the narrow eastern nearshore area is dominated by small-scale scour, while the narrow western nearshore zone shows small-scale sediment accumulation.

The modeled flood scenario predicts relatively deep (61 to 152 cm) erosion occurring in portions of the thalweg and, to a lesser extent (up to 61 cm), in adjacent channel margin areas (Map 3.1-9b). Some of the deeper portions of the navigation channel downstream of RM 5.5 and some off-channel areas show net deposition following the high-energy event (Map 3.1-9a). This includes the outer portions of Willamette Cove and narrow swaths along the eastern and western nearshore areas from RM 5 to 5.5 and up- and downstream of RM 6.

3.1.5.2.5 RM 3 to 5

The river widens again below RM 5 to an average cross-sectional area of 65,000 ft^2 (Map 3.1-10). The bathymetry is dominated by a deep (up to –70 ft NAVD88) dredged area on the eastern half of the channel between RM 4 and 5, which gradually shoals to the typical -40 ft depth CRD downstream of the International Terminal Slip. The time-series bathymetry indicates that the majority of the riverbed in the main channel undergoes minor net shoaling (30 cm or less) with swaths of more significant sediment accumulation along east and west channel edges and nearshore areas, especially between RM 4 and 5 (Map 3.1-7). The isolated areas of scour that are evident in some nearshore areas are likely due to anthropogenic factors; some dredging is also evident at the Port's Terminal 4 slips. The hydrodynamic model predicts low to moderate bed shear stresses, with relatively lower bed shear in the deeper upstream portion of this river segment and along the channel margins (Map 3.1-11).

Surface sediments are dominated by silts with some exceptions. The International Terminal Slip is mostly sand, apparently due to dredging or other anthropogenic factors (Map 3.1-12). A cross-channel swath at RM 3.2 leading into Multnomah Channel is also dominated by sandy surface sediments.

The modeled flood scenario predicts erosion on the order of 30 cm in portions of the navigation channel and in the channel margin and nearshore areas downstream of RM 3.4 (Map 3.1-9b). Up to 61 cm of erosion is predicted for an isolated cell in the eastern nearshore at RM 3.1. However, deposition during the flood event is predicted to reduce or eliminate the net erosion observed in many cells in this reach (Map 3.1-9a). Model results show net deposition exceeding 30 cm in much of the channel and nearshore area from RM 4 to 5, and to a lesser extent in the RM 3 to 4 segment, including small depositional zones along the western nearshore area just upstream of Multnomah Channel and just upstream of RM 3 in the eastern nearshore zone (this is a predicted nearshore shoal that continues to RM 1.5).

3.1.5.2.6 RM 1.9 to 3

As noted in Section 3.1.3.3.2, a significant fraction of the downstream LWR flow moves down the Multnomah Channel; the reduced LWR discharge volume downstream of Multnomah Channel results in markedly reduced bottom shear in the LWR (Maps 3.1-8 and 3.1-11). In addition, the main stem of the LWR continues to widen in this reach as it bends to the northeast, to an average cross-sectional area of approximately 67,000 ft^2 (Map 3.1-10). Maximum bed shear values are very low, particularly on the inside curve of the bend (Map 3.1-11). This is the lowest energy main channel reach in the Study Area. This is reflected in the observed surface sediment texture, which is predominantly fine-grained, and in the shoaling observed in the channel and east of the channel throughout this reach between 2002 and 2009. The area to the west of the channel boundary in this reach shows little net change over this time period.

The modeled flood scenario predicts very little erosion, with deposition dominating the area virtually from bank to bank (Maps 3.1-9a and b).

The downstream boundary of this reach (RM 1.9) is the lower end of the project-defined Study Area and so is jurisdictionally rather than hydrodynamically defined. Hydrodynamically, this reach appears to extend downstream to approximately RM 1.6.

As noted above and in Table 3.1-5, with the exception of the RM 10 to 11.8 and 5 to 6.9 reaches, the 10-mile Study Area is dominated by relatively fine-grained, low energy reaches where sediments are accumulating. Using the bathymetric change data discussed in Section 3.1.4.2, estimates of the mass change (in metric tons, MT) across the entire Study Area were generated by calculating the total volume of either accretion or erosion and overlaying the surface sediment texture data on those volumes (assuming densities of 0.7 g/cm^2 for cohesive sediments, silts and finer, and 1.2 g/cm^2 for non-cohesive sediments, sands and coarser). These estimates were generated for the 6-year

period from 2003 through 2009, as well as subsets of that time frame. Table 3.1-6 shows the results of these calculations for the entire Study Area. The overall depositional character of the Study Area is evident from these estimates, with estimate of mass accumulation averaging approximately 2 x 10^5 MT on an annual basis from 2003 to 2009 (Table 3.1-6).

3.1.5.3 Downstream Reaches

There are two reaches downstream of the Study Area that are briefly described below:

- Downstream Reach (RM 0 to 1.9)
- Multnomah Channel (from the LWR to the Sauvie Island Bridge).

3.1.5.3.1 Downstream Reach (RM 0 to 1.9)

The remaining river segment downstream of the Study Area extends to the Willamette's confluence with the Columbia River. Bed-shear stresses are low to moderate, increasing from about RM 1.6 downstream as the river narrows and becomes more dynamic as it reaches the Columbia (Map 3.1-11). Net shoaling (to 60+ cm) was observed along the eastern channel edge and east of the channel to around RM 1.5. (This is a continuation of the pattern seen upstream of RM 1.9; this is the furthest downstream extent of significant sediment deposition in the LWR channel.) Net deepening (60 cm or less) occurred from 2002 to 2009 in a narrow strip outside the channel along the western nearshore area, particularly in the final one mile of this reach, possibly representing natural channel migration along the outside bend of the river.

Surface sediments transition from silts to sands at approximately RM 1.5 and remain predominantly coarse-grained to the Columbia. The modeled flood predicts erosion to occur throughout most of this area, generally up to 30 cm, but up to 61 and 152 cm in areas downstream of RM 1 (Map 3.1-9b). Some net deposition is predicted to occur in this reach in the center of the navigation channel just upstream of RM 1 and along the eastern shoreline (Map 3.1-9a).

3.1.5.3.2 Multnomah Channel (LWR to the Sauvie Island Bridge)

Multnomah Channel between the LWR and Sauvie Island Bridge sees relatively high flows and bottom shear forces. The channel is notably shallower than the main stem of the LWR, and the flow moving down the channel is constricted (Map 3.1-5). Sandy sediments dominate the channel and the area immediately adjacent to it in the LWR (Map 3.1-12). Time-series bathymetric change data from 2002 to 2009 is not available for the Multnomah Channel. The modeled flood scenario includes the entrance and uppermost portions of the Multnomah Channel and indicates little or no change in the riverbed elevations in this area.

3.1.6 Habitat

The majority of the Study Area is industrialized, with modified shoreline and nearshore areas. Wharfs and piers extend out toward the channel, and bulkheads and riprap

revetments armor portions of the riverbank. Active dredging has produced a uniform channel with little habitat diversity. However, some segments of the Study Area are more complex, with small embayments, shallow water areas, gently sloped beaches, localized small wood accumulations, and less shoreline development, providing some habitat for a suite of local fauna.

The City of Portland updated its natural resource inventory of the Willamette River in 2008 (City of Portland 2008d). The natural resource inventory update (NRIU) is intended to be a refinement of the information presented in two prior documents: the *Willamette River Inventory: Natural Resources* (Adolfson et al. 2000) and Metro's (Portland-area regional government agency) inventory of regionally significant fish and wildlife habitat published in 2005 (Metro 2005). To date, the City has updated the proposed draft natural resource inventory of the North Reach of the Willamette River and plans to update the Central and South Reaches in the future. The proposed draft NRIU defines the North Reach as the 12 miles (RM 0 to 12) of river extending from the Broadway Bridge to the Columbia River, which encompasses all of the Study Area (RM 1.9 to 11.8). It is a proposed draft and subject to revision.

The inventory accounts for special habitat areas (SHAs), defined as areas that contain or support special status fish or wildlife species, sensitive/unique plant populations, wetlands, native oak, bottomland hardwood forests, riverine islands, river delta, migratory stopover habitat, connectivity corridors, grasslands, and other unique or unusual habitat features (Map 3.1-13; City of Portland 2008d). It also includes federally designated critical areas for salmonids. The inventory qualitatively ranked riparian corridors and wildlife habitat areas by applying a geographic information systems (GIS) model that evaluated vegetation and water-related information. Wildlife habitat areas were ranked based on connectivity to patches, connectivity to water, interior area, and patch size. Riparian corridor function was ranked based upon six classes of attributes, including riparian movement corridor, large wood/channel dynamics, food web, flow/flood storage, microclimate/shade, and bank function/water quality (City of Portland 2008d). The City of Portland incorporated information collected by Adolfson et al. (2000) in the most recent draft version of the NRIU and has published maps depicting the relative ranks of wildlife habitat areas and riparian corridor function (Map 3.1-14). This inventory is currently a proposed draft inventory and subject to further review and revision. In the City's NRIU (2008d), all in-water habitat is designated as an SHA for salmonids in the North Reach and is therefore ranked high. However, this ranking is not a representation of habitat quality—the inventory does not evaluate features that determine the variability of aquatic habitat or function to the same extent that it does terrestrial habitat—but an indication that the river supports federal- or state-listed species.

This section describes the general types and quality of aquatic habitat available to ecological species in the LWR.[10] The habitats for each ecological receptor group are described in greater detail in the BERA (Appendix G).

3.1.6.1 Open-Water Habitat

The LWR is characterized by a developed navigation channel and shoreline. The LWR historically had large amounts of off-channel habitat in the form of floodplain lakes such as Ramsey, Doane, and Guild's lakes. After industrialization, only a few shallower backwater sites (e.g., Willamette Cove, Swan Island Lagoon, individual slips), as well as a tributary (Columbia Slough) and a secondary channel (Multnomah Channel) remain (Map 3.1-15). The deep open water provides foraging habitat for fish and wildlife that feed in the water column. Piers and other structures in the open water provide additional habitat for certain species such as smallmouth bass. Shallow-water habitats provide refuge for juvenile salmonids and other fishes, as well as greater foraging opportunities for birds and mammals. Friesen et al. (2004) found that juvenile salmon were present in every month sampled from May 2000 to July 2003. Juvenile salmon were captured more frequently during winter and spring than during other seasons. Coho and steelhead were generally present only during winter and spring.

Historically the LWR was dominated by shallow water habitat, with approximately 80 percent of the river with depths less than 20 ft CRD. Dredging and alteration of the river channel have reversed these ratios, and the river is now 20 percent shallow water and 80 percent deep (Map 3.1-16; City of Portland 2009b). Shallow-water habitats, such as those preferred by some foraging wildlife (e.g., otter and mink), are now largely limited to the narrow strip between the shoreline and the navigation channel, which generally is vulnerable to disturbance and anthropogenic alteration due to its proximity to shore. Remaining pockets of shallow water habitat include areas such as Willamette Cove, Swan Island Lagoon, International Terminals Slip, Wheeler Bay, Shaver, Balch Creek Cove, Triangle Park, the mouth and channel of Multnomah Channel, and the Sauvie Island shoreline.

There are three types of benthic habitats in the open water of the LWR:

1. Unconsolidated sediments (sands and silts) in the deeper water (greater than approximately 20 ft CRD) of the navigation channel and lower channel slopes
2. Unconsolidated sediments (sands and silts) in shallow water depths (less than 20 ft CRD) in gently sloping nearshore areas (e.g., beaches and benches) and on the upper channel slopes
3. Developed shoreline (e.g., rock riprap, sheet pile, bulkheads, piers).

In addition, very limited areas of rock and rock outcrop are present in the LWR. The navigation channel habitat is subject to variable (daily [tidal], seasonal, and annual)

[10] Habitat information was compiled from observations made during field investigations conducted for the RI and risk assessments, as well as supporting studies conducted by others and historical information, as cited.

hydrodynamic forces, the impacts of navigation, natural sediment deposition, bed load transport/erosion, and periodic navigational dredging. These forces vary spatially through the system, largely as a function of the channel cross-sectional area, resulting in the presence of both relatively stable and unstable sedimentary environments and patchy infaunal and epibenthic communities that are characteristic of the local physical regime. The physical sedimentary regimes are a function of the local riverbank morphologies, and sheltered areas away from anthropogenic disturbance should support well-developed infaunal invertebrate communities that are characteristic of large river systems. Conversely, exposed nearshore areas, particularly around berths, docks, and boat ramps, likely have limited benthic communities due to the greater physical disturbance in these areas. Tidal and seasonal water level variability and nearshore disturbances (e.g., boat wakes) have a much larger effect in shallow water than they do in deeper water. The hard surfaces of the developed shoreline provide habitat for an epibenthic community.

Groundwater discharge to the river represents a minor contribution to the hydrologic characteristics of the river and the associated habitat. Groundwater has a limited effect on the ecological uses of the shallow, nearshore habitat except where contaminants from industrial activities have been released in upland areas and have been transported to the river. These effects are further evaluated in the BERA.

3.1.6.2 Bank and Riparian Habitat

The most common bank types occurring in the Study Area are riprap, sandy and rocky beach, unclassified fill, and seawall (Map 3.1-17).[11] In 2008, the City of Portland reported that vegetated riprap (25 percent), unclassified fill (21 percent), and beach (23 percent) were the dominant bank types in the North Reach (Broadway Bridge to the Columbia River; City of Portland 2008d). The bank types classified in the NRIU were identified based on physical characteristics and were not associated with a specific range of shoreline elevations. The riprap or rocky bank type is usually fairly steep with no or very narrow adjacent shallow water habitat present. These areas are usually exposed to heavy wave action and strong currents. The sandy bank type with little to no vegetation is characterized by gently sloped beaches (i.e., sand banks are rarely steep). However, this bank type is often adjacent to steep riprapped shorelines or developed uplands that are frequently exposed to heavy wave action and faster moving water. The rocky or sandy bank types with a mix of native and invasive vegetation are common within the Study Area. These bank types range from gently to steeply sloped beaches and, similar to the sandy bank type without vegetation, are often adjacent to steep uplands, although the uplands are either of sandy or rocky substrate. The rocky or sandy bank types are generally located in areas with less development and a lack of bank hardening, such as in Swan Island Lagoon, the Multnomah Channel, Kelley Point Park, and Sauvie Island.

[11] Classifications on Map 3.1-17 are based upon an ODFW 2000–2003 study (Vile and Friesen 2004) and are known to be outdated or incorrect in some locations.

The type of riverbank present in the Study Area is expected to influence fish species occurrence and use of a given area. Riverbanks with large woody debris and riparian vegetation that provides cover and creates small shallow pools will likely be used by juvenile salmonids and other small fish species (Bjornn and Reiser 1991, Sedell and Froggatt 1984). Areas with limited wood accumulations include the beach adjacent to Freightliner Corp., Kelley Point Park, and Mar Com. Friesen et al. (2004) found that in the LWR, coho preferred beach habitat and rock outcrops and avoided riprap and artificial fill, and the abundance of all species was low at seawall sites. The riprap and rocky substrate are the preferred habitats of sculpin and smallmouth bass (Farr and Ward 1992, SEA et al. 2003, Wydoski and Whitney 2003). Sculpin are predominately present in the shallow water habitats, and smallmouth bass are present in areas with moderate current. The shallow backwater pools and slow-moving areas of the river provide habitats for juvenile largescale suckers (yearling and subyearling) and peamouth (Wydoski and Whitney 2003). The peamouth remains nearshore during winter months and moves to deeper waters in the summer months. The shallow waters with abundant plants and woody debris available for cover are the preferred habitats for largemouth bass.

Numerous aquatic and shorebird species such as cormorants and spotted sandpipers use the habitats in the LWR. The upland environment near the LWR is primarily urban, with fragmented areas of riparian forest, wetlands, and associated upland forests (Map 3.1-18). Historical development and filling of channels and wetlands has left only small strips or isolated pockets of riparian wildlife habitat, with the exception of areas such as Harborton Wetlands, Oaks Bottom, Forest Park, and Powers Marine Park. Therefore, although isolated wildlife habitat areas along the LWR corridor exist, linkages to the larger landscape, such as Forest Park, are limited to a few areas. Forest Park, the largest of these upland habitat areas, is generally isolated from the LWR by the industrial corridor with the exception of a few small controlled watercourses. The barrier presented by the industrial corridor is unlikely to significantly inhibit the movement of birds between the river and the upland forest; however, it poses a significant barrier to the movement of other types of wildlife, such as amphibians and small mammals, which, as a result, experience limited or no connectivity to the river.

The City of Portland updated its NRIU in 2008, and it is intended to update both the documentation and mapping of the regional Habitats of Concern identified in Metro's inventory for the North Reach (2005), as well as the City's earlier inventory (Adolfson et al. 2000, City of Portland 2008d). The River Plan North Reach Discussion Draft (City of Portland 2008b) identified 20 "restoration sites," which were areas along the river that currently could provide important ecological functions, and that could be enhanced through additional restoration. The 2008 NRIU identified 20 SHAs within the North Reach (which encompasses the Study Area), including the entire Willamette River; portions of the Columbia Slough; Johnson Creek; Tryon Creek; urban nesting sites, such as bridges and chimney roosts; bluff areas; grasslands at Powell Butte; native oak assemblages; bottomland hardwood forests; and wetlands (City of Portland 2008d). SHAs were mapped at a coarse level and include areas identified as federally designated

Critical Habitat. The SHAs are not to be interpreted as areas of high quality habitat; rather, they are an indication of habitat features or geographic resources.

Potential general wildlife habitat areas in the Study Area are shown on Map 3.1-19. These include the sites identified in the Adolfson et al. (2000) inventory or were based on field observations made during the shorebird habitat reconnaissance (Windward 2004, pers. comm.) or site bathymetry. In the City of Portland's earlier version of the Willamette River corridor natural resource inventory (Adolfson et al. 2000), 15 sites of habitat value for fish, reptiles, amphibians, and wildlife were identified. These habitat sites are known to be utilized by numerous aquatic birds and semi-aquatic mammals. Notable habitat sites in the Study Area included the South Rivergate corridor at the north end of the Study Area, the Harborton forest and wetlands, Willamette Cove, the railroad corridor, and the Swan Island beaches and lagoon on the southern end (Adolfson et al. 2000). Other habitat sites identified in the general area were Kelley Point, at the confluence of the Willamette and the Columbia rivers, and the Ross Island and Oaks Bottom Complex around RM 16.

3.2 HUMAN ENVIRONMENT

3.2.1 Historical and Current Land Use

This section provides an overview of Portland Harbor's waterfront land and harbor use. Over the past 100 years, major physical alterations have modified the river hydrodynamics and changed the configuration of the river. Map 3.2-1 shows the configuration of the river and the existence of nearby lakes in 1888 (USC&GS 1888). Shoreline changes are presented by decade on Maps 3.2-2a–e. The first map shows the 1888 shoreline and the remaining maps represent a series of nine historical snapshots of the shoreline starting in the year 1936 and ending in 2007. The LWR is currently highly urbanized and industrialized. Some remnant natural areas remain and support habitat for aquatic and terrestrial wildlife.

Significant physical modifications to the river coincided with the development and industrialization of the harbor. Modifications included redirection and channelization of the main river, draining of seasonal and permanent wetlands and lakes in the lower floodplain, extensive filling of wetlands along the shoreline, conversion of agricultural lands, and periodic dredging to maintain harbors and the navigation channel.

Commercial and industrial development in Portland Harbor accelerated in the 1920s and again during World War II, which reinvigorated industry following the Great Depression. Before the war years, industrial development primarily included sawmills, manufactured gas production (MGP), bulk fuel terminals, and smaller industrial facilities. During the war years, a considerable number of ships, minesweepers, and tankers were built at military shipyards located in Portland Harbor. Additional industrial operations during the shipyard years included oil gasification, wood-treatment, agricultural chemical production, battery processing, ship loading and unloading, ship maintenance and repair (e.g., sandblasting, scaling, repair, painting,

refueling), and rail car manufacturing. Many of these operations continue today. Coincident with the development and use of Portland Harbor for these industrial purposes were a number of fires that occurred at wood products industries, manufacturing plants, or other waterfront facilities that were constructed predominantly of wood (Oregonian 1958; 1966a,b; 1967).

The development of the harbor centered on several industrial sectors, which are described in the following sections. Each sector discussed below contains a map showing the historical and current facilities included in the industrial sector (Maps 3.2-2 through 3.2-12). The approximate location of facilities is shown on the maps, based on the current ownership of the property. Mapping of actual facilities or operations was not attempted. Only a few sites in each category are discussed. Table 4.2-1 contains information regarding DEQ's ECSI site numbers and names. Information about these industrial sectors was obtained from the Conceptual Site Model Update and associated site summaries (Integral 2007a; Integral and GSI 2005a,b,c), DEQ's ECSI database, the Oregon Historical Society, historical Commission of Public Docks (CPD; aka Dock Commission) maps, EPA sources, and *Oregonian* newspaper articles.

3.2.1.1 Ship Building, Dismantling, and Repair

Ship-related activities in Portland Harbor include ship building (1800s–present), ship repair (1800s–present), and ship dismantling (1960s–1979). In the early 1900s, ship building plants in the harbor constructed various types of wooden and steel vessels, including ocean-going and river boats. A 1919 Dock Commission map (CPD 1919) lists seven facilities producing wooden boats, four facilities producing steel boats, and two outfitting companies. The Grant Smith-Porter shipyard at the present-day Mar Com North parcel launched 25 wooden cargo ships in 1918 to support the Emergency Fleet Corporation (EFC) during World War I. This shipyard was the most prolific company in the EFC wooden ship program at this time. Prior to WWI the steel ship building industry in the Pacific NW was not extensive due to the distance from steel-producing centers. Only one shipyard, just upstream from the Portland Harbor area, the Columbia River Shipbuilding Yard at RM 14W, produced steel ships for the EFC effort (Hopkins 1994). By 1935, the number of ship building facilities decreased to two: Albina Engine and Machine Works at RM 11E, and Willamette Iron and Steel Company (WISCO) at RM 10W (CPD 1935).

Ship building accelerated again during the World War II years. Map 3.2-3 shows the general location of historical shipyards visible on aerial photographs taken between 1936 and 1969. Approximate areas of the former shipyards include RM 4E (Oregon Shipbuilding Corp.), RM 5.6E (U.S. Shipping Board), RM 6.7E (St. Johns Dry Docks, also called the Port of Portland Dry Docks on the CPD [1935, 1945] maps), RM 7.4E (Peninsula Ship Building Co.), RM 9E Swan Island (U.S. Maritime Commission), RM 9W (American Ship Dismantlers), RM 10W (WISCO), and RM 11E (Albina Engine and Machine Works).

As the demand for new ships increased, industrialist Henry Kaiser built two large shipyards, including the Oregon Shipbuilding Corp. shipyard at the present-day International Slip and the Swan Island shipyard. The Gunderson Brothers Engineering Group (RM 8.5–9.2W) also increased its plant's capacity to build small vessels. Besides Liberty ships, these facilities built small aircraft carriers, T-2 tankers, and a variety of landing craft that delivered troops, tanks, trucks, and supplies to combat zones. The population of Portland increased by a third as people moved into the city to work in the shipyards (Portland Tribune 2009; Oregon Historical Society 2002).

After the war ended, the salvaging of Liberty ships continued to fuel Portland's economy. Zidell Exploration Company salvaged many of the Liberty ship parts, except for the bows which were reinforced with concrete when they were built. Many of these ship bows were buried along the west side of the Willamette just north of the Broadway Bridge. Ship dismantling and scrapping also took place at other facilities in Portland Harbor, such as Consolidated Builders, Inc., a Kaiser affiliate, which scrapped decommissioned troop landing ships at Swan Island for a short time, and American Ship Dismantlers located at the present-day Gunderson site (Portland Tribune 2009) and in the International Slip.

Ship repair also occurred at several facilities. Ship repair and related maintenance was conducted at the former St. Johns Dry Docks (aka Port of Portland Dry Docks) at Willamette Cove between 1903 and 1953, at Albina Engine and Marine Works between 1904 and 1971, and at Mar Com between 1905 and 2004. Other sites with historical ship repair activities include the Triangle Park and U.S. Moorings sites.

Ship-related activities continue at a much smaller scale in Portland Harbor today, with most work focused on ship maintenance and repair. At Vigor Industrial on Swan Island (formerly Cascade General), current activities at the dry docks include hull repair, maintenance, painting, and other dry lay-up ship repair tasks. The U.S. Moorings site (RM 6W) continues to do vessel repair. More than 2,000 marine vessels have been built at the Gunderson facility since the 1960s, including ocean-going barges, conventional deck barges, double-hull tank barges, railcar/deck barges, dump barges, and barges for aggregates and other heavy industrial products.[12] Houseboats and sailboats are currently being built at RM 5.8W.

Chemicals such as volatile organic compounds (VOCs), SVOCs, PAHs, PCBs, TPH, copper, zinc, chromium, lead, mercury, phthalates, and butyltins are identified as common contaminants associated with ship building, salvaging, and repair in studies by the National Shipbuilding Research Program (NASSCO 1999) and EPA's Office of Compliance (EPA 1997b), as well as EPA's Industrial Stormwater Fact Sheet Series (EPA 2006l). The anti-fouling paint applied to ships during World War I contained significant amounts of both zinc oxides and mercury oxides (Williams 1911).

[12] Gunderson Marine: http://www.gbrx.com/Marine_Barges_Home.php?expandable=2.

3.2.1.2 Wood Products and Wood Treating

The wood product industry has a long presence in Portland Harbor and has included wood-treating facilities (1944–1991), sawmills (1800s–2001), and plywood manufacturers (1905–1977), each with its own unique COIs. A 1919 Dock Commission map (CPD 1919) shows eight docks in the harbor devoted to lumber, the largest of which was Peninsula Lumber Company located at RM 7.4E on the M&B site and a portion of the Triangle Park site. Wood products created during this time included wooden barrels and box shooks (or box parts), which were shipped to fruit-growing areas or filled with local produce and railed to their destinations. The lumber industry grew exponentially during the war years, when barges pulling floating logs were a common sight in Portland Harbor. By 1977, the last sawmill in Portland was dismantled (MacColl 1979).

Lumber mills and wood treatment facilities operated at various locations within the Study Area historically, primarily RM 6.9 to 7.2E (Map 3.2-4). One of the largest sites was M&B, which produced treated wood for over 45 years. Other facilities that had wood-treating operations include West Oregon Lumber Co. (located at the Owens Corning-Linnton site) and Kingsley Lumber (at the Georgia Pacific-Linnton site). Wood-treating products used at these facilities include creosote/diesel oil mixtures, pentachlorophenol (PCP)/diesel oil mixtures and associated dioxin contaminants, and a variety of water- and ammonia-based solutions containing arsenic, chromium, copper, and zinc (Integral and GSI 2005a,b,c; EPA 2004b, 2006d).

The present-day Georgia-Pacific Linnton facility (RM 3.6W) was formerly occupied by a sawmill, creosote plant, a lumber storage facility, and, before the site was mothballed in 1997, a wood chip transfer facility. The historic sawmill was owned by the Kingsley Lumber Company and ceased operations in the late 1960s. A sawmill also operated at the current Mar Com South property for almost 50 years. An additional sawmill operated in the central parcel of the Willamette Cove site from the 1950s until the early 1960s (Integral et. al 2007). The Alder Creek Lumber Company in the Multnomah Channel has been the site of lumber-related activities (log storage, sawmill, lumber, planing) since its development in 1959 (Integral and GSI 2005a,b,c). While most of the byproducts of these operations were organic materials, COIs typically associated with saw mills include metals, TPH, and PAHs (EPA 2006d).

Various pesticides and fungicides have been used in glues and surface treatments in the plywood manufacturing process (Stellman 1998); insecticides were applied to forest lands and could be on logs stored in the river by wood products industries (U.S. Forest Service 1964; Eaton et al. 1949; Moore and Loper 1980). The first plywood panels to be manufactured from western woods were made in St. Johns, Portland, at the Plywood Manufacturing Co. in 1905 (PPA 1967). The last plywood manufacturer in the harbor, Linnton Plywood, closed in 2001. The St. Johns Lumber Company (aka Portland Lumber Mill, Portland Manufacturing Co.) operated on the present-day Crawford Street and BES Water Pollution Control Laboratory (WPCL) sites from the 1930s until 1974 when the mill was demolished (Integral and GSI 2005a,b,c). A plywood manufacturing

plant was also located on the west parcel of Willamette Cove upland facility (RM 6.3E) from 1901 until 1963 when it became a lumber mill. Building materials such as lumber, plywood, and laminated veneer lumber products were produced at Linnton Plywood and the western parcel of Willamette Cove. Linnton Plywood used phenol-formaldehyde resin, sodium hydroxide, and petroleum hydrocarbons, such as oil, diesel, and kerosene in its plywood manufacturing process (Integral and GSI 2005a,b,c). COIs associated with plywood manufacturing include VOCs, SVOCs, TPH, and metals (EPA 2006d), as well as possibly pesticides and fungicides, as described above.

3.2.1.3 Chemical Manufacturing and Distribution

Within the Study Area, some facilities manufactured chemicals and some stored, repackaged, and/or distributed chemicals. Chemical plants, including Arkema and Rhone Poulenc (RM 6.8–7.5W) that manufactured pesticides and herbicides, were in place as early as 1941 (Map 3.2-5). The Arkema facility was an organic and inorganic chemical manufacturing facility that produced sodium chlorate, potassium chlorate, hydrochloric acid, and DDT at various times until operations ended in 2001. At the former Rhone Poulenc facility, fertilizers and organic and inorganic pesticide formulations, sodium arsenite liquids, organochlorine insecticides and chlorophenoxy herbicides, acid/esters, and bromoxynil products were produced during its 47-year history.

The Great Western Chemical Company (aka Quadra Chemicals Western and Brenntag Pacific on the present-day McCall Oil site) at RM 7.9W produces water treatment chemicals, dry and liquid industrial cleaning agents and sanitizers, oxygen scavengers, and steam-line treatment chemicals (Integral and GSI 2005a,b,c). Other manufacturers in the Study Area included Premier Edible Oils (edible oil), West Coast Adhesives (phenolic resins), JR Simplot (urea), Ash Grove Cement (cement), Master Chemical (janitorial cleaners), Mammal Survey & Control Service (rodenticides), Mt. Hood Chemical Corp. (cleaning supplies), and McWhorter Inc. (varnish, paint, and resins). These sites are also identified on Map 3.2-5.

A number of facilities packaged, stored, and/or distributed chemicals. Van Waters & Rogers (aka Univar) handled a wide range of industrial chemicals, including organic solvents, acids and bases, ammonia, and other materials, until it ceased operations in 1988. Other chemical distributors included Great Western Chemical at RM 9.2W on the Chase Bag site (chemicals unknown), Wilbur Ellis (pesticides and herbicides), Ashland Chemical (primarily solvents), and McKesson (Mt. Hood Chemical Property – food additives, pharmaceuticals, and mineral acids).

COIs associated with chemical manufacturing operations can vary, depending on the operations, but could include pesticides, herbicides, VOCs, dioxins/furans, and metals (EPA 2006f).

3.2.1.4 Metal Recycling, Production, and Fabrication

Recycling, production and fabrication, and plating of metals occurred at several locations within the Study Area. Each of these processes has its own unique set of COIs, which are discussed below.

Metal salvage and recycling facilities operated in the Study Area (Map 3.2-6) at RM 4E (Schnitzer Steel – auto and appliance dismantling), RM 5.8W (Marine Finance), RM 7.2W (Gould/NL Lead), RM 7.3W (Schnitzer-Doane Lake), RM 8.5W (Calbag Metals), RM 8.8W (Gunderson – auto dismantling), RM 9.5E (Portable Equipment Salvage), RM 9.7W (Schnitzer Steel Recycling Yard on NW Yeon), RM 9.8W (Nudelman & Son), and RM 10.3W (Calbag-Nicolai). The metals recycling business includes cutting, torching, segregating, storing, and distributing metals, as well as recovering metals from wire. The Gunderson facility has manufactured and refurbished rail cars since 1913. Rail cars were also refurbished at the ACF Industries property (RM 3.7W) for almost 23 years (Integral and GSI 2005a,b,c). Ship dismantling and scrapping facilities, which also fall into this category, are discussed in Section 3.2.1.1. COIs commonly found in waste streams from these types of facilities include VOCs, TPH, PCBs, phthalates, cyanide, and a variety of metals (EPA 2006i,j).

Metal production and fabrication currently takes place in the Burgard Industrial Park, and several sites in the RM 8 to 10.3W reach, including Dura Industries, NW Copper Works, American Machine & Gear (RM 9.8W), and two non-ECSI sites, Portland Bolt & Manufacturing (RM 9.6W) and The Willard Storage Battery Company, which manufactured storage batteries at the Chase Bag site from approximately 1952 to 1958. COIs associated with metal production and fabrication include metals, PAHs, and TPH (EPA 2006e,h). For high-temperature applications such as die-casting machines, hydraulic oil with PCBs was often used (EPA 2004a).

The Columbia American Plating site operated as a commercial plating (primarily zinc) facility between 1975 and its closure in 2003. COIs associated with metal plating industries include VOCs, PAHs, TPH, cyanide, and metals (EPA 2006h).

3.2.1.5 Manufactured Gas Production

MGP operations took place between 1913 and 1955 at RM 6.5W (Map 3.2-7). Portland Gas & Coke constructed an oil MGP facility, known as Gasco, on the subject property. The plant initially produced town gas and pressed lampblack briquettes that were sold in the Portland area as fuel. In 1923, the gasification process was modified to optimize aromatic generation and light oil recovery for use as motor fuel. Tar recovery and refining were incorporated into the process in 1925 to provide tar for use as a road binder. During the 1930s, the plant expanded, and in 1941 a coking plant began production of electrode grade coke and high BTU gas (Integral et al. 2007). The Pintsch Compressing Company Gas Works at RM 11.7W operated between 1890 and the mid-1930s and manufactured compressed gas from crude oil for railroad train lighting. Just upstream from the Study Area, the Portland MGP site operated at RM 12.2W between 1860 and 1913.

COIs associated with manufactured gas operations include VOCs, SVOCs, PAHs, TPH, metals, and cyanide (Integral et al. 2007).

3.2.1.6 Electrical Production and Distribution

Electrical transformers and capacitors are associated with nearly all of the major industries in the harbor. Transformers and capacitors historically contained and may continue to contain PCBs. There are seven current and one historical substation in the Study Area (Map 3.2-8). The Portland General Electric (PGE) Harborton Substation at RM 3.3W currently consists of an operating 115-KV switchyard and distribution substation for electrical power regulation and transmission (Integral and GSI 2005a,b,c). PGE also operates Substation E (RM 10.4W), and other substations at Swan Island (RM 9.3E), Siltronic (RM 6.4W), and Willbridge Terminal (RM 7.5W). The PacifiCorp Albina substation is located at RM 11E, which is associated with a high-voltage cable crossing at RM 11.3. The remaining active substation is operated by Schnitzer Steel (RM 4E, near the International Slip). BPA operated a substation at the Arkema site and conducted site cleanup after decommissioning the substation.

Electrical equipment repair, servicing, and salvaging operations were found on the east bank from RM 11.3 to 11.5 (Tucker Building, Westinghouse, and PacifiCorp Albina Properties), at RM 3.7W (ACF Industries), RM 9.5E (Portable Equipment Salvage), RM 9.5W (GE Decommissioning), and RM 10W (GE facility at NW 28th Ave; Toxic Substances Control Act [TSCA] site). The GE Decommissioning facility handled dielectric fluids containing PCB concentrations greater than 50 ppm. These fluids were drained from customer-owned equipment into temporary aboveground holding tanks or drums. Often these fluids were returned to the equipment following service or repair (Integral et al. 2007). The GE facility at RM 10W conducted repairs from the mid-1990s to 2001, including decommissioning of custom transformers.

COIs associated with electrical production and distribution include PCBs, TPH, and PAHs (Pfafflin and Ziegler 2006).

3.2.1.7 Bulk Fuel Distribution and Storage and Asphalt Manufacturing

Bulk fuel facilities have a long history in Portland Harbor. By 1936, most of the facilities currently in place between RM 4 and 8 on the west side had already been established (Map 3.2-9). These facilities include ARCO's BP Terminal, Kinder-Morgan, Willbridge Terminal, Christenson Oil, ExxonMobil, Texaco Equilon, and Foss/Brix Maritime. Five additional facilities were located on the east side of the river in the RM 3 to 5 reach. These facilities have handled a variety of petroleum products, including lubricating and specialty oils, bunker fuel, diesel fuel, gasoline, ethanol, gasoline additives (e.g., methyl tert-butyl ether [MTBE], ethylene dibromide [EDB], ethylene dichloride [EDC], lead), aviation fuel, and various lubricants. Petroleum product pipelines are also found throughout the Study Area. Another petroleum product, asphalt, is manufactured at several facilities on the west side of the river, including Owens Corning Linnton (RM 3.8W), GS Roofing (RM 7.5W), McCall Oil (RM 7.9W), Chevron Asphalt Refinery (RM 8W), and Trumbull Asphalt (RM 9.1W).

The Municipal Paving Plant at RM 10.9E was constructed in 1928 and operated intermittently until its permanent closure in 1966.

COIs typically associated with bulk fuel storage operations and asphalt operations include VOCs, SVOCs, PAHs, TPH, and metals (EPA 2006g).

3.2.1.8 Steel Mills, Smelters, and Foundries

As shown on Map 3.2-10, the harbor hosted foundries at RM 2.8E (Consolidated Metco), RM 9.7W (Schmitt Forge), and RM 11.4W (Gender Machine Works). Steel foundries are currently operating at RM 10.4W (ESCO Main Plant) and RM 10.5W (ESCO Plant 3). Lead smelters were located at RM 7.2W (Gould), at RM 9W (National Lead/Magnus Smelter), and at RM 11.6W (RiverTec Property). In addition to lead smelting, facility operations at the Gould site included lead-acid battery recycling, zinc alloying and casting, cable sweating, and lead oxide production. The present-day Wilhelm Trucking site at RM 9.6W was the former location of a lead bearing rehabilitation plant. Steel mills were located at RM 2.4E (Evraz Oregon Steel Mills [EOSM]) and at RM 8.3W (former Oregon Steel Mill operation at Front Ave LP). Only the EOSM property is currently operating, confined to steel production, steel processing, and related ancillary operations.

Besides metals, other COIs associated with these types of operations include TPH and PAHs. PCBs were a component of hydraulic fluid for high temperature applications (machining, die casting) where fire resistance was important. PCBs were also a component of heat transfer fluid used in big applications like heat exchangers and recirculating cooling systems (EPA 2004a, 2006h).

3.2.1.9 Commodities Maritime Shipping and Associated Marine Operations

At the turn of the 19th century, Portland Harbor accommodated steamboats and railroads that transported farm products and natural resources from Idaho, eastern Washington, and Oregon. This cargo was then loaded onto sailing ships for markets in Asia, the eastern and western United States, and Europe. Portland Harbor also served as the gateway for incoming trade goods to the region. On the 1919 Dock Commission map (CPD 1919) there were docks dedicated to the distribution of shingles, cans, asphalt, sand and gravel, cereal, flour, grain, and vegetable oil.

The Port of Portland facilities have had a prominent presence in the maritime commodities shipping industry in Portland Harbor since 1891. Over the years, export/import of agricultural products, dry/liquid bulk products, forest products, and other bulk commodities have passed through Port facilities. Currently, the Port operates two deep-water marine terminals within the Study Area that handle thousands of tons of cargo each week. Major exports by volume handled at Terminals 2 and 4 (Map 3.2-11) include wheat, soda ash, potash, and compressed hay. The Port's operations in Portland Harbor constitute the third-largest export center for grain in the world and the largest

wheat export port in the United States (Williams 2007). Major imports include automobiles, steel, and limestone.

Other privately owned commodity shipping facilities in the harbor include or have included the grain handling operations at CDL Pacific Grain (RM 11.4E) and Centennial Mills (RM 11.3W), edible oils at the former Premier Edible Oils facility (RM 3.6E), scrap metal export at International Terminals (RM 4E), cement import and distribution at Glacier NW (RM 11.3E), as well as other non-ECSI sites. JR Simplot in the South Rivergate Industrial Park (RM 3E) has been distributing anhydrous ammonia and solid and granular urea to the Pacific Northwest since 1968. Goldendale Aluminum at RM 10E was used as an offloading facility (alumina, electrode binder pitch, and grain) from 1957 until 2001 (Integral and GSI 2005a,b,c). Sand and gravel operations have occurred at the Ross Island Sand & Gravel facility at RM 11.1E since at least 1919 (LWG 2007).

Supporting maritime activities include over-water tug and barge moorage, maintenance and repair facilities, overwater bunkering and lightering, tug-assisted and independent maneuvering of vessels in and around marine facilities, and stevedoring (loading and discharging) product at vessels.

Incidental spills into the river from commodities marine shipping include organic materials, VOCs, PAHs, and TPH (EPA 2006k).

3.2.1.10 Rail Yards

Rail tracks, yards, and terminals are located in the Portland Harbor area. Rail yards are found at approximately RM 9.8 to 11.1E (Union Pacific Railroad [UPRR] Albina Yard), RM 8.6 to 9.5W (Portland Terminal Railroad [PTRR] Guilds Lake Yard), RM 8.1W (Burlington Northern Santa Fe Railway Co. [BNSF] Willbridge Switching Yard), and RM 4.6E (UPRR – St. Johns Tank Farm) (Map 3.2-12). These rail yards support the interstate railroads, BNSF and UPRR. Primarily operating as switching yards, some rail yards offer locomotive fueling and servicing, railcar maintenance, and trailer-on flatcar storage. Railcar switching yards (RM 8.1W – BNSF Willbridge Switching Yard) are locations where trains are assembled and disassembled (and this type of operation typically does not result in releases or produce waste streams.) Historical rail yard operations were located at and around RM 11.6W (BNSF Hoyt Street Railyard, and UPRR Union Station operations). BNSF owned and operated the Hoyt Street Railyard from the early 1900s until 1988. The rail yard has been abandoned and dismantled, and many of the blocks in the site have been developed as condominiums and commercial businesses. Historical rail car maintenance operations were located at RM 3.7W (ACF Industries).

COIs associated with fueling activities and freight car repair operations at rail yards could include VOCs, SVOCs, TPH, PCBs, and metals (EPA 2011; DEQ 2011c).

3.2.2 Historical Land Use

This section summarizes the major historical land use, fill placement, and shoreline and overwater operations. As part of the Conceptual Site Model Update (Integral and GSI 2005a,b,c) and the Round 2 Report, historical aerial photographs were reviewed to evaluate general trends in land use along the Willamette River waterfront. Mosaic TIFF (tagged image file format) images created by the Port of Portland from scanned historical aerial photographs of the river and waterfront were also reviewed, as were more recent aerial photographs (see Maps 3.2-13a–f). The oldest historical aerial photographs available for this harbor-wide review were taken in 1936. Based on the pace of land development observed during the preliminary review of all of the aerial photo mosaics, six of the photo mosaics (1936, 1948, 1961, 1974, 2000, and 2007) were selected for broader-scale depiction of changes in land usage (Maps 3.2-14 through 3.2-19). For most years selected, aerial photo images were available for the entire river waterfront from the Columbia River to Ross Island.

Fill placement is shown on Maps 3.2-20a–f. Detailed information on the fill placement activities can be found in Table 3.2-1. Information used to construct this table was obtained from the aerial photographs, site summaries prepared during the RI,[13] and the City of Portland. The descriptions of subsurface soils in site investigation reports suggest that much of the fill placed in these areas consists of Willamette River sediment/sand/gravel dredged offshore of the respective facilities or in the immediate vicinity. Other sources of fill include dredged material from Multnomah Channel and the Columbia River. The source of the fill, if known, is identified in Table 3.2-1.

Overwater structures, such as wharfs, piers, floating docks, and pilings, were built largely to accommodate or support shipping traffic and remain common. These structures along the shoreline are clearly visible in the aerial photographs provided in Maps 3.2-21a–t.

In summary, industrial and commercial development along the river began in the mid- to late-1800s in scattered areas such as downtown Portland, St. Johns, Linnton, and Macadam. Portland Harbor remained largely undeveloped through the late-1800s, but as urban development in the downtown area at the beginning of the 20th century pushed industrial development downriver, businesses began to relocate to the current industrial area of the harbor. The west side of the river was settled and developed first.

3.2.2.1 Historical Land Use Changes by River Mile Segments

The most notable changes for the major reaches in the Study Area are described in the following subsections. These reach breaks are defined based on changes in the LWR's physical characteristics. General land use changes for the east and west banks of each reach are discussed, including historic riverbank fill placement and changes in overwater structures.

[13] Site summaries were presented in the draft Conceptual Site Model Updates (Integral and GSI 2005a,b,c) and in the Round 2 Report (Integral et al. 2007).

3.2.2.1.1 RM 9.5 to 11.8

In 1936 the waterfront hosted a lumber mill, grain elevators, cargo docks, oil and coal exporters, and ship building and ship repair facilities. Rail yards between RM 10 and 12 were present on both sides of the river in 1936 and were more fully developed by 1948 (Map 3.2-13a). By 1961, industrial development had expanded on both sides of the river and log storage areas were present along riverbanks (Map 3.2-13b). Relatively few changes occurred from 1961 to 2000, with the exception of the completion of Interstate 5 and Interstate 405 (Maps 3.2-13c–e). By 2007, dock structures were added along the west bank and a few parcels were converted from commercial to industrial or residential use (Map 3.2-13f).

From approximately RM 9.5 to 10, the original shoreline on the east bank formed a cove. In the 1970s this area was filled (Map 3.2-2c). Significant channel narrowing due to infill on the west bank is observed from 1888 to 1936 (Map 3.2-2a). Beginning on the east side, the riverside area near RM 11.2 to 11.4 where Glacier NW is currently located (plus adjacent nonriparian properties) was the site of the former Albina Engine and Machine Works property, where ship construction and repair was conducted for the U.S. Navy and the War Shipping Administration (see Map 3.2-3). Albina Engine and Machine Works was founded in 1904 as a riverfront repair yard. During WWII, the shipyard facility was expanded to encompass 16.8 acres and included six shipways, welding and pipe shops, paint storage and shops, warehouses, two outfitting docks, plate storage yards, burning slabs, and a pickling plant. Thirteen ship building contracts were undertaken for the Navy, including the construction of submarine chasers, landing craft and support ships, fuel oil and gasoline barges, and degaussing vessels. Operations in the general area since the 1970s have included a mix of industries, including transformer substations and electrical equipment manufacture, repair, and storage.

The shipways were filled beginning in the 1950s and completed by 1963. Most of the riverside buildings associated with the shipyard were demolished. The first new buildings on the former shipyard property appeared in the late 1970s. A portion of the former shipyard was used for expansion of the Pacific Power and Light Albina Substation beginning in the late 1940s.

Docks have been located in the area of the Albina Rail Yard (RM 10–11) and the Glacier facility (RM 11.3) from 1936 to the present day. From the review of aerial photographs, it appears the existing docks at the CDL Pacific Grain facility (RM 11.4) were constructed sometime between 1957 and 1966 (Map 3.2-20f). A large overwater structure called the Irving Dock was present at this location prior to construction of the present-day CDL Pacific Grain dock, as shown in both the aerial photographs and 1924 Sanborn maps reviewed by Integral. A large dock first appears in the 1961 aerial photo at RM 11.8E.

Along the west bank from RM 9.8 to 10.3, encompassing the present-day Terminal 2 and Sulzer Pumps properties, WISCO operated a shipyard for an unknown period up

until 1949 (Map 3.2-3). In 1941-1942, the WISCO facility was expanded with public funds from the Defense Plant Corporation. The reconfigured facility was 79 acres in total area, with government ownership of approximately 36 acres. Combined, these facilities provided a complete shipyard for launching and outfitting steel ships. Many of the manufacturing operations associated with the shipyard were located on the current Sulzer property (RM 10.3), which included outfitting operations, a sheet metal fabrication shed, a cable storage building, a machine shop, a paint shop, a coppersmith shop, and the main industrial building. WISCO operations conducted on what is now Terminal 2 consisted of three shipways with four attendant craneways located at the southern (upstream) end of the property; these shipways were subsequently filled.

Significant changes occurred along the west bank with dredging of a slip at the WISCO shipyard in the mid-1940s (RM 10); the creation of the Albina Ferry slip (Slip No. 1) at Municipal Terminal 1 (RM 10) in 1914 and Slip No. 2 in 1923; filling of the western shoreline downstream of Terminal 2 (RM 10.6) in the 1950s and 1960s; filling of the Terminal 1 South slip in the early 1900s; and filling of the Terminal 2 upstream slip by 1987 (Map 3.2-13e). Beginning with the 1936 aerial photograph, a large tank appears on the west bank at RM 12, but is no longer there in photographs taken after 1957.

Overwater features in this reach include the docks along the western shoreline at the former Municipal Terminal 1 and current Terminal 2 (RM 10 and 10.6), and an oil transfer pipeline (south of present-day Sulzer Pumps) at RM 10.4 (Map 3.2-20e). The oil transfer pipeline was used by Portland General Electric for transferring Bunker C oil from vessels to tanks at a nearby power plant. Some of these docks remain in place but are no longer in use. Most overwater activity associated with the docks in this reach appears to have occurred in the 1940s and 1950s, when the docks were used for loading lumber, paper products, grain, gravel, and coal. From the 1930s through the 1960s, log moorage rafts were present at approximately RM 9.2 and RM 10.

3.2.2.1.2 RM 8 to 9.5

This stretch of the river has undergone significant change through the years, as is shown in the six photo mosaics (Map 3.2-13a–f). Swan Island (RM 8.3 to 9.2 on the east bank) was originally a sandbar and marsh separated by two channels of the Willamette River. Prior to 1920, the eastern channel was the river's main channel. The eastern channel was deeper than the western channel, which was wide and shallow with a shoal that hindered boat passage. In the early to mid 1920s, the west channel was deepened and widened in places to facilitate navigation (the west channel was opened to navigation in 1926). In 1927, the diversion of the river's main channel from the east side to the west side of the island was completed through the construction of a causeway at the island's upstream end (creating a lagoon out of the east-side channel called Swan Island Lagoon). The filling of Swan Island, performed by the Port of Portland, was mostly completed by the 1920s before construction began on the airport in 1926.

Mocks Bottom, is located in the upland area east of the Swan Island Lagoon. Once a swampy slough, Mocks Bottom was filled by the Port and the USACE to build roads and facilitate industrial development. About half of Mocks Bottom had been filled by 1961 and filling was complete by 1974 (Map 3.2-20d). Although some industrial facilities had developed along the shoreline by 1961, less than half of the area had been developed by 1974. The area was fully developed by 2007 with industry related to truck manufacturing, shipping and transportation, marine salvage, and military uses.

The Swan Island peninsula has a long history of commercial and industrial operations that continue today. The Swan Island Municipal Airport functioned until operations moved in 1940 to a location that is now part of the Portland International Airport. Between 1942 and 1949, the U.S. Maritime Commission leased Swan Island from the Port of Portland and contracted with the Kaiser Company to construct a shipyard and associated facilities. The shipyard facilities were used to build T-2 tankers used during WWII. A Kaiser affiliate, Consolidated Builders, Inc., conducted ship dismantling between 1947 and 1949. After the war, the area was redeveloped and used for ship repair purposes. The redeveloped facilities were used by various ship repair contractors and their subcontractors. In addition, facilities were leased to a number of industrial tenants who conducted a range of activities, including steel fabrication and storage, wood products manufacturing, equipment manufacturing, maritime supply sales, printing, chemical and soap storage, war surplus storage, fire extinguisher service and storage, paint storage, aluminum oil tank manufacturing, service station operation, sheet metal work, roofing supply storage, and general office storage. The eight shipways constructed during the military era were filled with dredged materials between 1950 and 1962. The current configuration of dry docks at the end of the peninsula and berths along Swan Island Lagoon and the Willamette River was largely completed by 1979. Some filling also occurred in the northwestern portion of the shipyard area in the late 1970s.

Up until the 1960s the west side of the river was mostly undeveloped and was used for log raft storage. The present-day Shell Equilon dock occupied the west bank at RM 8.8 in 1936. Operations at Gunderson (RM 8.7 to 9.2) began as early as 1942, and most of the present-day site was constructed by 1966 (Map 3.2-20d); activities have generally included barge and rail car manufacturing. Ship building operations began at Gunderson in the 1960s and are still in operation today. During the 1960s and 1970s, a portion of the Gunderson facility was used by American Ship Dismantlers for ship scrapping. Overwater activities occurred at the barge launchways in Area 2 and the outfitting dock in Area 3. A dock structure and an oil transfer pipeline were located historically at the McCall Oil site (RM 8.2) prior to filling in the late 1960s. Fill was placed along the Gunderson shoreline beginning in the 1950s.

On the west side of the river in this reach, Guild's Lake was a shallow, marshy area located from the present-day Guilds Lake Railroad Yard (referred to by DEQ as "Lake Yard") westward to St. Helens Road. Map 3.2-1 shows the location of the lake in 1888. Filling of Guild's Lake began in approximately 1879 and was partially completed in

1913 by private entities using soil hydraulically sluiced from nearby hillsides, providing space for the rail yard and an industrial center (Oregon Historical Society 2002). The Port of Portland continued filling in the 1920s, using materials dredged from the Willamette River. The filling of the Guild's Lake area was planned in connection with the West Swan Island project, where the channel was diverted from the east to the west side of the island. Construction of the Guilds Lake Yard, which is owned by PTRR, began in 1916.

3.2.2.1.3 RM 5 to 8

The 1936 photo mosaic (Map 3.2-13a) shows that the east side of the river was largely undeveloped from RM 5 to approximately 5.7 until the period between the 1960s and 1970s. Early features include docks at M&B (RM 7), Willamette Cove (RM 6.7) and downstream of Mar Com (RM 5.7). The eastern bank between RM 6.5 and 6.9 was primarily filled in the 1910s and 1920s to create the central and eastern parcels of the Willamette Cove upland facility. Upstream of RM 6.9, the eastern bank remained relatively unchanged until the 1970s, when the downstream end of the property presently known as Triangle Park (RM 7.4) was filled to create a dock and berth area. From 1888 to 1936, shoreline development is most notable from RM 5.9 to 6.4, due to the construction of the St. Johns Bridge at RM 5.9 and timber processing facilities on the eastern bank at RM 6.2 and on the western shore at RM 6 and 6.4. From 1888 to 1936 the eastern bank shows widening due to development in the vicinity of timber processing plants, including the M&B site (RM 7.1), and narrowing due to installation of the railroad crossing at RM 6.9 (Map 3.2-2a).

The Mar Com facility, which ceased operations in 2004, was situated on land that had been used for ship building and vessel repair since approximately 1905. The central parcel of the Willamette Cove facility was also used for ship repair on dry docks and related ship maintenance between 1903 and 1953. Upland shops and structures and in-water dry docks were used by independent contractors working for various vessel owners. During wartime, U.S. Government contractors utilized the dry docks for military ship outfitting and repair. Several of these dry docks have since been removed from this stretch of the river (e.g., Mar Com, Willamette Cove). Dock structures at the former M&B facility were removed during the recent Superfund cleanup of this site.

The 1936 photograph of the west side of the river (Map 3.2-13a) shows the Willbridge Terminal (RM 7.5), U.S. Moorings (RM 6), and Gasco (RM 6.2) facilities with very little other development. Most of the shoreline change occurred on the west side of the river from the 1940s to the 1960s. Fill was placed along the eastern shoreline of RM 5 to 5.7 from the 1950s through the 1970s. By 1975, fill was also placed along the western shoreline and a larger low-lying area at what is present-day Siltronic (RM 6) and Gasco property. Fill materials for both sides of the river included quarry discards and dredge materials. At the Gasco and Siltronic properties, MGP materials were also included in the fill. At the Arkema site (RM 7.2), fill consisted of plant debris composed of asphalt, concrete, pipe, soil, and fill from other sources.

The historic 80-acre Doane Lake[14] and associated wetlands were situated in the upland area of this western stretch of the river (see Map 3.2-1). The lake was divided in 1908 when the Railroad Bridge and southbound rail lines were constructed, and again in 1968 when the northbound rail line was constructed. The 6-acre lake area between these two rail lines is called North Doane Lake. The portion of the lake north of the Railroad Bridge was filled between the 1960s and 1970s for industrial development. The portion of the lake south of the Railroad Bridge was used for waste disposal by adjacent industries. Fill material began to be placed in this remnant lake in the 1960s, and it was almost completely filled by the late 1990s when the Gould Superfund site completed remediation.

The western shore shows narrowing from RM 6.9 to 7.4 due to upland development and installation of the railroad crossing. Arkema maintained two dock structures for receipt of evaporated sea salt, which contained sodium chloride, and shipping of inorganic chemicals produced onsite. Operations ceased in 2001, and the facility has been dismantled, but the dock structures remain. Petroleum products have been loaded and unloaded at the Willbridge Terminal since the early 1900s.

3.2.2.1.4 RM 3 to 5

Major facilities on the east side of the river started in the early 1920s and included cargo handling, a flour mill, warehousing, and bulk fuel storage. Tank farms were developed on the west bank in approximately 1918 (present-day Kinder Morgan Linnton Terminal) and were expanded in the 1960s to be the predominant land use. The Owens-Corning Linnton facility installed several petroleum product tanks for use in their asphalt production in 1981. Other early west-shore industries included lumber mills, toy manufacturers, a creosote plant, and lumber storage. The PGE Harborton substation at RM 3.1W was constructed in 1985. Both sides of the river were fully industrial by the 1970s.

The most important shoreline changes in this reach occurred along the eastern shoreline from RM 4.2 to 4.6 (Map 3.2-20b). In the late 1910s and early 1920s, the mouth of Gatton Slough was filled (discussed in the following section), and three slips were dredged forming the Municipal Terminal No. 4 area (present-day Slips 1 and 3 and Wheeler Bay). Between approximately 1948 and 1958, the middle slip (Wheeler Bay) at Terminal 4 (which was never completed) was backfilled and Slip 3 was widened. The Port of Portland's auto storage facility at Terminal 4 was developed in the 1960s and the early 1970s, by placing sand fill to bring the site up to an elevation above the flood level. In the early 1970s, the sand fill was graded and the automobile storage yard and the steel dock and steel yard were constructed (Integral et al. 2007).

The Burgard Industrial Park (RM 4E) was the location of a large shipyard operated by the Oregon Shipbuilding Corporation. The deep-draft International Terminal Slip was created during the 1940s, and portions of the marshy, low-lying areas on the site were

[14] http://www.portlandonline.com/bes/watershedapp/index.cfm?action=DisplayContent&SubWaterShedID=1&SubjectID=3&TopicID=26&SectionID=1

filled. Over 450 ships were built on this property from 1941 to late 1945. Ship breaking activities were reported in 1946 (Oregonian 1946). The year in which shipyard was dismantled has not been presented in documents reviewed, but the shipways were filled between the early 1960s and 1972. Post-shipyard industrial uses included metal fabrication, log rafting, and upland log storage. The property was converted for use in 1972 as a metals scrap yard. Automobile shredding operations began in 1980.

Conspicuous historical overwater features within this reach include docks associated with ship building and repair, lumber mills, petroleum product distribution, moorage, and cargo unloading. Port of Portland Terminal 4 tenants that currently (or historically) handle soda ash, new automobiles, and liquid bulk materials from their docks are located on the eastern shoreline. Metal scrap delivery occurs at docks in the International Terminal Slip (RM 3.7). Along the western shoreline, there are bulk petroleum distribution docks (ARCO; RM 4.9) and sand and gravel unloading/loading overwater activities (Columbia River Sand & Gravel; RM 4.5).

3.2.2.1.5 RM 1 to 3

Little change to the shoreline occurred in this vicinity of the river until fill materials were placed at the present-day EOSM site (RM 2.1E) from the early 1940s to the 1960s; additional filling of the riverbank occurred in the 1970s using EOSM slag materials, onsite soils, dredge material, and imported materials (Map 3.2-20a). Within the larger Rivergate industrial area, approximately 500 acres of the historic Ramsey Lake, located between Smith and Bybee lakes and the Willamette, were filled with dredge material from the 1960s to the 1980s. As shown on Map 3.2-1, this lake and floodplain historically covered approximately 650 acres and included a seasonal stream called Gatton's Slough that flowed west to the Willamette and a channel connecting it to the Columbia Slough to the east (USC&GS 1888). A dredge/fill map compiled from USACE data shows dredge material from the Post Office Bar and the mouth of the Willamette being placed in the Rivergate industrial area (Port of Portland 1981; USACE 1973).

The primary overwater features along the eastern shore of this reach are docks for distribution of chemicals and petroleum products. From 1936 until the 1960s, the eastern shoreline was utilized for log raft storage. In the 1940s, a dock was constructed at what is now the EOSM site for the transport of oil and bilge water to an upland oil sump. The current dock at Ash Grove is first present in the 1966 aerial photograph (Map 3.2-13c). By 1975, new docks associated with EOSM, JR Simplot, and Port of Portland Terminal 5 are present along the RM 1 to 3 reach.

The only industrial feature on the western bank of the river in this area is Alder Creek Lumber Company (RM 2.9).

3.2.2.1.6 Multnomah Channel

Besides the Alder Creek lumber yard at the mouth of the Multnomah Channel, the only other conspicuous facilities in this stretch of the channel are Fred's Marina, the Multnomah Yacht Club, and the ESCO landfill.

Since 1959, floating logs have been delivered to the dock area at the Alder Creek Lumber property near the mouth of the channel. Houseboat and boat moorages and marinas line Multnomah Channel's southern bank, opposite the ESCO landfill, forming a continuous string that extends as far as one mile. Approximately 200 of these houseboats and sailboats are used as permanent residences (DEQ 2009c).

Fred's Marina has occupied its site since the 1940s (Parsons Brinckerhoff 2004). Presently, the marina contains a boat ramp, fuel dock, a boat trailer storage area, and over 200 slips. A designated dredged material disposal site is located upland directly east of the marina. This disposal site is for the containment of material dredged from the marina and vicinity that is deemed suitable for upland placement. The Multnomah Yacht Club has been in operation since 1961; prior uses of the property are unknown. The ESCO landfill does not have any operations on the shoreline. No further information on historical shoreline and fill placement activities was found.

3.2.3 Current Land Use

Portland Harbor is located within a broader region characterized by commercial, residential, recreational, and agricultural uses. A portion of the land adjacent to Portland Harbor is located within the Guild's Lake Industrial Sanctuary Plan (City of Portland 2001a) area (from the St. Johns Bridge at RM 5.8 to 10.7, along the west shore). Land use along the Willamette River within the harbor includes marine terminals, various manufacturing facilities, and commercial operations, as well as public facilities, parks, and open spaces. A description of the industrial and marine development in the harbor is presented in Section 3.2.1. As shown on Maps 3.2-22a–e residential areas on the west side include the Linnton neighborhood in pockets west of St. Helens Road between RM 4.3 and 5W, and in the mixed use Pearl District neighborhood in the vicinity of RM 12W. Most of the residential land use on the east side is above the bluff, except for the St. Johns neighborhood, which extends closer to the river between RM 5.7 and 6.8E.

Maps 3.2-22a–e illustrate current land use zoning within the LWR and upper Multnomah Channel and show sites located within Study Area drainage basins. Site parcels and sites inventoried in DEQ's ECSI database are identified on the maps. Waterfront properties are also labeled. Current and previous facility names for these sites are listed in Table 3.2-2.

The shoreline has had significant physical modifications, including structures built to stabilize the riverbanks for development, as shown on the aerial photographs in Maps 3.2-21a–t. Riprap is the most common bank-stabilization measure, although upland bulkheads and rubble piles are also used. Seawalls are constructed primarily of treated

timbers or concrete to control periodic flooding. Scattered natural areas also are found along the shoreline (Maps 3.2-21a–t).

The current overwater structures, such as wharfs, piers, floating docks, and pilings, were built largely to accommodate or support shipping traffic. These structures along the shoreline are clearly visible in the aerial photographs provided in Maps 3.2-21a–t. Numerous public and private outfalls, including stormwater and CSO outfalls, enter both shores of Portland Harbor, and are described further in Section 3.2.4.

Submerged lands are primarily owned by the Oregon Department of State Lands (DSL). The DSL submerged lands boundary can be ordinary high water (OHW), ordinary low water mark (OLW), or arbitrary deed lines specific to each riverfront property. Notable exceptions to DSL ownership include portions of the submerged and submersible lands at the Port of Portland Terminal 4, most of the dry dock area and riverside berths at Swan Island, which are owned by Shipyard Commerce Center LLC, and the International Terminal Slip owned by Schnitzer Steel Industries. DSL owns approximately 94 percent of the submerged lands in the Study Area. There are also areas within the Study Area below OLW not owned by DSL or the Port (DEA 2011).

The remainder of Section 3.2 discusses the historical and current conveyance systems in the harbor and the channel authorization and dredge and sediment cap activities within the Study Area..

3.2.4 Historical and Current Conveyance Systems

This section describes the historical and current conveyance systems in the Study Area, including both municipal and non-municipal systems. Non-municipal systems are either private or part of other public systems, such as Oregon Department of Transportation (ODOT) or the Port of Portland. Potential COIs associated with historical and current conveyance system discharges are discussed in Sections 4.3.1.2 and 4.4.1.3, respectively. Potential current and future loads from stormwater are discussed in Section 6.1.2.

3.2.4.1 General Description of Conveyance Systems

There are three types of conveyance pipes in Portland Harbor. The following is a description of the types of flows each pipe carries and its discharge points.

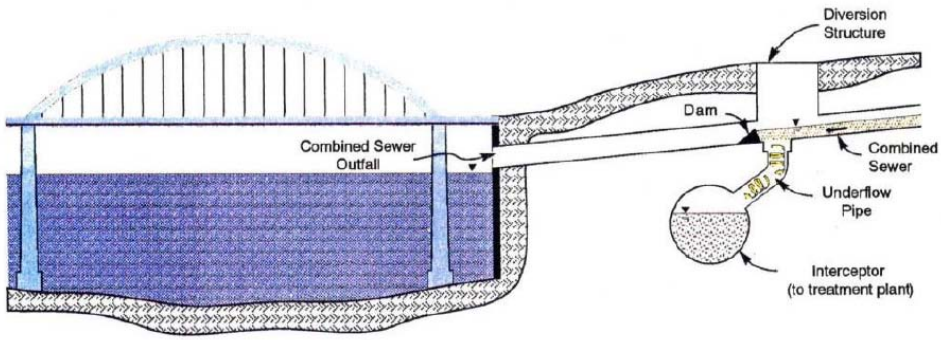
Sanitary Pipes: These pipes convey sanitary wastes from domestic and industrial sources, and may also carry industrial wastewater. Flows in these pipes discharge to the Columbia Boulevard Wastewater Treatment Plant (CBWTP), which discharges to the Columbia River at two discharge points (RM 105.5 and 105.6), approximately 4 miles upstream of the Willamette River confluence. Sanitary pipes are typically part of the City's collection system, although there are sanitary pipes on private property that connect to the municipal system.

Stormwater Pipes and Other Point Discharges: Stormwater conveyance systems typically consist of ditches, swales, storm drains, inlets and catch basins connected to an outfall through pipes or lines. Flows in these pipes typically discharge to the river, although some may discharge to lakes or infiltration facilities. Stormwater pipes can be part of the municipal collection system or part of a non-municipal system.

Combined Pipes: Combined pipes convey sanitary wastes from domestic and industrial sources and stormwater, and may also carry industrial wastewater. Historically, these flows typically discharged to the river. After combined pipes were connected to the sanitary interceptors, the outfalls draining these pipes were either converted to CSO outfalls or to storm-only outfalls. Historically, combined pipes were both municipal and non-municipal, but currently most combined systems are part of the City's collection system. Additional detail on how combined pipes function and the types of CSO outfalls is provided below.

These pipe types are currently configured into three general types of conveyance systems in the Study Area, as shown in Figure 3.2-1: 1) separated systems, 2) combined systems that discharge to the CBWTP (with no discharge to the river), and 3) combined systems with overflow diverters designed to reduce discharge to the river to a maximum of four times per winter and once every three summers (City of Portland 2001b). Separated systems have stormwater-only lines that discharge to the river and sanitary-only lines that discharge to the treatment plant. In combined systems, the stormwater and sanitary lines join and flow in a combined line. Most of the Study Area is currently served by separate storm lines and separate sanitary sewers. Only a limited portion of the area is served by the combined system, and not all of the combined system has the ability to overflow to the river (see Section 3.2.4.4). Stormwater and combined systems are further described below.

Historically, municipal and non-municipal combined and storm pipes discharged directly to the river. When sanitary interceptors were constructed, parallel to the riverbanks, they intersected the municipal combined trunk lines. The combined trunk lines were connected to the interceptor system. Diversion structures in the combined pipes, essentially dams, direct flows that exceed the interceptor capacity to overflow to the river as illustrated below.



The overflow system was built to prevent the interceptor system from being overwhelmed during a storm event. This overflow system allows flows to breach the diversion dams and discharge the combined storm and sanitary sewage flows through an outfall to the Willamette River. Any sewer connections at points in the trunk lines above the diversion structures would overflow to the river only when rainfall caused an exceedance of the approximately three-times-dry-weather flow capacity of the system (City of Portland 1969). Discharges to connections at points in the trunk line downstream of a diversion or connections directly to the interceptor could not overflow to the river during a CSO event. On average, a typical CSO contains about 80 percent stormwater and 20 percent wastewater. Once the interceptors were put in place, the outfalls were referred to as CSO outfalls (CH2M Hill 1992; City of Portland 1952b, 2001b; Stevens & Thompson 1964).

Diversions from the combined system to the river (i.e., CSO events) occur before flows reach the interceptor. Once flows have entered the interceptor, these flows are directed to the treatment plant. During construction of the interceptor system, pump stations were added, some of which included emergency overflow lines that were connected to outfalls, which are known as sanitary sewer overflow (SSO) outfalls. Thus, after the interceptor connections were made, any sewer connections made directly to the interceptor pipe could not overflow to the river unless there was an emergency failure at a pump station. Figure 3.2-2 shows the location of the interceptors and the SSO outfalls.

Historically, most non-municipal combined pipes were along the shoreline, between the interceptors and the river. When an interceptor was constructed, some property owners separated their wastewater from their onsite combined pipes and directed all or some of it to the interceptor.

Starting in 1991, additional reductions of CSO events were implemented by the City of Portland to meet its NPDES permit limits. Once these additional controls were implemented, the outfall was called a Controlled CSO. Additional information on CSO controls implemented during this time is provided in Section 3.2.4.4.

3.2.4.2 Historical Municipal Conveyance Systems

This section summarizes historical information regarding the City of Portland's (City) municipal conveyance system development in the Study Area. There are three major time periods discussed below. The initial period (1880–1947) was when most of the conveyances were combined systems. From 1948 to 1955, interceptors were installed and connected to most of the City's combined system (converting these outfalls to CSOs) and some separated systems were constructed. The final period was from 1956 to 1990, when sanitary service was extended to the northwest area and more separated systems were created. See Current Conveyance Systems (Section 3.2.4.4) for system development changes after 1990.

3.2.4.2.1 1880–1947

The first municipal sewers in Portland Harbor were constructed by the cities of Portland, Linnton, and St. Johns, starting in the 1880s. These sewers were located in the downtown areas of each of these cities. The Linnton and St. Johns outfalls were transferred to the City of Portland when these cities were annexed into Portland, in 1905 and 1915, respectively. These combined sewers collected both surface drainage and sanitary wastes (including domestic and industrial discharges) and, as all private and public sewers did during this time period, discharged directly to the Willamette River (City of Portland 1966b).

In 1936, 48 municipal outfalls directly discharged to the Willamette River (City of Portland 1936), located between RM 17 and RM 4. Nineteen outfalls discharged within the Study Area; these were located in the downtown core (between RMs 9.6 and 11.8 on both sides of the river), in the St. Johns area (between RMs 5.2 and 5.9 on the east shore), and in the Linnton area (between RMs 4.3 and 5.3 on the west shore).

During and immediately after WWII, the City operated six additional combined conveyance systems. Two of these were constructed by the City to serve residential-only areas in St. Johns (OFs 48 and 49). Two were constructed by the Federal Housing Authority, built to serve temporary housing for wartime workers (OFs 18 and 19). Two were constructed by private parties to drain their site and then were transferred to the City under Public Works permits (OFs 20 and 21). OF-20 was abandoned in 1949 and flows were redirected to OF-19. Table 3.2-3 identifies the locations of outfalls, including those constructed during this time period.

Initial scoping of the Sewage Disposal Project included several interceptor sewers and a treatment plant discharging to the Columbia River (Smith 1936). Construction of this project began in 1947 (City of Portland 1952b).

3.2.4.2.2 1948–1955

During this time period, the CBWTP and the northeast and downtown interceptors were constructed, and combined flows from the municipal trunk lines were connected to the interceptors. As shown in Table 3.2-3, within most of the City CSO basins in the Study Area, industrial areas were separated so that industrial wastewater would not overflow

to the river during CSO events. The exception to this was the industrial area in the north part of downtown on the west shore, where separation was more difficult because the area had already been heavily developed. Primary treatment (solids removal) was also added to the two Linnton outfalls as part of the effort to reduce loading to the river. Also during this time period, the City constructed new outfalls or accepted existing outfalls.

In 1947 the City Sewage Disposal Project began the construction of two interceptor lines (the east side and the west side) and a treatment plant (Oregon State Sanitary Service Authority; OSSA 1964). Construction of the interceptor lines diverted most flows to the newly-constructed CBWTP.

The interceptor system in Portland Harbor, as of 1952, is presented on Figure 3.2-3 (City of Portland 1952b); additional work that continued through 1954 is not shown in this figure. The first unit of the interceptor sewer system (serving northeast Portland) was completed in 1947. The CBWTP and the interceptor system on the east side of the Willamette River were completed in 1952. The eastside interceptor system extended from the southern limits of the city north to the treatment plant but did not include the Rivergate area, which was outside the city limits at that time (City of Portland 1952a). Of the eastside CSO outfalls, OF-43 through OF-53 discharged to the Portland Harbor Study Area.[15] During construction of the interceptor system, the City separated the sewers serving most of the industrial areas near the riverfront by building separated sanitary and industrial wastewater sub-basins connected directly to the interceptor, and separated stormwater systems that discharged directly to the river.

Some of these outfalls still had a combined system upgradient of the nearshore industrial separated area that served primarily residential areas. After the interceptors were installed, some properties that formerly discharged through private outfalls directly to the river had the opportunity to connect to the City's sanitary system (either to the new sanitary system installed or directly to the interceptor).

Also during this time, the City constructed a local treatment facility to provide primary treatment (solids removal) for the two Linnton outfalls (OFs 23 and 24) that served primarily residential areas (City of Portland 1999).

By September 1955, the City had completed construction of the westside (including downtown) interceptors incorporating outfalls designated OF-1 through OF-17 (OSSA 1953; 1954a,b; 1955). Of these west side outfalls, OF-11 through OF-17[16] discharge to the Portland Harbor Study Area. Portions of the industrial area that had been connected to the combined system were connected to a separate sanitary sewer that discharged directly to the interceptor.[17] The areas with no separate storm and sanitary systems

[15] OF-43 is at RM 11.4 and OF-53 is at RM 5.2.
[16] OF-11 is at RM 11.4 and OF-17 is at RM 9.6.
[17] Based on City as-built drawings for the interceptor and other separation projects. As-built drawings are available on the City's website: http://PortlandMaps.com.

available continued to discharge to the combined system or discharged directly to the river. The interceptors and associated facilities reduced the volume of untreated sewage discharging to the Willamette from the City's combined system.

3.2.4.2.3 1956–1976

Once the eastside and downtown interceptors were installed, the only areas not served by City sanitary sewer system in Portland Harbor were the northwest (from Guild's Lake to Linnton), Swan Island, Mocks Bottom, and Rivergate areas (see Figures 3.2-3 and 3.2-4). The latter two areas were mostly undeveloped except for a few shoreline facilities. Conveyance systems in the northwest area were operated by private and City systems, and conveyance systems in the Swan Island area were operated by the Port of Portland.

During the time period of 1956 to1976, the City implemented a number of system changes to further reduce combined sewer discharges to the river, including programs to assure that properties connected to the City system were discharging to the appropriate pipe. The City constructed several separate sanitary and storm systems in areas that were undeveloped or were served by non-municipal combined systems. Also during this period, the City completed the northwest interceptor system, which separated stormwater and sanitary sewers in the northwest industrial areas and converted the Linnton neighborhood service areas to CSO areas. By 1975, the City's service area included both separated systems and CSO systems, including new separated systems in the Mocks Bottom/Swan Island and St. Johns areas to support industrial development.

Although the east side interceptor was operational by 1952, studies performed by the OSSA in 1953 identified three outfalls upstream of Portland Harbor where bypass of sewage to the river occurred during dry weather periods (Stevens & Thompson 1964). For example, Stevens & Thompson (1964) estimated as much as 17 million gallons a day (dry weather flows) discharged from diversion structures south of the Sullivan Gulch Pumping Station upstream of Portland Harbor. After completion of the eastside and downtown interceptors, the City implemented a number of efforts to further reduce discharges to Portland Harbor, including the following (City of Portland 1966a):

- Expanding existing systems
- Adding pumping stations to pick up low areas below the interceptors
- Reconstructing or adjusting diversion structures (e.g., raising dams, adjusting orifice diameters, raising or lowing weirs)
- Re-routing combination sewers to discharge above rather than below diversions, thus eliminating dry-weather (sanitary) flows from entering the river
- Where separated systems were constructed, requiring property owners to separate and reroute site sanitary and storm discharges to the appropriate system
- Requiring installation of pretreatment systems for industrial wastes

- Design of the Linnton-Guild's Lake sewerage system to provide facilities for diversion of dry weather flow for existing public sewers and to allow industry to connect to the City system, thus eliminating private industrial and sanitary outfalls to the river
- Design of a stormwater and sanitary system on Swan Island to replace the existing combined system.

In those areas where the City provided separate sanitary and stormwater systems, most properties rerouted their sanitary and stormwater discharges to the appropriate City conveyance. In 1967, the City established additional efforts to ensure compliance with the City's requirements for the separation of storm and sanitary wastes and connection to an approved sewer (City of Portland 1967a). These efforts included review of connection and plumbing records, field inspections, and dye testing to verify that property owners had appropriately rerouted their discharges within the City system. Only stormwater and uncontaminated cooling water could be discharged to the storm system, and only sanitary waste and approved industrial wastewater could be discharged to the sanitary sewer system. For example, in the fiscal year 1967-1968, 77 properties were investigated in Portland Harbor and 23 of these were required to reroute either their stormwater or wastewater (City of Portland 1969). By 1976, investigations and rerouting of wastewater from storm lines to the sanitary sewer was completed (City of Portland 1976).

In the 1960s, the City replaced an existing non-municipal system with a separated sanitary and storm system on Swan Island and Mocks Bottom. In 1969, the City constructed sanitary lines to the International Slip area. In 1972, the City also built a storm system in the St. Johns area to allow development to occur in a previously undeveloped area (depicted in as-built drawings on Portland Maps).

A 1964 study by Stevens & Thompson focused on the sewer systems in the Guild's Lake-Linnton area in northwest Portland and in the area served by the east side interceptor (Stevens & Thompson 1964). Eight combined sewage/stormwater collection systems were identified in the Guild's Lake-Linnton area on the west side of the Willamette River, all within the current Study Area of the Portland Harbor Superfund Site. Most of the collection systems were noted to drain areas dominated by industrial and commercial activities and discharge directly to the Willamette River. This area encompassed drainage to outfalls currently designated OF-17 through OF-24A. Stevens & Thompson also indicated that there were 12 private outfalls in this area discharging industrial wastes (several discharging only cooling water) directly to the Willamette (Stevens & Thompson 1964).

Stevens & Thompson estimated future volumes in the system to assess design parameters for the needed new diversion structures. The analysis compared the 1964 capacities for different interceptor lines/areas against population and flow projections for 1980. Stevens & Thompson determined that increased flows in the southeast and northwest sections, roughly comprising outfalls OF-11 through OF-17 (west side, in the

Study Area) and OF-26 through OF-38 (east side, upstream of the Study Area), respectively, would exceed capacity of the interceptors during periods of maximum flow in the future. However, Stevens & Thompson determined that other sections of the system would be overloaded even if the volume of sewage allowed to bypass diversions in these outfalls were reduced (i.e., if more sewage was diverted to the treatment plant). Based on the results of this study, Stevens & Thompson recommended new and renovated facilities to alleviate overloading and meet the Sanitary Authority's capacity requirements (Stevens & Thompson 1964).

In 1968, the City initiated sewer projects to direct sanitary sewage discharges directly to the CBWTP, including industrial wastewater discharges that were discharging directly to the river in the Guild's Lake-Linnton area. The 1968 projects included construction of a pumping station, the Portsmouth Tunnel, which crossed under the river to the CBWTP, and the northwest interceptor. Construction was completed the following year for the tunnel, pump station, and a portion of the northwest interceptor (from Guild's Lake to the railroad bridge) (City of Portland 1969). By 1973, the northwest interceptor (from the railroad bridge to Linnton) had been completed.

In summary, all City outfalls served by the northwest interceptor were separated, except for the two small residential combined basins in Linnton. Much of this northwest area was not previously served by City outfalls but rather discharged industrial waste directly to the river or to Doane Lake. The interceptor provided a means for industries to eliminate discharges to water bodies by routing their wastewater to the interceptor, although some industries continued to discharge to the river, including some under permits from the State.

3.2.4.2.4 1977–1990

During this time period, the City conducted studies to determine design parameters for future CSO controls and continued implementing programs to detect illicit connections. The City also constructed additional stormwater systems to provide storm service for areas that were being redeveloped or to improve existing drainage. The City incorporated into the municipal system some existing storm and sanitary lines in the newly annexed Rivergate area.

In 1972, the City estimated that the amount of combined sewage overflowing to the Willamette River and Columbia Slough was over 10 billion gallons per year (City of Portland 2001b).

In 1977, the CRAG undertook a study of the greater Portland area to evaluate municipal and industrial wastewater and urban stormwater, including the quality of the overflows from the City of Portland CSO system (CRAG 1977). The study provided a baseline for reevaluating Portland's CSO system. (Several of the reports cited below state different numbers of outfalls in the combined system; this most likely is due to the combination of two outfalls into one or the elimination of some outfalls during the time these reports were completed.) At the time of the CRAG study, there were 43 CSO

outfalls in the City's entire Willamette River CSO system (25 on the east side and 18 on the west side), each draining a basin. Of the 43 outfalls discharging to the Willamette River, 16 discharged within the Study Area, and 27 were upstream of the Study Area up to RM 17.2. The resulting report contained descriptions of each outfall drainage basin, including acreage served, land-use type, pipe size, interceptor, diversions, and details on specific diversions, where applicable. The report also distinguished, in acreage, the type of collection system for each drainage area/outfall. Table 3.2-4 summarizes that information (CRAG 1977).

The CRAG study calculated the average annual runoff of suspended solids, settleable solids, biological oxygen demand (BOD), ammonia, phosphorus, and bacteria from CSOs in the City's entire CSO system, based on historical rainfall data. The study area focused on municipal outfalls discharging to the Willamette River that were within the City of Portland's boundary, and provided discharge estimates for 1975 and projected land use and conditions for the year 2000. Table 3.2-5 lists the 1975 results for suspended solids from the lowest downstream location to the highest upstream location measured.

In 1986 the City issued a Sewer Outfall Report, the purpose of which was to gather information to design future pollution abatement programs (City of Portland 1986). In the 1986 report, the City stated that the CSO system included 57 CSO outfalls, with 44 of the CSOs discharging to the Willamette River (16 of which were in the Portland Harbor Study Area), and 13 discharging to the Columbia Slough (City of Portland 1986). An outfall inspections program was instituted to include observations of the outfalls during the dry season to identify the condition of the outfall and to determine if the dry weather flow had any sanitary component. Dry weather flows could include groundwater infiltration, permitted and non-permitted process water (such as cooling water discharges and landscape irrigation), or illicit connections of sanitary discharges downstream of diversions. These dry weather flows were, and continue to be, analyzed for bacteria to determine if there is any sanitary contribution, and flow volumes are estimated where dry weather flows were evident (City of Portland 1986).

3.2.4.3 Historical Non-Municipal Conveyance Systems

As described in Section 3.2.2, most historical industrial development occurred along the shoreline. At least through the 1960s, very few shoreline facilities were connected to the municipal systems; nearly all managed their own stormwater and wastewater. Most of these discharges were combined (i.e., included stormwater and sanitary and/or industrial wastewater) (OSSA 1950; DOI 1967; Stevens & Thompson 1964). Figure 3.2-4 shows the areas not served by the interceptors in 1963; these areas were predominantly shoreline properties (OSSA 1963). Many of the non-municipal conveyance systems served a single property, but there were several larger shared conveyance systems noted in historical reports such as the Oregon Terminals (currently International Slip) (OSSA 1963) and Swan Island (CH2M Hill 1957).

The Rivergate area was undeveloped until about the 1960s. Separate sanitary and storm systems were installed by the Port to provide conveyance services to this developing area. The Rivergate area was annexed into the City of Portland beginning in 1979 and was completed in 1989, and some of these conveyance systems were transferred to the City after annexation.

OSSA conducted several surveys of industrial dischargers in the Harbor (OSSA 1963, 1966) and beginning in 1967, required waste permit applications to be submitted by dischargers (OSSA 1967c). Additional information about the State discharge permits is provided in Section 4.3.1.4.

3.2.4.4 Current Conveyance Systems

This section focuses on the current physical characteristics of stormwater runoff and conveyance systems, outfalls, combined sewer systems, and NPDES permits in the LWR in the Study Area. Information presented in this section is primarily from the Programmatic Work Plan (Integral et al. 2004) and the *Programmatic Source Control Remedial Investigation Work Plan for the City of Portland Outfalls Project* (CH2M Hill 2004), as well as additional LWG analyses.

3.2.4.4.1 Outfalls

Within the Study Area, outfalls have been installed by a variety of entities, including private landowners, the Port of Portland, the State of Oregon, and the City of Portland. Most of the outfalls currently convey primarily stormwater, although historically some also conveyed industrial and sanitary discharges.

Some outfalls also currently convey nonstormwater discharges. Some nonstormwater discharges, such as noncontact cooling water, must be permitted, while other nonstormwater discharges, such as landscape irrigation, are exempt under federal regulation. As discussed below in Section 3.2.4.4.3, some outfalls include a CSO component as well.

The City of Portland identified over 400 potential public and private outfalls along both shores of the Study Area (City of Portland 2006d). Using site-specific information and field reconnaissance, the LWG independently verified these outfalls and researched areas that potentially had additional outfalls. Incorporating results of the field reconnaissance, a total of 436 total outfalls were identified; of these approximately 313 are active, 44 are inactive, 30 are abandoned, 15 have been removed, 27 are unknown outfalls, and 7 were determined to not be outfalls (Integral 2008a).

The types of outfalls are defined below:

- **Active** = outfall is currently in use
- **Inactive** = outfall pipe exists, and is not filled, plugged, or disconnected, but discharge is presently not occurring

- **Abandoned** = outfall pipe exists but it is filled, plugged, or disconnected, and discharge is not occurring
- **Removed** = outfall pipe has been removed
- **Unknown** = despite best efforts, the status of some outfalls cannot be determined.

Attributes for some outfalls in the data set remain flagged despite repeated attempts by LWG to verify during field work or due to conflicting information from the facility and the City.

The location and status of the outfalls within the Study Area are shown on Maps 3.2-23a–m.

3.2.4.4.2 Stormwater Runoff and Conveyance Systems

Stormwater enters the river via stormwater conveyances, overland flow, and infiltration to groundwater. Stormwater conveyance systems typically consist of ditches, swales, storm drains, inlets, and catch basins connected to the outfall through pipes or lines.

Overland flow of stormwater occurs at some locations immediately adjacent to the river. In many of these areas, the extent to which rainwater falling on pervious ground near the river shoreline results in runoff versus infiltration into the ground is unknown. In some impervious shoreline areas, stormwater appears to be transported to the river via overland flow, with little chance for infiltration into the ground. A preliminary assessment of outfall drainage basins conducted for the Round 2 Report indicated that the area drained by overland flow appears to be relatively small compared to the area in which stormwater is discharged via outfalls. Given the difficulties of defining all stormwater conveyance drainage basins along the river, the proportion of overland flow to the river has not been further quantified for this RI. Nevertheless, this pathway may represent a significant contaminant transport pathway route, especially as it relates to riverbank erosion.

Additionally, stormwater can enter the river indirectly via infiltration into pervious ground (or through dry wells, sumps, and other infiltration facilities), where it is then mixed with groundwater and discharged to the river as groundwater. Groundwater discharges are further discussed in Section 3.1.2 and Section 4.

Most of the stormwater from the west side of the river drains from Forest Park, an area which consists mostly of undeveloped parkland. Streams from Forest Park generally enter underground pipes at the base of the West Hills, near U.S. Highway 30. At this point, the highway stormwater drainage often enters these same conveyance systems. This runoff is comingled with industrial stormwater runoff as it moves through industrial properties between the park and the river. On the east side of the river, there are few open channel drainages, and most of the stormwater is discharged via conveyance systems. Most properties adjacent to the river on both sides do not

discharge through shared conveyance systems but directly discharge to the river via their own stormwater conveyance systems and outfalls or overland flow.

Just under half of the stormwater drainage to the Study Area is through shared conveyance systems; open space comprises about 60 percent of these basins. These systems are further discussed in Section 4.4.1.3 and include shared conveyance systems owned by the City, by Burgard Industrial Park, and by ODOT; multiparty outfalls with unknown ownership; and Saltzman Creek.

In some locations, stormwater is captured by the City of Portland combined conveyance systems and is routed to CBWTP.

Section 4.4.1.2 further discusses the stormwater basins and the types of stormwater discharges, including a map showing a categorization of the different drainage types within the Study Area (i.e., shared conveyances, direct discharge, no discharge, and uncertain drainage).

Figure 3.2-5 shows the hydroboundary, the approximate overall area draining stormwater to the Study Area. The delineation of the overall drainage basin area between RM 1 and 11.8 was provided by the City of Portland (2006c). An analysis of stormwater flow contributions to overall river flows estimated that the Portland Harbor area runoff volume contributions are between 0.06 percent for the wet year conditions (1997) and 0.08 percent for dry year conditions (2001) of the total Willamette River flow. The average annual runoff volume for the Portland Harbor is 0.06 percent of the total Willamette River flow (City of Portland 2006c)

3.2.4.4.3 Combined Sewer Systems

This section focuses on the current City of Portland CSO system. Sanitary interceptor lines run south to north through the main trunk lines, paralleling the riverbanks. Interceptors are large lines that collect sanitary and combined flows and direct them to the treatment plant. Some combined lines have diverters that allow excess flow to discharge to the LWR during heavy storms for a portion of the rainfall event; this is called a CSO. The diverters are designed to protect the interceptor from excess stormwater inflow by diverting the peak portions of the flow. Combined systems without CSO diverters direct all sanitary and stormwater flow to the treatment plant.

In 1991 the City and State signed a Stipulation and Final Order (SFO) that would require the City to eliminate CSO discharges that violated applicable water quality standards. The following year, the federal government issued a CSO policy that was not as stringent as the SFO.[18] The City and State entered into a collaborative process to study the environmental and socioeconomic impacts of the requirements of the SFO; the process also considered additional system characterization information not available in

[18] The 1994 Federal CSO Policy states that an 85 percent volume reduction is an adequate level of control to meet water quality standards (Federal Register, Vol. 59, No. 75, 18692, April 19, 1994), while the City of Portland level of control was a 94 percent reduction (Combined Sewer Overflow Management Plan [CH2MHill 1992]).

1991 (CH2M Hill 1994). As a result of this process, DEQ and the City entered into an amended SFO (ASFO) in 1994 that adopted a standard of control that continued to be more stringent than that required by the 1992 federal CSO Strategy (CH2M Hill 1994). The AFSO granted the City until 2011 to reduce the CSO events to the Willamette from about 100 events per year to four events per year in winter and one event every three summers (City of Portland 2001b). CSO discharges are organized into "events" based on the storm conditions that cause the overflows. Each statistically independent storm event causes one CSO event. A storm in Portland is considered a single statistically independent storm only if there has been a minimum of a preceding 24-hour period with no rain (CH2M Hill 1992, 1994).

At the time the AFSO was signed in 1994, it was estimated that the CSO system discharged an average of 4.8 billion gallons of stormwater (~80 percent) and untreated sewage and pretreated industrial waste (~20 percent) to the river between RM 4 and 17 (CH2M Hill 1994). The discharges occurred through 42 outfalls to the Willamette River, some of which overflowed nearly every time it rained (150 days), while others only overflowed 30 days per year (City of Portland 2001b). The City estimated an average of 50 CSO events (encompassing up to a total of 112 days) per year in the entire CSO system (City of Portland 1998).

Around this same time, it was determined that dry weather discharges of untreated sewage (including pretreated industrial wastewater discharges) were also still occurring in some portions of the City system due to periodic failure of the system to function properly, vandalism, illicit discharges, blockages caused by a variety of sources, and groundwater infiltration, which resulted in the discharge of untreated sanitary sewage through CSO outfalls directly to the Willamette River. These dry weather discharges involved relatively small volumes and are different than wet weather CSO events, which occur when the combined sanitary sewage and stormwater flows exceed the system's capacity during rain events. The City completed improvements to the CSO system between 1992 and 1996, and signed a SFO with DEQ in 1996 in which it agreed to eliminate the dry weather discharges (DEQ 1996).

To achieve the requirements of the 1994 ASFO, the City prepared a CSO Management Plan with recommendations to address wet weather overflow discharges, including the following:

- Implementation of "Cornerstone Projects" focused on reducing the volume of stormwater to the system
- Implementation of storage and treatment facilities to eliminate the CSO discharges to the Columbia Slough as required in the SFO
- Implementation of storage and treatment facilities along the Willamette River ("Big Pipe project") to control the CSO discharges as required by the ASFO.

The City is now in the final year of the 20-year project, with completion planned for 2011 (City of Portland 2008a). The current City outfall system is discussed in Section

3.2.4.4. As described in that section and in Table 3.2-3, the CSO abatement projects include one or more of the following for each outfall:

- Completely separating storm and sanitary to create stormwater-only outfalls with stormwater treatment prior to discharge, where possible
- Completely sealing and abandoning outfalls or diversions to prevent overflows
- Reducing stormwater flows to the CSO system to minimize flow through the system during a storm event, such that the system meets the AFSO standard
- Increasing the storage capacity for the CSO system to reduce the frequency of overflows to meet the AFSO standard.

The primary means for increasing the storage capacity is through construction of the West Side Tunnel (completed in 2006) and the East Side Tunnel (anticipated to be completed in 2011). The ASFO requires the City to control 16 CSO outfalls by 2006 and all remaining CSO outfalls by 2011. The City has met all requirements of the ASFO to date and is on schedule to complete the CSO abatement program in 2011. The goal of the abatement projects is to meet the AFSO design standard to control CSO discharges to an average of four events in the winter (November 1 to April 30) and one event in three summers (May 1 to October 31; City of Portland 2005b). Since completion of the West Side CSO Tunnel Project in 2006 there have been three winter seasons and a total of five events for all Willamette River outfalls that were controlled by 2006—fewer than an average of two events per winter since completion of construction of the West Side tunnel. The overflow points in the Study Area were outfalls OF-11 and OF-47.

The abatement projects, including the West Side Tunnel and the final selected design for the East Side Tunnel, are projected to meet the CSO system demands and AFSO design standard by 2011 and continue through the year 2025. This projection is based on the assumption that other City programs will continue to implement mitigation measures to reduce stormwater flow to the overall CSO system by initiating projects not listed in the report (e.g., infiltration basins, green roofs, and other such stormwater reduction measures). The City noted that additional efforts would be required to control CSO demands beyond 2025 (City of Portland 2005b). The configuration and dates of the abatement project for the separate and combined sewer systems is shown on Figure 3.2-2.

The City of Portland is now in the final year of the CSO abatement program. An abatement is defined as including both full and partial separation techniques. The location and status of CSO outfalls, including a summary of abatements completed and planned within the Study Area (City of Portland 2008a), is provided in the Table 3.2-3 and shown in Map 3.2-23a–m. The abatements listed include the following:

- Controlled to 3-year summer/4-per-winter storms
- Sealed off, which means that no sewage discharge can occur.

A previously combined system that has been fully separated sends sanitary wastewater to the CBWTP and stormwater to the LWR through separate conveyance systems. The stormwater may be diverted to a different outfall, or it may use the former CSO outfall. In the case of the latter, the CSO outfall becomes an MS4 outfall. Table 3.2-3 identifies which situation applies to each fully separated outfall. A partially separated CSO system conveys the combined sanitary and industrial wastewater and significantly reduced stormwater to the CBWTP except during extreme wet weather events when a portion of the combined flow overflows to the Willamette River due to capacity limitations.

Table 3.2-3 also shows the combined or CSO outfalls that were either converted to stormwater-only outfalls or abandoned before the City's 20-year abatement program was initiated.

3.2.4.4.4 NPDES Permits

Discharges to the river are regulated by a variety of NPDES permits, including multiple general NPDES stormwater permits (e.g., 1200-Z, 1200-C); the City of Portland, Port of Portland, and Multnomah County Municipal Separate Storm Sewer System (MS4) NPDES discharge permits; the City of Portland NPDES wastewater discharge permit (primarily for the discharge from the CBWTP to the Columbia River, but also including combined sewer overflows and sewer system overflows into the LWR); individual stormwater NPDES permits; and individual wastewater NPDES discharge permits. ODOT also has its own MS4 NPDES discharge permit for runoff from state highways. These permits are described further in Section 4.4.1.4. Many stormwater discharges are not regulated through MS4 or NPDES permits.

3.2.5 Historical and Current Harbor Use

3.2.5.1 Navigational Channel Authorization History

Congress authorized the LWR federal navigation project through the Rivers and Harbors Act in June 1878. Its purpose was to deepen and maintain parts of the Columbia and Willamette rivers to a 20-ft minimum depth. The USACE maintains the channel for both rivers, both of which have been deepened at various intervals since that time. Most significantly, the authorizations affecting the LWR depth occurred as follows: –25 ft CRD in 1899, –30 ft CRD in 1912, –35 ft CRD between 1930 and 1935, and, finally, –40 ft CRD in 1962.

The current project authorization, as modified by Congress in 1962, encompasses 11.7 miles of the Willamette River in Portland and 103.5 miles of the Columbia River below Vancouver, Washington. Work on the authorized –40-ft-deep CRD channel from Portland and Vancouver to the Pacific was completed in 1976. The Willamette River channel from the Broadway Bridge (RM 11.7) to the mouth (RM 0) varies in width from 600 to 1,900 ft, with an average width of approximately 1,700 ft.

In 1999, Congress authorized the Willamette River (and Columbia River) deepening to –43 ft CRD. The existing 600-ft-wide, 40-ft-deep Willamette River navigation project channel would be deepened from RM 0 to RM 11.6 (USACE 1999).

3.2.5.2 Dredging and Capping Activities

This section presents Portland Harbor dredging and capping activities since 1997. This date corresponds to the oldest data used in the presentation and evaluation of analytical data in this report. This section also notes ongoing and upcoming dredging projects in Portland Harbor.

In certain areas of Portland Harbor, periodic dredging is necessary to maintain the authorized depth of the navigation channel, as well as to maintain operational depths at docks and wharfs. Major changes in the river's bathymetry from 1888 to 2001 are depicted on Map 3.1-6. This map shows how the original shoreline was altered and filled, where material was excavated to create new uplands, and how most of the original channel has been deepened by at least 10 to 20 ft to reach the authorized federal navigation channel depth of –40 ft CRD. Historically, periodic dredging was needed to maintain this depth in two major shoaling areas, between RM 8 and 10, particularly in the western half of the channel, and from RM 2 to 2.5 in the eastern portion of the channel. The navigation channel has not been dredged since January 1997, although dredging at various docking facilities has occurred on an as-needed basis (Map 3.2-24). Currently, maintenance dredging has been suspended until issues are resolved regarding dredging within the boundaries of the Portland Harbor Superfund Site. The lack of maintenance dredging over the past 10 years has resulted in significant shoaling of the channel. Many areas of the channel are now less than 40 ft deep, which is a significant navigation hazard to large cargo ships that require a minimum draft of 40 ft. A critical area of shoaling in the river that needs immediate attention is Post Office Bar at RM 2. According to the USACE (2010a), this dredging is proposed to take place in the summer of 2011.

Dredging projects undertaken since 1997 by the Port of Portland, USACE, the City of Portland, and private parties are listed in Table 3.2-6. This table is an update of a similar compilation provided in the Programmatic Work Plan (Integral et al. 2004) and the Round 2 Report. The dredging projects that are italicized in the table indicate recent projects for which a USACE public notice has been issued, but specific information about dredging dates and amounts was not available in time for this report. Note that the issuance of a permit does not mean that the project was implemented or that the volume of dredged material indicated in the table was dredged. Furthermore, the table does not distinguish between single events and multi-year permits. Map 3.2-24 shows the locations of dredging and capping operations between RM 1 and 11.8, since the most recent USACE-sponsored dredging of the federal navigation channel in January 1997.

Since 1997, the Port of Portland has performed maintenance dredging at its marine Terminals 2, 4, and 5 (see Table 3.2-6). Maintenance dredging has also been performed

by Schnitzer Steel Industries, Inc. (Schnitzer berths in International Terminal Slip, RM 4), Chevron (Willbridge Terminal, RM 7.5), the City of Portland (Portland Fire Bureau Station 6 Dock, RM 9.7), the former Goldendale Aluminum Company (Goldendale Aluminum facility dock, RM 10), and CDL Pacific Grain (Irving Elevator Terminal, RM 11.4). The City of Portland project also included cap placement, as noted below. Brief descriptions of these dredging projects are provided below:

- Schnitzer performed maintenance dredging of its berths located inside the International Terminal Slip in 2004 under two separate permits. Approximately 77,000 yd^3 of material was dredged from Berths 1, 2, and 3 under Permit #199100099. Maximum target dredge depths were –42, –38, or –24 ft CRD, depending on the location within the slip. Outside the slip, Schnitzer dredged approximately 61,000 yd^3 of material from Berths 4 (to –42 ft CRD) and 5 (to –36 ft CRD) under Permit #199200812. The permits for both projects allow for biannual maintenance dredging through January 31, 2009 (USACE 2004a,b).
- In 2001, Chevron Products removed approximately 15,000 yd^3 of material from both sides of its pier at Willbridge Terminal. The dredging was performed under a maintenance dredging permit issued in 1997. Sediments were removed to a target dredge depth of –40 ft CRD (PNG 2001).
- The former Goldendale Aluminum Company conducted maintenance dredging at its dock in 2000. Dredging volumes were not provided, but material was removed to –38 ft CRD (CH2M Hill 2000).
- The City of Portland performed maintenance dredging of the Portland Fire Bureau Station 6 Dock in 2005. The area approaching the dock was dredged to –12 ft CRD, and the area adjacent to the dock was dredged to –10 ft CRD. Altogether, 4,130 yd^3 of dredged material was removed. In accordance with the permit, both areas were capped to bring the bottom grade to between –10 and –11 ft CRD. Approximately 1,190 yd^3 of capping material was used (CH2M Hill 2005).
- CDL Pacific Grain performed maintenance dredging at two separate locations at the Irving Elevator Terminal in 2009. The dredge volume was approximately 1,430 yd^3 (NRC 2009). In 2001, approximately 5,000 yd^3 was removed to a permitted depth of –40 ft CRD (Harding ESE 2001).
- The dock area offshore of Glacier NW (RM 11.3E) was dredged in 2004, but no as-built drawings are available to determine the volume removed and the exact footprint.
- As part of the Terminal 4 Early Action removal, approximately 13,000 yd^3 of contaminated sediments were dredged from Slip 3 in 2008 (discussed further below).

Maintenance dredging is planned for the dock areas offshore of Gunderson, the Portland Shipyard (Cascade General), and at ConocoPhillips and Chevron properties in the Willbridge Terminal complex.

Dredging and/or capping have also been completed or are in process as part of remedial actions at selected Portland Harbor locations. Interim removal action activities at Terminal 4 are underway and are scheduled to occur in two phases. The first phase, which was completed in the fall of 2008, included dredging of approximately 13,000 yd^3 of contaminated sediment and placement in an offsite disposal facility, isolating contaminated sediment in the back of Slip 3 with a cap made of an organoclay-sand mix, and re-contouring the slope of the bank along Wheeler Bay and planting native vegetation to minimize erosion and improve stability. The second phase of the Terminal 4 project includes a combination of dredging, capping, and monitored natural recovery (MNR) in areas not completely addressed by the first phase, as well as construction of a confined disposal facility in Slip 1. The second phase design and implementation of the Terminal 4 removal action will be implemented after the Portland Harbor Record of Decision (Port of Portland 2011).

At the ARCO BP terminal, a new steel sheetpile wall was installed in 2007 to stabilize the facility and prevent migration of contaminants to the river. The following year, the concrete revetment riverward of the new sheetpile wall was removed, along with 13,293 yd^3 of underlying and nearshore contaminated sediment, which was disposed of offsite. Clean fill was placed in the excavated area (DEQ 2010a).

Two in-river sediment capping projects (M&B and Gasco) have taken place since 2003. M&B was a remedial action project following a ROD, and Gasco was an interim removal action. Both projects are described below.

Sediment cap construction activities at M&B, a former wood treating facility, were completed in September 2005. (Subsequent modifications to the cap were performed in October 2005 and July 2007.) The cap's shoreward boundary extends from the south end of the property north into Willamette Cove (RM 6.8). Its offshore boundary extends up to approximately 700 ft from the shoreline. In Willamette Cove, the cap extends offshore up to approximately 600 ft. Approximately 23 acres of contaminated sediments were capped with 2 ft of sand. More highly contaminated areas were capped with 5 ft of sand. In addition, multiple areas of the cap overlying seeps were constructed with a total of 600 tons of organoclay, a bentonite or hectorite clay altered to be hydrophobic. The cap design incorporated different types of armoring (i.e., articulating concrete block mats and rock) in the nearshore areas to reduce erosion (DEQ 2005).

In 2005, pursuant to an EPA Administrative Order, approximately 15,300 yd^3 of tar and tar-contaminated sediment were removed by dredging from the riverbank and nearshore area adjacent to the Gasco facility and disposed of at the Chemical Waste Management landfill in Arlington, Oregon. After the removal action, an organoclay mat was placed along an upper-elevation band of the shoreline dredge-cut. This mat was secured with

placement of cap sand and quarry spalls over the clay mat. The remainder of the removal area (0.4 acres) received 1 ft of cap sand and 0.5 ft of erosion protection gravel. In addition, 2.3 acres of the area surrounding the removal area received 0.5 ft of "fringe cap" sand material. The removal action also created a depression into which potential seepage could be captured and localized for future response. Construction activities took place between August and October 2005 (Parametrix 2006).

3.2.6 Human Access and Use

This section describes the current understanding of the physical and biological setting of the Study Area as it pertains to potential human uses, including specialized groups that may use the river for various activities. Most of the demographic information relating to the Study Area is based on historical background and documented human uses. This information is used to determine potential receptor populations and to develop the general CSM.

Portland Harbor and the Willamette River have served as a major industrial water corridor for more than a century. Industrial use of the Study Area and adjacent areas has been extensive. The majority of the Study Area is currently zoned for industrial land use and is designated as an "Industrial Sanctuary" on the Portland Comprehensive Plan Map (City of Portland 2006b). The Portland industrial sanctuary policy is designed to encourage the growth of industrial activities in the city by preserving industrial land. In addition to industrial use zoning designation, the City of Portland citywide zoning map (January 2009) displays several other zoning designations for smaller portions of the Study Area including: open space (e.g., Cathedral Park and Willamette Cove); general employment (mixed use allowed though primarily an industrial use focus); and multi-dwelling residential (e.g., University of Portland). The zoning codes apply to lands along the river and not to the actual river itself.

As shown in Map 3.2-22a-e, the Guild's Lake Industrial Sanctuary Plan (GLISP), which covers one portion of the Study Area zoned for industrial use, is intended to preserve and enhance industrial land in the Guild's Lake area, generally bounded by Vaughn Street on the south, the St. Johns Bridge on the north, Highway 30 on the west, and the Willamette River on the east (City of Portland 2001a). Over many decades, public and private investments in infrastructure, such as marine, rail, and highway facilities, as well as investments in industrial physical plants, have occurred within this area. The stated purpose of the GLISP is to maintain and protect this area for heavy and general industrial uses. The plan's objectives were adopted as part of Portland's Comprehensive Plan to ensure preservation of this land use over the next 20 years.

Much of the shoreline in the Study Area includes relatively steeply sloped banks. Some sections of the shoreline are also covered with riprap or constructed bulkheads, and other human-made structures such as piers and wharfs extend out over the water in various locations. The Willamette Greenway Plan addresses the quality of the natural and human environment along the Willamette River and generally includes all land adjacent to the river, public lands near the river, and land necessary for conservation of

significant riparian habitat. (The Willamette Greenway Plan was adopted by City Council November 5, 1987, Ordinance 160237.) The Greenway Plan is intended to "protect, conserve, enhance, and maintain the natural, scenic, historical, economic, and recreational qualities of lands along Portland's rivers" (Portland City Code Chapter 33.440). The plan supports industrial uses within Portland Harbor while at the same time looks to increase public access to the river. As a result, recreational use within the Study Area may increase at certain locations in the future.

People interact with the riverine environments in a number of ways. Worker activities that may include contact with sediments and surface water at industrial and commercial facilities in the Study Area are limited in the shoreline areas due to the sparse beach areas and high docks associated with most of the facilities.

In addition, the LWR provides many natural areas and recreational opportunities, both within the river itself and along the riverbanks. Within the Study Area, Cathedral Park, located under the St. Johns Bridge, includes a sandy beach area and a public boat ramp and is used for water skiing, occasional swimming, and waterfront recreation. Recreational beach use also may occur within Willamette Cove, which is a riverfront natural area; in Swan Island Lagoon; and on the southern end of Sauvie Island, which is within the Study Area. Swan Island Lagoon includes a public boat ramp. Additional LWR recreational beach areas exist on the northern end of Sauvie Island and in Kelley Point Park, both of which are outside of the Study Area. Willamette Cove is planned to be turned into an open greenspace that will serve as an extension of the Willamette River Greenway. Potential human use beach areas in the Study Area are shown in Map 3.2-25.

Diving activity also occurs in the LWR. In the Study Area, the majority of divers are expected to be commercial divers. Some diving for scientific purposes, including some aspects of site characterization for this RI, has occurred in the Study Area. In a memo dated September 2, 2008, the EPA provided information on the types of diving conducted in the Portland Harbor area. According to the EPA, "diving is done by several groups of people including: the public for recreation and gathering of biota for consumption; the sheriff's office for investigations and emergency activities; and, commercial divers for a variety of purposes, including marine construction, underwater inspections, routine operation and maintenance, and activities related to environmental work. The majority of divers are expected to be commercial divers."

The St. Johns Town Center is a mixed-use district that extends to the waterfront on the east side of the Willamette River at the St. Johns Bridge. The recent St. Johns/Lombard Plan (City of Portland 2004) includes a proposed redevelopment of this area near the Willamette River. The Riverfront Subdistrict included in the St. Johns/Lombard Plan is currently zoned as Open Space and as a Central Employment (EX) zone. The development standards of the Central Employment (EX) zone are intended to ensure that the Riverfront Subdistrict is developed in a manner consistent with adjacent areas

and to support existing industry by limiting uses that may be less compatible with industry (City of Portland 2004).

The exact extent to which commercial fishing occurs within the Study Area is currently not known. No reports of commercial fisheries for anadromous salmonids on the Willamette River have been found. A commercial crayfish fishery exists in the LWR. Crayfish landings must be reported to ODFW by water body and county. Per ODFW, the crayfish fishery is not considered a large fishery (Grooms 2008, pers. comm.). Based on ODFW's data for 2005–2007, no commercial crayfish landings were reported for the Willamette River in Multnomah County.

Non-commercial fishing is conducted throughout the LWR basin and within the Study Area, both by boaters and from locations along the banks. Limited interviews by ATSDR suggest that the groups most likely to be catching and eating fish from the LWR are immigrants from Eastern Europe and Asia, African-Americans, and Hispanics. The same source also suggests that the most consumed species are carp, brown bullhead (a catfish), crappie, and smallmouth bass (ATSDR 2002a). Other sources (CRITFC 1994) suggest that Native Americans fish in the Willamette River. The LWR provides a ceremonial and subsistence fishery for Pacific lamprey (particularly at Willamette Falls) and spring Chinook salmon for Native American tribes. Many areas in the LWR are also important for cultural and spiritual uses by local Native Americans. There is also an active recreational fishery for salmon, steelhead, and sturgeon in the LWR.

Transients have been observed along the LWR, including some locations within the Study Area. The observation of tents and makeshift dwellings during RI sampling events confirms that transients were present along some riverbank areas. Transients are expected to intermittently utilize this area in the future. Transients may be using the LWR as a source of drinking water. Conversations were conducted with transients about their consumption of fish or shellfish from the Willamette River as part of a project by the Linnton Community Center (Wagner 2004, pers. comm.). The transients that were contacted reported harvesting and consuming various fish species as well as crayfish and clams. It should be noted that the most common clam species in the Portland Harbor is an invasive species, the harvesting of which is illegal. Many of the individuals indicated that they were in the area temporarily, move from location to location frequently, or have variable diets based on what is easily available. However, the interviews did not quantify the frequency or duration of transient presence along the shoreline.

Under OAR 340-041-0340, Table 340A, the designated beneficial use of the LWR includes public domestic water supply after adequate pretreatment to meet drinking water standards. There are no known current or anticipated future uses of the LWR within Portland Harbor as a domestic water supply. According to the City of Portland, the primary domestic water source for Portland is the Bull Run watershed, which is supplemented by a groundwater supply from the Columbia South Shore Well Field

(City of Portland 2008d). In addition, the Willamette River was determined not to be a viable water source for future water demands through 2030 (City of Portland 2008d). Upstream of Portland Harbor, the City of Wilsonville does use the LWR as a domestic water source following treatment.

NOTE TO REVIEWERS:
Section 4 text has been reorganized since the Draft RI submittal. Text from the Draft RI Section 4.1 Historical Land Use was moved to Section 3 and shows as strike out here in its original location. Some paragraphs in Section 4.3.1 Direct Discharge – Industrial Wastewater, Stormwater, and CSOs (under historical sources) and Section 4.4.1 Direct Discharge – Industrial Wastewater, Stormwater, and CSOs (under current sources) have been moved within these sections to improve readability. This moved text shows as strike out in its original location but is not shown as redline in its new location for ease of review.

4.0 IDENTIFICATION OF SOURCES

As part of the Portland Harbor RI/FS, the LWG is required to identify sources that are contributing to contamination of the in-water portion of the Study Area. The SOW (EPA 2001a and amendments) states:

> Respondents will identify sources of contamination to the in-water portion of the Site, including those sources identified based on information obtained through DEQ summaries . . . (Section 7)
>
> Respondents will identify source areas that are contributing to contamination to the in-water portion of the Site. Although DEQ is primarily responsible for the control of upland contaminant sources to the Site, as part of the RI/FS, Respondents shall evaluate the distributions of sediment contaminants and, if appropriate (e.g., if the sediment data suggests the presence of an ongoing source), make recommendations to EPA and DEQ if the need for further investigation or control of sources is identified. EPA and DEQ will utilize this information in making source control adequacy determinations. Because upland sites represent many of the known contaminant sources, coordination with upland investigations and DEQ source control efforts will be required. (Section 7.4)

The National Contingency Plan Section 300.430(d) provides the requirements of an RI, and Section (d)(2)(4) states that the investigation [should] include the extent to which the source can be adequately identified and characterized.

The LWR and its adjacent upland areas have been used for industrial, commercial, and shipping operations for nearly a century. During this time, contaminants associated with those practices were released from various sources through migration pathways to the LWR sediments, which may pose risk to receptors. Many private- and agency-led environmental investigations have been conducted, both in the LWR and on adjacent upland properties, to further characterize the nature and extent of contamination in the LWR, as well as to identify potential sources of contaminants that could continue to enter the LWR.

The primary focus of this section is the discussion, by pathway, of the historical and current sources that contributed to in-river contamination within the Study Area. Section 4.1 discusses the potential migration pathways of sources to the LWR. The site

summary process, which was initiated in 2004, and its role in identifying sources and pathways within the Study Area as well as DEQ's Milestone Report are described in Section 4.2. Historical sources of contamination are presented in Section 4.3, and current sources of contamination, including ongoing secondary releases, and controls are presented in Section 4.4. Potential sources outside the Study Area from other reaches of the LWR and above Willamette Falls are identified in Section 4.5. Section 4.6 discusses DEQ's role in indentifying, evaluating, and controlling sources of contamination that may affect the LWR and the status of source control measures at upland sites in the Portland Harbor JSCS.

Although this section and its associated tables identify many specific sources of contamination, neither this section nor this RI report generally is intended by the LWG as an exhaustive list of current or historical sources of contamination. Identification and evaluation of potential sources is still ongoing. For example, EPA is still receiving and reviewing responses to information it requested in 2008 pursuant to CERCLA §104(e), and the LWG has not reviewed the majority of this information. All historical sources may never be known, and current sources likely will continue to be discovered into the future. However, sufficient information about likely significant historical and current sources is available sufficient for preparation of the FS to begin.

Sources are one component of the CSM for the Portland Harbor. Other components of the CSM are presented elsewhere in this RI:

- Historical and current land and harbor use is presented in Section 3.
- The distribution of select indicator chemicals in environmental media in the Study Area is presented in Section 5.
- The methods and information used to evaluate fate and transport of contaminants from potential sources is evaluated in Section 6.
- Human health and ecological risk evaluations are used to identify receptors potentially exposed to unacceptable risk in Sections 8 and 9.
- A description of the relationship between all of these elements (i.e., contaminant distribution, sources, fate and transport, and risk) completes the CSM in Section 10.

4.1 POTENTIAL MIGRATION PATHWAYS

Contaminants released to media such as air, soil, ground water, surface water, or impervious surfaces may migrate to the LWR via the following pathways[1]: direct discharge, overland transport, groundwater, riverbank erosion, atmospheric deposition, overwater activities, and upstream watershed. This section briefly highlights these

[1] This section addresses external contaminant sources and migration pathways to Study Area sediments. Internal processes, such as bedded sediment resuspension and erosion, are important internal contaminant fate and transport mechanisms and these are discussed in RI Sections 6 and 10 and evaluated in the FS.

potential pathways to the LWR. In-depth information on specific sources and migration pathways is provided in Sections 4.2 through 4.5.

4.1.1 Direct Discharge

In general, direct discharge systems include municipal or other publicly owned drainage systems, privately owned and managed drainage systems, and sanitary/combined sewer systems. Based on a survey in 2002, approximately 300 individual points that discharge into the Portland Harbor Study Area have been identified. These individual points, often referred to as outfalls, are defined here as locations of discharge of stormwater, combined sanitary sewage and stormwater, and/or industrial wastewaters transported via a collection system, although most of the latter two are now routed through the sanitary sewer and no longer discharge directly to the waterway. The classifications of direct discharges include CSOs, emergency overflows, and public and private storm drain systems.

4.1.2 Overland Transport

Contaminated surface soils exposed in the upland areas can be carried directly to the river in uncontrolled flow of water from a site (i.e., not a point discharge). Overland transport was likely to have been more important historically, prior to the development of extensive stormwater conveyance systems within the Study Area. However, specific historical information on overland runoff is lacking for most sites.

4.1.3 Groundwater

Groundwater flow in the greater Portland Basin within the Study Area is generally towards the LWR, although the direction varies locally depending on the nature of subsurface materials, hydrostratigraphy, and proximity to the LWR. Near the LWR, tidal action can greatly alter groundwater flow directions, rates, and water quality. Contaminated groundwater may enter directly into the Portland Harbor Study Area via discharge through sediments or bank seeps, or it may infiltrate into storm drains/pipes, ditches or creeks that discharge to the river. Contaminant migration may occur as NAPLs or as dissolved phase transport.

4.1.4 Riverbank Erosion

Contaminated riverbank soil may release contaminants directly to the Portland Harbor Study Area through bank erosion. Unprotected shoreline banks are susceptible to erosion by wind, river flows, wave action, tidal changes, and surface water runoff. Shoreline armoring and vegetation reduce bank erosion. Bank slope is also a factor; steeper banks are more susceptible to erosion.

4.1.5 Atmospheric Deposition

Contaminants are emitted to the air from both point and nonpoint sources. Point sources include a variety of industrial facilities. Nonpoint sources include mobile emissions from motor vehicles, marine vessels, and trains, as well as common materials (e.g., plastics) through off-gassing. Contaminants emitted to the air may be transported over

long distances, generally in the direction of the area's prevailing winds. They can be deposited from the atmosphere to land and water surfaces through wet deposition (precipitation) or dry deposition (as particles). Air pollutants can be deposited to water bodies through either direct or indirect deposition. Direct deposition occurs when contaminants are deposited onto the surface of a water body. Indirect deposition occurs when contaminants are first deposited on land and then transported to the water body via surface water or stormwater runoff.

4.1.6 Overwater Activities

Contaminants from overwater activities (e.g., sandblasting, painting, unloading, maintenance, repair, and operations) at riverside docks, wharves, or piers; discharges from vessels (e.g., gray, bilge, ballast water); fuel releases; and spills have the potential to impact the LWR.

4.1.7 Upstream Watershed

Upstream sources include or have included sewers, stormwater runoff, and direct discharge of industrial wastes; agricultural runoff; and aerial deposition of global or regional contaminants on the river water surface and drainage areas within the Willamette Valley. For a discussion of contaminants within the upstream watershed see Section 4.5.

4.2 SITE SUMMARIES AND SOURCE TABLE

A goal of the RI Report is to "identify sources of contamination to the in-water portion of the Site, including those sources identified based on information obtained through DEQ summaries" (Section 7 of SOW). Sources of contamination are often hazardous substances contained in drums, storage tanks, surface impoundments, waste piles, and landfills, and the widespread use of pesticides/herbicides, fertilizers, and other chemicals. Heavily contaminated media (such as soils and groundwater) may also be considered secondary sources of contamination, especially if the original source (such as a leaking tank) is no longer releasing contaminants.

Sources can be either historical or current in origin. Historical sources have released COIs to the river in the past, but no longer have an upland source to control. Current sources are releases of COIs from historical or current activities that are migrating to the river through a migration pathway that needs to be controlled. Current sources are defined as those that are active after 2004, the year that LWG began reviewing upland source information.

DEQ is the lead agency responsible for the cleanup of known or potentially contaminated upland sites, which are tracked in DEQ's ECSI database. It is important to note that sources of contamination in Portland Harbor may include ECSI and non-ECSI sites.

4.2.1 Site Summaries

Site summaries are the primary vehicle for assembling information on upland sources for the Portland Harbor RI/FS, and are integral to the CSM for the Study Area. Summaries have been prepared for upland ECSI sites that were generally located within 0.5 mile of the LWR between RM 2 and 11. Map 4.2-1 depicts the locations of the ECSI sites within the Study Area.[2]

Each site summary describes general ECSI site information (location, physical description); owner history; current and historical site uses; potential sources (overwater activities, recent and historical spills); the nature and extent of contaminants in soil, groundwater, surface water, and sediment; stormwater and wastewater permit information; and a summary of cleanup actions. Site summaries have been updated periodically, primarily from information on file with DEQ.

Site summaries were originally published in 2004-2005 (Integral and GSI 2005a,b,c), and a subset of the summaries was updated in 2007 (Integral 2007a). The status of the ECSI sites within the Study Area is tabulated in Table 4.2-1. For each site listed in the table, the following information is shown:

- The site name and ECSI number
- The site status (e.g., remedial investigation, expanded preliminary assessment, not in DEQ cleanup program)
- The site summary documents prepared (e.g., site summary, site summary addendum, no site summary prepared) and dates of documents.

It is important to note that site summaries have not been prepared for all ECSI sites within the Portland Harbor hydrographic basin. Not all ECSI sites that might be potential sources of contamination to the Study Area are included within the geographic boundaries described in the paragraph above. Also, potential sources of contamination may exist that have not been reported or included in the ECSI database. For example, ECSI sites have been newly identified based on DEQ stormwater investigations. The PTRR Guilds Lake Yard located between RM 8.6 and 9.5 (ECSI #100) and the City of Portland outfalls located in the Study Area between RM 2.7 and RM 9.8 (ECSI #2425) were not part of the site summary process described in this section. The City's CSOs are discussed in Section 4.4.1.3. Other ECSI sites may be either connected to an outfall more than 0.5 mile from the river or located upriver or downriver of the original Study Area boundaries. In addition, site summaries have not been prepared for all sites associated with parties that have received general notice letters.

The site summaries are based on a review of information in the associated DEQ ECSI files and other readily available site information, including, in the case of LWG-

[2] ECSI sites in Map 4.2-1 are as of February 2011. The locations shown on the map are based on the DEQ GIS data augmented by GIS data provided by the City of Portland. The locations of the sites on the two GIS data files can be different.

member sites, information provided by the site owner. It is important to note that the development of site summaries and the source information presented here is highly dependent on whether a site is involved in DEQ's cleanup program and the degree of investigation and data generation, as well as the level of detail provided in DEQ's ECSI database. As shown on Table 4.2-1, several sites adjacent to or near the Study Area are not in the cleanup program, and it is likely that many sites, particularly those that are the location of historical facilities that operated outside the boundaries of current sites, are not fully addressed in DEQ files. As a result, this section does not represent a complete inventory of sites and operations that contribute or have contributed to contamination in Portland Harbor. These limitations on source information primarily affect historical sources, and the understanding of current sources is adequate for the purposes of the FS.

4.2.2 Source Table

The summaries and addenda are the basis for the pathway information for upland sites provided in Table 4.2-2 (formerly Table 5.1-2 of the Round 2 Report), otherwise known as the source table. This table summarizes ECSI sites that are being investigated by DEQ through its site cleanup program, a few of which are non-ECSI sites (e.g., TSCA-led cleanups). A chronology of modifications to the site summaries and source table is provided below:

- In 2003, summary descriptions focused on the groundwater pathway were prepared for the ECSI sites (GSI 2003a).
- Following discussions with EPA and its partners in early 2004, updated site summaries were prepared in 2004-2005 (Integral and GSI 2005a,b,c). This iteration provided information on all pathways potentially contributing to in-river contamination.
- A second update of site summaries was prepared in 2006 (Integral 2007a). At the request of the LWG, DEQ (Anderson 2006a,b, pers. comm.) provided information concerning the status of cleanup actions and regulatory decisions at sites not owned by members of the LWG (i.e., non-member sites). New or completely revised site summaries were prepared for sites that had not previously been described, for sites where EPA-requested revisions had not previously been submitted, and for sites where significant new work warranted a complete revision. Addenda to existing summaries were prepared for sites where significant additional information was provided on one or two pathways.
- In 2007, based on recent sampling that indicated elevated PCBs in sediment and surface water, attention was focused on the east side of the river from RM 11 to 11.6 in the vicinity of the former Albina Shipyard and the current Glacier NW and Cargill facilities. A site summary was prepared for this stretch of the river and was added to Table 4.2-2.
- When the Study Area was expanded in 2008 to include RM 1.9 to 11.8, 20 additional ECSI sites were added to Table 4.2-2. Pathway information and COIs

at these sites were obtained from the ECSI database; DEQ files were not reviewed and no site summaries were prepared.

- In January 2008, EPA commented on pathway information provided in Table 5.1-2 of the Round 2 Report. The LWG responded to those comments and EPA commented on this response.
- In December 2008, Table 4.2-2 was further updated for LWG member sites and updated with minor revisions for non-member sites, as a result of the detailed review of the table for CSM indicator chemicals and development of Section 10 source tables. With a few exceptions, Table 4.2-2 reflects site conditions as of September 2008.
- In January 2009, the overwater release pathway for both historical (H) and current (C) impacts was modified based on the updated spill table (Table 4.2-3). The distinction between historical and current impacts is January 1, 2004.
- The completeness of the wastewater pathway on in-water media was not evaluated; however, sites with individual wastewater permits are identified in Table 4.2-2 and historical wastewater discharges are discussed in Section 4.1.3.4.
- In February 2009, DEQ's LUST database and other Internet records were searched for confirmed releases at sites with no groundwater data. If a release was confirmed, the COI box was entered as NS (x,y)—no sampling, confirmed releases include chemicals x and y—but the status remained current, insufficient data to make a determination (C-c).
- Consistent with its agreement with EPA, the LWG used the September 2010 version of the Milestone Report and meetings/communications with EPA/DEQ on the table entries as the means for populating a source inventory table for the FS and updating the RI source table (Table 4.2-2). This process recognizes that it is EPA and DEQ that are collecting additional information and are thus able to make judgments on such matters as the completeness or reliability of information from non-LWG parties. LWG members also provided updates for their sites.[3] The LWG did retain the distinction between current and historical status for the overwater release pathway as above, which differs from the status in the Milestone report for some sites.

In Table 4.2-2, a contaminant is listed as a pathway COI if it was detected in sampled media, identified as having been released to site media, identified as a site COI, or documented to have been released directly to the river from site operations. The LWG

[3] The information in Table 4.2-2 of the RI is a compilation of public information available from site owners and operators and from DEQ, and is based upon information provided through September 2010, the date of DEQ's most recent Source Control Milestone Report. In some instances DEQ provided specific language to the LWG. The LWG has not independently verified all information provided by DEQ, and to the extent parties and DEQ disagree with language in this table, those differences will be worked out as part of the DEQ Source Control process.

has not separately screened the results against DEQ's JSCS values or any other screening criteria. Note that the LWG and non-LWG stormwater sampling data (as described in Section 4.4.1.2) were not reviewed or screened for the purposes of this table.

COIs for a pathway in Table 4.2-2 were assigned one of four categories (a–d), as defined below, for **both** historical (H) and current (C) impacts:

Category a. **Documented evidence of a complete transport pathway**—Data demonstrate that the pathway is complete; DEQ, the responsible party, or both concur that the pathway is complete.

Category b. **Likely a complete pathway**—Data suggest that the pathway is complete, but in the absence of confirming data (e.g., investigations are incomplete, nearshore wells are not yet installed, overwater operations are present and active). DEQ, the responsible party, or both have not concurred that pathway is complete. Although DEQ and responsible party evaluations are considered, LWG's analysis may support a different conclusion.

Category c. **Insufficient data to make determination**—Either a release has been documented but there has been no sampling of the potentially affected media, or a release has been documented but transport pathways have not been investigated, or no sampling has been conducted at the site or for a given pathway. Although DEQ and responsible party evaluations are considered, the LWG may have, for the purposes of the CSM, assumed that the pathway is complete.

Category d. **Not a complete pathway**—Information indicates with reasonable certainty one of the following:

- The relevant media for a given pathway likely are not affected by site-related COIs (e.g., site-related COIs are not detected in groundwater)
- A current or historical complete pathway as defined above likely is not present (e.g., riverbank is not present at a site away from the river, COIs were not detected in downgradient groundwater)
- The pathway is determined by DEQ to be insignificant in the Table 1 of the Milestone Report (DEQ 2010c).

The overall importance and relative contribution of the pathway is not evaluated in Table 4.2-2. DEQ's Milestone Reports (see Section 4.6) rank sites and pathways in terms of priority for investigation and cleanup, but the ranking is not chemical-specific.

For each potential migration pathway that is known or likely to be complete (categories a and b), Table 4.2-2 also shows whether the site's impact is current (C) or historical

(H). The overwater pathway is designated H-a or C-a when a release has been documented in the DEQ Emergency Response Information System (ERIS) database, U.S. Coast Guard (USCG) records, the Port's 104(e) responses, or other similar documentation. If no spills have been reported for a facility that had or has active overwater operations, the pathway was modified to H-b or C-b.

For the groundwater pathway, Table 4.2-2 includes a column for the presence of NAPL. A "yes" is shown where the pathway is known or likely to be complete. A "no" is shown where the pathway is known to be incomplete. A question mark is shown if the presence or absence of NAPL cannot be evaluated because of insufficient data.

4.2.3 DEQ's Milestone Report

To help readers track the assessments tabulated in Table 4.2-2, Table 1 from DEQ's (2010c) Milestone Report is reproduced here as Appendix B. The table, which was considered in the development of Table 4.2-2, provides information on the status of DEQ's source control evaluations, decisions, and measures for ECSI sites within the original Study Area. The DEQ table does not list the new ECSI sites in the expanded Study Area (RM 11–11.8) or recently identified sites within the shared stormwater conveyance basins.

An important difference between DEQ's and LWG's evaluation of sources is that DEQ focuses on current and potential sources of pollution to the river, whereas LWG also considers historical inputs when information is readily available. As a result, DEQ may identify a source as "insignificant" based on the current condition, while LWG may characterize the same source as a known or potentially complete pathway because of historical conditions. Additionally, DEQ prioritizes pathways (high, medium, low) but does not identify COIs. The LWG identifies COIs for each pathway, but does not prioritize. LWG's evaluation may also differ where DEQ identifies a source as insignificant but there are no data (e.g., no groundwater sampling) supporting DEQ's conclusion.

4.3 HISTORICAL SOURCES WITHIN THE STUDY AREA

Historical releases from upland or overwater activities likely contributed to the majority of the observed chemical distribution in sediments within the Study Area. Table 4.2-2 provides an assessment for the upland ECSI sites of whether the predominant impact for each of the pathways was historical or current. All the pathways have a historical component and many can be attributed entirely to historical operations or releases (e.g., historic discharge of waste to Doane Lake and historical MGP effluent ponds). This section discusses by pathway the major historical operations that contributed to in-river contamination within the Study Area. Note that in this context, the term "pathway" refers only to the physical transport of a contaminant of interest to the Study Area. It does not include identification of exposure points, receptors, or exposure routes. Please note that DEQ's upland source control work is focused on current sources of contamination to the river.

4.3.1 Direct Discharge – Industrial Wastewater, Stormwater, and CSOs

This section provides a general overview of historical water quality issues related to direct discharges to the Willamette River, both within Portland Harbor and upstream. Direct discharges and associated COIs specific to Portland Harbor, including industrial wastewater, stormwater and CSOs/SSOs, are described in separate subsections below, in addition to the regulatory history of these discharges.

Based on what is known about the historical operations of many sites along the Willamette River, the historical direct discharge of waste materials to the Willamette River is perhaps the primary source of the observed sediment contamination. In the early 1900s, rivers in the United States were generally used as open sewers, which was also true for the Willamette (Carter 2006). The growing city's untreated sewage, as well as process water from a variety of industries, including slaughterhouses, chemical plants, electroplaters, paper mills, and food processors, was discharged directly into the river. Stormwater runoff carried pollutants from nonpoint sources, including agricultural fields, outdoor industrial activities, oil spills, and rubber and oils from parking lots. By the 1930s, the water pollution was so severe that workers refused to work on riverside construction projects because of the foul odors and the risks to their health. Loggers even went on strike because they did not want to handle the scum accumulated on logs (Blalock 2008). Potential pollutants associated with these activities likely included VOCs, SVOCs, PAHs, petroleum hydrocarbons, metals, pesticides, and herbicides.

From 1926 to 1929, the U.S. Public Health Service collected samples from seven mid-river locations on the Willamette River. Although the studies focused on bacteria and oxygen levels from domestic waste, the sewage discharges also included municipal, industrial, and commercial waste products and stormwater runoff. Conclusions from the study indicated that although oxygen levels were sufficient to support fish life in all but two months of low water in late summer, the overall water quality probably was "not sufficiently pure" to justify recreational uses at any time (Laurgaard 1929, pers. comm.).

As noted in Section 3.2.4.2.2, the interceptors and associated facilities installed in the 1950s reduced the volume of untreated sewage discharging to the Willamette from the City of Portland. OSSA (1955) concluded that in spite of the fact that considerably less raw sewage was being discharged in to the river by the City of Portland [than the year before], the degree of pollution in the harbor was "approximately the same" in 1954 in terms of dissolved oxygen (DO) and BOD levels. The OSSA noted, however, that based on weekly measurements from seven stations between Willamette Falls and S.P. and S. Railroad Bridge,[4] the BOD levels at Willamette Falls were the highest in the

[4] The reference to the S.P. and S. Railroad Bridge appears to be BNSF Railroad Bridge located at RM 6.9. The 1957 OSSA report says (Page 6), "Stream samples were collected once each week … located above the Willamette Falls and extending 20 miles downstream to the S.P and S. Railroad Bridge." Further, OSSA (1957) references the S.P. and S. Railroad Bridge at river mile 7.0 (Page 2).

OSSA study and that the DO at the S.P. and S. Railroad Bridge were the lowest. This OSSA observation indicates that there were sources with high BOD upstream of Willamette Falls. A special survey of Portland Harbor disclosed that significant quantities of untreated domestic sewage and industrial wastes were being discharged into the river from at least 20 separate outfall sewers and estimated that oxygen demand from industrial sources was three times greater than that from domestic sewage (OSSA 1957). A 1958 OSSA report estimated that "89 percent of the total oxygen depleting pollution load, prior to treatment, is from industrial sources and only 11 percent from domestic sewage" in the Willamette basin (OSSA 1958). A 1967 federal report on Willamette River water quality stated that low DO was due in large measure to the discharge of untreated wastes of pulp and paper mills upstream of the City of Portland (DOI 1967). Another federal report summarized the water quality problems in the rivers of the Northwest (Figure 4.3-1) and identified the major sources of pollution as reservoir management procedures, agriculture, and factory wastes (DOI 1968). Stevens & Thompson (1964) estimated that the maximum dry-weather flow of 6.3 million gallons per day was discharging through eight City outfalls and 12 private outfalls. This could have been a combination of sewage, industrial waste, cooling water waste, stream flow, and groundwater.

Valuable insight into the magnitude of historical releases is provided by Glen D. Carter, an aquatic biologist employed between 1956 and 1988 by the OSSA, a forerunner to Oregon's DEQ. By the time he was hired in 1956, "fish kills were common in the river, massive rafts of decaying algae floated downstream, and a thick layer of bacterial slime covered much of the river bottom and shoreline. Rotting vegetation, bacterial slime, and countless dead fish produced highly unpleasant sights and odors. Large deposits of sewage sludge accumulated around sewage outfalls" (Carter 2006). In water quality tests performed during this period, fish often suffocated within minutes after being exposed to the water (Carter 2006).

In summary, most historical discharges to the river included a mix of industrial wastewater, domestic sewage, and stormwater.

4.3.1.1 Industrial Wastewater

Historical industries discharged a variety of contaminants and COIs based on site-specific land use. Land use activities are described in Section 3.2.1. Activities and discharges specific to historical industrial wastewater are described below.

The outfalls and drainage basins for the WWII shipyards in the area (e.g., the Oregon Shipbuilding Corporation's shipyards) were most likely separate stormwater drainage and sanitary sewer systems consisting of multiple outfalls that discharged directly to the Willamette (Bridgewater 2000). Swan Island also had combined and separated storm and sanitary systems that discharged directly to the river in 1957 (Port of Portland 2006). Other industries that lined the banks of the river most likely had direct sanitary and industrial discharges as well. Potential contaminants found in stormwater, industrial wastewater, sanitary sewer overflows, and overland sheet runoff were likely

associated with sandblasting, metal plating and surface finishing, painting, fiberglass construction, leaking hydraulic or pump equipment, transformers, paints, sealants, dust suppression activities, and machining and metal working activities at the shipyards. These contaminants likely included VOCs, SVOCs, PCBs, PAHs, petroleum hydrocarbons, metals, cyanide, and butyltins (EPA 1997b).

Also, in the early decades of the last century, it was routine practice for chemical plants to dump waste tars and sludges along or directly into the river. Chemical plants, petroleum terminals, lumber and steel mills, and various other industrial operations within the harbor also directed untreated industrial wastewaters to the river. Pollutants potentially associated with these plants and other industries included herbicides, pesticides, PCDD/Fs, mercury and other metals, VOCs, SVOCs, PAHs, and PCBs.

Some of the releases resulted from combined stormwater/industrial wastewater discharges. Examples of industries with historically complete pathways in the harbor include the former M&B site, where wastewater and non-contact cooling water were discharged directly into the Willamette River between 1945 and 1969 (PTI 1992). The MGP effluent ponds along the Gasco and Siltronic riverbank had drainage or overflow features leading to the river, which may have resulted in the former tar body in the river (HAI 2005a,b). At the former Rhone Poulenc facility, treated and untreated stormwater/wastewater and waste materials were historically discharged to Doane Lake, where they commingled with stormwater and releases from Gould/NL Industries, Schnitzer/Air Liquide, and ESCO (AMEC 2002). Historical aerial photographs indicate that the former Doane Lake periodically discharged to the LWR through a historical drainage ditch. Historically, surface water collecting in a topographically low area of the Linnton Oil Fire Training Grounds may have periodically discharged to the Willamette River via a series of drainage features.

The OSSA surveyed industrial discharges from private outfalls to Portland Harbor for three years and estimated dry weather flow (i.e., non-stormwater) of 9.9 million gallons per year (mgy) in 1963, 8.92 mgy in 1964, and 0.5 mgy in 1965[5] (OSSA 1966). As discussed in Section 3, once the interceptors were installed in the municipal system, many industries routed their industrial wastewater to the sanitary system. A survey of industrial users and wastewater characteristics was conducted in 1974 (City of Portland 1974). Fifty-seven industries were identified in Portland Harbor; 33 of these discharged to the municipal sanitary system, 17 discharged to a CSO system, and 7 did not connect to a municipal system (Table 4.3-1). Of the 7 industries not connected to a municipal system, 5 discharged directly to the river (some with pretreatment) and 2 did not.

4.3.1.2 Stormwater

As described above, many of the historical direct discharges were combined flows of stormwater, industrial wastewater, and sanitary wastewater. Stormwater has historically and currently run off to the river through outfalls and as sheet flow. Most of the flows

[5] Survey did not include outfall sewers in the Guild's Lake-Linnton area in 1965.

from shoreline properties were discharged through non-municipal outfalls or sheet flow, while flows from non-shoreline properties discharged through municipal, private, or other public agency outfalls.

As discussed in Section 6, based on LWG studies, it is clear such stormwater picks up COIs as it flows across industrial and commercial properties with outdoor process activities, across transportation corridors and residential neighborhoods that have vehicular traffic and parking, and even across open spaces that are subject to atmospheric deposition.

In summary, historically, a large majority of the land draining to Portland Harbor was undeveloped and no COIs are associated with this land use except chemicals associated with air deposition, although concentrations and loading are not expected to be significant (see Section 6). Of the developed land, the industrial land use was dominant. COIs associated with industrial wastewater (described above) could also be found in runoff from these sites, as stormwater comes into contact with industrial operations. As shown on Maps 3.2-14 through 3.2-19, residential, commercial, and major transportation land uses historically comprised a smaller percentage of the overall drainage.

Twenty-five sites have been identified as having a known complete historic migration pathway (Table 4.2-2). Another 38 sites have been identified as having likely complete historical pathways for stormwater but lack confirmatory data. Historical stormwater information does not exist for most of the historically and currently present sites discharging to the Portland Harbor.

4.3.1.3 CSOs/SSOs

As discussed in Chapter 3, prior to the construction of the interceptor system beginning in the early 1950s, 19 municipal and numerous non-municipal sewers discharged sewage (including industrial waste) and stormwater directly to the Willamette River within the Study Area (City of Portland 1936). Once interceptors and pump stations were installed, all the municipal outfalls were converted to storm-only outfalls or to CSO outfalls, which discharged less frequently to the river. Many of the industrial areas draining to the municipal system were separated when interceptors were constructed so industrial waste that had previously discharged to the municipal system was connected directly to the interceptors (i.e., could not overflow to the river). In some cases, dry weather flow (industrial or sanitary waste) continued to discharge to the river from municipal outfalls after the interceptors were constructed until the City implemented additional programs to assure that diversion structure performance was maximized, properties had appropriately connected their pipes to the municipal system, and connections were rerouted (see Section 3.2.4.2.3).

Historically, direct measurement of contaminants in CSO discharges focused on DO, TSS, bacteria, and BOD. Assumptions about COIs associated with historical discharges from the municipal sewer system (including direct discharges prior to construction of

the interceptor system, wet-weather CSO events after construction of the interceptor system, and dry weather overflows through the CSO outfalls) can be made based on the types of industries and activities (e.g., transportation corridors, parking) that discharged to the system and whether those industries and activities discharged to the combined system at a location that could overflow a diversion structure, as well as from pretreatment records.

As summarized in Section 3.2.4.2, most of the municipal CSO outfalls in the Linnton, St. Johns, and Albina (across from downtown) areas served primarily residential customers, with some commercial land use; COIs in those discharges would be similar to those for stormwater and domestic and commercial sewage. The other municipal CSOs present in the harbor were located in the downtown and north downtown areas, which included industrial, commercial and residential land uses. COIs associated with industries in CSO basins are dependent on the type of industry discharging to the combined system.

Detailed information on specific industries discharging industrial wastewater to the CSO sewer system prior to the 1980s is limited. However, some historical documents provide information about the types of industries discharging to the system during this time. The City of Portland Waste Discharge Permit application to OSSA identified all industries with a significant waste load; 11 industries were located in Portland Harbor. Seven of these industries did not connect to municipal treatment facilities (i.e., not connected to interceptor) and two industries were connected to a separated sanitary system (City of Portland 1967b). Industrial wastewater from the remaining two industries (an industrial laundry and a flour mill) discharged to the municipal combined system, which could reach the river during CSO events. No toxic wastes were reported for these industries (typically during this time only pH, BOD, and suspended solids were reported).

A 1974 Survey of Industrial Users (City of Portland 1974) indicated that facilities were discharging wastewater to the CSO system. The industry types are associated with metals, transportation, laundries, food, rubber, and bag manufacturing, and a flour mill.

Since the majority of CSO discharge consists of stormwater, COIs in overflows could also be associated with contaminants exposed to stormwater. Based on the 1967 and 1974 industrial surveys (City of Portland 1967b, 1974), COIs associated with these industrial wastewaters and/or stormwater within CSO basins may have included metals (iron/steel manufacturing, electroplaters), solvents (various manufacturing industries), PAHs (combustion emissions, road tar, treated wood), and PCBs (transformers, paints, rubber, and plasticizers after 1930).

CSO discharges also include a component of domestic sewage. The constituents in domestic sewage are primarily fecal bacteria and nutrients (which can decrease dissolved oxygen). Studies of domestic sewage from other cities have also found low concentrations of contaminants including PAHs, PBDEs, phthalates, and selected VOCs

(Palmquist and Hanæus 2005; Rowsell et al. 2010; Gasperi et al. 2008; Pham and Proulx 1997; North 2004; Song et al. 2006; Rule et al. 2006; Wilke et al. 1996). Although no data are available for Portland Harbor, domestic sewage that may occasionally enter the Study Area through CSOs may contain trace amounts of contaminants from consumer products and other compounds reflecting the ubiquitous presence of some industrial chemicals at low concentrations in urban environments.

4.3.1.4 Direct Discharge Regulatory History

In the mid-20th century, cities and industries began efforts to improve the quality of wastewater discharged to the Willamette. Flood control reservoirs created by the federal government increased summer flow in the river, which contributed to the dilution of wastes. The State of Oregon and the City of Portland regulated wastewater discharges before the enactment of the Clean Water Act (CWA) in 1972, but there was minimal regulation of stormwater discharges before the passage of CWA amendments in 1987. State and municipal regulatory programs relevant to wastewater (sewage and industrial) and stormwater discharges are discussed below.

4.3.1.4.1 Sewage and Industrial Wastewater State Permits

At the state level, the Water Purification and Prevention of Pollution Law,[6] one of the first comprehensive state water pollution laws in the country,[7] was passed in 1938. The following year saw the creation of the OSSA, which began implementing wastewater treatment requirements. In 1967, the state legislature required a permit for sewage and wastewater discharges from any sewer system and imposed liability for pollution-related injury to fish and wildlife or their habitat.[8] In 1967, the OSSA issued water quality standards for the Willamette River. For the area including Portland Harbor, the water quality standard required that the daily average DO concentration could not be less than 5 mg/L. The standards included "not to exceed" concentrations for several metals and total dissolved solids, the latter of which could not exceed 100 mg/L (OSSA 1967a). Also in 1967, OSSA issued an Implementation and Enforcement Plan to detail the facilities or actions needed to achieve compliance with the standards, a time schedule for such compliance, the controls and surveillance to be used in measuring compliance, and the measures to be taken for ensuring compliance. Part of this plan included identification of major municipal and industrial waste sources; in the Portland Harbor area, this included the City of Portland and the industries identified in Table 4.3-2 (OSSA 1967a).

OSSA meeting minutes from 1967 and 1968 indicate that a number of these industries were approved for discharge permits from OSSA. The minutes also identified other industrial facilities to be permitted, including Northwest Sand & Gravel, Willamette Western-River St, Ash Grove Lime and Portland Cement, Centennial Mills, Phillips

[6] Oregon Laws 1939, c. 3.
[7] Oregon Dept. of Environmental Quality, Administrative Overview, 2 (Mar. 2003), *available at* http://arcweb.sos.state.or.us/recmgmt/sched/special/state/overview/19970007deqadov.pdf (last visited May 6, 2009).
[8] Oregon Laws 1967, c. 426.

Petroleum, Richmond Tank Car Manufacturing, Pacific Building Materials (OSSA 1967b), Oregon Steel Mills (aka Gilmore Steel) (OSSA 1968b), Shipper's Car Line (OSSA 1968a), and Oregon Steel Mills at Rivergate (OSSA 1968c). In many cases, the permits were issued on a temporary basis in order to collect additional information to develop permit conditions. When the northwest interceptor was completed in the early 1970s, many of these industries in the northwest area connected to the municipal sanitary interceptor, thus preventing these wastes from entering the river except during an SSO event.

The focus of OSSA was predominantly on pollutants for which water quality standards were established, and no information was found cataloging toxic waste materials in historical discharges. Beginning in 1973, industrial and municipal point source dischargers were required under the CWA to obtain NPDES permits for their wastewater and process water discharges. NPDES permits for wastewater and process water are administered by the DEQ and set effluent limits, monitoring requirements and other conditions on the discharges. The requirements can be individual, written for a specific facility, or general, applicable to a group of dischargers having similar characteristics.[9] In some cases, stormwater discharges were also included in these individual NPDES permits.

The City of Portland combined sewage overflows that discharge to the Willamette River are regulated under an NPDES permit for the CBWTP.

4.3.1.4.2 Industrial Pretreatment Municipal Permits

At the municipal level, the City of Portland regulated industrial wastewater discharges (which could be either continuous or batch discharges) to the City sanitary and combined systems. The City of Portland's specific authority to prohibit discharges of contaminants to the Willamette River within the City of Portland dates to at least 1942.[10] Revisions to the City Code in 1960 prohibited discharges to the public sewer of specific materials, including gasoline and other petroleum products, solvents, acids, and toxic wastes, and required pretreatment of industrial wastes prior to discharge to the public sewer.[11] Restrictions on the discharge of commercial and industrial wastes were added and preliminary treatment was required for a number of contaminants before discharging wastewater to the municipal system.[12] Between 1969 and 2006, the City Code was amended ten more times to further limit discharges of untreated wastes to the City's storm and sanitary sewer systems.

Starting in the 1960s, the City began working with industries connected to the City system to reduce discharges of wastes to the City of Portland's sanitary or combined sewer, such as requiring the installation of pretreatment systems for industrial wastes (City of Portland 1966b). Wastewater monitoring typically included pH, BOD, and

[9] http://www.epa.gov/npdes/pubs/101pape.pdf.
[10] City of Portland Charter, § 9-604(22) (1942)
[11] City of Portland Ordinance No. 111595 (1960).
[12] *Id.*

suspended solids; other monitoring parameters (heavy metals, phenols, cyanide) were added, depending on wastes identified by the industry (City of Portland 1980). Regulation was through wastewater permits or administrative enforcement actions (City of Portland 1980; SAIC 1987).

In order to identify industries that needed pretreatment controls, the City compiled a list of businesses and Standard Industrial Classifications (SIC) in Portland whose discharges could have an effect on the treatment plant or receiving waters. As of 1972, the City had identified 300 industries that had discharges potentially damaging to the treatment system (code compliance violator) and 55 high strength (BOD or TSS) dischargers. Evaluation of the discharges was prioritized to address the most critical industry groups first, such as the metal plating industry (City of Portland 1980).

Early pretreatment efforts had less impact in the Study Area, since there were few CSO areas that contained industrial properties and their discharges (see Section 3.2.4.2.3), but they likely reduced overall discharges to the Willamette River. As discussed in Sections 4.3.1.1 and 4.3.1.3, only 2 of the 11 industries identified in 1967 and 17 of the 57 industries identified in 1974 were connected to a CSO system. All identified industries discharged at or upstream of RM 9.8 (wastewater from Centennial Mills would actually overflow upstream of the Study Area). All but 1 of the industries were located on the west side of the river (see Table 4.3-1).

EPA first issued regulations for the National Pretreatment Program in 1978 and revised these regulations in 1981 (EPA 1983). The federal pretreatment program required publicly owned treatment works with greater than 5 million gallons per day design flow to establish a pretreatment program as a condition of their NPDES permits.

The City submitted a report to DEQ in 1980 outlining its pretreatment program and identifying modifications needed to comply with the new federal requirements (City of Portland 1980). The City of Portland's NPDES pretreatment program for the CBWTP was approved in March 1983 (SAIC 1987). In the early 1980s, 260 industrial users were discharging to the City's interceptor system throughout the city and from six outlying areas (SAIC 1987). Based on the 1985 Industrial User Survey, there were 62 industrial users identified in Portland Harbor; 41 of these discharged to a municipal sanitary system, 15 discharged to a municipal CSO system, and 6 did not discharge industrial waste to any City system (City of Portland 1986). The City used these industrial surveys to identify potential facilities subject to federal categorical and prohibited discharge standards or to local standards to comply with the new federal pretreatment requirements.

4.3.1.4.3 Stormwater

Stormwater discharges had very little control and/or monitoring before the passage of the CWA. Industrial and municipal stormwater discharges were specifically addressed in the amendments to the CWA of 1987, and EPA stormwater rules became effective in

1990.[13] These rules ultimately required stormwater permits for industrial dischargers, discharges from construction activities, and discharges from municipal separate storm sewer systems serving urban areas.[14] Many properties are unregulated with respect to stormwater discharges because they are not required to have an NPDES permit based on SIC code.

The NPDES permitting program was extended to require point sources of stormwater to obtain an NPDES stormwater permit. DEQ administers several types of NPDES stormwater permits in Oregon, covering municipal, industrial, and construction-related operations. Municipal entities that discharge in the Study Area are regulated by MS4 NDPES stormwater permits; industrial dischargers that discharge into the Study Area are regulated by 1200-Z NPDES stormwater or individual NPDES permits; and discharges from construction activities are regulated under 1200-C or 1200-CA NPDES stormwater permits. Municipal and industrial permittees that discharge in the Study Area are listed in Table 4.3-3 and shown on Map 4.3-1.[15]

4.3.1.4.4 NPDES Municipal Stormwater Permits

Phase I of the NPDES Stormwater Program, developed in 1990, requires permits for stormwater discharges from medium and large MS4s serving populations of 100,000 or more.[16] In accordance with the regulatory requirements,[17] an MS4 must implement stormwater management programs to reduce the discharge of pollutants "to the maximum extent practicable."

4.3.1.4.5 NPDES Industrial Stormwater General Permits

Point source stormwater discharges from certain types of businesses and industries—such as manufacturers, hazardous waste treatment facilities, and publicly owned treatment works[18]—are regulated by NPDES Industrial General Stormwater Permits, which were first issued by DEQ in 1991. Industrial activities that are subject to permitting requirements are determined by Standard Industrial Classification codes listed in the federal regulations 40 CFR 122.26(b)(14) and (15). These activities include:[19]

- Heavy manufacturing (such as paper mill, chemical plants, petroleum refineries and steel mills)

[13] 55 Federal Register 47,990 (November 16, 1990).
[14] *Id.*
[15] ECSI sites shown in Map 4.3-1 are as of June 2011. The locations shown on the map are based on the DEQ GIS data augmented by GIS data provided by the City of Portland. The locations of the sites on the two GIS data files can be different.
[16] *Id.*
[17] 40 C.F.R. § 122.26(d) (1990).
[18] For the full list, see EPA, Categories of Industrial Activity that Require Permit Coverage, http://cfpub.epa.gov/npdes/stormwater/swcats.cfm (accessed May 5, 2009).
[19] http://cfpub.epa.gov/npdes/stormwater/swcats.cfm

- Light manufacturing (such as food processing, printing and publishing, electronic manufacturing)
- Coal and mineral mining and oil and gas exploration and processing
- Hazardous waste treatment storage and disposal facilities
- Landfills, land application sites, and open dumps with industrial wastes
- Metals scrap yards, salvage yards, automobile junkyards, and battery reclaimers
- Steam electric power generating plants
- Transportation facilities that have vehicle maintenance, equipment cleaning, or airport deicing operations
- Treatment works treating domestic sewage with a design flow of 1 million gallons a day or more
- Other facilities subject to federal stormwater effluent discharge standards in the 40 CFR Parts 405-47.

However, some categories (e.g., mineral extraction industry, transportation and light industry) have special conditions or exceptions that may exclude a facility from the stormwater permitting requirements. Also, stormwater discharges associated with the wholesale, retail, commercial, or service industries are exempt.

The NPDES stormwater program requires a regulated facility to develop a stormwater pollution control plan that identifies pollutant sources and specifies best management practices (BMPs) to minimize impacts on stormwater quality.

In 1994, the City of Portland entered into a memorandum of agreement (MOA) with DEQ to administer industrial stormwater NPDES permits for discharges to the City's MS4. In 1999, the MOA was revised to cover all industrial NPDES stormwater permits in the City's urban services boundary. The City administers the general 1200-Z permits in the Portland Harbor and inspects sites for compliance; DEQ maintains responsibility for enforcing permit conditions.

4.3.1.4.6 NPDES Construction Stormwater Permits

DEQ's 1200-C and 1200-CA stormwater permits cover construction activities. NPDES 1200-C Stormwater Discharge Permits, first issued by DEQ in 1991, are required for any construction activities that disturb five or more acres of land to control erosion and reduce sedimentation in waterways. In 2002, the threshold for construction activities was lowered to include projects that disturb one or more acres of land.

While the development and implementation of stormwater regulations have resulted in significant reductions in uncontrolled releases to the river, both permitted exceedances and unpermitted releases continue to occur.

4.3.2 Overland Transport

Contaminated surface soils exposed in the upland areas can be carried directly to the river in uncontrolled runoff (e.g., non-channelized or non-piped stormwater runoff). Overland transport was likely to have been more important historically, prior to the development of extensive stormwater conveyance systems within the Study Area. However, specific historical information on overland runoff is lacking for most sites. At former shipbuilding facilities with shipways, the upland site drainage patterns were conducive to the migration of contaminants to the river through stormwater sheet runoff (EPA 1997b).

Overland transport has been identified as a complete historical pathway for only three ECSI sites within the Study Area: Gasco, Gunderson, and M&B. The historical overland transport pathway has been identified as likely complete at eight other ECSI sites, but confirmatory data are lacking. As with other historical pathways, very little information is available on the details of operations and COIs, and it is more than likely that there were many more sites contributing COIs to this pathway.

4.3.3 Groundwater

Contaminated groundwater may have entered the river historically via discharge through sediments or bank seeps, or it may have infiltrated into storm drains/pipes, ditches, or creeks that discharge to the river. Contaminant migration may have occurred as NAPLs or as chemicals dissolved in the groundwater itself. Though insufficient data are available to evaluate the historical groundwater pathway at most sites reviewed (Table 4.2-2) significant contaminant migration via the historical groundwater pathway has been identified at a small number of upland ECSI sites within the Study Area. At a limited subset of these sites, the upland groundwater may have loaded upland chemicals to the local transition zone, including sediment and pore water. Because several of these sites are considered current sources of contamination as well, they are discussed in detail in Section 4.4.3. At M&B—the historical pathway was complete, but recent groundwater source control efforts have been effective at reducing or eliminating the impacts from this pathway.

4.3.4 Riverbank Erosion

As discussed in Section 3.2, the majority of the shoreline sites in Portland Harbor have been filled to extend the land surface into the former river channel. In many areas, the fill approaches 30 ft in thickness. Materials used for fill are at the riverbank and may or not be protected from river flows and erosion (see Maps 3.2-20a–f). The most common fill materials are hydraulically placed sands and silts dredged from the river. Though there are no records of the quality of dredge material used for fill, it is likely that the fill materials included contaminated dredge spoils at some locations.

Surface soils can be eroded directly into the river (especially from unarmored or unprotected banks) by in-water forces due to fluctuations in river levels, currents, floods, boat wakes, and propeller wash from ship activities. Over the past 150 years,

the Willamette River has experienced numerous floods. Most recently during the floods of 1964 and 1996, the river fully occupied its historical floodplain in the lower, narrower portion of the river and much of the mid-river portion as well.

In some locations, low-lying contaminated riverbank soils can be prone to erosion, and potentially contribute to sediment contamination in the river. These low-lying bank areas are particularly prone to erosion during periodic flooding events. The occurrence and relative importance of riverbank contamination is not well characterized for all parts of the Study Area, but is a focus of DEQ's Joint Source Control investigations.

Because of the limited historical data, riverbank erosion has been identified on Table 4.2-2 as a "known" historical pathway for six ECSI properties within the Study Area: Arkema; Gasco; M&B; Port of Portland Terminal 4 Slip 1 and 3; and EOSM.[20] This identification is based upon the detection of elevated concentrations of COIs in riverbank soils. Ten additional ECSI sites are likely complete historical pathways for riverbank erosion but lack confirmatory data, and 38 sites lack enough information as to determine the completeness of the pathway.

4.3.5 Atmospheric Deposition

Air pollution comes from both natural and manmade sources and can be in the form of either gasses or particulates. Historical air pollution sources were much greater prior to the creation of the Oregon Air Pollution Control Agency in 1952 and the Clean Air Act of 1970, which required air pollution controls. A partial list of principal historical anthropogenic pollution sources in Portland Harbor include chemical plants, manufactured gas plants, petroleum and natural gas storage, flour mills, asphalt plants, incinerators, metal smelters, boiler furnaces, ship repair/refurbishing, recycling activities, and mobile sources such as motor vehicles, marine vessels, locomotives, and aircraft.

Regional sources included automotive emissions, lead smelters, pesticide application, combustion sources, volcanoes, and energy generation. Chemicals commonly acknowledged to play an atmospheric source role in urban river settings within the broader geographic region of the Pacific Northwest include PCBs, PCDD/Fs, PAHs, and mercury. For example, extensive examination of the role of atmospheric deposition of such chemicals has been performed for the Columbia River Basin (EPA 2009a). From the study, it has been found that:

- Atmospheric deposition from sources inside and outside the region is thought to be a major pathway for mercury;
- Incineration and atmospheric deposition bring PCBs from distant sources which are then contributed to the basin.

[20] See also Map 4.6-1a.

Global atmospheric transport and subsequent deposition has also been documented as a significant transport mechanism for PCDD/Fs (Commoner et al. 2000; Augusto et al. 2004).

Table 4.2-2 does not address atmospheric deposition. Information on the importance of this pathway is provided in Sections 6 and 10.

4.3.6 Overwater Releases

Historically, overwater releases were common occurrences for industries on the banks of the Willamette that relied on maritime shipping to get commodities to and from market. Overwater releases are important contributors to in-water contamination at sites that have long histories of overwater operations and product transfers. Historical overwater activities were discussed by river mile in Section 3.2.2.1.

Table 4.2-3 lists documented overwater spills for the ECSI sites within the Study Area based on information from DEQ, the USCG, the Port of Portland, and the National Response Center's (NRC) centralized federal database of oil and chemical spills. Records for 1995 to 2006 were available from DEQ, detailed reports of spills from 1990 to present were available from federal sources, and summary information for spills between 1982 and 1989 was obtained from the NRC online database.[21] Releases that did not meet reporting requirements in effect at the time of occurrence may not be included in these databases. Information on spill locations, particularly in the earliest reports, is often very general (e.g., only the river mile is provided). Spill information is also provided in the site summaries. Table 4.3-4 provides information on additional spills in the Study Area, primarily from vessels, that are not associated with known ECSI sites. Information on these spills was obtained from the NRC incident database, Oregon State Fire Marshal database, USCG pollution reports, and from Appendix F of the Portland Shipyard Supplemental Preliminary Assessment (Ash Creek and Newfields 2006).

As noted above, records of overwater releases are generally not available prior to the 1980s. It is expected that very little was done to prevent overwater spills during the ship building activities or the dismantling activities that occurred after WWII. Historical, non-documented spills were likely to have been associated with refueling, load/offloading of hazardous materials, and overwater ship maintenance. It was common practice before controls were put in place for ship repairers and repair facilities to allow sand blast grit to go directly from docks and dry dock to the river by paint scraping and abrasives blasting directly overwater and by lowering the dry dock and allowing any materials on the surface of the dry dock to wash into the Willamette River.

The overwater release pathway is complete historically for approximately 29 ECSI facilities and is a likely complete pathway at 14 ECSI facilities within the Study Area. Dates of documented overwater releases are listed in Tables 4.2-3 and 4.3-4 and, as

[21] National Response Center Database, http://www.nrc.uscg.mil/

discussed in Section 4.2, any spills that occurred prior to January 1, 2004, are considered historical. (Spills that occurred after January 1, 2004, are considered current overwater releases and are discussed in Section 4.4.6.) Of these facilities, some of the largest spills have occurred at bulk fuel facilities (e.g., ARCO, Kinder Morgan Linnton, Willbridge Terminal), commodity shipping facilities (Goldendale Aluminum), and ship repair facilities (Schnitzer Steel, Cascade General). Types of spills include diesel, Bunker C fuel, asphalt, lube oil, hydraulic fluid, crude oil, sandblast grit, ballast/bilge water, waste oil, and generator fuel. Spills not related to ECSI sites and primarily related to vessels are shown in Table 4.3-4. Nearly 1,000 spills have been identified in Tables 4.2-3 and 4.3-4, with dates ranging back to 1914, with most spills reported since 1980. Examples of the larger documented spills include a 50,000-gallon release of heavy oil from a U.S. Navy hull scrapped by Zidell Explorations Inc. in the Linnton area in 1973; 25,000 gallons of diesel released to the river near ARCO in 1997 and another 15,000 gallons in 2005; 8,000 to 11,000 gallons of Bunker C released near the Goldendale site in 1991 by Taiwan Navigation; 3,297 gallons of aviation fuel released near the Shell dock in 1979; 2,500 gallons of gasoline released near the Willbridge Terminal in 1971; and a 1980 release of 225 gallons of pesticides released to a storm drain by the Pax Company of Utah.

4.4 CURRENT SOURCES WITHIN THE STUDY AREA

Current sources of in-water contamination within the Study Area are discussed in this section. Many current sources are related to historical activities, although the historical activity may no longer take place. Identifying current sources is critical to understanding remedy effectiveness and recontamination potential for the FS and subsequent cleanup. Information presented in the following subsections varies in detail because of differences in the level of understanding and quantitative investigation of the various pathways associated with the upland sites. The direct discharge pathway in Section 4.4.1 and groundwater pathway information in Section 4.4.3 represent the latest refinement of analyses that have been underway for several years. Information on overland runoff, riverbank erosion, atmospheric deposition, and overwater releases is limited, and these potential sources are described in general terms.

4.4.1 Direct Discharge – Industrial Wastewater, Stormwater, and CSOs,

Pollutants from commercial, industrial, private, or municipal outfalls are being discharged directly to the Study Area. Some discharges are permitted under the CWA NPDES program, while some non-municipal outfalls and pipes are not permitted or the status is unknown. Discharges from outfalls include industrial wastes, stormwater runoff, and CSOs. The City has tracked SSOs (emergency overflows from sewage pump stations) since 1996: there are no records of SSO events in Portland Harbor. Permitted discharges include treated industrial wastes, stormwater runoff, treated groundwater from remediation sites, CSOs, and SSOs.

4.4.1.1 Industrial Wastewater

Additional information on wastewater discharges, including a list of the hazardous substances being discharged to the river, the concentrations and loads per day, information on mixing zones, and the COIs that may pose risk to the river, is included in Section 6.

4.4.1.2 Stormwater

The following sections provide a brief description of the stormwater basins, the types of stormwater discharges, potential sources, currently available data, and a review of the current stormwater and wastewater permits within the Study Area.

Maps 4.4-1a–d present the following information:

- Stormwater and CSO and SSO outfalls
- Stormwater piping
- Streams discharging to the Study Area.

The maps also contain a characterization of the Study Area showing areas:

- With shared conveyances (e.g., City and Schnitzer-International Slip outfalls)
- With direct discharge (either through outfalls or sheet flow)
- Known to have no stormwater discharge, such as a site where there is specific information that the site/area only has infiltration and no ability to discharge stormwater (e.g., PGE-Harborton, which has a berm around it so no stormwater runoff occurs)
- With uncertain drainage.

4.4.1.2.1 Summary of Stormwater at ECSI Sites

DEQ began in approximately 2004 to include stormwater evaluations as part of source control evaluations under the DEQ/EPA JSCS program for Portland Harbor. Of the 114 facilities in the Study Area with permitted stormwater discharges (see Section 4.4.1.4), most have not yet conducted a stormwater source control evaluation or are recently in the process of conducting one. Of those ECSI sites for which stormwater source control evaluations have been completed, stormwater discharge has been determined to be a complete and current pathway at nine sites and a likely complete pathway at 24 sites. For a site to have a complete or likely complete stormwater pathway, COIs have been identified in site-reported stormwater data. No screening of stormwater COIs has been performed by LWG. However, as noted in Section 4.4.1.2.3, JSCS screening values for stormwater were exceeded in every land use sampled for at least some chemicals based on the LWG sampling program discussed below.

4.4.1.2.2 Potential Sources to Shared Conveyances Draining Stormwater from Multiple Properties

Just under half of the stormwater drainage to the Study Area is through shared conveyance systems, although open space is about 60 percent of those basins. The majority of these outfalls are not monitored nor were they sampled for the RI/FS. To qualitatively evaluate potential COIs from these systems, the LWG evaluated COIs from ECSI sites within the basins, public records, and, where available, from sampling data. Table 4.4-1 identifies 39 City outfalls, 8 outfalls for Burgard Industrial Park, 4 owned by ODOT, and 15 unknown multiparty outfalls, including multiple properties that discharge to Saltzman Creek. In addition, information on the outfall structure is included (e.g., location, affiliated organization, outfall size, outfall material, outfall status, basin area). Analytes shown in bold font are permitted under the NPDES 1200Z permit (City of Portland 2010). Note that inactive outfalls are included in the table for completeness, but are not considered to be current sources. ECSI sites within each basin and sites immediately upstream of the outfall on the main stem of the river are identified in the table.

For each of these sites, COIs were determined either through review of site summaries, public records, or DEQ's ECSI database. Sites for which the stormwater pathway has been independently investigated (e.g., DEQ's site discovery process or JSCS pathway evaluations are being conducted) are identified in Table 4.4-1. Sites that have not had stormwater pathway evaluations (which typically include sites where cleanup occurred before the JSCS, inactive sites, and sites in former CSO basins) are shown as Other Potential Sources in Table 4.4-1. There was insufficient information on these sites to include them in Table 4.2-2, but they are included in Table 4.4-1 to provide a list of potential historical sources. The non-municipal shared conveyance systems draining to the Study Area basin areas for these systems are typically not defined so potential sources in these basins are unknown. As described in Section 4.4, Table 4.4-1 provides a compilation of known and potential sources but is not an exhaustive list of current or historical sources of contamination. Identification and evaluation of potential sources is still ongoing.

COIs were identified through investigations at or adjacent to sites draining to 21 outfalls and included PCBs, TPH, metals, VOCs, PAHs, phthalates, and DDx at one or more outfalls (see Table 4.4-1; DEQ 2009a; Anchor 2006a, 2008a; Anchor QEA 2009; GeoDesign 2008, pers. comm.; MWH 2009; City of Portland 2006a, 2009a; PES 2008; SES 2008; Evren Northwest 2007; CH2M Hill 2008; Consolidated Metco 2008, pers. comm.).

4.4.1.2.3 Summary of Stormwater Sampling

Stormwater sampling data are presented below from two sources. The LWG sampling program data are used in Section 6 to generate estimated stormwater loads to the Study Area for the purposes of fate and transport modeling and recontamination analysis. The non-LWG stormwater data were provided by DEQ in early 2008 for sites collecting data under the JSCS program and are presented for reference purposes in this section

but will not be used in estimating stormwater loads, as directed by EPA. Stormwater sample locations and analyses are summarized in Section 2.2.6, with tabular detail for LWG and non-LWG collected stormwater data in Appendix C1.

LWG Sampling Program

In November 2006, EPA and LWG determined that stormwater data were needed to complete the RI/FS, and that such data would need to be collected in the 2006–2007 wet-weather season to fit within the overall RI/FS project schedule. They convened a Stormwater Technical Team, which included representatives from EPA, DEQ, and LWG, to develop the framework for a sampling plan. The sampling framework described in the FSP was developed by the Stormwater Technical Team and is based on an EPA memorandum dated December 13, 2006 (Koch et al. 2006). This framework was discussed and approved by Portland Harbor managers from EPA, DEQ, the Tribes, and LWG on December 20, 2006.

The Stormwater Technical Team evaluated a range of stormwater data collection technical approaches and selected those that are described in the framework and elaborated on in the Stormwater Sampling Rationale. Selection was based on 1) the ability to meet the objectives for data as agreed to by the Portland Harbor managers; and 2) practicability in terms of schedule, cost, and feasibility.

As also discussed in Section 6.1.2, representative samples from five general categories of land use (heavy industrial, light industrial, residential, major transportation, and parks/open space), as well as samples from non-representative locations, were included to obtain a practical and sufficient data set to estimate stormwater loading to the Study Area. Stormwater runoff and stormwater quality are affected by land use and the associated activities that occur within each drainage basin. Estimates of the predominant land use classifications for the overall Study Area by drainage basin, per the City of Portland's GRID model (City of Portland 2008, pers. comm.; see RI Report Appendix E, Attachment E-2) are as follows:

- Parks and Open Space/Vacant—approximately 57 percent of total site drainage
- Light Industrial—approximately 8 percent
- Heavy Industrial—approximately 25 percent
- Residential/Commercial—approximately 8 percent
- Major Transportation—approximately 2 percent.

Maps 4.4-2a–d indicate the distribution of land uses through the Study Area. Further discussion on how various City zoning classifications were grouped into land uses is included in Section 6.1.2. Generally, areas adjacent to the river are dominated by industrial land uses. The largest combined areas of Heavy Industrial land use are on the east bank from RM 1 to 5 and on the west bank from RM 7 to 10. From RM 8 to 10 on the east bank of the river is the largest area of Light Industrial land use. Extensive areas

of Parks and Open Space land use occur slightly away from the west bank from approximately RM 1 to 10. Similarly, much of the area east and away from the river from RM 5 to 12 is Residential/Commercial land use. Although Major Transportation thoroughfares extend throughout the Study Area, the largest areas tend to be at the upper reaches of the Study Area.

Stormwater composite water and sediment samples were collected from a subset of drainage basins/outfalls within each land use category in the Study Area. These locations were sampled by LWG during two sampling efforts in the spring/summer of 2007 (Round 3A) and the fall/winter of 2007–2008 (Round 3B), Port of Portland (Terminal 4 composite water and sediment trap samples at outfalls 52C and 53), and City of Portland (OF-53 composite water samples). One additional site (GE Decommissioning) was sampled by GE during the same time frame. Results from the GE investigation will also be used in the overall LWG stormwater data set. The stormwater composite water and sediment trap data were collected in accordance with the Round 3A Stormwater FSP and Addendum (Anchor and Integral 2007a,b) and its companion document, the Round 3A Stormwater Sampling Rationale (Anchor and Integral 2007c), and analyzed in accordance with the QAPP Addendum 8 (Integral 2007b).

Data were collected during a total of 15 storm events, with each outfall sampled an average of three times. Flow-weighted composite samples from each location were collected over the course of a storm event using ISCO automatic samplers to obtain event mean concentrations of chemicals. The objective was to get a composite sample that represents the water quality over the entire storm hydrograph. Sediment traps were left in place for 3 to 7 months during two separate sampling periods. These two measurements were collected to provide data to support two independent means of estimating stormwater chemical loads as explained further in Appendix E. Due to the limited time span of sample collection and the known variability of stormwater, these data should be considered to represent a “snap shot” of stormwater entering the Study Area during the sampling period.

Table 4.4-2 provides summary statistics for nature and extent indicator chemicals for stormwater collected by the LWG. Appendix C1, Table C1-1, provides summary statistics for composite water and sediment traps for all stormwater chemicals analyzed during LWG stormwater investigations. Summary statistics for the LWG data include all LWG data, plus Terminal 4 catch basin and stormwater data (including City outfalls that are not part of Terminal 4), and GE Decommissioning stormwater data used for the stormwater loading analysis provided in Section 6. All data are Category 1. For purposes of showing summary statistics, representative and non-representative sampling locations were combined together. An analysis of representative land use locations versus non-representative locations was conducted as part of the estimation of actual stormwater loads (applying estimated concentration to estimated runoff volumes) to the Study Area and is detailed in Section 6.1.2.

Concentrations of indicator chemicals, such as total PCBs, total PAHs, DDx pesticides, non-DDx pesticides, bis(2-ethylhexyl)phthalate (BEHP), hexachlorobenzene, and metals, in the LWG stormwater sampling results were compared by land use. For the vast majority of indicator chemicals, including composite water and sediment data collected for total PCBs, total PAHs, DDx and non-DDx pesticides, and metals, samples taken from Heavy Industrial land use locations had the greatest concentrations. Exceptions include isolated metals (i.e., lead) in Light Industrial sediment trap data. Analyte concentrations collected from Open Space and Residential land uses were generally lower than other land uses. JSCS values for stormwater were exceeded in every land use sampled for at least some chemicals. The analysis of this data in terms of projected loads (which takes into account acreage of the various land use types) is contained in Section 6.

Non-LWG Sampling Program

In addition to the LWG stormwater data, at LWG's request, DEQ provided stormwater data in early 2008 for sites that had thus far collected data under the JSCS program. Table 4.4-3 provides a summary of the locations, sampling dates, data quality, and parameters analyzed. Table 4.4-4 provides summary statistics for nature and extent indicator chemicals for stormwater collected by non-LWG parties. Appendix C1, Table C1-2 provides summary statistics for all stormwater chemicals collected during non-LWG stormwater investigations. Summary statistics for the non-LWG stormwater data are limited to data collected after January 1, 2004, and before early 2008, and are Category 1 data.

Note that the methods and procedures used to collect non-LWG samples vary from the LWG stormwater sampling program and thus concentrations would be expected to vary. For example, some of the water samples contained in the non-LWG data set are grab samples collected during one part of a storm event as opposed to flow-weighted composite samples collected by the LWG sampling program.

In addition, Table 4.2-2 summarizes ECSI sites being investigated by DEQ through its site cleanup program and Table 4.4-3 provides specific information regarding the characterization of stormwater in Portland Harbor; not all properties listed on Table 4.4–3 are listed on Table 4.2-2.

A comparison of the mean of the detected and non-detected non-LWG stormwater data and the LWG stormwater data indicated that analyte concentrations for non-LWG data were generally in the range of LWG data except for the instances mentioned below. For water data, non-LWG data were generally greater than LWG data for metals and non-DDx pesticides. For sediment trap data, non-LWG data were generally greater than LWG data for non-DDx pesticides and BEHP.

4.4.1.3 CSOs

In 1990 the City of Portland modeled approximate annual volumes for historical combined sewer overflows in preparation for development of a facilities plan for its

Combined Sewer Overflow Plan. Estimated combined sewer overflow volumes in Portland Harbor are shown in Figure 4.4-1. Based on these modeled volumes, in 1970 approximately 1.6 billion gallons of combined stormwater and wastewater (sanitary sewage and some industrial wastewater) overflowed in the Portland Harbor Study Area. By 1990 the overflow volume had decreased to approximately 925 million gallons annually.

By 2001 the overflow volume was reduced to approximately 628 million gallons annually, as a result of the elimination of several outfalls, downspout disconnections, some sewer separation projects, and infiltration of stormwater to sumps in some areas served by combined sewers. In 2006 the West Side CSO Tunnel Project was completed and the annual CSO volume was reduced to approximately 195 million gallons. Upon completion of the East Side CSO Tunnel Project in 2011, the annual estimated CSO volume in the Portland Harbor area will be approximately 20 million gallons.

Contaminants in CSO discharges identified in a 1997 DEQ report for sampling are bacteria,[22] copper, and lead (DEQ 1997). A review of Annual Pretreatment Reports to DEQ was conducted to determine other potential COIs. Table 4.4-5 shows only those industries that discharge pretreated industrial wastewater to a portion of the combined sewer system that can overflow in the Portland Harbor Study Area. All of these permittees discharge to a CSO located at or upstream of RM 9.8, and most are located on the west side of the river, where industrial areas were not separated when the interceptors were installed.

Eleven of the 32 permitted industries have permits, but they do not discharge to the City system. Fifteen industries were required to have permits based on the potential to exceed local limits, and their industrial activities are related to food and beverages, laundries, rubber processing, bag manufacturing, photographic processing, press and printing, and transportation. COIs[23] based on the permit discharge limits for these 15 industries include metals, oil and grease, and volatile organics. Four permits are for discharge of groundwater (either from remediation sites or construction dewatering) to the combined system, and COIs are identified as oil and grease, BTEX, and metals. Three permitted discharges are for metals-related industries, and COIs are identified as metals, oil and grease, cyanide, and total toxic organics. Industrial dischargers are required to list all potential pollutants in their permits even if they do not pretreat and discharge those constituents. The City prohibits discharge of many toxic wastes to its combined and sanitary system, including PCBs and pesticides.[24]

A CSO is composed of approximately 80 percent stormwater and 20 percent sanitary and pretreated industrial wastewater and, therefore, CSO water quality can also be

[22] Bacteria is not a risk-based COI in Portland Harbor.
[23] BOD and pH listed in permits are not included as COIs because they are not risk-based COIs in Portland Harbor
[24] Portland City Code Chapter 17.34; Industrial Wastewater Discharges Administrative Rules ENB-4.03. These are current discharge prohibitions. General discharge prohibitions have been in City Code since the 1960s and the City Charter since as early as 1942.

affected by the exposure to stormwater and contaminants in domestic sewage (see Section 4.3.1.3. Table 4.4-1 (see italicized sites) shows the identified potential stormwater sources in the CSO basins and associated COIs.

4.4.1.4 Stormwater and Wastewater Discharge Permits

Many types of stormwater and industrial wastewater permits are issued within the Study Area uplands. Stormwater permits include discharges of industrial and municipal stormwater and stormwater runoff from construction activities. Wastewater includes permits for process water, oil/water separator discharge, petroleum hydrocarbon cleanup wastewater (tank cleanup and groundwater treatment), vehicle and equipment washwater, boiler blowdown, filter backwash, cooling water, heat pump wastewater, log pond drainage, noncontact geothermal exchange water, and rinsewater of various types, and CSO and SSO discharges. Permitted wastewater discharges are generally required to be treated before discharge.

As discussed in Section 3.2.4, stormwater and wastewater enter surface waters via pipes, culverts, ditches, catch basins, and other types of channels. In the Study Area, both stormwater and treated wastewater generally enter the river via constructed conveyance systems and outfalls. All wastewater discharges and stormwater discharges from certain types of facilities require a NPDES permit.

DEQ issues two types of NPDES permits: general and individual. General permits are issued to dischargers with similar operations and type of waste. Individual permits are issued to facilities whose processes or wastewater/stormwater flows merit unique monitoring requirements. There are 14 individual industrial wastewater permit holders amongst 13 facilities (Columbia River Sand and Gravel – Linnton Facility holds two permits) discharging to the Study Area. There are no municipal wastewater treatment plant discharges in the Study Area. However, the 2002 NPDES permit for the CBWTP permitted the City of Portland to discharge CSO and pump station overflows (SSOs) into the Study Area from designated outfalls. The 2002 permit is currently in effect. The Port of Portland, ODOT, Multnomah County, and the City of Portland discharge stormwater under MS4 permits, which include discharges to the Study Area.

As of February 2011, there were approximately 114 general NPDES stormwater and 14 general NPDES wastewater permitted discharges to the Study Area, as listed in Table 4.3-3. Map 4.3-1 shows permitted facilities and the type of permit. Note that multiple permits may be associated with a single outfall. The number of NPDES-permitted discharges by type of permit is shown below:

NPDES Permit Type	Number of Permits (as of 2/2011)
General Permits for cooling water/heat pumps **(GEN01)**	8
General Permits for boiler blowdown **(GEN05)**	2
General Permits for treatment of groundwater **(GEN15A)**	4

NPDES Permit Type	Number of Permits (as of 2/2011)
General Permits for stormwater **(GEN12A,C,Z)**	86
Individual Permits for facilities not elsewhere classified that dispose of primary smelting/refining of metals not elsewhere classified **(NPDES-IW-B08)**	1
Individual Permits for facilities not elsewhere classified that dispose of process wastewater (includes remediated groundwater) **(NPDES-IW-B14)**	1
Individual Permits for facilities not elsewhere classified that dispose of process wastewater **(NPDES-IW-B15)**	6
Individual Permits for facilities not elsewhere classified that dispose of non-process wastewater **(NPDES-IW-B16)**	4
Municipal Separate Storm Sewer System Discharge Permit **(NPDES-DOM-MS4-1)** including CBWTP	1
Individual Permits for facilities that dispose of sewage (50 MGD or more) **(NPDES-DOM-A1)**	1

Individual permits are specifically tailored to an individual facility's unique discharge and contain more than just unique monitoring requirements. They also contain specific limitations or conditions for that facility. Individual permit limits may be based on either effluent concentrations or total loadings, incorporating factors such as mixing zones or available technologies. Thirteen facilities within the Study Area have individual permits and are denoted with a footnote in Table 4.2-2. Discharge monitoring requirements, effluent limits, and information on mixing zones are provided for these 14 individual permits in Table 4.4-6. The vast majority of permitted discharges to the Study Area (by number of permits) are for industrial stormwater discharges under general permits (NPDES GEN12Z). Instead of flow or chemical limits, these permits specify benchmark concentrations to help permittees evaluate the effectiveness of their stormwater management practices. Table 4.4-7 lists the permit discharge requirements for each type of general permit. Monitoring parameters for NPDES GEN12Z are limited to pH, oil and grease, TSS, copper, lead, zinc, and sometimes E. coli. The monitoring data generated under these permits provide some data regarding metals and TSS but are otherwise are of limited value in identifying sources. General stormwater permits are limited to a handful of parameters, most of which are not related to Portland Harbor COCs. Individual wastewater permits are specific to the individual process at the facility. Therefore, the data collected from general stormwater and individual wastewater permit facilities are not a good gauge of Study Area contaminants.

Other tools that have been used to control active discharges include industrial process changes, pollution prevention practices, and technology-based effluent controls. These tools, in addition to the development and implementation of stormwater regulations, have resulted in significant reductions in uncontrolled releases to the river. However,

not all industrial operations and many other operations near the Study Area (wholesale, retail, commercial or service industries) are not currently regulated.

4.4.2 Overland Transport

Overland transport has been identified as a complete and current pathway at two facilities: Gunderson and Triangle Park. This pathway is likely complete at Gasco. These sites or portions of these sites lack stormwater conveyance systems, and stormwater either infiltrates the ground or discharges to the river via sheet runoff. This pathway is rarely investigated and likely occurs at other sites in the Study Area.

4.4.3 Groundwater

Based on the conceptual understanding of the regional hydrogeology (see Section 3.1), groundwater discharge to the river is expected to occur over most of the Study Area. However, this does not mean that all upland areas represent sources of contamination to the river via the groundwater pathway. Understanding the groundwater pathway as a source of contamination to the river requires an understanding of the distribution of upland plumes in relation to the river and the hydrogeologic factors affecting the migration and discharge of groundwater and groundwater contaminants to the river.

4.4.3.1 Assessment of the Groundwater Pathway during the Remedial Investigation

In cooperation with the EPA and DEQ, the LWG initiated the groundwater pathway assessment (GWPA) for the Study Area in 2003. The GWPA consisted of categorization of upland groundwater ECSI sites based on detailed file reviews and consultation with DEQ site managers; selection of a subset of high-priority sites for inclusion in GWPA field investigations; agreement by EPA to those sites for investigation; performance of these field investigations; and detailed, site-specific evaluation of the results of these investigations, using multiple lines of evidence to reach conclusions with respect to the existence of complete groundwater transport pathways to the LWR and their potential significance as a source of contamination to TZW and sediment in the LWR. The elements and primary findings of the GWPA are summarized below. Detailed documentation of the GWPA is provided in Appendix C2.

4.4.3.1.1 GWPA Upland Site Categorization and Selection

As part of GWPA planning efforts conducted from 2003 through early 2005, the LWG conducted a review of data available from DEQ ECSI files for all chemicals analyzed in groundwater for sites between RM 2 and 11, including sites bordering the river and sites with documented groundwater plumes in certain areas up to approximately 0.5 mile inland from the river (GSI 2003a). This review identified 113 sites that were initially categorized according to their likelihood to represent a potential source of COIs to Portland Harbor via the groundwater transport pathway. The initial site categorization was presented in GSI (2003a) and was updated in the draft GWPA Technical Memorandum (GSI et al. 2004). The 113 upland sites were categorized as follows:

- **Category A –** Category A included 21 sites that met the following three criteria (GSI et al. 2004):
  - An upland source of COIs is present.
  - COIs have been detected in upland groundwater.
  - A groundwater pathway from the upland site to the river is complete or is reasonably likely to be complete. This criterion is met when COIs present in upland groundwater are either confirmed or, based on professional judgment, believed to have a reasonable potential to discharge to the river (via sediments, the transition zone, surface water, or a combination thereof).

  All Category A sites were considered for inclusion in the Round 2 Groundwater Pathway Assessment. Representatives of the EPA, DEQ, and the LWG met on January 7, 2005, to review available data for each of the 21 Category A sites. This review considered several factors, including the nature and extent of upland groundwater plumes, the presence or absence of detectable levels of upland groundwater COIs in nearshore wells, the presence or absence of NAPL in nearshore wells and/or in shoreline seeps, site-specific hydrogeological conditions (including stratigraphic evidence and/or direct observations of preferential flow paths, springs, or seeps), and the potential role of infrastructure (e.g., storm drain lines and other outfalls) in creating possible preferential flow paths. Based on this review, EPA, in consultation with DEQ's upland site managers, selected 12 high-priority sites for which sufficient evidence of a complete groundwater transport pathway to the LWR existed to carry them forward into the site-specific scoping process for the Round 2 GWPA. The 12 high-priority sites are listed below:

  - EOSM
  - Time Oil Northwest Terminal
  - Schnitzer Investment Corporation (Premier Edible Oils Site)
  - Kinder Morgan Linnton Terminal
  - ARCO Terminal 22T
  - ExxonMobil Oil Terminal
  - Gasco
  - Siltronic
  - Rhone Poulenc

- Arkema
- Willbridge Terminal
- Gunderson.

EPA and DEQ determined that there was insufficient evidence to conclude that groundwater plumes associated with the remaining nine Category A sites were migrating to the LWR, and these sites were eliminated from further evaluation in the GWPA.[25]

- **Category B –** Category B included 83 sites for which the potential for groundwater COIs to reach the river could not be confirmed based on available data. Category B sites were referred to Oregon DEQ for possible further evaluation of the potential for groundwater-related COIs to discharge to the river. During GWPA planning, Oregon DEQ did not identify a need for further evaluation in the RI/FS process of potential upland groundwater plume transport to the LWR at any Category B sites. Thus, none of the Category B sites were carried forward into the GWPA.
- **Category C –** Category C included nine sites where available site-specific groundwater data indicated at the time that groundwater COIs either are not present or are not likely to reach the river. Category C sites also were not evaluated further in the GWPA.

A further, more detailed review of available data was conducted by the LWG for each of the 12 Category A sites that were carried forward into the scoping phase of the Round 2 GWPA. The following criteria were established to determine an appropriate course of action for each of the 12 high-priority Category A sites shown in Figure C1.3-1 (Appendix C2):

Criterion 1: Existing offshore groundwater sampling data indicate that a potentially complete transport pathway exists for groundwater COIs to reach the transition zone.

Criterion 2: Existing shoreline sampling data from groundwater wells or seeps indicate a reasonable likelihood of a complete transport pathway for groundwater COIs to reach in-river exposure points.

Criterion 3: Existing observations of NAPL seepage to the river indicate that a complete transport pathway may exist for groundwater COIs to reach in-river exposure points.

[25] The nine Category A sites that were not carried forward for the GWPA were McCall Oil; M&B; UPRR Albina Railroad; Port of Portland Terminal 4, Slip 3; Cascade General; Triangle Park; Foss Maritime; Mar Com; and Marine Finance.

Criterion 4: Shoreline groundwater seeps containing COIs are known to be present and represent a potentially complete exposure pathway for human receptors.

Sites that met one or more of the above inclusion criteria were included in the GWPA program. Sites for which insufficient data were available to determine if any of the inclusion criteria are met were referred to Oregon DEQ for additional uplands groundwater characterization (to be conducted by the upland party under DEQ oversight) to support a determination of whether in-water investigation of the groundwater transport pathway would be needed to support the RI/FS.

Based on the screening against these criteria, the EPA and DEQ determined nine of the 12 high-priority Category A sites should be included in the Round 2 GWPA field investigations. A summary of the evaluation of each site against the inclusion criteria is presented in Table C2.3-1 of Appendix C2, and complete summaries for the nine selected GWPA sites, including general background, hydrogeology, and the nature and extent of COIs in groundwater, are presented in Appendices A-1 through A-9 of the GWPA SAP (Integral et al. 2005).

The three sites which were not included in the GWPA were EOSM, Time Oil Northwest Terminal, and Schnitzer Investment Corporation (Premier Edible Oils site). Two of these—EOSM and Time Oil Northwest Terminal—were excluded because further evaluation of the available data suggested the groundwater pathway was incomplete. At Premier Edible Oils, the LWG, EPA, and DEQ concurred that uplands characterization had not been performed at the time of the GWPA to determine whether or not a groundwater transport pathway from the upland site to the river was complete or reasonably likely to be complete. As a result, during the scoping process for Round 3B investigations, EPA and DEQ determined that any further evaluation of groundwater transport from the Premier Edible Oils site to the LWR would be conducted separately from the RI under DEQ's JSCS program. These decisions were finalized during a July 12, 2007 meeting of the EPA, DEQ, and LWG managers on Round 3B scoping. Additionally, none of the other sites that were initially classified as Category A, but that were not included in Round 2 GWPA field investigations for the reasons discussed above, were identified by EPA and DEQ during the Round 3 scoping process as warranting inclusion in Round 3 GWPA field investigations. As with Premier Edible Oils, EPA and DEQ determined that any future evaluation of the groundwater transport pathway to the LWR at any of these nine sites would be conducted separately from the RI under the DEQ's JSCS program.

Additional details on the site selection process for the GWPA, including background information on the 12 Category A sites that were carried forward for detailed scoping as described above, are provided in Appendix C2.

4.4.3.1.2 Distribution of Upland Groundwater COIs at GWPA Study Sites

For each of the nine sites included in the GWPA field investigations, a series of figures has been prepared displaying the distribution of NAPL (if present) and COIs in upland groundwater, based on available upland data:

Site Name	Selected COIs	Figure Numbers
Kinder Morgan Linnton	NAPL, total BTEX, total PAHs, arsenic	4.4-2a–d
ARCO	NAPL, total BTEX, total PAHs, lead, arsenic	4.4-3a–e
ExxonMobil	NAPL, total BTEX, arsenic, lead, zinc	4.4-4a–e
Gasco	NAPL, total BTEX, naphthalene, total cyanide	4.4-5a–d
Siltronic	NAPL, total BTEX, trichloroethene, cis-1,2-dichloroethene, vinyl chloride	4.4-6a–e
Rhone Poulenc	NAPL, 1,2-dichlorobenzene, trichloroethene, Silvex, arsenic	4.4-7a–e
Arkema	NAPL, chlorobenzene, perchlorate, total of 4,4'-DDx, chromium	4.4-8a–f
Willbridge Terminal	NAPL, total BTEX, total PAHs, total chromium	4.4-9a–d
Gunderson	1,1,1,-Trichloroethane, 1,1-dichloroethene, total lead	4.4-10a–c

The COIs presented in these figures are not intended to be inclusive of all contaminants detected in upland groundwater at the site; rather those presented represent the occurrence, distribution, and concentrations of select COIs for a given site.

4.4.3.1.3 Overview of Round 2 GWPA Field Investigations

The Round 2 GWPA field investigation was completed at the nine selected study sites in 2006. Results of the Round 2 GWPA, which are presented in Appendix C2 of this RI Report, were used to ascertain the status of the groundwater pathway using combined lines of evidence:

- Approximate zones of groundwater discharge offshore of the sites, identified from groundwater discharge mapping (site stratigraphy, upland groundwater contours and concentrations, sediment texture, Trident temperature data, and seepage meter results) and from analytical chemistry data for TZW and sediment
- Major ion signatures for upland groundwater, surface water, and TZW, evaluated using Piper diagrams
- Chemical concentrations in zones of groundwater discharge relative to zones of low or no groundwater discharge (concentration graphs by flow zone).

In some cases, the analyses presented in Appendix C2 using data collected during the GWPA are supplemented by information collected by upland site responsible parties during site-specific investigations to further characterize the groundwater pathway to the LWR at a given site. In addition, at the request of EPA, a follow-up investigation was conducted offshore of Gunderson's Area 1 during Round 3 to evaluate possible in-river discharge of a suspected chlorinated solvent remnant plume. The stratigraphic coring indicated that a complete flow pathway was not present, and LWG and EPA agreed that additional TZW sampling offshore of Gunderson Area 1 during Round 3 was not necessary.

4.4.3.1.4 Summary of Major Findings of the GWPA

Based on consideration of the multiple lines of evidence discussed above and presented in detail in Appendix C2, the GWPA reached the following overall conclusions with respect to the potential role of the groundwater transport pathway as potential source of contamination to sediments and TZW in the LWR for the nine GWPA study sites:

- **Kinder Morgan Linnton Terminal (GATX).** The combined lines of evidence suggest some possibility that low levels of PAHs in upland groundwater may be migrating to the transition zone in the groundwater discharge zone offshore of the Kinder Morgan Linnton site. For other upland groundwater COIs at this site, however, there is no evidence of a complete and significant transport pathway to the TZW environment. (See Section C3.1.5 of Appendix C2.)
- **ARCO.** Migration of chemicals in upland groundwater to the transition zone does not appear to significantly influence TZW and sediment. (See Section C3.2.5 of Appendix C2.)
- **ExxonMobil Oil Terminal.** The findings of the Round 2 GWPA suggest that BTEX and metals in upland groundwater at the ExxonMobil site may have been transported to the TZW via groundwater flow prior to the implementation of upland groundwater source control measures. It is also plausible that the chemicals detected in TZW samples collected at this site during the RI reflect chemical partitioning from sediment to pore water rather than transport from upland groundwater. (See Section C3.3.5 of Appendix C2.)
- **Gasco.** The findings of the Round 2 GWPA and NW Natural's in-water investigation at the Gasco site indicate a complete groundwater pathway for VOCs and PAHs to the transition zone. However, the relative contribution of the groundwater transport pathway to COI concentrations observed in the TZW is uncertain due to the masking effects of existing in-water sediment chemical sources. (See Section C3.4.5 of Appendix C2.)
- **Siltronic.** The pathways for chlorinated volatile organic compounds in the offshore zone and PAHs, BTEX, and TPH in the nearshore zone are complete. (See Section C3.5.5 of Appendix C2.)

- **Rhone Poulenc.** A complete pathway for transport of two upland groundwater COIs (1,2-dichlorobenzene and Silvex) to the transition zone is present. (See Section C3.6.5 of Appendix C2, as well as the November 19, 2010 RI/SCE Report for the Rhone Poulenc Portland Site; AMEC 2010.)
- **Arkema.** The pathway for transport of upland groundwater COIs to the transition zone within the nearshore and intermediate zones is complete. (See Section C3.7.5 of Appendix C2.)
- **Willbridge Terminal.** Based on concentrations and spatial patterns in TZW, a complete groundwater transport pathway from the upland to the transition zone does not appear to be present. (See Section C3.8.5 of Appendix C2.)
- **Gunderson.** Chlorinated solvents measured in nearshore TZW off Area 1 are likely the result of migration of upland groundwater COIs prior to installation of the remediation system extraction wells. (See Section C3.9.5 of Appendix C2.)

4.4.3.2 Study Area-Wide Summary of the Groundwater Transport Pathway

Table 4.2-2 presents the Study Area-wide understanding of the current groundwater transport pathway at DEQ ECSI sites based on late 2010 information from DEQ on the status of the sites and pathways. Maps 4.4-3a–h provide a river-mile-scale view of groundwater areas identified by DEQ to be affected by upland COIs in the vicinity of the Portland Harbor and the identified zones of in-river groundwater plume discharge, both interpreted and potential, based on the findings of the GWPA.

The groundwater pathway has been reasonably well-characterized at about half of the sites listed in Table 4.2-2, as summarized by category below:

- Documented evidence of a complete current pathway (a): 12 sites
- Likely a complete current pathway (b): 1 site
- Insufficient data to make determination (c): 80 sites[26]
- Not a complete current pathway (d): 27 sites[27].

In addition, groundwater discharging to stormwater pipes has been identified at 11 facilities. DEQ's evaluation of pathways, which has been reproduced in Appendix B (DEQ 2010b, Table 1), reaches similar conclusions with respect to the groundwater evaluation of the current status of the sites, with a few areas of potential disagreement:

- DEQ determined that the groundwater pathway was "insignificant" at several sites based on "screening" in an earlier version of the Milestone Report (i.e., July 2006); however, in Table 4.2-2, sites without groundwater investigations are classified as category c (insufficient data to make determination). These sites identified by DEQ include Alder Creek, Babcock Land Company, Chase

[26] Reflects EOSM's C-c pathway for metals only, see Table 4.2-2.
[27] Reflects EOSM's C-d pathway for TPH only, see Table 4.2-2.

Bag, Ryerson & Son, McWhorter Technologies, Olympic Pipeline, RK Storage, Schnitzer – Doane Lake, and Transloader International. LWG identifies all sites with no groundwater data as having insufficient data to make a determination. As noted in Section 4.4.3.1.1, sites for which insufficient data were available to determine if any of the GWPA inclusion criteria are met were referred to the Oregon DEQ for additional upland groundwater characterization.

- Conclusions about complete pathways reached in the GWPA differ from the late 2010 designations. Specifically, the GWPA did not confirm that complete and/or significant (in terms of influence on TZW and sediment chemistry) groundwater transport pathways exist at ARCO, ExxonMobil, Kinder Morgan, Willbridge Terminal, Premier Edible Oils, ST Services, Port of Portland Terminal 4, Slip 3, and Triangle Park, whereas the Milestone Report (DEQ 2010c) identifies complete pathways for these sites. The DEQ reduced the status of the groundwater pathway at ARCO and Willbridge Terminal from known (a) to likely (b).

4.4.4 Riverbank Erosion

Currently about 75 percent of the riverbanks within the Study Area are stabilized and armored with various engineered materials, including seawalls, riprap, structures, and engineered soil (Map 3.1-17). Riverbank erosion from unstabilized bank areas may represent an ongoing release mechanism in the Study Area. Riverbank erosion is identified on Table 4.2-2 as a "known" current pathway at six ECSI sites: Arkema, Gasco, EOSM, Gunderson, Triangle Park, and Terminal 4 (Slip 3).[28] This identification is based upon the detection of elevated concentrations of COIs in riverbank soils. Many other ECSI sites have not been evaluated as to the completeness of this pathway and if a bank has not been armored (e.g., sea walls and rip-rap) and there have been significant releases, it should be assumed that this pathway has the potential to release contamination to the Willamette River.

Today, riverbank stabilization and remediation plans are underway at several of these facilities.

4.4.5 Atmospheric Deposition

Similar to historical sources, current regional sources include automotive emissions, pesticide applications, and energy generation. Chemicals commonly acknowledged to play an atmospheric source role in urban river settings within the broader geographic region of the Pacific Northwest include PCBs, PCDD/Fs, PAHs, and mercury (see Section 4.3.5). Air pollution (e.g., vehicle and industrial emissions, other combustion products, fugitive dust, etc.) can enter the river directly through the processes of dry and wet deposition. Agricultural air pollution comes from contemporary practices which include clear felling and burning of natural vegetation as well as spraying of pesticides and herbicides. Atmospheric deposition is known to be a source of contamination

[28] See also Map 4.6-1a.

globally, and its relative importance in the Study Area in terms of atmospheric loading to the Study Area is evaluated in Section 6.

4.4.6 Overwater Releases

Overwater spills are unpermitted releases that occur directly into the waterway and can be intentional or unintentional. As discussed in Section 4.2.2, current overwater spills are those that have occurred since January 1, 2004. As shown in Table 4.2-3, documented spills have occurred since January 1, 2004 at approximately 22 facilities located within the Study Area. The nature of reported spills ranges widely, from dropped bottles to sheens of unknown origin to a 100-gallon spill of lubricating oil in April 2007, as a result of equipment failure at the Cascade General facility. Not all spills are reported or reflected in Table 4.2-3.

The activities most commonly associated with spills in the Study Area are product handling, overwater activities such as refueling, and vessel leaks:

- **Product handling**. Many facilities are now required to maintain spill prevention plans and have instituted practices to reduce spills.
- **Overwater activities.** Overwater activities, including ship repair or vessel refueling, are potential sources to surface water and sediment contamination. Regulations and BMPs have reduced such contributions in recent years. Spills during refueling are the most common type of overwater spill, but incidents during transfer of other materials (e.g., paint, hydraulic fluid, coal tar pitch) have also been reported. Furthermore, the operation of boat motors may contribute to surface water and sediment contamination.
- **Vessel leaks.** On average, 20 spills from vessels directly into the LWR are reported to the USCG each year (NRC 2010), nearly all of which are diesel fuel, gasoline, hydraulic oil, lubricating oil, or waste oil. Bilge and ballast water from vessels has also been released. A detailed list of vessel spills is included in Table 4.3-4. In addition to the types of releases above, the spills include sandblast grit, sewage, paint mixtures, sulfuric acid, and grain in volumes ranging from unknown sheen quantities to barrels. Reasons for spills vary, but are primarily related to equipment or operator error.

Utility crossings are a potential source of spills in the Study Area. One petroleum pipeline crosses the Willamette River within the Study Area. It is located between the Willbridge bulk fuel terminal and south end of Triangle Park (approximately RM 7.7). Gasoline lines cross the river at RM 2.8 and near the Sauvie Island Bridge in the Multnomah Channel. Two sewer lines cross the river, one at RM 7 and the other near RM 10. There are no records of spills or leaks from these crossings.

4.5 HISTORICAL AND CURRENT SOURCES OUTSIDE THE STUDY AREA

Point and nonpoint discharges within the Willamette River Basin are potential sources of contamination in sediment, surface water, and biota in the Study Area. Chemicals in discharges and runoff from diverse land uses in the basin eventually make their way to the river by the time it flows into the Study Area. Contaminant loading from sediment transport and water from upstream areas throughout the last century also contributed to the conditions currently observed in the Study Area.

4.5.1 Non-Study-Area Sources in the Lower Willamette River

Sources in the LWR, both downstream and upstream of the Study Area, may contribute to chemical deposition within the Study Area. The Study Area is at the downstream end of a large basin with a long history of industrial, municipal, and agricultural inputs. Significant agricultural runoff persists upriver, and together with inputs from other industries and cities upstream, as well as atmospheric deposition in the watershed, the river's chemical burden is already elevated before entering the Study Area.

These upstream and downstream areas are prone to flooding, as evidenced during the major flood events of the past century. Flooding contributes to in-water contamination by eroding contaminated riverbank areas and other surface soils, and potentially breaching historical wastewater containment ponds proximal to the river. Today, many riverbanks have been armored with seawalls, riprap, and other engineered materials. The 32-ft-tall seawall that extends approximately one mile from the Hawthorne Bridge to the Steel Bridge was constructed by the City from 1923 to 1929 as a bulwark against floods (Blalock 2008).

The downtown reach immediately upstream from the Study Area is described below (GSI 2009):

> The downtown reach of the Willamette River has been used and modified for more than 150 years. Various industrial activities have occurred on the banks of the river, including ship building and ship breaking, heavy manufacturing, pesticide formulating, manufactured gas production, power generation and distribution, lumber processing, and commodities importing and exporting. The river banks have been significantly modified and used for automotive transportation, particularly in the lower half of the downtown reach. Waterfront and upland facilities and roadways may have contributed contaminants to the Willamette River via direct discharges (e.g., stormwater and non-stormwater flows), groundwater discharges, overwater activities, overland runoff, or bank erosion.

The downtown reach has been the focus of a collaborative effort by DEQ, the City of Portland, ZRZ Realty Company, Portland General Electric, and TriMet to evaluate the potential presence of contaminants in sediment from RM 12 to 16, immediately upstream from the RI Study Area. The focus of this effort is to determine where source control measures or cleanup activities should be implemented at riverfront properties (as well as facilities discharging to shared conveyances) to minimize potential contamination (GSI 2009). Results of these investigations are discussed in Appendix H.

Shoreline facilities upstream of the Study Area that are included in DEQ's ECSI database are listed in Table 4.5-1, with locations shown in Maps 4.5-1a–d. Note that unless a pathway was identified as a complete pathway to the river in the ESCI database, LWG has not independently confirmed that sites listed on Table 4.5-1 and Maps 4.5-1a–d have or had a confirmed pathway to the river.

One of these sites, the former Portland MGP facility at RM 12.2W has been the focus of an upland and in-water source control investigation under a Consent Order with DEQ executed in April 2009. This facility operated between 1860 and 1913 for manufacturing gas from coal, carbureted water (water enriched with oil), and, briefly, oil. Initial review of sediment results shows that MGP-related contaminants including PAHs, VOCs, and cyanide are present in the Willamette River adjacent to the former gas manufacturing site, most notably in subsurface sediments adjacent to Block 5 of the former site. Limited impacts were observed in shallow upland wells (to 56 ft bgs) installed along the riverfront to assess the potential for ongoing impacts to the river from historical upland releases. Phase 2 follow-up investigations found elevated VOCs and PAHs in deeper upland groundwater zones (to approximately 130 ft bgs), in particular in the 65 to75-ft bgs zone, which appears to coincide with the fill/native sediment interface. The latest proposal is to install deeper wells at four riverside locations corresponding to the former Block 5 of the MGP operation, and adjoining locations to the north and south.

Another site, the former 17-acre Zidell property (owned by the ZRZ Realty Co.) is located on the west bank of the Willamette at RM 14. The Zidell site was used for ship building, ship dismantling, welding, and other ship activities from 1916 until the 1960s. Barge building activities are currently ongoing. As a result of these activities, the onsite soils and offshore sediment are contaminated with asbestos, metals, petroleum hydrocarbons, PAHs, butyltins, and PCBs above state-mandated cleanup levels. According to USACE (2010b), this property is the focus of a major cleanup effort involving the removal or capping of 17,000 yd^3 of contaminated soils and sediment, updates to the stormwater management system, removal of invasive species, removal of treated wood pilings and a floating dock, and bank stabilization. Cleanup activities began in the summer of 2011 (USACE 2010b).

At the PGE Station L site (RM 13.1 to 13.5E), PCB oils were generally used in electrical equipment from the mid-1930s to the 1970s. Soils, various structures onsite, and Willamette River sediments were found to be contaminated with PCBs that were released from transformers. Dredging and capping of river sediments was conducted from 1990–1991. The multi-layer cap was constructed with sand, gravel, and riprap to a thickness of greater than 6 ft. A Record of Decision for No Further Action at site was issued by DEQ in 1994 (DEQ 2011a). As a result of DEQ's 2009 downtown reach sediment evaluation, PGE is currently evaluating sediment and upland sources between RM 13.1 and 13.5 to determine if additional sediment remediation or source control is needed adjacent to the historical PGE Station L and Station F/Inman-Poulson Lumber mill site.

Maps 4.5-1a–d also show outfalls upstream of the Study Area. Outfalls in the downtown reach have not been fully mapped, and this is an ongoing effort by the City. These maps show outfalls that were identified by the City as of February 2010. These include municipal outfalls, including CSO outfalls, and other public and private outfalls. Table 4.5-2 lists currently available data on NPDES-permitted discharges from facilities upstream (to Willamette Falls) and downstream of the Study Area (Anderson 2006a,b, pers. comm.).

As described in Section 3, under certain river stages, flows, and tidal conditions, the influence of the Columbia River estuary causes periodic flow reversals in the Willamette River near its mouth and within Multnomah Channel. These flow reversals could transport sediment-bound chemicals from downstream reaches of the river into the Study Area.

The list of impaired waters in Oregon prepared under Section 303(d) of the federal CWA and its amendments includes the main stem and tributaries of the Willamette River. The 303(d) listings in the LWR (RM 0 to 24.8 as defined by DEQ)[29] include aldrin, DDT, DDE, dieldrin, iron, manganese, mercury, PCBs, PCP, PAHs, temperature, and bacteria. Johnson Creek, a tributary that enters at RM 18, is listed for toxic chemicals, including dieldrin, DDT, PAHs, and PCBs. DEQ has developed total maximum daily loads (TMDLs) for temperature, bacteria, dieldrin, and DDT in Johnson Creek to reduce these watershed contaminants.

4.5.2 Sources above Willamette Falls (Upper Willamette River)

Both point sources and nonpoint sources of contamination are present above Willamette Falls. Agriculture, forestry, urban land use, geologic features, and atmospheric deposition have contributed to conditions in Portland Harbor.

Table 4.5-3 lists historical sources within the Willamette River basin above the falls that were present in 1967, according to OSSA. The table shows the source of waste (both industrial and municipal), the receiving stream, the Willamette river mile of the effluent discharge, the present treatment and disposal of wastes, and action needed by OSSA. Examples of the types of sources within the basin at that time included domestic sewage (with primary and/or secondary treatment), glue wastes from plywood manufacturers, pulp process wastewater, slaughterhouse wastes, kraft mill wastes, metal plating wastes, dye and wool fibers from woolen mills, sulfite mill wastes, and silage wastes.

Presently, more than 750 permitted discharges enter the Willamette River upstream of Willamette Falls, including 10 municipal sewage treatment plants and several pulp, paper, lumber, and fiberboard manufacturers. Hundreds of facilities also have general permits for discharge of noncontact cooling water and filter backwash, gravel mining waste streams, and tank cleaning fluids. Industrial stormwater discharge permits are

[29] http://www.deq.state.or.us/wq/assessment/rpt0406/results.asp

held by facilities that handle paint, steel, metal plating, semiconductors, adhesives, and food products, as well as by landfills and transportation companies.

Most of the agricultural and forested land in the Willamette River Basin can generate nonpoint sources of pollution. The primary nonpoint source problem associated with forestry is accelerated sediment transport, but nutrients, fertilizers, and herbicides are also found in forest runoff. Erosion from agricultural lands in the Willamette Valley is the most commonly cited nonpoint source of pollutants in the upper reaches of the Willamette River Basin (Tetra Tech and E&S 1993), especially fertilizers, pesticides, and herbicides. In USGS studies of pesticides in the Willamette Basin (Wentz et al. 1998), the highest concentrations of organochlorine pesticides and PCBs were reported for three mostly agricultural sites. Historical mining is also an upriver source of mercury and lead. Historical discharges of dioxins from pulp and paper mills are relevant sources of contamination as well.

Nonpoint pollutants from the upper Willamette Basin (e.g., pesticides, PAHs, metals) also enter via runoff from residential, industrial, and commercial areas that do not require stormwater permits. Municipal stormwater permits are also held by cities in the upper Willamette Basin.

A fish advisory for mercury is in effect throughout the entire main stem of the Willamette River, due in part to runoff from natural volcanic sources, past mining activities, and atmospheric deposition in the upstream reaches of the Willamette River Basin.

DEQ's 303(d) list of impaired waters above Willamette Falls includes numerous tributaries of the Willamette River. The 303(d) listings in the main stem above Willamette Falls include aldrin, arsenic, DDT, DDE, dieldrin, iron, manganese, mercury, PCBs, DO, temperature, and bacteria. Most of the 303(d) listings for the upper Willamette River tributaries are for temperature and bacteria; other listings relate to nutrients, DO, turbidity, and pH. In addition, smaller creeks in the middle and upper Willamette sub-basins are listed for dieldrin, heptachlor, dichloroethylene, PCE, TCE, arsenic, copper, iron, lead, manganese, mercury, or zinc.

Based on the 303(d) list, DEQ has developed TMDLs for 11 of the 12 Willamette River sub-basins (Table 4.5-4; DEQ 2006). TMDLs are currently being developed for the Yamhill sub-basin. Temperature, bacteria, and mercury TMDLs have been issued for all Willamette River sub-basins and the main stem. A PCDD/F TMDL was developed by EPA in 1991 for the Willamette and Columbia rivers. Further reduction in watershed contaminants will likely occur as a result of TMDL implementation and other future watershed toxic reduction efforts.

4.6 SOURCE CONTROL MEASURES

Under the 2001 MOU, DEQ is the lead agency responsible for identifying and controlling upland sources of contamination. EPA is the lead agency for overseeing the

investigation and cleanup of the in-water portion of the Study Area. Together, these two agencies developed the Portland Harbor JSCS in 2004 with the goals of identifying, evaluating, and controlling sources of contamination that may affect the LWR.

Upland source control is necessary to allow cleanup of the river to proceed without the risk of recontamination. Source control measures are implemented at a given site to address ongoing sources of contamination whether or not the source is a result of a historical or current release. Currently, DEQ is investigating or directing source control work at over 80 upland sites in Portland Harbor (DEQ 2010b).

For DEQ, upland source control is an iterative process, where conclusions determined earlier may be refined by information gathered later in the process. The 2010 Milestone Report lists the following combination of tools that DEQ uses to control a source(s):

- Technical assistance.
- Cleaning up contaminated upland areas by removing highly contaminated soil areas, stabilizing or capping contaminated bank areas, treating or containing contaminated groundwater, and extracting contaminated sediment from storm sewer systems.
- Source control of active discharges using BMPs, industrial process changes, pollution prevention practices, and technology-based effluent controls. Compliance is achieved voluntarily or through administrative actions, including permits or enforcement.
- Source control of stormwater.
- Administrative actions and enforcement, such as licenses, permits, deed restrictions, requirements for site development plans, and enforcement actions, which may be necessary when administrative actions are violated.

Table 1 of the 2010 Milestone Report (reproduced as Appendix B) summarizes, for a given site, the status and type of source control activities, the basis for determining if source control is needed, and the schedule for implementing source control measures. Sites listed in the table are only those sites for which DEQ is actively overseeing upland investigations or source control activities (also including sites for which source control decisions have been made). Several ECSI sites are not included in the table because DEQ does not believe these sites are contributors to Willamette River contamination, because there is insufficient information to determine if the site is a contributor but the site has not entered DEQ's cleanup program, or because DEQ had not amended the Milestone Report to align with the expanded Study Area (e.g., ECSI sites in the RM 11 to 11.8 reach).

Information from Table 1 has been graphically displayed in Maps 4.6-1a–e for each of the major pathways of a particular site: riverbank erosion, groundwater, overland transport, overwater activities, and stormwater/wastewater. Sites on the maps are

shaded different colors to correspond with the status of the following DEQ source control activities:

- Red – Source control evaluation is ongoing
- Blue – Source control evaluation has not started yet
- Green – The source control evaluation is complete or under DEQ/EPA review
- Yellow – A "No Further Action" determination has been made for the site
- Gray – The pathway does not exist for a site
- White – Site is not included in Table 1 of the Milestone Report.

For each ECSI site on Maps 4.6-1a–e, a symbol is included that corresponds with DEQ's interpretation of the potential for that pathway to impact in-water media. The priority levels for sites and pathways, as described in the footnotes to DEQ's Table 1, are provided below:

> **High** = High priority pathways and sites are those where a complete migration pathway exists and the upland source is significantly impacting the river or poses a significant and imminent threat to the river based on initial evaluation of key source control prioritization factors (see p. 4-3 of the JSCS). A primary consideration is that one or more media (soil, water, air) significantly exceed applicable Screening Level Values (SLVs) at the point of discharge to the river (e.g., water at the end of a discharge pipe, or soil or material at the riverbank) or the most reliable and cost-effective data point (e.g., groundwater measured at the shoreline), or where a bioaccumulative chemical is detected at concentrations significantly above the SLV. In addition, if an upland source is violating DEQ narrative water quality criteria for the Willamette River, the site may be considered a high priority. High priority sites are expected to move forward with aggressive source control measures without delay or be subject to enforcement action.
>
> **Medium** = Medium priority pathways and sites are those where a complete contaminant migration pathway exists and the upland source is impacting the river or poses a significant and/or imminent threat to the river based on an initial evaluation of key source control prioritization factors (see p. 4-3 JSCS). A primary consideration is that one or more media exceed applicable SLVs, but not significantly, at the point of discharge to the river, or where a bioaccumulative chemical is detected at concentrations above the SLV. Although exceedance of SLVs does not necessarily indicate a site poses a significant and/or imminent threat or needs to immediately implement source control measures, it does indicate that the site may pose a threat to human health or the environment and that additional evaluation may be needed to determine if source control measures are required to prevent, minimize or mitigate the migration of hazardous substances to the river. If the site exceeds one or more SLVs, the need for further characterization or for implementation of source control measures will be based on a site-specific weight-of-evidence determination. Medium priority sites are expected to perform a weight-of-evidence evaluation to determine if source control measures are required.
>
> **Low** = Low priority pathways and sites are those where upland data indicate, based on an initial evaluation of key source control prioritization factors (listed on p. 4-3 JSCS), that the site likely poses a low threat to the river (e.g., concentrations are near or below

> SLVs) or where DEQ, in consultation with EPA, may issue an upland "No Further Action" (NFA) determination or lower the State's priority of the site for further upland investigation or remedial action under DEQ's cleanup authority. Source control measures will not be required at low priority sites unless determined necessary by the results of the Portland Harbor RIFS or ROD.

As of September 2010, the ECSI sites were categorized, according to DEQ's source control efforts, into the following categories:

- High-priority sites – 11
- Preliminary high-priority sites – 5
- Medium-priority sites – 24
- Low-priority sites – 23
- Priority to be determined – 3
- Sites with source control decisions – 24.

Additionally, DEQ and the City (under an Intergovernmental Agreement) are jointly working together to identify and control upland sources draining to the Study Area through City outfalls.

5.0 IN-RIVER DISTRIBUTION OF INDICATOR CHEMICALS

The distribution of ICs in sediment, in-river sediment traps, surface water, TZW and groundwater seeps, and biota is summarized in this section.[1,2]

COIs are contaminants expected to be present at a site based on a review of site information. Numerous chemical and physical parameters were identified as COIs for the Study Area and were subsequently analyzed and detected in sampled media (Table 5.0-1). Owing to the distribution of contaminant sources and the dynamic nature of the Willamette River, COIs have become comingled to some extent throughout the Study Area. To facilitate a clear and practical presentation of the nature and distribution of contamination in the Study Area for the RI, an IC list was identified from the extensive list of COIs to represent the nature and extent of the range of contaminants that potentially pose risk to human health and the environment in sediment, surface water, TZW, and biota. Although additional COIs beyond the IC list may pose risk to human health and the environment and are present within the Study Area, the IC list was generated, in consultation with EPA[3] in the spring/summer of 2008, based on the chemicals that emerged from the Round 2 risk screening and preliminary risk evaluation process and consideration of the following non-risk-based factors:

- Frequency of detection—Chemicals with a high frequency of detection (e.g., generally >50%) were selected as an IC.
- Cross media comparisons—Preference was given to chemicals that would allow comparisons across media.
- Representative chemicals—Several ICs were selected to represent a suite of compounds (e.g., BEHP was selected to represent the phthalate group).
- EPA requests—EPA requested the inclusion of several additional chemicals for sediment and biota based on its review of the Round 2 Report (Integral et al. 2007). EPA also requested the inclusion of certain other chemicals with widespread sources in the harbor (metals, PAHs, and TPH) during the IC lists development discussions in 2008.

Table 5.0-2 identifies the ICs[4] selected by this process for various uses in the RI. This includes nature and extent for abiotic and biotic media chemicals (Section 5); loading, fate, and transport (Section 6); fate and transport modeling (part of FS); and the site-wide CSM (Section 10).

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[1] The discussion of chemical nature and extent may include use of facility names and landmarks (e.g., bridges) for location references; mention of such names does not necessarily indicate a source or origin.
[2] Section 5 text, maps, figures, and tables are based on the data lockdown date of June 2, 2008. Data collected or received between June 2, 2008 and July 19, 2010 is presented in Appendix H.
[3] As documented in Appendix A5, Attachments A4, A7, A9, and A11.
[4] Contaminants for which PRGs are being developed through the FS process will not be finalized until late 2011. However, the LWG presumes that the ICs included in the RI will be consistent with the PRG list.

Once the baseline risk assessments were completed (the BHHRA is provided in Appendix F; the BERA in Appendix G), a list of contaminants posing potentially unacceptable risk was documented. These contaminants for the BHHRA and BERA are listed in Table 7-1 in Appendix F and Table 11-2 in Appendix G, respectively, and are included in Table 5.0-2. Additionally, chemicals in surface water and TZW sampling results that exceed tap water and surface water quality criteria/screening values (without taking into account any spatial or temporal averaging) are also included in Table 5.0-2; these contaminants will be carried into the FS for evaluation of chemical mobility. The water screening methods and results are provided in Appendix D3.3 and Anchor QEA (2011).

The nature and extent of four ICs or compound groups identified in the BHHRA and BERA as posing risk in the Study Area are believed to encompass the spatial extent of potentially unacceptable risks associated with the contaminants identified in the baseline risk assessments. These four chemicals, hereafter referred to as bounding ICs, are total PCBs, total PCDD/Fs (as both total dioxins/furans and dioxin/furan TCDD TEQ[5]), total DDx, and total PAHs. This is not intended to imply that other contaminants will not be evaluated in the FS and accounted for in the site remedy[6]. Other contaminants potentially posing risk are comingled with the bounding ICs, but because of their spatial distribution within the Study Area, only these bounding chemicals are discussed in detail in the Section 5 subsections that follow. For each medium, the nature and extent of an additional set of ICs is also described. This set varies by media but it includes all chemicals that are the focus of the comprehensive cross-media and fate and transport evaluations presented in Section 10 (CSM) of this RI. This set of CSM chemicals was selected in consultation with EPA to provide a relatively complete picture of the distribution, transport, and fate of contaminants in the Study Area across a range of physical, chemical, and biological processes, as well as known and potential sources. Finally, the nature and extent data for the balance of the ICs listed in Table 5.0-2 are presented on all tables, maps, and figures in Appendix D but not addressed in the narrative.

The following sections are organized by media and focus on the nature and extent of ICs in and immediately adjacent to the Study Area in sediments (Section 5.1), sediment traps (Section 5.2), surface water (Section 5.3), TZW (Section 5.4), and biota (Section 5.5).

Depending on the medium examined, the nature and extent discussion is supported by a variety of tabular and graphical materials: 1) plan-view and core maps for sediment showing the extent of each chemical's distribution, 2) summary statistics tables, 3) scatter-plot graphs depicting chemical concentrations by river mile, and 4) bar charts

[5] The dioxin/furan TCDD TEQ does not include dioxin-like PCB congeners.
[6] These four COCs largely represent the areal extent of unacceptable risk to humans and wildlife (birds and mammals). Other COCs are associated with unacceptable risk, but generally within the areas represented by these four COCs. The exception may be where toxicity test results indicate unacceptable risk to the benthic invertebrate community, but no specific chemical has been associated with toxicity.

and box-whisker plots for comparing values and distributions. Summary statistics for ICs are tabulated in the main text for sediment (Tables 5.1-1 and 5.1-2), in-river sediment traps (Tables 5.2-1 through 5.2-4), surface water (Tables 5.3-2 through 5.3-7), and biota (Tables 5.5-1 and 5.5-2). Summary statistics for all parameters analyzed in each medium are presented in tables included in Appendix D. The entire RI database is presented in Appendix A3, and the updated RI database is presented in Appendix H, Attachment H-2.

For each IC, the following summary statistics are tabulated: the frequency of detection; the minimum, maximum, mean, median, and 95th percentile; and the station locations of the maximum values. Two sets of statistics are presented for each IC. One set reflects only detected values and the other set shows detected and undetected values combined. The statistics have been compiled separately for areas inside the Study Area (RM 1.9–11.8, ≤13 ft NAVD88, exclusive of the Multnomah Channel) and areas outside of it. Summary statistics for sediments include both point samples and composite samples to provide a general understanding of IC concentration distributions. The discussion of the nature and extent of ICs in media provided in the remainder of this section is based on statistics calculated for detected concentrations only.

Where specific results are cited in the text (i.e., the concentration of a sample, median and 95th percentile values) qualifiers associated with that result are also cited, with one exception. The qualifier "T" is not cited as it generally indicates that the result was mathematically derived through summing multiple results (e.g., total PCB congeners equal the sum of the PCB congener results). The "T" qualifier may also indicate that a result is an average of multiple results for a single analyte (e.g., field replicates) or that a result was selected for reporting in preference to other available results (e.g., for parameters reported by multiple methods). The following qualifiers are cited with the results:

A – Total value is based on a limited number of analytes.

J – The associated numerical value is an estimated quantity.

N – Presumptive evidence of presence of material; identification of the compound is not definitive.

U – The material was analyzed for, but was not detected. The associated numerical value is the sample quantitation limit.

V – Median or 95th percentile was obtained through interpolation of data.

In certain cases, concentrations of closely related analytes were added together to create a group sum. When calculating group concentrations for this nature and extent evaluation, a value of zero was used for non-detected concentrations on an individual sample basis; other analyte summing approaches were used in the risk evaluations presented in Appendices F and G of this report for the BHHRA and BERA,

respectively.[7] The 2,3,7,8- TCDD TEQ values for dioxin-like PCB congeners and PCDD/Fs were calculated with WHO 2005 TEFs for mammals (Van den Berg et al. 2006). The cPAH BaPEq values were calculated using PEFs provided in EPA (1993). Tables in Appendix D1.6 present the constituent concentrations used in each group sum. Further information on summing methods is provided in Section 2.1.4.

5.1 INDICATOR CHEMICALS IN SEDIMENT

This section summarizes the surface and subsurface sediment data collected in the Study Area. The locations of all sediment samples in the nature and extent data set are shown on Maps 2.2-1a–y and 2.2-2a–t. The surface sediment data set includes all samples with intervals starting at 0 cm and extending to depths ranging to 40 cm bml.[8]

5.1.1 Sediment Data Set

The sediment RI data set is composed of all Category 1 LWG and non-LWG data collected within the Study Area (below +13 ft NAVD88) from May 1997 to December 2007. Sediments collected below +13 ft NAVD88 include both subaqueous sediment and beach sediment. Sediment data are used to characterize contaminant distribution and potential source effects, to provide data necessary for the risk assessments, to provide data for the FS, and to refine the understanding of the physical dynamics of the river system.

Summary statistics for grain size, TOC, and ICs in the surface and subsurface sediment samples from the Study Area are presented in Tables 5.1-1 and 5.1-2; the full data set is provided in Appendix D1.3. These summary statistics do *not* include results from locations that were dredged or capped subsequent to sample collection. However, post-dredged sediment samples are included in the summary statistics.

5.1.2 Indicator Chemicals in Sediment

The IC list for sediment is presented in Table 5.0-2. The selection of ICs was guided by the considerations detailed in Section 5.0. A total of 34 individual analytes and calculated chemical sums were identified as ICs for sediment. They are organized as follows:

- PCBs

[7] For the RI, the summation methods use zero for non-detects within sums. This allows a clear presentation of results for assessment of nature and extent, avoiding bias presented by the detection limits. This also prevents high detection limits from creating confusion in the evaluation of nature and extent. For the BHHRA and BERA, the summation method uses one-half the detection limit for non-detects within the sums. This was the agreement with EPA.

[8] The functional definition of surface sediments for this site is 0–30 cm based on physical system studies. However, the recorded lower depth of a set of sediment samples (i.e., shallow cores that begin at the mudline) in the nature and extent data set reached to 40 cm. These samples were grouped with the surface data set, thus extending the maximum depth for the surface horizon to 40 cm. Core samples that extended from the mudline to depths greater than 40 cm were grouped with the subsurface sediment data set.

  - Total PCBs*
  - PCB TEQ (ND=0)
- PCDD/Fs
  - Total PCDD/Fs* and TCDD TEQ (ND=0)*
- DDx
  - Total DDx (sum of 2,4'- and 4,4'-DDD, DDE, DDT)*
  - Total of 2,4'- and 4,4'-DDT
  - Total of 2,4'- and 4,4'-DDE
  - Total of 2,4'- and 4,4'-DDD
- PAHs
  - Total PAHs*
  - Total cPAH BaPEq values
  - Total LPAHs
  - Total HPAHs
  - Phenanthrene
  - Naphthalene
  - BAP
- Petroleum Hydrocarbons
  - TPH
  - TPH - Diesel-range hydrocarbons (DRH)
  - TPH - Residual-range hydrocarbons (RRH)
- SVOCs
  - BEHP*
  - Butylbenzyl phthalate
  - Pentachlorophenol
  - Hexachlorobenzene
- Pesticides
  - Total chlordanes*
  - gamma-Hexachlorocyclohexane (HCH)
  - Aldrin*

  - Dieldrin*
- Metals
  - Arsenic*
  - Cadmium
  - Chromium*
  - Copper*
  - Lead
  - Mercury
  - Nickel
  - Zinc*
- Organometallic Compounds
  - TBT*.

This section focuses on the distribution of a subset of 13 ICs in surface and subsurface sediment within the Study Area, which are indicated with an asterisk (*) in the above list (total PCDD/Fs and TCDD TEQ are grouped as one chemical in this count). Although numerous contaminants that potentially pose risk to human health or the environment are comingled across the site, these 13 ICs were selected, in consultation with EPA, based on the preliminary risk evaluations and other site information, and match the contaminants presented and discussed in the site-wide CSM (Section 10). Of these, four contaminant groups—total PCBs, total PCDD/Fs (including TCDD TEQ), total DDx, and total PAHs—have been identified as bounding ICs based on the risk evaluations and their nature and extent are discussed in depth here. The discussion focuses primarily on the following items:

- A description of the data set for each contaminant, including sample counts, concentration range, and frequency of detection.
- A discussion of the surface and subsurface concentration distributions in the Study Area organized by eastern nearshore, western nearshore, and navigation channel subareas.
- Additional information is provided for the four bounding ICs:
  - The vertical trends in sediment concentrations
  - The relationship of subsurface sediment to surface sediment
  - The nature and composition of these complex chemical groups and distribution patterns.

The discussion of the other 9 ICs in the subset is less comprehensive, omitting the data set description and referring instead to maps, tables, and figures to provide a complete picture of the distribution of these contaminants. The data for the remaining 21 sediment ICs is presented in Appendix D1. In Section 10, contaminant distributions across abiotic and biotic media and in relation to specific potential sources are presented in more detail for the subset of 13 ICs.

5.1.3 Description of Sediment Presentation Tools

The sediment RI data set is composed of all Category 1 LWG and non-LWG data collected within the Study Area (below +13 ft NAVD88) from May 1997 to December 2007. The sediment chemistry distributions are depicted in five graphical formats: surface plan-view concentration maps and subsurface core concentration maps (Maps 5.1-1 through 5.1-28), scatter-plot graphs (Figures 5.1-1 through 5.1-32), and histograms and stacked bar charts (Figures 5.1-33 through 5.1-47). At EPA's request, core plots showing a higher level of detail have been produced for the following ICs:

- Total PCBs
- Total DDx
- TCDD TEQ
- Total PAHs
- Total cPAH BaPEq values.

These more detailed core plot maps are presented in Appendix D1.2.

Maps and scatter-plot graphs of surface and subsurface concentrations for the remaining 21 ICs, plus percent fines and TOC, are included in Appendices D1.1 and D1.4, respectively. Appendix D1.3 provides statistical summaries of all sediment analytes.

Surface Chemistry Maps: The plan-view concentration maps present all surface sample data using color-coded dots that correspond to a concentration scale for that particular chemical. Non-detected concentrations are differentiated from detected concentrations on the surface maps by a dot in the center of the sample symbol (i.e. ⊙). The maps include data points from locations that were dredged or capped subsequent to the collection of the sample(s) (shown by a circle centered around the sample symbol [i.e., ◎]).[9,10] Data from these areas are presented to show spatial patterns of chemicals from a historical perspective. In addition, the surface maps include histograms showing the distributions and frequencies of the detected and

[9] For example, all data shown for locations *within the capped area* at the M&B site (see Maps 2.2-1i and 2.2-2i) are from surveys completed between 1999 and 2002, prior to capping. These data are shown on the surface and subsurface core plan-view maps and included in the map histograms; however, they are not included in the other sediment data presentations (i.e., scatter plots, histograms, and stacked bar charts, discussed below).

[10] Surface interval sample locations G088, G087, and G091 collected in 2004 in the International Terminals Slip were dredged subsequent to sampling. These locations were resampled in 2005 at C088, C087, and C091.

non-detected results. Data from all samples shown on the maps are included in the histograms.

The concentration ranges (or intervals) used in color-coding the chemical data shown on the maps (e.g., the threshold value for the red labels) were based on the frequency distributions (i.e., natural breaks) in the historical data set for these compounds and as approved or modified by EPA for use in the Programmatic Work Plan. These concentration range intervals were also used in the Round 2 Report.

Subsurface Core Maps: The core maps show the distribution of ICs with depth at the subsurface sediment sampling stations (these maps also include the surface sample data). Inset maps for densely sampled core areas are provided in most cases, unless the core samples in these areas were archived (i.e., for possible future chemical analysis). In these maps, the actual core station is marked with a triangle (i.e., △). The core segment divisions displayed on the maps are scaled to the thickness of each sample interval. Note that these maps do include cores from locations that were subsequently dredged or capped, as indicated on the maps. Cores taken post-dredging are also included on the maps. The subsurface concentration maps do not indicate samples where concentrations are based on partial sums (i.e., A-qualified data; the few cases where data are based on partial sums are from non-LWG studies).

Scatter Plots: Scatter plots of the distribution of analytes in surface and subsurface sediment per river mile are presented in Figures 5.1-1 through 5.1-32, and in Appendix D1.4. To aid in differentiating potential concentration trends, the data in these plots are further separated into eastern nearshore, western nearshore, navigation channel, and Multnomah Channel stations as defined by the federal navigation channel boundary. The areas falling into these categories are shown in Map 5.1-29. Unlike the plan-view maps, the scatter plots do *not* include data for samples from locations that have been subsequently dredged or capped.

Histograms: Histograms were developed to supplement the subsurface core maps and support examination of vertical trends in chemical concentrations with depth in the sediment column. The histograms compare the absolute magnitude of the ratios of surface and subsurface sediment chemical concentrations for the bounding ICs on subarea basis (e.g., RM 8-9 west of channel; see Figures 5.1-33, 5.1-39, 5.1-42, and 5.1-45). The ratios were calculated by dividing the mean of all detected surface sample concentrations in a given subarea by the mean of all detected subsurface core interval samples in that subarea. The absolute magnitude of the ratios is plotted on the histograms (i.e., where the subsurface mean is greater than the surface mean, the inverse of the ratio is plotted). The actual surface and subsurface concentrations for individual samples by river mile are shown on the scatter plots (Figures 5.1-1 through 5.1-32 noted above), and as with the scatter plots, the histograms do *not* include data from dredged or capped samples.

The y-axis in the plots is centered on a value of 0, which represents no numerical difference between the mean surface and subsurface concentrations for a given subarea. Values extending downward from the y-axis indicate areas where the mean of subsurface values exceeds the surface mean. Bars extending upward show where the surface sediment means are greater. The y-axis value indicates the absolute magnitude of the differences between the surface and subsurface means. In some instances, a ratio could not be determined because only surface sediments were analyzed for the bounding ICs in that subarea. Subareas included east, navigation channel, and west zones for each river mile in the Study Area, as well as downstream (RM 0–2), Multnomah Channel, and Swan Island Lagoon. Mean concentrations were also calculated for the Study Area as a whole (see leftmost column in each figure).

These histograms are useful in providing a visual summary of spatially averaged surface/subsurface trends throughout the Study Area and, therefore, for identifying ongoing versus historical sources of contamination. However, some caution is needed in interpreting the trends due to the biased nature of the RI sampling program (i.e., subsurface core samples were generally focused on known areas of contamination, whereas surface samples were distributed more widely). Consequently, these histograms should be examined in conjunction with the subsurface core maps in evaluating surface to subsurface trends for a specific IC and subarea. This is particularly true for the relative low density PCDD/F data plotted in Figure 5.1-39, where a single extreme data point can skew the calculated ratio.

Stacked Bar Charts**:** Stacked bar charts are designed to reveal potential distinctive patterns in the relative abundance of bounding IC components (e.g., homologs, isomers). The analyte components are shown in the stacked bars as a percent of the total concentration, while the total concentration of the IC is displayed as a line on a logarithmic scale. Station location labels are provided on the x-axis, and river mile is indicted on the secondary x-axis along the top of the chart. Subsurface figures display only the sample interval with the maximum concentration of the analyte, per core station location. The stacked bar charts do *not* include data from dredged or capped samples.

Any patterns apparent in the stacked bar charts must be interpreted with caution. Changes in chemical composition and apparent trends shown by the bar charts may be indicative of significant patterns (e.g., distinctive source contributions), or they may be within the range of normal data variability. Further, it should be kept in mind that the display of constituent components as a percentage of the total may, in some cases, amplify the visual impact of what are in fact small-scale changes in sample composition. It is important to emphasize that source identification and allocation are complex multivariate problems. The pattern shifts discussed in the following sections based on stacked bar chart presentations may be suggestive, but cannot be interpreted directly as or attributed to localized sources. Such a characterization would require rigorous quantitative forensic analysis, which is outside the scope of this RI/FS.

5.1.4 Sampling Methods

LWG surface sediment samples were collected in a consistent, repeatable manner with a stainless-steel, 0.3-m^2 hydraulic power-grab sampler. The maximum penetration of the power-grab sampler was approximately 30 cm. Non-LWG surface samples were collected using a variety of sampling devices, including ponar samplers, power grabs, Eckman samplers, box cores, and spoons. A limited number of non-LWG surface samples were collected from the mudline to depths of 30 to 40 cm bml, and these data are also included in the RI surface sediment data set.

Subsurface sediment was collected by the LWG using a customized vibracorer equipped with either 14-ft or 20-ft core tube. Non-LWG subsurface samples were collected by a variety of methods and depths, the most common being a vibracore, followed by a "driven tube." The driven tube could include the vibracore, impact, or gravity methods. Other coring samplers included gravity corer, macro-core sampler, impact corer, hand core, split-spoon sampler, Mudmole™, and steam auger.

5.1.5 Total PCBs in Sediment (Congeners and Aroclors)

This subsection summarizes the surface and subsurface distribution of total PCB concentrations in the Study Area, compares the PCB congener and Aroclor concentrations, and discusses PCB homolog and Aroclor patterns across the Study Area. For the purpose of sediment characterization, total PCB congener concentrations represent the sum of detected congener concentrations in a sample. In cases where no congeners were detected, the single highest detection limit of all congeners analyzed is used to represent the total value. Similarly, total PCB Aroclor values reflect the sum of detected Aroclors in a sample.

To simplify characterization of PCBs in the Study Area, the total PCB congener and total Aroclor data were combined into a single data set of total PCBs. These total PCB data were used to create Maps 5.1-1 and 5.1-2a–m. The total PCB data set consists of the result for total PCB congeners for each sample when available (with one exception), and the result for total Aroclors when no total PCB congener data are available. Priority was given to PCB congener data based on the greater specificity and accuracy of the laboratory method for congeners (see Appendix D1.5). The exception is that total Aroclor data were selected to represent total PCBs for Round 2A beach sediment samples because the beach samples were only analyzed for coplanar PCB congeners, which constitute a small fraction of the total PCBs. Congener analyses for the remaining LWG sediment samples included all 209 congeners. Total PCB concentration data for the Study Area are available for 1,184 surface and 1,325 subsurface samples. Most of the PCB data are based on Aroclor analyses (Tables 5.1-1 and 5.1-2). Maps 5.1-30 and 5.1-31 display the locations of surface and subsurface sediment samples analyzed for PCBs and indicate whether PCB congener data, Aroclor data, or both are available.

The distribution of maximum total PCB concentrations at each surface sediment sampling station throughout the Study Area is depicted on Map 5.1-1; concentrations with depth at subsurface stations are depicted on Maps 5.1-2a–m. The complete data set is plotted on scatter plots presented in Figures 5.1-1 and 5.1-2. Scatter plots for total PCB congeners and Aroclors are shown in Figures 5.1-3 through 5.1-6. The summary statistics values shown in Tables 5.1-1 and 5.1-2 for total Aroclors and total PCB congeners indicate overall higher sample concentrations of total PCB when summing congeners. The higher concentrations measured by summing congeners are not a result of differences in laboratory methodology, but rather are attributable to a more targeted sample selection process, i.e., samples selected for PCB congener analysis frequently targeted areas known or suspected to have relatively high PCB contamination.

5.1.5.1 Total PCB Congener and Aroclor Correlation

The relationship between total PCB congener and total Aroclor concentrations is discussed in detail in Appendix D1.5. Both methods represent the total PCB concentrations well, and summed total PCB concentrations are fairly comparable between methods in most cases. The surface sediment correlation (coefficient of determination) between same-sample congener and Aroclor totals was $r^2 = 0.761$, and the subsurface correlation was $r^2 = 0.476$. Plots of these regressions are presented in Appendix D1.5. For all data (sediment, sediment trap, and biota), r^2 was 0.70. PCB totals based on congeners and Aroclors did not correspond well for 11 sediment samples (i.e., an order of magnitude difference between the total congener and total Aroclor results); these are also described in Appendix D1.5. The evaluation of the relationship between PCB congener and PCB Aroclor concentrations in Appendix D1.5 indicates that total Aroclor data may overpredict total PCB congeners in concentrations below ~750 µg/kg total Aroclors and may result in underprediction above 750 µg/kg.

PCB congener data better represent total PCB concentrations than Aroclor data, as the congener method is less affected by “weathering,” non-PCB interferences, and subjective Aroclor identifications. For this reason, in this report, total PCB congener concentrations are given priority over total Aroclor concentrations when total PCB congener data exist for any given sample. Because measured total PCB concentrations are fairly comparable between methods in most cases (especially when measurement error is considered), it is useful to use Aroclor concentrations when no PCB congener data exist. Combining the PCB data in this way provides greater spatial and temporal coverage than using congener data alone.

5.1.5.2 Total PCBs in Surface Sediment

Total PCBs were detected in 940 surface samples within the Study Area (detection frequency of 79 percent), with concentrations ranging from 0.851 J µg/kg to 35,400 µg/kg (Table 5.1-1). Ninety-five percent of the concentrations in surface samples were less than 641 V µg/kg.

Total PCB concentrations in surface sediment varied along the Study Area. With few exceptions, concentrations were generally less than 100 µg/kg throughout the

navigation channel, whereas many areas in the nearshore zones contained concentrations greater than 100 µg/kg in surface samples (Figure 5.1-1). Total PCB concentrations exceeding 1,000 µg/kg in the scatter plots are indicated in red on Map 5.1-1. A prominent peak in the surface data from the western nearshore zone occurred at RM 8.8, where the highest surface concentration in the data set was detected (Station G453). The highest subsurface concentration was also found in this vicinity (Station C455; 30–153 cm bml).

5.1.5.3 Total PCBs in Subsurface Sediment

PCBs were detected in 862 subsurface samples (detection frequency of 65 percent) and ranged from 0.00138 J µg/kg to 36,800 µg/kg (Table 5.1-2).

Similar to surface sediment, total PCB concentrations in the subsurface also varied within the Study Area (Figure 5.1-2; Maps 5.1-2a–m). With few exceptions, concentrations were generally less than 100 µg/kg throughout the navigation channel. Areas with subsurface concentrations greater than 1,000 µg/kg generally occurred in areas with surface concentrations also greater than 1,000 µg/kg (Figures 5.1-1 and 5.1-2).

5.1.5.4 Total PCB Surface and Subsurface Sediment Relationships

Surface and subsurface sediment relationships were examined by calculating ratios of mean total PCB concentrations (i.e., surface/subsurface) for the Study Area; for the east, middle, and west sides of the Study Area; for Multnomah Channel; and for Swan Island Lagoon. Ratios compared all surface and all subsurface sample intervals and excluded non-detected data. The magnitude of surface and subsurface mean total PCB concentrations were then plotted in a histogram to illustrate general trends in surface and subsurface chemical distributions. Statistical summaries for river mile reaches are provided in Table 5.1-3.

Total PCB concentrations were higher in subsurface sediments within the Study Area as a whole (left side of Figure 5.1-33),[11] with localized exceptions. Where mean surface sediment total PCB concentrations are greater than mean subsurface concentrations the magnitude of difference is generally low (less than twice the mean subsurface concentration), with the exception of the navigation channel at RM 11 to 11.8. Higher surface sediment concentrations on the east side the navigation channel between RM 11 and 11.8 are also evident in the core plots (see Map 5.1-2m), indicating a probable ongoing source.

5.1.5.5 Patterns and Trends of PCBs in Sediment

This subsection includes a general description of the distribution of PCB homologs and Aroclors in sediment to provide information that may be used to infer the presence of

[11] Note that the magnitude of mean surface/mean subsurface concentration ratios above "0" in Figure 5.1-33 indicate higher mean surface sediment concentrations while those below "0" reflect higher mean subsurface sediment concentrations. In situations where only surface samples were analyzed (i.e., RM 2–3 west bank, and RM 10–11 west bank), a ratio could not be calculated and is indicated by an asterisk on the figure.

different sources and PCB transport within the Study Area. Aroclor distributions are compared to homolog distributions to evaluate the Aroclor identifications made by the laboratories.

PCB homologs are congener groups based on chlorination level (i.e., the number of chlorine atoms [1–10] bonded to the biphenyl molecule). All of the PCB congeners within each homolog group are isomers. Homolog groups are identified as monochlorobiphenyl (one chlorine atom [$C_{12}H_9C_l$]; monoCB) through decachlorobiphenyl (10 chlorine atoms [$C_{12}Cl_{10}$]; decaCB). Examples of the PCB congener content of Aroclors has been reported by several authors (e.g., Erickson 1997, Frame et al. 1996) and was used to present the PCB homolog content of Aroclors in Figure 5.1-34. Identification of PCB Aroclors at the analytical laboratory can be subjective if the PCB pattern in the sample does not closely reflect the Aroclor standards. This is frequently the case in environmental samples as a result of fate and transport processes, the presence of more than one Aroclor in a sample, and chromatographic interference. Differing sorption, solution, and volatilization rates for different congeners and degradation processes can lead to weathering of Aroclors in the natural environment. Varying degrees of weathering were observed in the sediment samples, with some samples exhibiting what appeared to be weathered Aroclor patterns and other samples exhibiting Aroclor patterns that closely resembled Aroclor standards. Complex mixtures of two or more Aroclors were also observed in many sediment samples. Additional discussion about weathering and comparison of PCB congener and Aroclor totals is provided in Appendix D1.5.

PCB homolog group sums were calculated by summing the constituent congener concentrations, following the summation rules for the RI data.[12] PCB homolog and Aroclor compositions for samples within the Study Area are presented as bar charts in Figures 5.1-35 through 5.1-38. The bar charts show the percent composition of individual PCB homologs and Aroclors for each sample. In this data presentation, non-detected (U-qualified) homologs and Aroclors were assigned a value of zero; therefore, they do not appear on the bar charts. The figures are organized to show the east zone, navigation channel, and west zone (relative to the top of the navigation channel boundary). The dominant homolog (i.e., the homolog group detected at the highest concentration) at each sampling location is displayed on Maps 5.1-32 and 5.1-33, and the dominant Aroclor at each sampling location (i.e., the Aroclor detected at the highest concentration) is shown on Maps 5.1-34 and 5.1-35. Subsurface homolog and Aroclor patterns are shown only for the depth interval with the highest PCB concentration at each location. The PCB composition at other depths may differ from that at the depth of maximum concentration.

[12] Calculated totals for each homolog are the sum of all detected constituent congeners. Non-detected congeners in a homolog group are assigned a value of zero. If all congeners in a homolog group were not detected, then the homolog total is assigned the highest detection limit among the constituent congeners and qualified with a U. For the bar chart presentations, U-qualified homologs and Aroclors are assigned a value of zero and, therefore, do not appear.

Spatial variations in apparent PCB patterns are evident throughout the Study Area, and areas of high PCB concentrations often exhibit homolog patterns that appear distinct from surrounding areas of lower PCB concentrations. These variances in the relative abundances of the homolog groups potentially reflect the differences in the sources of the PCBs and the transport and weathering processes affecting the Aroclors. Overall, the tetrachlorobiphenyl (tetraCB), pentachlorobiphenyl (pentaCB), hexachlorobiphenyl (hexaCB), and heptachlorobiphenyl (heptaCB) homolog groups are predominant in the Study Area, with localized exceptions (Figures 5.1-35a–c and 5.1-36a–c). In the eastern nearshore zone, the overall chlorination level of PCBs in the surface and subsurface sediments tends to appear higher upriver in the Study Area and lower downstream. Between RM 6.7 and 11.3 in the eastern nearshore zone, the highest concentration samples are chiefly composed of hexaCBs and heptaCBs. The chlorination levels appear lower at several locations of higher PCB concentrations, from about RM 2 to 4 east, with a dominance of trichlorobiphenyls (triCBs), tetraCBs, and pentaCBs. Homolog patterns in areas of high PCB concentration appear to be more variable in the western nearshore zone.

The PCB homolog patterns in subsurface sediment appear generally similar to surface sediment patterns at the 37 locations where PCB congener data are available for both surface and subsurface sediment. However, PCB homolog patterns in subsurface sediment appear different from surface sediment at RM 2.8 (sample location LW3-G609/LW3-C609), RM 3.7 (sample location LW3-C093), and RM 8.4 (sample location LW3-C393) in the eastern nearshore zone; RM 10.1 (sample location LW3-G747) in the navigation channel; and at RM 5.1 (sample location LW3-G184), RM 7.7 (sample location LW3-G401), and RM 9.6 (sample location LW3-G738) in the western nearshore zone. As noted previously, patterns displayed by the stacked bar charts may or may not be statistically significant and indicative of different potential sources. Detailed forensic analyses would be required to distinguish significant patterns in the data and potential source contributions.

Aroclors 1248, 1254, and 1260 were identified throughout the Study Area (Figures 5.1-37a–h and 5.1-38a–d). Aroclor 1221 was identified locally in surface sediments, and Aroclors 1242 and 1268 were identified locally in both surface and subsurface sediments; however, these Aroclors were not widespread (Figures 5.1-37a–h and 5.1-38a–d). Within the Study Area, Aroclor 1016 was identified in one surface sediment sample, and Aroclors 1232 and 1262 were identified in one subsurface sediment sample each. Aroclor patterns in the subsurface sediments were also generally similar to the surface sediment patterns.

5.1.5.5.1 Comparison of PCB Homolog Patterns with Reported Aroclors

For areas with total PCB concentrations greater than 1,000 µg/kg and infrequently reported Aroclors, a comparison of identified Aroclors to the PCB homolog groups was made by comparing the Aroclor homolog profiles as presented in Figure 5.1-34. The PCB homolog data, where available, generally supported the Aroclor identifications of Aroclors 1242, 1248, 1254 and 1260, with two notable exceptions.

Near RM 8.8 (western nearshore zone) Aroclors 1242 and 1248 dominated the PCBs in surface and subsurface sediment, and concentrations greater than 1,000 µg/kg of Aroclor 1260 in surface sediment and Aroclor 1254 in subsurface sediment were also reported. Aroclors 1242 and 1248 are generally difficult to differentiate on a gas chromatogram and may be reported differently by different laboratories or analysts. The PCB homolog distribution in this area supports the identification of Aroclor 1242, although it does not definitively rule out the additional presence of Aroclor 1248 (Figures 5.1-37a–h and 5.1-38a–d). TriCBs were notably abundant in the sediment, often in a pattern that resembled Aroclor 1242 more than Aroclor 1248. The presence of dichlorobiphenyls (diCBs) further supports the identification of Aroclor 1242. Overall, the homolog patterns were very similar for the two samples in this area with the highest concentration (LW2-G453 and LW2-GBT028), even though the Aroclors identified in these samples were different. The reporting of two Aroclors in this area by the laboratories appears to reflect the difficulty of Aroclor identification rather than a difference in the PCBs present in the samples.

In samples with total PCB concentrations greater than 1,000 µg/kg at RM 2.1–2.5 (eastern nearshore zone), Aroclors 1248, 1254, and 1260 were identified in the surface and subsurface sediments in this area, and Aroclor 1242 was identified in five of the subsurface sediment samples. The PCB homolog distribution in this area was consistent with the identification of Aroclor 1248 as the predominant Aroclor in the surface sediments. However, for some subsurface samples (e.g. LW2-C015-B) the homolog pattern was not consistent with the reported Aroclors. Aroclors 1254 and 1260 were reported as dominant in these samples, but the homolog profiles for these samples resemble the profile for Aroclor 1242 or 1248, with a potential contribution of Aroclors 1254 or 1260. The lack of agreement between the homolog profiles and Aroclor identifications suggests the influence of weathering effects.

Aroclors not commonly reported in the Study Area (i.e., identified in fewer than 100 sediment samples or Aroclors other than 1242, 1248, 1254, and 1260) were also evaluated using PCB homolog data as described in the following paragraphs.

Aroclor 1221 was reported in surface sediment in the eastern nearshore zone and eastern edge of the navigation channel between RM 9.3 and 10 at concentrations up to 109 µg/kg (Station G472). However, the PCB homolog pattern is not consistent with Aroclor 1221 in the two samples from this area that were analyzed for PCB congeners. MonoCBs and diCBs are the dominant homologs in Aroclor 1221 (Erickson 1997; Figure 5.1-34), but tetra- through heptaCBs dominated the homolog profiles in this area. The same homolog profile was also present at adjacent location BT031, which was sampled at a later date and analyzed by a different lab, and for which Aroclor 1221 was not identified. Based on the PCB homolog patterns, and the fact that Aroclor 1221 is rarely reported in environmental samples, the identification of Aroclor 1221 in this area appears questionable.

Aroclor 1221 was also identified in surface sediment at four isolated locations: in the eastern nearshore zone near RM 11, in the western edge of the navigation channel near RM 10.3, in the western nearshore zone at RM 7, and in the navigation channel near RM 8. PCB congeners were analyzed at all of these stations, and in all four cases, monoCBs and diCBs were not reported at sufficient levels to support the identification of Aroclor 1221. A focused review of Aroclor chromatograms confirmed the laboratory identifications, although some differences were noted in the PCB patterns in the samples relative to the standards. The differences in PCB patterns identified by the two methods may be the result of sample heterogeneity or another unidentified cause.

Aroclor 1268 was reported in surface and subsurface sediment in the eastern nearshore zone near RM 5.6 at concentrations up to 474 J µg/kg. PCB congener profiles generally confirm the presence of Aroclor 1268. Nonachlorobiphenyls (nonaCBs) and decaCBs are present in Aroclor 1268 (Figure 5.1-34) and were more abundant in both surface sediment locations and in one of the two subsurface sediment locations analyzed in this area than in areas without Aroclor 1268 detections.

Aroclor 1268 was also identified in isolated locations in several other areas. The Aroclor 1268 identifications were confirmed by the PCB homolog profile in surface sediments in the eastern nearshore zone at RM 3.7, off the mouth of the International Terminals Slip. PCB homolog profiles did not confirm the presence of Aroclor 1268 reported in surface and subsurface samples in the eastern nearshore zone near RM 4, the surface sediments in the eastern nearshore near RM 7.3, or in the subsurface sediments in the western nearshore zone near RM 7.4. Aroclor 1268 could not be evaluated at other locations because either no PCB homolog data were available, or Aroclor 1268 constituted a relatively small fraction of the Aroclor total.

Highly chlorinated PCBs were present at one location where Aroclor 1268 was not identified. At one subsurface location (C093-B) in the International Terminals Slip, nonaCBs and decaCB together accounted for approximately 25 percent of the PCB congener total, suggesting the presence of Aroclor 1268. A review of the Aroclor chromatogram for this sample confirmed the laboratories' Aroclor identifications. Again, the differences in PCB patterns identified by the two methods may be the result of sample heterogeneity or another unidentified cause.

Aroclors 1232 and 1016 were each identified in only one sample. Aroclor 1232 was identified in subsurface sediment at location PSY36C in the navigation channel near RM 8, and Aroclor 1016 was identified in surface sediment sample PP01M105 near the east bank of Swan Island Lagoon at approximately RM 8. The unique Aroclor constituted a small fraction of the total PCB Aroclors at both locations. PCB homolog data are not available at either location to corroborate the identifications; PCB homolog data at locations near these samples do not show evidence of Aroclors 1232 or 1016. The identity of these Aroclors is questionable.

Aroclor 1262 was also identified in only one sample, a subsurface sediment sample in the navigation channel at location LW3-C760 near RM 10.5. No PCB homolog data are available at or near this location and the Aroclor identification could not be confirmed.

5.1.6 Total PCDD/Fs and TCDD TEQ in Sediment

This section discusses the nature and extent of PCDD/Fs in surface and subsurface sediment samples collected within the Study Area. Also discussed is the distribution of TCDD TEQ concentrations and observed trends in the relative abundance of PCDD/F homologs in surface and subsurface samples. TCDD toxicity with respect to 2,3,7,8-TCDD was calculated from concentrations of PCDD/F congeners designated by the WHO as similar in mechanism of toxicity to 2,3,7,8-TCDD (Van den Berg et al. 2006). Each WHO-designated congener is assigned a specific TEF indicating its degree of toxicity compared to 2,3,7,8-TCDD, which is given a reference value of 1.

The distribution of total PCDD/Fs and TCDD TEQ concentrations at each surface sampling station throughout the Study Area is depicted in Maps 5.1-3 and 5.1-5, respectively; concentrations with depth at subsurface stations are depicted in Maps 5.1-4a–m and 5.1-6a–m, respectively. The complete data set is plotted on scatter plots presented in Figures 5.1-7 through 5.1-10.

The selection of sediment samples for PCDD/F analysis is typically based on a highly biased approach at locations near known or suspected sources. As a result, there are relatively fewer data points for these analytes and the resulting TCDD TEQ data in the RI sediment database than for other chemicals (for example, the PCDD/F data set is approximately one-fifth the size of the PCBs and DDx data sets). This is particularly true in areas away from suspected sources, such as the navigation channel. The existing PCDD/F data are sufficient for RI purposes; however, as will be pointed out in this section and later in Section 10, the fewer number of data points for PCDD/Fs in some areas limits the level of detail to which the extent of chemical distribution may be resolved, and introduces the need for caution in interpreting the surface to subsurface trends shown by the histograms (Section 5.1.6.5) and in making conclusions regarding the spatial patterns of the composition of PCDD/Fs in sediment (Section 5.1.6.6).

5.1.6.1 Total PCDD/Fs in Surface Sediment

Total PCDD/Fs were detected in all 216 surface sediment samples in which this suite of chemicals was analyzed. Surface concentrations ranged from 2.48 to 264,000 pg/g (Table 5.1-1). Ninety-five percent of the surface data was below 5,620 JV pg/g.

The data show concentrations greater than 2,000 pg/g (indicated in red on Map 5.1-3) total PCDD/Fs at several locations along the eastern and western nearshore zones. Limited surface PCDD/F data are available for the navigation channel and spatial resolution is somewhat limited. However, of the channel samples that were analyzed, most concentrations were less than 500 pg/g, and a pattern is evident of relatively high concentrations in nearshore areas compared with markedly lower levels in the adjacent channel areas.

5.1.6.2 Total PCDD/Fs in Subsurface Sediment

Of the 241 subsurface samples analyzed, total PCDD/Fs were detected in all but one of the samples. Concentrations ranged from 0.0578 J pg/g to 218,000 pg/g (Table 5.1-2).

PCDD/F concentrations greater than 2,000 pg/g in subsurface sediment were generally found at the same surface locations with PCDD/F concentrations greater than 2,000 pg/g along the eastern and western nearshore zones (Maps 5.1-4a–m).

Core sample C455 (30–152 cm bml) collected along the western shoreline at RM 8.8 contained the highest subsurface concentration of total PCDD/Fs in the data set (Table 5.1-2). This station also contained the maximum total PCBs and total chlordanes concentrations in the Study Area.

Limited subsurface PCDD/F data are available for the navigation channel; of the samples that were analyzed, most concentrations were less than 100 pg/g (Maps 5.1-4a-m).

5.1.6.3 TCDD TEQ in Surface Sediment

A total of 217 surface samples were selected for analysis of WHO-designated PCDD/Fs, with a frequency of detection of 100 percent. The resulting calculated TCDD TEQs show a wide range of values, from 0.00803 J pg/g to 14,100 J pg/g in surface sediment (Table 5.1-1). Ninety-five percent of the surface data were below 43.2 JV pg/g.

The data show that TCDD TEQ values vary spatially along the length of the Study Area (Figure 5.1-9). In general, values were higher in the western nearshore zone than in the eastern nearshore and navigation channel zones. The most significant peak in the data in the western nearshore occurred between approximately RM 6.8 and 7.3, where data points are relatively dense in comparison to the rest of the Study Area.

Limited data for WHO-designated PCDD/Fs are available for sediments in the navigation channel (Map 5.1-5). TCDD TEQ surface values within the channel were relatively low, with the exception of two samples with relatively elevated concentrations along the eastern edge of the navigation channel between RM 6.6 and 6.7.

5.1.6.4 TCDD TEQ in Subsurface Sediment

WHO-designated PCDD/Fs were detected in approximately 93 percent of the 245 subsurface samples in which they were analyzed. The resulting calculated TEQs show a wide range of values, from an estimated 0.000262 J pg/g to an estimated 7,480 J pg/g in subsurface sediment (Table 5.1-2; Figure 5.1-10).

The maximum subsurface TCDD TEQ value was found at Station SD092 (0–90 cm vertically composited sample) at RM 7.2W (Map 5.1-6h).

Within the navigation channel, TCDD TEQ values over 10 pg/g (indicated in red in Maps 5.1-6a-m) were calculated only at one subsurface station near the western channel boundary at RM 6.6 (33.3 J pg/g in the interval from 132 to 243 cm bml at Station C314; Figure 5.1-10).

5.1.6.5 PCDD/F Surface and Subsurface Sediment Relationships

The magnitude of the mean surface and subsurface sediment concentrations for PCDD/Fs is shown on Figure 5.1-39. Summary statistics are presented in Table 5.1-4. The methods used to develop these presentations are described in Section 5.1.5.4. It should be noted that fewer data points are available for PCDD/Fs than for the other bounding chemical groups (PCBs, DDx, and PAHs). This relatively small sample size combined with the effect on the mean of one or two elevated values in a given subarea limits the interpretability of Figure 5.1-39; the actual mapped data patterns shown in Maps 5.1-4a-m are also discussed here.

The surface/subsurface mean ratios show that total PCDD/F concentrations were slightly higher in surface sediments within the Study Area overall (left side of Figure 5.1-39), but the magnitude of the ratios are small, indicating that the average surface and subsurface concentrations were comparable across most of the Study Area. This pattern is supported by Maps 5.1-4a–m, which show that most areas lack strong or consistent vertical concentration gradients. Some exceptions to this include the area under and just upstream of the Railroad Bridge at RM 6.9, where surface layers show higher concentrations than at depth (Map 5.1-4g) and the northwest corner of Willbridge Terminal where higher levels are evident at depth (Map 5.1-4h). This suggests a current source or sources at the former location and an historical source or sources at the latter. Elsewhere in the Study Area, significant changes in the level of PCDD/F inputs over time are generally not indicated.

5.1.6.6 Patterns and Trends of PCDD/Fs in Sediment

PCDD/F homolog compositions for samples within the Study Area are presented as bar charts in Figures 5.1-40a–c and 5.1-41a–c. The bar charts show the percent composition of the individual PCDD/F homologs for each sample. The figures are organized to show the east zone, navigation channel, and west zone (relative to the navigation channel boundary). Subsurface homolog patterns are shown only for the depth interval with the highest total PCDD/F concentration at each location. The PCDD/F composition at other depths may differ from that at the depth of maximum concentration.

OCDD is generally the dominant homolog (>50 percent of the total concentration) present in surface and subsurface sediments throughout the Study Area, with HpCDDs present to a significant but lesser degree. The other homolog groups generally constitute 20 percent or less of the total concentration. Exceptions where PCDD/F homolog distributions appear to vary significantly (possibly reflecting isolated areas of differing sources or weathering patterns) are clustered throughout the Study Area, with the largest clusters occurring along the western nearshore area around RM 6.8 and near

RM 7.4, where data points are relatively dense in comparison to the rest of the Study Area.

The apparent variations in PCDD/F homolog compositions do not always reflect variations in total PCDD/F concentrations. Near RM 6.7E and from RM 6.8 to 7.4W, samples with high total PCDD/F concentrations are marked by a high proportion of furans relative to other areas. However, in general, samples with high PCDD/F concentrations have homolog profiles that match the prevailing pattern of OCDD and HpCDD dominance. Also, the surface and subsurface homolog distributions did not appear to vary greatly for any given location.

Finally, and as noted previously, definitive statements about spatial patterns in PCDD/F composition cannot be made based on the stacked bar charts, particularly for PCDD/Fs, which were analyzed based on a highly biased sampling design; this was not a primary goal of the RI sampling program. Detailed forensic analyses would be required to determine the existence of statistically significant patterns in the data.

5.1.7 Total DDx in Sediment

The distribution of concentrations of total DDx and its constituent compounds DDD, DDT, DDE in the Study Area sediment is summarized in this section. Observed trends in DDx isomers in surface and subsurface samples are also discussed. Frequencies of detection of total DDx were approximately 89 percent for surface samples and 81 percent for subsurface samples.

The distribution of total DDx concentrations at each surface sampling station throughout the Study Area is depicted in Map 5.1-7; concentrations with depth at subsurface stations are depicted in Maps 5.1-8a–m. The complete data set is plotted on scatter plots presented in Figures 5.1-11 and 5.1-12. The individual total DDT, DDD, and DDE concentrations (totals of the 2,4’- and 4,4’-isomers) are depicted in Maps D1.1-3 through D1.1-8 in Appendix D1.1.

Some of the results of the component isomers that were summed in the total DDx concentrations were N-qualified (Section 2.1.3.2). Additionally, some of the DDx isomer data are uncertain and potentially biased high because of the analytical interference from the presence of PCB congeners in the sample. The N qualifier indicates that the quantity is estimated because there is only presumptive evidence that the chemical compound exists. When an individual isomer result is N-qualified, the N qualifier is carried forward to the reported summed total. For individual isomers, the percentages of N-qualified sediment data range from zero (2,4’-DDE) to approximately 30 percent (2,4’-DDD) of both the surface and subsurface data. N-qualified total DDx concentrations range from 0.051 NJ µg/kg to 84,900 NJ µg/kg in surface sediment and from 0.054 NJ µg/kg to 51,800 NJ µg/kg in subsurface sediment.

5.1.7.1 Total DDx in Surface Sediment

The results of the 1,210 surface samples that were analyzed for both the 2,4'- and 4,4'-isomers of the DDx compounds are depicted on Map 5.1-7. Detected concentrations ranged from an estimated 0.051 NJ µg/kg to an estimated 84,900 A µg/kg in surface sediment (Table 5.1-1; Figure 5.1-11). Ninety-five percent of the samples were less than 470 JV µg/kg.

Areas of total DDx concentrations greater than 100 µg/kg occurred at several locations scattered along the nearshore zones and channel margins (Figure 5.1-11). The most prominent area of total DDx concentrations greater than 100 µg/kg occurred along the western shoreline between RM 6 and 7.5. Concentrations above 10,000 µg/kg (shown in red on Map 5.1-7) were found only in surface sediment near the western shore at RM 7.5. The maximum concentration was found at Station OSS002 in this vicinity. Upstream along the western shoreline at RM 8.8, DDx was detected in a single sample at a concentration greater than 1,000 µg/kg.

With the exception of four samples, samples collected within the navigation channel were less than 100 µg/kg. Concentrations greater than 100 µg/kg were located at RM 5.6, 6.5, and 11.3 (Figure 5.1-11).

The DDD, DDE, and DDT components show generally similar patterns of distribution, though relative concentrations vary (Maps D1.1-3, D1.1-5, and D1.1-7).

5.1.7.2 Total DDx in Subsurface Sediment

Of the 1,291 subsurface samples analyzed for total DDx, detected concentrations ranged from an estimated 0.058 J µg/kg to an estimated 3,643,000 A µg/kg in subsurface sediment (Table 5.1-2).

The maximum subsurface concentration was found in the interval 323 to 384 cm bml at Station WB-24 at RM 7.2E. Of the samples collected within the navigation channel, the cores with concentrations greater than 100 µg/kg corresponded to areas of similar concentrations within the nearshore (Figure 5.1-12; Maps 5.1-8a–m).

5.1.7.3 Total DDx Surface and Subsurface Sediment Relationships

The magnitude of total DDx mean surface and subsurface sediment concentrations is shown on Figure 5.1-42. Summary statistics are presented in Table 5.1-5. The methods used to develop these presentations are described in Section 5.1.5.4.

With the exception of one area (RM 11-11.8), the mean subsurface concentrations of DDx are higher than the surface concentrations throughout the Study Area (Figure 5.1-42). The magnitude of the ratios are generally low, mostly around 5 or less; however, at RM 7-8 W, the mean subsurface levels greatly exceed the surface mean, indicating a large historical source or sources that has been markedly reduced over time. The only portions of the Study Area where surface sediment total DDx concentrations are higher than subsurface sediments are from RM 11–11.8E and in the navigation

channel, possibly suggesting a current source or sources, but the magnitude of these ratios is relatively low. The surface/subsurface trends revealed by Figure 5.1-42 are supported by the data plotted in Maps 5.1-8a-m.

5.1.7.4 Patterns and Trends of Total DDx in Sediment

DDx patterns appeared to vary widely across the Study Area, as shown in Figures 5.1-43a–h (surface sediments) and 5.1-44a–d (subsurface sediments). Selected potential trends are summarized below. The bar charts in these figures include samples that may lack results for the 2,4'-isomers of a DDx compound if these were not analyzed (see Appendix D1.6). The DDx patterns are incomplete for these samples. The discussion of subsurface sediment trends is based on the evaluation of DDx patterns only for the depth interval with the highest concentration at each location, presented in Figures 5.1-44a–d. The DDx composition at other depths may differ from that at the depth of maximum concentration.

As noted above, total DDx concentrations exceeded 10,000 µg/kg in one area located near RM 7 in the western nearshore zone. The main constituent of the surface sediments in this area was the 4,4'-isomer of DDT (Figure 5.1-43g), while the 4,4'-isomers of DDT and DDD were the main constituents of the subsurface sediments (Figure 5.1-44d). This pattern may indicate degradation of DDT to DDD in deep anoxic sediments.

The relative concentrations of the DDx isomers appear highly variable from station to station across the Study Area in both surface and subsurface sediment samples. However, a few general trends were observed:

- The 4,4'-isomer concentrations were greater than those for the 2,4'-isomers of the DDx constituents overall. In some locations, the 2,4'-isomers were more abundant than their 4,4'- counterparts, particularly 2,4'-DDD and, less frequently, 2,4'-DDT. 2,4'-DDE was rarely detected and was dominant only in samples with relatively low concentrations.
- Overall, samples with the highest concentrations tended to display a dominance of DDT and/or DDD isomers, particularly below RM 8.
- There was a broad potential trend observed in the western nearshore DDx patterns. Both surface and subsurface sediment samples collected upstream of RM 8 generally had relatively low total DDx concentrations (typically between 10 and 100 µg/kg) and a large DDE component, whereas samples collected between RM 7.5 and 6.9 had relatively higher total DDx concentrations (generally higher than 100 µg/kg) and were dominated by DDT, and samples downstream of RM 6.9 had relatively lower total DDx concentrations (generally less than 100 µg/kg) and a larger apparent DDD component. These upstream/downstream trends were also evident in the navigation channel, which generally paralleled the western nearshore trends from RM 12 to about RM 4 (except where embayments such as Willbridge are crossed). In contrast, the

eastern nearshore patterns were more like those of the western nearshore zone above RM 8.

5.1.8 Total PAHs in Sediment

The distribution and composition of total PAH concentrations in Study Area sediment are summarized in this section. The data set of total PAH concentrations includes 1,603 surface samples and 1,545 subsurface samples. Frequencies of detection of PAH compounds were high, approximately 99 percent in surface samples and 95 percent in subsurface samples.

Map 5.1-9 shows the distribution of total PAH concentrations at each surface sampling station throughout the Study Area; concentrations with depth at subsurface stations are depicted in Maps 5.1-10a–m. The complete data set is plotted on scatter plots presented in Figures 5.1-13 and 5.1-14.

5.1.8.1 Total PAHs in Surface Sediment

The concentration range of total PAHs in surface sediment varied widely, from 3.3 J µg/kg to 7,260,000 µg/kg (Table 5.1-1). Ninety-five percent of the 1,603 surface samples were less than 66,600 JV µg/kg.

The data, particularly above RM 6.5 where samples are more abundant, showed variable concentrations throughout the Study Area. Except for several areas of relatively higher concentrations, total PAH levels were generally 1,000 µg/kg or less in channel and nearshore zones of the main stem of the river (i.e., outside Swan Island Lagoon) in the upper portion of the Study Area between RM 7 and 11.8 (Figure 5.1-13; Map 5.1-9).

Scattered areas of concentrations greater than 1,000 µg/kg were found throughout the Study Area, but the highest concentrations (>30,000 µg/kg; indicated in red on Map 5.1-9) were most commonly found in the eastern and western nearshore zones between approximately RM 4 and 7.5 (Figure 5.1-13).

PAH concentrations above 30,000 µg/kg were also found in surface sediment in the navigation channel adjacent to the RM 5.2–6.8 reach.

5.1.8.2 Total PAHs in Subsurface Sediment

Of the 1,545 subsurface samples analyzed for total PAHs, the concentration range varied widely, from 0.15 J µg/kg to 53,300,000 µg/kg in subsurface sediment (Table 5.1-2).

Similar to surface sediment, scattered areas of concentrations greater than 1,000 µg/kg in subsurface sediment were found throughout the Study Area, and concentrations greater than 30,000 µg/kg were most commonly found in the eastern and western nearshore zones between approximately RM 3 and 7.5 (Figure 5.1-14; Maps 5.1-10a–m).

PAH concentrations above 30,000 µg/kg were also found in subsurface sediment in the navigation channel adjacent to and downstream from RM 6.5.

5.1.8.3 Total PAHs Surface and Subsurface Sediment Relationships

The magnitude of mean total PAH surface/subsurface sediment concentrations is shown on Figure 5.1-45. Summary statistics are presented in Table 5.1-6. The methods used to develop these presentations are described in Section 5.1.5.4.

The surface/subsurface mean ratios show that total PAH concentrations were slightly higher in subsurface sediments within the Study Area as a whole (left side of Figure 5.1-45).

In areas where mean subsurface total PAH concentrations were greater than mean surface total PAH concentrations, the highest magnitude was at RM 6-7 in the navigation channel followed by RM 11–11.8 E. In areas where mean surface sediment total PAH concentrations were greater than subsurface concentrations the highest magnitude was at RM 5–6 in the navigation channel. Inspection of Map 5.1-10g indicates that the high subsurface mean at RM 6–7W is driven by some high PAH concentrations (>30,000 µg/kg) in core samples collected at the channel edge off of the Gasco early action area. The relatively high surface/subsurface mean ratio in the channel from RM 5–6 is driven by several high concentration (>30,000 µg/kg) surface-only samples in the channel.

5.1.8.4 Patterns and Trends of Total PAHs in Sediment

The distribution of detected PAHs at each location is presented in Figures 5.1-46a–j (surface sediment) and Figures 5.1-47a–f (subsurface sediment). In order to simplify the bar charts, PAHs were grouped according to the number of fused aromatic rings in the PAH. A list of individual PAHs included in the sum for each of these groups is provided in Table 5.1-7. Only PAHs analyzed for LWG samples are summed. Of these PAHs, two-ring PAHs include only naphthalenes (i.e., naphthalene and 2-methylnaphthalene). LPAHs include PAHs with two or three rings (green and yellow segments), and HPAHs include PAHs with four to six rings (purple, red, and blue segments). Only the depth interval that contained the highest total PAH concentration is shown in Figures 5.1-47a–f (i.e., the subsurface charts represent a variety of depths based on the interval of the maximum concentration) and evaluated below.

Surface sediments within the Study Area are generally dominated by HPAHs, primarily four-ring PAHs, with localized exceptions. Five-ring PAHs are the second most abundant HPAH, followed by six-ring PAHs. Three-ring PAHs are the principal LPAH in surface sediments, with two-ring PAHs generally being a minor component of the surface sediment PAH profile. Surface sediments from the western nearshore zone appeared to exhibit higher proportions of LPAHs than sediments from the eastern nearshore zone and the navigation channel, but follow the general trend of HPAH dominance. Some areas of high total PAH concentration have PAH profiles that appear to differ from the prevailing trend of HPAH dominance, the most notable between

RM 6.2 and 6.9 in the western nearshore zone. In this area increased contributions from two and three-ring PAHs, and a corresponding reduced six-ring PAH abundance is also observed. Similar high PAH profiles are observed between RM 6.4 and 7.2 in the eastern nearshore zone and between RM 5.6 and 7.4 in the navigation channel.

Subsurface sediments appear to have greater contributions from two- and three-ring PAHs than the surface sediments, but generally exhibit similar PAH profiles to the surface sediments.

The proportions of individual PAH compounds varied throughout the Study Area, reflecting PAH contributions from multiple types of hydrocarbon sources, as well as weathering and degradation. Hydrocarbon source types include pyrogenic (e.g., tars and creosote), petrogenic (e.g., fresh oil), and a variety of combustion processes and natural biological production processes. PAHs characteristic of these source types include alkylated PAH compounds, which are especially useful in distinguishing between pyrogenic and petrogenic source types. Alkylated PAH data are available for selected sediment and clam tissue samples and are presented in the SCRA database and summary statistics in Appendix D; however, these data are not discussed in the RI report. Once again, detailed forensic analyses would be required to identify significant differences among the PAH patterns within the sediment data set and to distinguish contributions from potential source types.

5.1.9 Additional Indicator Chemicals in Sediment

This section discusses the occurrence and distribution of nine additional ICs in sediment within the Study Area. The narrative in this section is less comprehensive than the preceding sections, omitting the data set description, surface/subsurface relationships, patterns and trends, and referring instead to maps, tables, and figures to provide a general picture of the distribution of those chemicals. The nature and extent data for the remaining 21 sediment ICs listed in Table 5.0-2 are presented on all tables, maps, and figures in Appendix D.

5.1.9.1 Bis(2-ethylhexyl)phthalate in Sediment

Concentrations of BEHP were ≤1,500 µg/kg in the majority of samples analyzed (Maps 5.1-11 and 5.1-12a–m, see frequency plot inset; Figures 5.1-15 and 5.1-16). Limited areas with concentrations greater than 1,500 µg/kg were found at several locations within the Study Area. Frequencies of detection were 61 percent for surface samples and 39 percent for subsurface samples. Ninety-five percent of the surface samples were below 2,230 JV µg/kg (Table 5.1-1).

Clusters of concentrations greater than 1,500 µg/kg occurred in the surface data set from the eastern nearshore, in Swan Island Lagoon, and between RM 3.8 and 4.1 in the International Terminals Slip and along the riverfront (Maps 5.1-11 and 5.1-12a–m). The highest surface concentration detected in the Study Area was found at Station G367 at the mouth of Swan Island Lagoon.

Additional isolated occurrences of concentrations greater than 1,500 µg/kg were found. With few exceptions, these concentrations occurred outside the navigation channel, in the eastern and western nearshore zones. The most notable exception is the western side of the navigation channel at RM 10.3, where the highest subsurface concentration in the Study Area was found in the interval of 0–195 cm bml at Station WR-VC-110. A similarly elevated subsurface concentration was detected in the channel at RM 8 near the mouth of Swan Island Lagoon.

5.1.9.2 Total Chlordanes in Sediment

Overall, detected concentrations of total chlordanes were below 5 µg/kg throughout most of the Study Area (Maps 5.1-13 and 5.1-14a–m, see frequency plot inset) and, with few exceptions, were generally lower along the navigation channel (Figures 5.1-17 and 5.1-18; Tables 5.1-1 and 5.1-2). Sediment concentrations greater than 5 µg/kg were detected at several locations throughout the Study Area, but occurred most extensively along the western nearshore zone between approximately RM 6 and 7.4 (Maps 5.1-13 and 5.1-14a–m).

Frequencies of detection were 64 percent and 55 percent, respectively, for surface and subsurface samples. Approximately ninety-five percent of the surface samples were below 12.2 JV µg/kg. The maximum surface concentration was found at Station G355 (RM 7.3W). Another cluster of concentrations greater than 5 µg/kg in surface and subsurface samples occurred at RM 8.8W and at the head of the International Terminals Slip (RM 3.7E; some areas have subsequently been dredged). A sample collected at RM 8.8 contained the highest subsurface concentration of total chlordanes in the Study Area (Station C455 in the interval of 30–152 cm bml). This same core sample contained the maximum concentrations of total PCBs and total PCDD/Fs in the Study Area.

Except for elevated detections from RM 10 to 10.2 and RM 11.2 to 11.6, peaks in the navigation channel were typically located near elevated concentrations in the nearshore.

5.1.9.3 Aldrin and Dieldrin in Sediment

The insecticides, aldrin and dieldrin, have similar chemical structures and are discussed together here because aldrin quickly breaks down into dieldrin in the environment. Detected concentrations of both chemicals were generally less than 1 µg/kg (Maps 5.1-15 through 5.1-18, see frequency plot inset), though higher concentrations occurred at several locations scattered along the nearshore zones and navigation channel (Figures 5.1-19 through 5.1-22).

Frequencies of detection for aldrin were low, 23 percent for surface samples and only 13 percent for subsurface samples (Tables 5.1-1 and 5.1-2). Ninety-five percent of the surface samples were below 10.6 JV µg/kg. The most prominent area of detected aldrin concentrations was at RM 7.4, where the maximum surface (Station G355) and subsurface (Station C356, 136–256 cm bml) concentrations were detected.

Concentrations greater than 1 µg/kg extended downstream along the western shoreline to approximately RM 5.2 (Maps 5.1-15 and 5.1-16a–m).

Two other prominent aldrin peaks are shown by the data, one indicated by concentrations exceeding 100 µg/kg in surface and subsurface samples at RM 8.8, the other indicated by concentrations approaching 100 µg/kg in the subsurface from approximately RM 6 to 6.5 (Figures 5.1-19 and 5.1-20). Some of these data are N-qualified.

Aldrin concentrations above 1 µg/kg were detected at several locations within the navigation channel. The maximum concentration of aldrin within the navigation channel occurred in the interval of 30 to 137 cm-bml at core Station C206 (approximately RM 5.6 near the east bank). This same interval also recorded high concentrations for PAHs, arsenic, chromium, copper, lead, mercury, zinc, and PCBs. Aldrin concentrations above 1 µg/kg were also detected in the navigation channel in the RM 11.2–11.5, 10–10.2, and 8.6–8.9 reaches.

Frequencies of detection for dieldrin were even lower, 21 percent for surface samples and only 7 percent for subsurface samples. Ninety-five percent of the surface samples were below 5.93 JV µg/kg. Dieldrin concentrations above 1 µg/kg were detected in subsurface samples collected along the eastern nearshore between RM 3.8 and 4, where the highest subsurface concentration in the Study Area was found at Station C092 (30–152 cm bml; N-qualified) at the head of the International Terminals Slip (Maps 5.1-17 and 5.1-18a–m). Dieldrin concentrations above 1 µg/kg were also detected in several cores collected in the navigation channel downstream from the Broadway Bridge. Along the western shoreline, dieldrin concentrations above 1 µg/kg were detected in the RM 8.5–8.8 reach, at RM 7.4, and between RM 5.7 and 6.6 (Figures 5.1-21 and 5.1-22). The maximum surface sediment concentration of dieldrin in the Study Area was found at Station G453 located in the western nearshore zone at RM 8.8.

5.1.9.4 Arsenic in Sediment

Overall, detected concentrations of arsenic were below 10 mg/kg throughout most of the Study Area (Maps 5.1-19 and 5.1-20a–m, see frequency plot inset) and were generally below 5 mg/kg in most of the navigation channel. Frequencies of detection for arsenic were 92 percent for surface samples and 96 percent for subsurface samples (Tables 5.1-1 and 5.1-2). Ninety-five percent of the surface samples were below 8.98 V mg/kg. Clusters of concentrations greater than 10 mg/kg were identified in several areas (Maps 5.1-19 and 5.1-20a–m; Figures 5.1-23 and 5.1-24).

The highest concentration among surface sediment samples was detected at RM 2.3 along the eastern bank (Station RB08). The highest subsurface concentration was found in the interval of 150–236 cm bml at Station C708 near the mouth of Swan Island Lagoon.

5.1.9.5 Chromium in Sediment

Chromium concentrations were relatively low (<50 mg/kg) throughout the majority of the Study Area (Maps 5.1-21 and 5.1-22a–m, see frequency plot inset), including the navigation channel. Detection frequencies were nearly 100 percent for both surface and subsurface samples (Tables 5.1-1 and 5.1-2). Ninety-five percent of the surface samples were below 55.8 V mg/kg.

Clusters of chromium concentrations greater than 50 mg/kg were identified in several areas along the eastern and western shorelines (Maps 5.1-21 and 5.1-22a–m; Figures 5.1-25 and 5.1-26). The maximum chromium surface concentration was found at Station RB06 in the RM 2.2 vicinity. The maximum subsurface concentration was found at Station HA-42 (46–61 cm bml) at RM 8.9.

5.1.9.6 Copper in Sediment

Copper data showed concentrations greater than 60 mg/kg (Maps 5.1-23 and 5.1-24a–m, see frequency plot inset) at many locations along the eastern and western nearshore zones, but few in the navigation channel. Detection frequencies were nearly 100 percent for both surface and subsurface samples (Figures 5.1-27 and 5.1-28; Tables 5.1-1 and 5.1-2). Ninety-five percent of the surface samples were below 172 V mg/kg.

As shown in Maps 5.1-23 and 5.1-24a–m, copper was greater than 60 mg/kg in isolated groupings at several locations along the eastern and western shorelines. A surface sample at RM 11.2 (Station UG01) contained the highest concentration of copper. The maximum subsurface copper concentration was found at Station C384 (30–128 cm bml), at the mouth of Swan Island Lagoon.

5.1.9.7 Zinc in Sediment

Zinc was found at concentrations greater than 200 mg/kg (Maps 5.1-25 and 5.1-26a–m, see frequency plot inset) at many locations along the eastern and western nearshore zones, but few in the navigation channel. Detection frequencies were 100 percent for both surface and subsurface samples (Figures 5.1-29 and 5.1-30; Tables 5.1-1 and 5.1-2). Where the latter elevations appeared, extent was generally limited (Maps 5.1-25 and 5.1-26a–m). Ninety-five percent of the surface samples were below 375 V mg/kg.

The highest surface (HA-43) and subsurface (HA-42; 46–61 cm bml) concentrations were found between RM 8.2 and 9.2. Station HA-42 also recorded the highest chromium concentration in the Study Area.

5.1.9.8 Tributyltin Ion in Sediment

Sediment samples at selected locations were analyzed for butyltins. Bulk sediment TBT concentrations varied among locations (Maps 5.1-27 and 5.1-28a–m; Figures 5.1-31 and 5.1-32). Frequencies of detection for TBT were 94 percent for surface samples and 59 percent for subsurface samples (Tables 5.1-1 and 5.1-2). Ninety-five percent of the surface samples were less than 851 V µg/kg. Concentrations greater than 1,000 µg/kg (Maps 5.1-27 and 5.1-28a–m; see frequency plot inset) were measured in

surface samples near the western nearshore zone at RM 9 and near the entrance to the International Terminals Slip. Most concentrations greater than 1,000 µg/kg were found near RM 8 in areas surrounding Swan Island and immediately downstream of Swan Island. The maximum surface concentration of TBT was found at the mouth of the International Terminals Slip (Station SD012). Station PSY30C (121–152 cm bml) at RM 8.2 near Swan Island contained the highest subsurface concentration of TBT in the Study Area.

5.1.10 Summary of the Nature and Extent of Indicator Chemicals in Sediment

PCBs, PCDD/Fs, DDx, and PAHs were found across the Study Area, but concentrations varied by orders of magnitude. Concentrations of these chemicals were lowest in the navigation channel and highest in localized, generally well-defined, nearshore areas along both the west and east river banks.

The surface to subsurface mean concentration ratios indicated that PCBs were higher in the subsurface than surface in the Study Area as a whole. A notable exception was RM 11–11.8, where mean PCB concentrations were notably higher in the surface sediment than in the subsurface sediment. In general, the overall chlorination level of PCBs in the surface and subsurface sediments in the eastern nearshore zone tended to be higher upriver in the Study Area and lower downstream. Homolog patterns in areas of high PCB concentration tended to be more variable in the western nearshore zone. These variances in the relative abundances of the homolog groups may reflect differences in the sources of the PCBs.

For PCDD/Fs, the surface to subsurface ratios were generally close to one, indicating that the surface and subsurface concentrations were comparable across the entire Study Area. Exceptions to this pattern occurred in the area under and just upstream of the Railroad Bridge at RM 6.9, where surface concentrations were higher, and in the northwest corner of Willbridge Terminal, where subsurface concentrations were higher. OCDD was the predominant homolog (>50 percent) in surface and subsurface sediment within the Study Area.

The surface to subsurface ratios for DDx within the Study Area indicated that concentrations were higher in the subsurface than in the surface layer. Along the west bank from RM 7 to 8, where total DDx levels were highly elevated, the average subsurface concentration greatly exceeded the overall surface concentrations. The 4,4'-isomer of DDT dominated the surface sediment profile here, while DDT and DDD dominated the subsurface profile, possibly indicating degradation of DDT to DDD in the deeper anoxic sediments. Overall, the 4,4'-isomers were more abundant than the 2,4'-isomers within the Study Area.

The surface to subsurface ratios for PAHs indicated that concentrations were higher in the subsurface than in surface sediments. A notable exception occurs at RM 5–6 where the mean channel surface concentration was considerably higher than the mean

subsurface concentration. The proportions of individual PAHs varied throughout the Study Area, possibly reflecting PAH contributions from multiple sources, as well as weathering and degradation. Three-ring LPAHs and four-ring HPAHs dominated the western shoreline profile between RM 5.4 and 6.6 where the highest PAH concentrations were found.

5.2 INDICATOR CHEMICALS IN SEDIMENT TRAPS

This section summarizes the sediment trap data for the LWG rounds of investigation. The geographic locations of all sediment trap stations are presented on Map 2.2-3. The following subsections present histograms and stacked bar charts to support discussion and evaluation of the nature and extent of selected ICs. Additional tabular and graphical summaries of the sediment trap data set are included in Appendix D2.

The primary purpose of sediment trap sampling was to gather data for the evaluation of FS alternatives (Anchor 2006c). In addition, the sediment trap investigation was designed to capture anticipated spatial and temporal variability of suspended sediment mass, fill data gaps related to the nature and extent of potential sources, and support the preparation of the BERA (Anchor 2006c).

5.2.1 Sediment Trap Data Set

This section focuses on the concentrations of 15 ICs associated with samples from in-river sediment trap samples collected within the LWR. These samples were analyzed to measure the sediment trap mass accumulation and concentrations of sediment-bound contaminants that enter the Study Area from upstream sources, contaminant concentrations associated with regional sources within the Study Area, and concentrations of sediment-bound contaminants that migrate downstream from the Study Area. Additional information on LWR hydrology, sediment accumulation, and the role of fine sediments provided to aid with interpretation of the chemical data, is presented in Figures 5.2-1 through 5.2-5. Distributions of selected ICs and patterns of bounding ICs—total PCBs, total PCDD/Fs, total DDx, and total PAHs—are shown in Figures 5.2-6 through 5.2-10 and 5.2-14 through 5.2-30, and are summarized in Tables 5.2-1 through 5.2-4. Figures 5.2-11 through 5.2-13 present an analysis of PCB homolog data across all stations; details of this analysis are discussed in Section 5.2.5.

5.2.2 Indicator Chemicals in Sediment Traps

The IC list for sediment is presented in Table 5.0-2. A total of 34 individual analytes and calculated analyte sums were identified as ICs for bedded sediment in Section 5.1. Specific chemicals and chemical sums selected as ICs for sediment traps were identical to those selected for sediments, with one exception: total PCB Aroclors and total PCB congeners are presented separately in this section, whereas total PCBs in Section 5.1 were evaluated in a single presentation (i.e., total Aroclors were selected when total PCB congeners were not available). The ICs for sediment identified below pose the most widespread or significant potential risks to human health or the environment in Portland Harbor Study Area sediment; however, other contaminants present and comingled throughout the Study Areas may also pose risk. Additional discussion of the identification and use of the ICs in the RI and their relation to contaminants with potentially unacceptable risk in the baseline risk assessments and FS is presented in the introduction of Section 10.

The ICs for sediment traps are organized as follows:

- PCBs
  - Total Aroclors*
  - Total PCB Congeners*
  - PCB TEQ (ND=0)
- PCDD/Fs
  - Total PCDD/Fs*
  - TCDD TEQ (ND=0)*
- Total DDx
  - Total DDx (2,4'- and 4,4'-DDD, DDE, DDT)*
  - Total of 2,4'- and 4,4'-DDT
  - Total of 2,4'- and 4,4'-DDE
  - Total of 2,4'- and 4,4'-DDD
- PAHs
  - Total PAHs*
  - Total LPAHs
  - Total HPAHs
  - Total cPAH BaPEq values
  - Phenanthrene
  - Naphthalene
  - BAP
- Petroleum Hydrocarbons
  - TPH
  - TPH - DRH
  - TPH - RRH
- SVOCs
  - BEHP*
  - Butylbenzyl phthalate
  - PCP
  - Hexachlorobenzene

- Pesticides
  - Total chlordanes*
  - gamma-HCH
  - Aldrin*
  - Dieldrin*
- Organometallic Compounds
  - TBT*
- Metals
  - Arsenic*
  - Cadmium
  - Chromium*
  - Copper*
  - Lead
  - Mercury
  - Nickel
  - Zinc*.

Fourteen of the ICs, which are indicated with an asterisk (*) in the above list, are discussed in this section (total PCDD/Fs and TCDD TEQ are grouped as one chemical in this count). As indicated in Section 5.1, although numerous contaminants that potentially pose risk to human health or the environment are comingled across the site, 13 ICs were selected for sediments, in consultation with EPA, to match the contaminants identified for the site-wide CSM (Section 10). In this section, PCB congeners and PCB Aroclors are described separately, bringing the total up to 14. The remaining ICs are not discussed in this section, but their concentrations, as well as percent fines and TOC, in sediment traps are displayed graphically in Appendix D2.1 (see Figures D2.1-1 through D2.1-23).

Discussion of the ICs addressed in Section 5.2 focuses primarily on the following elements:

- A description of the data set for each analyte, including frequency of detection and concentration range
- The sampling locations and periods (sampling quarters) with elevated analyte concentrations and any apparent spatial or temporal gradients among the data set
- An evaluation of analyte concentrations found in the Study Area compared to concentrations found at locations upstream of the Study Area

- Brief comparisons between analyte concentrations in east-west paired samples.

As noted previously, the bounding ICs—total PCBs, total PCDD/Fs, total DDx, and total PAHs—are discussed in more detail. Patterns in the relative abundance of each contaminant's components (e.g., homologs, isomers) are examined to evaluate the potential for source identification among spatial and temporal dimensions.

5.2.3 Description of Sediment Trap Presentation Tools

The sediment trap chemistry distributions are depicted in several graphic display formats. The two primary graphical formats are histograms showing concentrations for each location and grouped by sampling quarter (Figures 5.2-6, -7, -14, -16, -17, -19, and 5.2-22 through 5.2-30), and stacked bar charts depicting patterns of bounding ICs (Figures 5.2-9, -10, -15, -18, -20, and 5.2-21).

The blank spaces in the histograms within station groups signify that the volume of material collected for the quarter was not sufficient for analysis or the sediment trap was lost. Sample analyses resulting in non-detects are flagged in the histograms to distinguish them from cases where results are not available. Scales for IC concentrations (y-axis) were selected to emphasize higher concentrations yet visually distinguish comparatively low concentrations. In some cases, values above scale maximums are labeled with the sample concentration.

Stacked bar charts are designed to reveal potentially distinctive patterns in the relative abundance of bounding IC components. These graphs provide a line showing concentrations of the IC (i.e., totals) on a logarithmic scale, but do not display concentrations of the analyte components (e.g., homologs, isomers). The analyte components are shown as a percent of the total concentration. As discussed in Section 5.1, patterns displayed in these charts must be interpreted with caution. The changes in chemical composition and apparent trends shown by the bar charts may be indicative of unique patterns (e.g., distinctive source contributions), or they may be within the anticipated range of data variability. Further detailed forensic analyses outside the scope of this RI/FS are required to determine statistically significant variations among the data and contributions from various potential sources.

Other graphic displays used to assist with data interpretation include two scatter plots (Figures 5.2-3 and 5.2-8) with lines to fit the data and accompanying regression equations. Natural log-transformed PCB congener concentrations are regressed on natural log-transformed Aroclor concentrations in Figure 5.2-8 to display the relationship between PCB results obtained using different analytical methods. At EPA's request, Figures 5.2-11, 5.2-12, and 5.2-13 were generated to facilitate a discussion of the relationships among PCB homolog concentrations and deposition rates. The relationship between sediment accumulation rates and the percentage of fines (i.e., silt and clay, particles ≤62 μm) is shown in a scatter plot of the un-transformed data sets (Figure 5.2-3). Plots of sample grain size distribution are shown in Figure 5.2-4. A line graph (Figure 5.2-1) is used to display the Willamette River

daily discharge hydrograph for the entire sediment trap sampling period, with quarterly sampling periods identified by different colors. This hydrograph also displays average historical daily discharges for a 36-year period (1972–2008). Figure 5.2-5 shows the quarterly distribution of the daily Willamette River discharge combined with sediment accumulation rates (also depicted in Figure 5.2-2), and percent fines (also depicted in Appendix D2.1, Figure D2.1-22).

5.2.4 Overview of Sample Collection Effort

The Round 3A in-river sediment trap sampling consisted of four rounds of sample collection over the course of a year's deployment in the LWR. A detailed description of field efforts associated with sediment trap deployment and recovery is included in the sediment trap data report (Anchor and Integral 2008) and in the respective FSRs for the initial deployment and each collection quarter (Anchor 2006d; 2007a,b,c; 2008c).

Sediment traps were deployed at 16 locations in the LWR from late 2006 through late 2007 (see Map 2.2-3). Twelve of the locations were within the Study Area between RM 1.9 and 11.5. One sediment trap was deployed just downstream of the Study Area at RM 1.8, two were located just upstream of Ross Island at RM 15.6 and 15.7, and one was located in Multnomah Channel. Paired sediment traps were deployed and maintained on opposite sides of the river at approximately RM 1.9, 6, 11.5, and 15.7. A total of 52 sediment trap samples were collected and analyzed per the protocols used in Rounds 2A and 2B. Sediment traps were deployed, sampled, and retrieved during the following periods:

Event	Date
Initial deployment of sediment traps	October 30 to November 2, 2006
Quarter 1 (Winter) sample collection	January 30 to February 2, 2007
Quarter 2 (Spring) sample collection	April 30 to May 2, 2007
Quarter 3 (Summer) sample collection	August 8, 16, and 17, 2007
Quarter 4 (Fall) sample collection and final retrieval of sediment traps	November 13 and 14, 2007

Table 5.2-5 lists river miles, water depths, and location coordinates for each station. Sampling dates and sample recovery information for each deployment are shown in Table 5.2-6.

In total, 52 field samples were collected. Several factors precluded the collection and analysis of all 64 field sample attempts (16 locations x 4 quarters) plus accompanying QC samples as described in the FSP (Anchor 2006c). Traps were lost at three stations during the initial deployment period (Quarter 1) and at one location during Quarter 2. In addition, eight trap retrievals during Quarter 2 and Quarter 3 had sediment accumulation too low to warrant collection. Among the samples collected, an analytical prioritization scheme was used to determine the parameter schedule for samples with limited mass. PCB congener analysis had the highest priority and grain-size analysis

had the lowest priority. A detailed discussion of the decision rules for analytical priorities is included in the data report (Anchor and Integral 2008).

5.2.4.1 Sample Collection Methods and River Conditions

Sediment traps were deployed and maintained on both sides of the river at approximately RM 1.9, 6, 11.5, and 15.7. Individual sediment traps were deployed and maintained at seven other locations throughout the Study Area and at one location in Multnomah Channel. The number and locations of sediment traps and the frequency of recovery and redeployment were designed to capture anticipated spatial and temporal variability of suspended sediment mass and to investigate the potential accumulation of suspended sediment chemical constituents in suspected depositional areas.

Sediment traps were initially deployed October 30 to November 2, 2006. Thereafter, sediment traps were recovered, sampled, and redeployed on a quarterly schedule until final recovery and sampling on November 13 and 14, 2007.

Hydrologic data used to assess flow patterns during sampling were obtained from the USGS stream flow station located upstream of the Morrison Bridge (Willamette River at Portland, gage no. 14211720). The highest flows during sampling occurred during Quarter 1, with a median daily discharge of 79,000 cfs (Figure 5.2-1). This period was characterized by variable flows, reaching twice the historical average discharge on a number of separate events. Much lower than normal discharge periods (≤50 percent of average) also occurred twice during Quarter 1, the latter depression stretching into Quarter 2. The discharge record for Quarters 2 and 3 (median discharges of 31,000 cfs and 10,000 cfs, respectively) did not demonstrate the variability that characterized Quarter 1. In general, sampling during Quarters 2 and 3, and at least a portion of Quarter 4 (median discharge of 11,000 cfs), occurred during river flows that were very similar to historical averages. Discharge data from the last half of Quarter 4 (October 2007 through mid-November 2007) are considered estimates due to uncertainty about the accuracy of the rating curve used at the Portland location for flows less than 20,000 cfs.

5.2.4.2 Rates of Sediment Accumulation

Net sediment accumulation rates at each station/quarter were calculated from the height of the sediment column in the traps and from the specific gravity and moisture content of the material. The highest rates of accumulation occurred during Quarter 1, with the largest accumulation in the sediment traps placed at RM 11.3 and 15.6 (Figure 5.2-2); sediment accumulation rates were lower in the sediment traps placed downstream of RM 11.3. As mentioned previously, traps were lost at stations ST014 (RM 7.5), ST006 (Swan Island Lagoon), and ST016 (RM 9.9) during Quarter 1.

Medium-coarse silt made up approximately 50 percent of the trapped material during each quarter, although the highest sediment accumulation rates generally corresponded with a comparatively low percentage of fine material in the sediment traps. Figure 5.2-3 shows rates of accumulation as a function of percent fines. Trend lines shown for

the data set as a whole ($R^2 = 0.41$), as well as for the individual quarters (R^2 ranging from 0.01 to 0.46), suggest inverse linear relationships between accumulation rates and percent fines for this data set are weak. TOC showed relatively small differences among samples, with concentrations ranging from 1.1 to 3.5 percent and approximately 75 percent of the measured values between 2 and 3 percent (Appendix D2.1, Figure D2.1-23).

Because sediment trap samples do not constitute temporally discrete samples (i.e., they represent a continuous collection over a three-month period), river conditions during sampling can only be discussed meaningfully in seasonal terms. Accumulation rates of trapped sediment may have been substantially affected by instantaneous events, such as high water resulting from heavy rainfall, but the impact of these isolated events cannot be quantified based on the existing data or the sampling methodology employed. Further, there were instances in which sediment traps found to contain insufficient accumulated material for analysis were redeployed with the previous quarter's deposited material. In two cases (ST001 Quarter 3, and ST013 Quarter 3) traps retrieved the following quarter were found to have a shorter column of sediment in them than they had when they were deployed (Table 5.2-6). In the case of ST013, a quarterly deposition rate of zero was used in Quarter 3 data presentations.

Figure 5.2-4 displays the grain size distributions for all sediment trap samples analyzed. As shown, samples from each station generally showed similar grain size distributions, except for an increase in the coarse-grained fraction (i.e., sand) during the winter quarter (Quarter 1) at stations ST008, ST009, and ST010, and during the fall quarter (Quarter 4) at ST007. The higher rate of sediment accumulation and the entrainment of sandy material in the sediment traps placed between RM 11.5 and 15.7 during Quarter 1 may be due in part to the frequency of higher flow events that occurred during this period (Figure 5.2-5). The distribution of flows shows that the maximum daily flows during Quarter 2 were approximately the same as median Quarter 1 flows, and approximately 75 percent of the Quarter 1 daily discharge levels were higher than any of those recorded during Quarters 3 and 4. A lower accumulation of trapped sediments, particularly at upriver stations, occurred during Quarters 3 and 4 when comparatively low-flow events were typical.

5.2.5 Total PCBs in Sediment Traps (Congeners and Aroclors)

PCB congener analysis was conducted for all 52 sediment trap samples; 48 of these samples were also analyzed for PCB Aroclors. PCB congeners were detected in all 52 samples, with total PCB congener concentrations ranging from 3.14 J µg/kg to 11,100 J µg/kg (Figure 5.2-6). PCB Aroclors were detected in 30 of the 48 samples analyzed, with total Aroclor concentrations ranging from 7.1 J µg/kg to 2,600 µg/kg (Figure 5.2-7).

The relationship between total PCB congener and total PCB Aroclor concentrations is shown in Figure 5.2-8 and discussed in detail in Appendix D1.5. Although the PCB concentrations in sediment trap samples correlated well for the two methods,

concentrations of total PCBs measured as congeners were higher overall than total PCBs measured as Aroclors. The methods used for analysis of PCB congeners and Aroclors are fundamentally different and would be expected to yield moderate differences in total PCB concentrations, as described in Appendix D1.5. In addition, among detected Aroclor results for the sediment trap samples, almost half of the individual concentrations (21 of 50 results) were below the method reporting limit (MRL). There is less confidence that these estimated Aroclor concentrations are as accurate and precise as concentrations above the MRL. Ten additional Aroclor results were reported at concentrations less than two times the MRL. These values are also expected to be less precise than concentrations that are farther above the noise level of the system and well within the calibration range.

The accuracy and precision of Aroclor data are also affected by PCB "weathering." An examination of the Aroclor chromatograms indicates that many of the Aroclor patterns in the sediment trap samples were weathered, and that the PCB patterns in the samples did not match the Aroclor standards well. This affects the laboratory's ability to identify, as well as quantify, the Aroclor(s) in the sample and is an inherent limitation to the Aroclor method. Because of the uncertainties in the Aroclor data related to concentrations near or below the MRL and to "weathering," the following summary of PCB patterns and trends in sediment trap samples gives preference to PCB congener data.

PCB concentrations were generally higher in sediment traps located between RM 6.7 and 11.3 compared to other locations. PCB concentrations upstream of RM 11.3 did not exceed 24 µg/kg for total Aroclors or 13 µg/kg for total congeners.

Increasing concentrations generally occurred with each successive period at locations downstream of ST007 (RM 11.3E), a trend that was clear in the PCB congener data but not apparent for Aroclors, possibly due to the lower number of samples with detectable Aroclor concentrations. PCB concentrations at Station ST007 during low- and medium-flow periods (Quarters 2, 3 and 4) were elevated two to three orders of magnitude above concentrations at other locations for the respective periods. ST007 PCB congener concentrations during the highest flow period (Quarter 1) were much lower than during subsequent quarters, and only slightly above concentrations at downstream locations during the same period. Aroclors concentrations at ST007 were similarly elevated relative to other locations during Quarters 3 and 4, but unlike congeners, the total Aroclor concentration during Quarter 2 was one-half the Quarter 1 concentration.

Aside from large differences in PCB levels between ST007 (RM 11.3E) and its paired location ST008 (RM 11.5W), there was little relative difference in PCB concentrations between other cross-river sample pairs. Total PCB congener concentrations in the Study Area samples were all higher than the average PCB concentrations from upstream locations—one-to-five fold greater than upstream concentrations, in most cases. These trends were generally also reflected in the Aroclor data.

The distribution of detected PCB homologs and Aroclors is shown in Figures 5.2-9 and 5.2-10, respectively. TetraCBs, pentaCBs, hexaCBs, and heptaCBs were the predominant homolog groups in the sediment trap samples. Aroclors 1254 and 1260 accounted for most of the reported Aroclors, which is consistent with the homolog composition. Aroclor 1254 consists primarily of pentaCBs followed by tetraCBs and hexaCBs, and Aroclor 1260 contains mostly hexaCBs and heptaCBs (Figure 5.1-34). DiCBs and triCBs were detected in most samples and are consistent with patterns displayed by weathered Aroclors 1254 and 1260.

In addition to 1254 and 1260, Aroclors 1221 and 1242 were reported in eight instances. Aroclor 1221 was reported once, in the Quarter 3 sample from ST011 (RM 3.5E) at a concentration just above the method detection limit (MDL) and with qualifiers "NJ" (i.e., tentatively identified and estimated). Aroclor 1242 was identified in seven samples from Quarter 2, all at concentrations below the MRL of approximately 10 µg/kg, but was not identified during other sampling periods. Considering the low levels at which these Aroclors were detected and the regular occurrence of weathered Aroclor components among samples, it is conceivable that the partially dechlorinated PCBs in these samples led to misidentifications as Aroclors 1221 and 1242.

The higher concentrations and apparent increased dominance by hexaCBs and heptaCBs in samples from ST007 during Quarters 2, 3, and 4 in Figure 5.2-9 suggested that the dominance of higher chlorinated PCBs may increase with increasing concentrations. At EPA's request, the data were further analyzed to examine this possibility. A covariance matrix[1] of the sediment trap data set was constructed and investigated, both graphically and quantitatively. The data were log-transformed, centered and scaled to conform to the assumptions of normality and equal variance required for the parametric correlation metric used (Pearson's R).

The congener results showed a very uniform correlation structure, indicating that the relative homolog ratios were statistically similar among all samples regardless of total concentration. All of the PCB homologs had strong (Pearson's R in the 0.9 range) and significant positive relationships throughout the entire data set (Figure 5.2-11). Only decaCBs (otherwise a minor contributor to the total PCB concentration in all samples; Figure 5.2-9) had slightly lower correlations (Pearson's R in the 0.6 range) to the other nine homologs. Taken together, these results strongly suggest that across the entire range of total PCB concentrations within this data set, the relative proportions of all 10 PCB homolog groups are essentially constant.

Four homolog groups (tetraCBs, pentaCBs, hexaCBs, and heptaCBs) clearly dominate the PCB composition of all sediment trap samples (Figure 5.2-9), constituting 80 to 90 percent of total PCBs throughout the data set. Though the three samples (from Quarters 2, 3, and 4) collected at station ST007 visually appear to have slightly more hexaCBs and heptaCBs than others, quantitative analysis of these data indicates that the overall

[1] A regression analysis was not selected because homolog concentrations and total PCB concentrations are not independent variables, i.e., both are calculated from the same congener data set.

composition of these ST007 samples falls in line with the rest of the data set (Figures 5.2-11 and 5.2-12).

Also at EPA's request, the relationship between PCB homolog concentrations and the calculated net deposition rate of the sediment trap samples was examined (the deposition rate data variable is coded "DEPRT" in Figure 5.2-11). None of the PCB homologs, or total PCB concentrations, was significantly correlated to the solids deposition rates in the sediment traps (Figures 5.2-11 and 5.2-13).

PCB homolog distributions in sediment trap samples were generally consistent from quarter to quarter at each location resulting in a high degree of similarity among the samples in this data set. The PCB composition of all samples was dominated by tetraCBs, pentaCBs, hexaCBs, and heptaCBs (80-90 percent of total), but minor localized or temporal variations were also noticeable. These small-scale deviations may be indicative of potential localized sources, but also may be within the range of normal data variability. Detailed forensic analyses would be required to determine whether or not these minor variations represent statistically significant departures from the overall PCB pattern, which may be linked to local sources. Variations apparent in the stacked bar charts and a comparison of the paired sediment trap samples are discussed below:

- The Quarter 1 sample at ST009 (RM 15.7E) had a slight increase in the proportion of pentaCBs and hexaCBs, along with consistent contributions from tetraCBs and octachlorinated biphenyls (octaCBs). Samples at this station from the other three quarters had slightly greater contributions from diCBs, triCBs, tetraCBs, and octaCBs. ST010 (RM 15.6W) is located west of ST009, and the Quarter 1 sample from this station showed a slightly greater proportion of pentaCBs compared to other quarters, but to a lesser degree than was seen at ST009.
- HexaCBs and heptaCBs appeared to contribute increased proportions to the PCB concentrations at ST007 (RM 11.3E) as total concentrations increased in Quarters 2, 3, and 4. The proportion of octaCBs was also slightly higher in these samples, and the proportion of tetraCBs and pentaCBs was slightly lower, in comparison to those from upstream locations. This small-scale change in PCB pattern, combined with the high concentrations, may indicate the presence of a localized source of PCBs in this vicinity during Quarters 2, 3, and 4.[2] The PCBs in Quarter 1 at this location were different from the subsequent quarters, with relatively higher percentages of triCB and tetraCB fractions, relatively lower hexaCB, octaCB and heptaCB fractions, and a much lower total PCB concentration. However, the overall composition profile and magnitude of total PCB concentration at ST007 are, in fact, closer to the other samples in this data set.

[2] Station ST007 is located between two facilities with active, deep draft docks (i.e., between RM 11.3E and 11.4E).

- The PCB patterns at ST008 (RM 11.5W), across from ST007, appeared slightly different from ST007, and the PCB concentrations were lower; ST008 does not appear to exhibit significant influence from ST007 in terms of PCB composition.
- A small shift of the pattern of higher chlorinated PCBs was consistently seen at ST006 (Swan Island Lagoon), ST015 (RM 9.7W), and ST016 (RM 9.9E). All samples from these stations were predominated by pentaCBs, hexaCBs, and heptaCBs.
- ST013 (RM 6.7E) in Willamette Cove was composed primarily of moderately chlorinated homologs (tetra- through heptaCBs), with relatively high contributions from octaCBs in all three quarters sampled. While the PCB homolog distribution was consistent for all samples from this location, the total PCB congener concentration in the Quarter 4 sample was five times higher than during Quarter 1.
- The pattern of the Quarter 1 sample at ST004 (RM 6.0E) appeared slightly shifted towards higher contributions of hexaCBs and heptaCBs, whereas samples from other quarters at this station more closely matched the tetraCB, pentaCB, hexaCB, and heptaCB distribution of the other stations. ST005 (RM 6.0W), opposite ST004, had a relatively higher proportion of tetraCBs and pentaCBs, and lower abundance of heptaCBs compared to ST004.TriCBs made up a larger proportion of the samples from ST011 in Quarters 1 and 2 in comparison to other locations. The octaCB proportion appeared elevated in the Quarter 4 sample from this station. Quarter 3 and 4 samples from ST011 had total PCB congener concentrations two to three times higher than the Quarter 1 and 2 samples.

5.2.6 Total PCDD/Fs in Sediment Traps

At least one dioxin (PCDD) or furan (PCDF) homolog constituent was detected in each of the 48 samples analyzed (Figure 5.2-14). Total PCDD/F homolog concentrations ranged from 5.16 J to 6,100 J pg/g.

There was no apparent spatial gradient or trend in total PCDD/F concentrations. The highest PCDD/F homolog concentration (6,100 J pg/g) occurred during Quarter 4 at ST006 (Swan Island Lagoon), and was elevated 1 to 2 orders of magnitude above concentrations at most other locations. However, the incidence of elevated PCDD/Fs at ST006 other than during Quarter 4 could not be assessed since no samples from previous quarters were analyzed at this location. PCDD/F spikes of 1,820 J pg/g and 1,250 J pg/g occurred during Quarter 3 at ST007 (RM 11.3E) and at ST002 (RM 1.8W), respectively, representing the next highest levels found. Relatively high concentrations were also seen in Quarter 4 samples from ST014 (RM 7.5W; 1,060 J pg/g) and ST007 (745 J pg/g), and Quarter 1 samples from ST001 (RM 1.9E; 563 pg/g) and ST011 (RM 3.5E; 535 pg/g).

Study Area locations generally had total PCDD/F concentrations higher than average concentrations from the upstream locations. However, total PCDD/F concentrations from the two upstream locations were not similar to each other, with concentrations from ST009 (RM 15.7E) averaging more than six times those from ST010 (RM 15.6W). Some ST009 samples had comparatively higher total PCDD/F concentrations than concurrently sampled Study Area locations.

The highest PCDD/F concentrations among stations generally occurred during Quarters 4 and 3. Stations ST007 and ST009 contained higher total PCDD/Fs than their counterparts during these periods. However, the east (ST004) and west (ST005) pair at RM 6 had nearly identical concentrations during all quarters analyzed.

Homolog patterns for sediment trap samples showed OCDD as the dominant homolog in almost all sediment trap samples, with HpCDDs present to a substantial degree in most samples (Figure 5.2-15). PCDFs generally accounted for less than about 20 percent of the total PCDD/Fs in most of the samples. OCDF typically accounted for the highest furan concentrations, followed by HpCDFs.

Samples with homolog patterns apparently different from the prevailing pattern were observed at several locations, but these patterns are obscured because U-qualifiers were applied during data validation in most cases; non-detects are treated as zero in the graphs of chemical patterns. However, the Quarter 2 sample from ST008 (RM 11.5W) was notable for its relatively high proportions of OCDF and relatively low proportions of OCDD compared to other samples.

The following samples have incomplete PCDD/F patterns because some of the PCDD/F homolog data were qualified as undetected during validation (Anchor and Integral 2008): ST010, Quarter 2 (HpCDDs undetected) and Quarter 4 (HpCDFs and OCDF undetected); ST003 (Multnomah Channel), Quarter 2 (HxCDDs, HxCDFs, HpCDDs, HpCDFs, and OCDD undetected); ST008, Quarter 4 (HpCDFs and OCDF undetected); and ST016 (RM 9.9E), Quarter 4 (HpCDFs undetected).

Samples with total PCDD/F concentrations >500 pg/g—ST001 and ST011 from Quarter 1; ST002 and ST007 from Quarter 3; and ST014, ST006, and ST007 from Quarter 4—had apparently similar homolog patterns.

5.2.6.1 TCDD TEQs in Sediment Traps

Measurable TCDD TEQs were found in all samples at concentrations ranging from 0.05 J pg/g to 16.3 J pg/g (Figure 5.2-16). The highest TCDD TEQ was found in the Quarter 4 sample from ST006 (Swan Island Lagoon). Since Quarter 4 was the only time PCDD/Fs were analyzed at this location, it is difficult to gauge the occurrence of similar PCDD/F concentrations during other periods or the potential that there is a localized source. TEQs ≥1 pg/g were also found at ST007 (RM 11.3E) during Quarters 3 and 4; the Quarter 3 sample from ST002 (RM 1.8W); ST014 (RM 7.5W)

during Quarter 4; and Quarter 1 samples from ST001 (RM 1.9E), ST011 (RM 3.5E), and ST005 (RM 6.0W), where the Quarter 2 sample was also >1 pg/g.

TEQ concentrations derived from PCDD/Fs closely track total PCDD/F homolog concentrations, both geographically and seasonally. The close parallel is due to the resemblance in homolog patterns among samples. TEQ concentrations are generally two-to-three orders of magnitude lower than total PCDD/F homolog concentrations due to the abundance of higher chlorinated 2,3,7,8-substituted dioxins and furans, which have less toxic equivalency than tetra- and penta-chlorinated congeners.

5.2.7 Total DDx in Sediment Traps

DDx compounds were detected in all but one sediment trap sample (Figure 5.2-17). Concentrations of total DDx ranged from 0.98 J µg/kg to 150 µg/kg in samples with detectable concentrations.

The highest total DDx concentration occurred during Quarter 4 at ST007 (RM 11.3E) and was an order of magnitude higher than most other samples. Quarter 4 samples from ST005 (RM 6.0W) and ST006 (Swan Island Lagoon) had the second and third highest concentrations, 33 J µg/kg and 31 J µg/kg, respectively. At most locations, total DDx concentrations were highest during Quarter 4.

Among locations, samples collected from sediment traps at RM 6 to 11.3 generally had the highest total DDx concentrations. Downstream of RM 6, total DDx concentrations appeared to be similar within quarters or to decrease slightly with distance downstream. The maximum concentration found downstream of RM 6 (15 J µg/kg) was low relative to maximum concentrations from stations located at RM 6 through 11.3.

Total DDx concentrations in Study Area samples were generally higher than those from upstream locations. Overall, 35 of the 43 (81 percent) Study Area samples had higher total DDx concentrations than average concentrations from concurrent upstream samples. Differences between Study Area and upstream samples were most pronounced during Quarters 3 and 4, when the combined average Study Area total DDx concentration was six-fold that of the combined average concentrations from upstream locations. By contrast, Study Area samples from the first two quarters had a combined average total DDx concentration 34 percent higher than the combined average concentration of upstream samples.

Paired samples from opposite sides of the river at RM 1.8–1.9 (ST002/ST001) and at RM 15.6–15.7 (ST010/ST009) had similar total DDx concentrations, whereas high Quarter 4 total DDT levels were noted at ST005 and ST007 compared to their respective counterparts, ST004 (RM 6.0E) and ST008 (RM 11.5W; Figure 5.2-17).

The relative concentration of DDD, DDE, and DDT varied widely among samples. Among the 2,4'- and 4,4'-isomers, the 4,4'-isomers generally predominated (Figure 5.2-18). On average, 4,4'-DDT comprised the largest proportion of total DDx

(29 percent of total), followed by 4,4'-DDE (26 percent), and 4,4'-DDD (21 percent). Among the 2,4'-isomers, 2,4'-DDT was found at the highest percentage of total DDx (12 percent on average), followed by 2,4'-DDD (10 percent), and 2,4'-DDE (one percent).

Stations ST007 and ST015 (RM 9.7E) had the highest proportion of 2,4'-DDx compounds due to comparatively high percentages of 2,4'-DDD and 2,4'-DDT. Station ST013 (RM 6.7W) also had a high percentage (52 percent) of 2,4'-DDT in the Quarter 4 sample. Upstream samples were primarily composed of 4,4'-DDx isomers.

The DDD isomers exhibited variability in both their concentration and relative contributions at most stations; no seasonal or geographic trends were evident. DDT isomers appeared to have the highest relative concentrations during Quarter 1, although no geographical trends were apparent. A more seasonal trend was observed for DDE isomers, where they were reported at detectable concentrations in all but two of the samples from Quarters 2 through 4, yet were detected in only three of the 13 samples from Quarter 1. Relative contributions of DDE compounds to total DDx were generally highest during Quarter 4.

Patterns of relative concentrations of DDx constituents among samples are somewhat confounded by elevated detection limits and interferences. Of the 300 DDx results, detection limits were elevated in 18 percent, and another four percent were classified as non-detects due to contamination in the associated laboratory or field blanks (Anchor and Integral 2008). The elevated detection limits could obscure low concentrations of DDx. Because non-detected results are treated as zero in Figure 5.2-18, true patterns of relative concentration among the various isomers and congener compounds may also be obscured. In addition, 9 percent of the results were qualified as tentatively identified and estimated (NJ) during data validation due to poor confirmation, and another 15 percent were estimated (J) as a result of the confirmation data.

Total DDx at ST007 Quarter 3 and 4 samples (24 and 150 µg/kg, respectively) were composed entirely of 2,4'-DDD. However, 2,4'-DDD reported by the laboratory in these cases may be artifacts of PCB interference. High concentrations of Aroclor 1260 were also detected at this station in these samples (1,800 and 2,600 µg/kg respectively). Aroclors have the potential to interfere with the analysis of DDx due to co-elution of Aroclor peaks with DDx compounds. While the 2,4'-DDD results were not N-qualified during data validation, the results for other isomers (2,4'-DDT and 4,4'-DDT in both Quarters 3 and 4, and 2,4'-DDE in Quarter 4) were N-qualified. It is possible that the reported results are affected by Aroclor interferences.

5.2.8 Total PAHs in Sediment Traps

PAHs were detected in all samples analyzed, with concentrations of total PAHs ranging from 77 J µg/kg to 11,000 µg/kg (Figure 5.2-19). Total PAH concentrations at ST005 (RM 6.0W) were elevated an order of magnitude above most other locations during Quarters 1, 2, and 4 (PAHs were not analyzed at ST005 during Quarter 3). In general,

total PAH concentrations were higher at locations between RM 3 and 6, including Multnomah Channel (ST003) which had a relatively high Quarter 4 level (2,700 J µg/kg). Total PAH concentrations upstream of RM 9.7 were generally lower, with only two samples exceeding 500 µg/kg: 640 µg/kg at ST007 (RM 11.3E) during Quarter 3 and 1,300 J µg/kg at ST010 (RM 15.6W) during Quarter 1.

The highest PAH concentrations within stations tended to occur during Quarters 3 and 4, but additional seasonal differences among stations were not apparent. Most samples analyzed from Study Area locations had total PAH concentrations exceeding average upstream concentrations during concurrent sampling periods. Overall, 33 of the 43 (77 percent) Study Area samples had total PAH concentrations higher than concurrent samples from upstream locations, with the exception of Quarter 1, where the average total PAH concentration of upstream samples (700 µg/kg) was higher than most Study Area samples. During subsequent quarters, however, concentrations upstream were lower (combined average of 190 µg/kg), and 91 percent of Study Area samples had total PAH concentrations an average of six times greater than those from upstream.

Paired samples generally did not display substantial differences in total PAH concentrations, with two notable exceptions. There were large differences between total PAH concentrations at ST005 and its cross-river counterpart (ST004, RM 6.0E) during all quarters where samples from both stations were analyzed. Total PAH concentrations during Quarter 1 at ST010 were also higher than the concurrent sample from ST009 (RM 15.7E).

An evaluation of the PAH composition among samples was done by combining the 17 individual PAH compounds into categories determined by their number of fused rings, as described in Section 5.1.8.4. Fused ring patterns did not appear to vary substantially among samples, with four-ring PAHs as the predominant component, followed by those with five rings (Figure 5.2-20). HPAHs (PAHs with four or more fused rings) generally accounted for more than 80 percent of total PAHs. LPAHs (PAHs with two or three fused rings) accounted for a maximum of one-third total PAH concentrations, and exceeded 20 percent of total PAHs in only nine of 49 samples analyzed. A bar chart of all site PAHs (Figure 5.2-21) shows that the four-ring PAHs consist primarily of fluoranthene and pyrene and the five-ring PAHs consist primarily of BAP and benzo(b)fluoranthene. These four PAHs account for approximately 50 percent of the total PAHs in all samples.

The PAH distributions at ST005, the location with the highest total PAH concentrations, were similar to the prevailing pattern. Samples with PAH distributions that appeared to differ somewhat from the prevailing pattern were observed at several locations:

- The Quarter 2 sample from ST009 (RM 15.7E) exhibited apparent increased contributions from five-ring PAHs, mainly BAP.
- At ST002 (RM 1.8W) the Quarter 2 and 3 samples exhibited apparent increased contributions from five-ring and six-ring PAHs. However, the distribution of

the individual PAHs within the five-ring and six-ring PAH groups appeared similar to other samples.

- At ST003 (Multnomah Channel), apparent increased contributions from five-ring and six-ring PAHs were observed in the Quarter 2 and 4 samples. The distribution of the individual PAHs within the five-ring and six-ring PAH groups appear similar to other samples.

The evaluation of the LPAH distributions is confounded by the fact that eight results for naphthalene (a two-ring PAH) from Quarter 2 were restated as undetected because of laboratory blank results (Anchor and Integral 2008). The levels of LPAHs in these Quarter 2 samples may be under-represented. With that caveat, samples from Quarter 1 at ST010 (RM 15.6W), Quarter 3 at ST011 (RM 3.5E), and Quarter 4 at ST003 (Multnomah Channel)—all with total PAH concentrations greater than 1,000 µg/kg (1,300 J µg/kg, 1,600 J µg/kg, and 2,700 J µg/kg, respectively)—are notable for their low contribution (less than 20 percent) of LPAHs to the total PAHs.

In cases where LPAHs constitute more than 20 percent of the total, naphthalene accounts for 60 to 85 percent of the two-ring PAHs. Phenanthrene accounts for over 50 percent of the three-ring PAHs, with anthracene accounting for an additional 10 to 30 percent of the three-ring PAHs. Samples with more than 20 percent LPAHs include: ST009, Quarters 1 and 3; ST010, Quarter 4; ST008 (RM 11.5W), Quarters 1 and 2; ST007, Quarter 3; ST015 (RM 9.7W), Quarter 3; ST013 (RM 6.7E), Quarter 1; and ST004, Quarters 1 and 2.

5.2.9 Additional Indicator Chemicals in Sediment Traps

5.2.9.1 Bis(2-ethylhexyl)phthalate in Sediment Traps

BEHP was detected in all samples analyzed at concentrations ranging from 35 µg/kg to 1,600 µg/kg (Figure 5.2-22). BEHP concentrations at ST006 (Swan Island Lagoon) were elevated an order of magnitude above most other locations, although BEHP at ST010 (RM 15.6W) approached the Quarter 4 level from ST006 (480 J µg/kg vs. 710 µg/kg). Concentrations downstream of Swan Island Lagoon varied less within and among stations and sampling periods than did BEHP concentrations upstream of Swan Island Lagoon. BEHP concentrations upstream of Swan Island Lagoon were generally highest during Quarters 3 and 4, whereas the highest concentrations downstream of Swan Island Lagoon generally occurred during Quarters 1 and 2.

Average upstream BEHP concentrations at ST009 (RM 15.7E) and ST010 were generally lower than Study Area locations during concurrent sampling. During Quarter 4, however, the BEHP concentration at ST010 (480 J µg/kg) was higher than at all Study Area locations except ST006.

BEHP concentrations in paired samples appeared closely matched at RM 6 and locations downstream. Upstream of RM 6, paired samples appeared less similar, with

BEHP concentrations at ST007 (RM 11.3E) and ST010 (RM 15.6W) generally higher than their respective counterparts (ST008 and ST009).

5.2.9.2 Total Chlordanes in Sediment Traps

One or more chlordanes were detected in approximately 70 percent of samples analyzed (Figure 5.2-23). Detectable concentrations of total chlordanes ranged from 0.22 J to 3.7 NJ µg/kg, although reporting limits for non-detects ranged from 0.63 to 460 µg/kg.

The highest total chlordanes concentration was found at ST008 (RM 11.5W) during Quarter 1. Other comparatively high concentrations (>3 µg/kg) were found during Quarter 4 at ST011 (RM 3.5E) and during Quarter 3 at ST006 (Swan Island Lagoon). Total chlordane concentrations were highly variable within and among locations and within and among seasons. Therefore, spatial and seasonal gradients or trends were not apparent.

Overall, Study Area total chlordane concentrations were higher than upstream concentrations. The maximum total chlordanes in all upstream samples was 1 J µg/kg, whereas nine of the 14 Study Area stations had at least one sample with greater than 1 µg/kg total chlordanes. West-side samples from RM 6 (ST005) and 11.5 (ST008) had higher levels during Quarter 1 than their respective east-side sediment traps, ST004 and ST007, but similarities or differences between paired samples were difficult to assess due to the number of non-detects and the vast range of reporting limits.

5.2.9.3 Aldrin and Dieldrin in Sediment Traps

Aldrin and dieldrin, two closely related organochlorine pesticides, were detected in five samples each (Figures 5.2-24 and 5.2-25). None of the samples analyzed contained detectable levels of both aldrin and dieldrin. All of the detected concentrations of aldrin were at similar levels (0.61 NJ µg/kg to 1.1 NJ µg/kg), with the highest concentration found at Station ST005 (RM 6W). Detected dieldrin concentrations were more variable, with concentrations ranging from 0.15 NJ µg/kg at ST003 (Multnomah Channel) to a maximum of 4.9 µg/kg at ST006 (Swan Island Lagoon). Two of the five dieldrin detections were at the upstream location ST009 (RM 15.7E).

Four of the five (80 percent) aldrin detections occurred during Quarter 1, while all of the detectable dieldrin concentrations were found during Quarters 3 and 4. The infrequency of detections did not allow for assessment of a possible geographical concentration gradient or trend. Of the three instances where either aldrin or dieldrin was detected at a paired sample location, corresponding cross-river locations had no detectable concentration. All detected aldrin concentrations occurred at or below RM 7.5, while dieldrin was detected in Multnomah Channel (ST003), RM 3.5E (ST011), Swan Island Lagoon (ST006), and upriver at RM 15.7E (ST009).

5.2.9.4 Arsenic in Sediment Traps

Arsenic was detected in all samples analyzed at concentrations ranging from 1.48 J mg/kg to 7.01 mg/kg (Figure 5.2-26). There was relatively little variation in

concentrations among samples, with approximately 80 percent of the values between 3 mg/kg and 6 mg/kg.

The highest arsenic concentration was found at Station ST011 (RM 3.5E) during Quarter 4. There was little difference between paired samples. The highest levels were generally found during Quarter 4, particularly downstream of RM 9.

Most concentrations of arsenic from Study Area stations were similar to or slightly above arsenic concentrations in upstream locations, except during Quarter 3. Arsenic levels in Study Area samples rarely varied from the arsenic levels at upstream stations by more than a factor of two.

5.2.9.5 Chromium in Sediment Traps

Chromium was detected in all samples analyzed at concentrations ranging from 10.8 mg/kg to 59.5 mg/kg (Figure 5.2-27). There was relatively little variation in concentrations among samples, with approximately 80 percent of the values between 28 and 40 mg/kg.

The highest chromium concentration was found during Quarter 3 at Station ST009 (RM 15.7E), the location furthest upstream. However, the highest concentrations within stations tended to occur during Quarter 1. There were no locations with levels of chromium consistently higher than all others, and there was little difference in chromium concentrations between paired samples.

Although the highest chromium concentration was found in a sample from an upstream location during Quarter 3, most chromium concentrations from Study Area stations were the same or slightly above chromium concentrations typically found in upstream locations. Chromium levels in Study Area samples rarely varied from average chromium levels at upstream stations by more than a factor of two.

5.2.9.6 Copper in Sediment Traps

Copper was detected in all samples analyzed at concentrations ranging from 15.2 mg/kg to 93.6 mg/kg (Figure 5.2-28). There was relatively little variation in concentrations among samples, with approximately 80 percent of the values between 34 mg/kg and 58 mg/kg.

The highest copper concentration was found at Station ST006 (Swan Island Lagoon) during Quarter 3. The second highest concentration (75.1 mg/kg) was found at the upstream location ST009 (RM 15.7E), also during Quarter 3. The highest concentrations among sampling periods per station often occurred during Quarter 4, particularly at stations from RM 3.5 through 6.7. There was little difference in copper concentrations between paired samples.

Although the second highest copper concentration was found in a sample from one of the upstream locations, most copper concentrations from Study Area stations were

slightly above copper concentrations in upstream locations, except during Quarter 3, as mentioned above. Copper levels in Study Area samples rarely varied from the copper levels at upstream stations by more than a factor of two.

5.2.9.7 Zinc in Sediment Traps

Zinc was detected in all samples analyzed at concentrations ranging from 71.6 mg/kg to 319 mg/kg (Figure 5.2-29). There was relatively little variation in concentrations among samples, with approximately 80 percent of the values between 99 mg/kg and 170 mg/kg.

The highest zinc concentration was found during Quarter 3 at Station ST006 (Swan Island Lagoon). ST006 was the only station that appeared to contain comparatively high localized concentrations, although only data from two quarters were available from this location. Among sampling periods, the highest zinc concentrations per station most often occurred during Quarter 4. Quarter 1 generally had the lowest zinc levels among sampling periods. Samples from paired stations had nearly identical zinc concentrations during concurrent sampling periods, except for the relatively large difference between ST010 (RM 15.6W; 125 mg/kg) and ST009 (RM 15.7E; 71.6 mg/kg) during Quarter 4.

Most concentrations of zinc from Study Area stations were slightly above zinc concentrations in upstream locations, except during Quarter 3. Zinc levels in Study Area samples rarely varied from the zinc levels at upstream stations by more than a factor of two.

5.2.9.8 Tributyltin Ion in Sediment Traps

TBT was detected in approximately 80 percent of the samples analyzed. Detectable concentrations of TBT ranged from 0.48 J µg/kg to 81 µg/kg (Figure 5.2-30).

TBT concentrations at ST006 (Swan Island Lagoon), for the two quarters that data were available for that station, were elevated an order of magnitude above other locations. Concentrations within locations were generally highest during Quarter 4, and concentrations during all sampling periods were generally highest downstream of Swan Island Lagoon. The median TBT concentration downstream of RM 9 was 3.8 µg/kg, with only one sample below the reporting limit (0.17 µg/kg, at ST005). Upstream of RM 9, the median TBT concentration was 0.48 J µg/kg, assuming non-detects equal zero, and nearly one-half of the samples were undetected at reporting limits of 0.12–0.87 µg/kg.

Differences between paired samples were difficult to assess due to the number of non-detects. The largest observed within-pair differences occurred at RM 6, where TBT concentrations from east-side (ST004) samples were higher than those from the west side (ST005) during Quarter 1 (4.6 µg/kg vs. non-detect at 0.17 U µg/kg) and Quarter 2 (4.9 µg/kg vs. 1.5 J µg/kg). However, this situation was reversed during

Quarter 4, with higher concentrations found at ST005 (13 µg/kg) compared to ST004 (3.9 µg/kg).

In general, TBT levels in the Study Area were higher than TBT levels upstream of the Study Area. However, since only one of six samples from ST010 (RM 15.6W) and ST009 (RM 15.7E) had a detectable TBT concentration, the degree of elevation upstream cannot be meaningfully quantified.

5.2.10 Summary of Nature and Extent of Indicator Chemicals in Sediment Traps

PCBs, PCDD/Fs, DDx, and PAHs were found in sediment traps throughout the Study Area and at upstream locations, with concentrations of each IC varying by two-to-three orders of magnitude. In general, samples with the highest IC concentrations were found in sediment traps located from RM 6 through RM 11.3. Specific locations with the consistently greatest PCB and PAH concentrations were easily distinguishable; it was less obvious if maximum PCDD/Fs and DDx consistently corresponded to specific locations over the course of the study.

Total PCB concentrations (as congeners) ranged from 3.14 J to 11,100 J µg/kg, with the highest PCB concentrations consistently found on the east side of the river at RM 11.3. Samples from RM 11.3E during low- and medium-flow periods (Quarters 2, 3, and 4) had total PCB concentrations elevated two-to-three orders of magnitude above those at other locations, with concentrations generally diminishing with distance downstream. Total PCB concentrations in sediment traps located above Ross Island (RM 15.6–15.7) were generally lower, with an overall median concentration one-third of those from Study Area sediment traps.

For PCDD/Fs in sediment traps, it was more difficult to identify a distinct location with the highest total concentrations. Total PCDD/F homolog concentrations ranged from 5.16 J pg/g to 6,100 J pg/g, with the highest concentration found in the solitary sample from Swan Island Lagoon. There did not appear to be a consistent geographical gradient in PCDD/F concentrations, and intermittently elevated concentrations (relative to other samples) occurred at RM 11.3E, 1.8W, and 7.5W. Study Area samples had an overall median total PCDD/F concentration approximately 40 percent higher than the overall median of upstream samples.

Total DDx concentrations in sediment traps were also variable enough to preclude assigning the highest concentrations to a single location. Concentrations of total DDx ranged from 0.98 J µg/kg to 150 µg/kg, with the highest concentration found at RM 11.3E. However, the unusual composition of DDx in samples from RM 11.3E (100 percent 2,4’-DDD during Quarters 3 and 4) coupled with possible analytic interference from Aroclors calls into question the validity of the DDx results from this location. Like other ICs, total DDx was higher in Study Area samples, with an overall median concentration double that of upstream samples.

Total PAH concentrations at RM 6W were elevated an order of magnitude above most other locations. Concentrations of total PAHs ranged from 77 J µg/kg to 11,000 µg/kg and, in general, were highest at locations from RM 3 through RM 6, including Multnomah Channel. Although one sample from RM 15.6W had comparatively high total PAH concentrations during one of the sampling periods, the overall median concentration in Study Area samples was triple that of samples from upstream sediment traps.

5.3 INDICATOR CHEMICALS IN SURFACE WATER

This section summarizes the surface water data for the LWG rounds of investigation. The geographic locations of all Round 2A and 3A surface water sampling locations are presented on Map 2.2-4. Maps 5.3-1a–c present the surface water sampling locations symbolized according to sampling method. The following subsections present hydrographs, histograms, scatter plots, line plots, and stacked bar graphs to support discussion and evaluation of the nature and extent of selected ICs. Additional tabular and graphical summaries of the surface water data set are included in Appendix D3.

5.3.1 Surface Water Data Set

The LWG conducted two rounds of surface water investigation for this RI. The Round 2A and 3A surface water sampling programs consisted of seven field collection events that occurred between November 2004 and March 2007. The surface water study was designed to characterize surface water chemical concentrations and flow conditions of the river during three different flow regimes: low river flow (low-flow; <50,000 cfs), high river flow (high-flow; >50,000 cfs), and stormwater-influenced flow (sampling during active runoff in the Study Area). The threshold discharge rate of 50,000 cfs was selected because it is the river discharge at which significant transport of streambed sediment begins (Willamette Basin Task Force 1969). The seven events are listed below:

- November 2004 (Round 2A, low-flow)
- March 2005 (Round 2A, low-flow)
- July 2005 (Round 2A, low-flow)
- January 2006 (Round 3A, high-flow)
- September 2006 (Round 3A, low-flow)
- November 2006 (Round 3A, stormwater-influenced)
- January 2007 (Round 3A, high-flow[1]).

A summary of the sampling events, including dates of collection, flow rates, and relative flow conditions, are presented in Table 5.3-1. Average discharge rates (recorded as cfs) for each event are based on measurements collected by the USGS at the stream flow station located upstream of the Morrison Bridge at RM 12.8 (Station 14211720). Flow measurements from the USGS gauge at this station are collected every 30 minutes and were used to calculate flow rates for each of the seven sampling

[1] The January 2007 high-flow event was cancelled after two days of sampling due to unexpected change in flow conditions. Sampling recommenced on February 21, 2007 once high-flow conditions (>50,000 cfs) were once again observed.

events.[2] It should be noted that discharge rates below 20,000 cfs measured at this station are considered to be unreliable by the USGS. Therefore, the average discharge rates calculated for the low-flow events should be considered estimates.

Surface water data from the Study Area (RM 1.9–11.8) and areas immediately upstream and downstream of the Study Area were generated during the Round 2A and 3A collection programs. Summary statistics for ICs for all sampling events are provided in Tables 5.3-2 through 5.3-7. In addition, summary statistics for all chemicals measured in surface water are presented in Tables D3.2-1 through D3.2-6.

5.3.1.1 Round 2A and 3A Sample Collection Conditions

Surface water samples were collected at 23 target locations from RM 2 to 11 in the LWR during three Round 2A sampling events in 2004 and 2005. Single-point samples were collected by peristaltic pump at all locations. Additional samples were collected by employing the high-volume XAD sampling method (description of XAD sampling method in Section 5.3.1.2 below) at seven of the 23 locations, including three cross-sectional river transects and four discrete locations. All high-volume samples were collected using an Infiltrex 300 water collection system which pumped water through an inline 0.5-micron glass fiber pre-filter and then through an XAD-2 resin column. Each filter and the resin column was extracted and analyzed separately to determine chemical concentrations in the particulate and dissolved phases of each sample, respectively. The Round 2A surface water study is described in Section 2.1.3.4 of the Round 2 Report (Integral et al. 2007). The Round 2A Surface Water SCSR (Integral 2006j) provides details regarding the sampling program, sample collection procedures, and laboratory analyses.

During the Round 3A sampling events, surface water was collected at 18 target locations from RM 2 to 16 in 2006 and 2007. A transect station located at the upper end of Multnomah Channel (RM 2.9) was added to the program to provide a better understanding of the flux of chemicals exiting the Study Area via Multnomah Channel; and a transect station at RM 16 was added to assist with the analysis of upstream sources and loading into the Study Area. Peristaltic and high-volume samples were collected from 18 stations, including 6 transects and 12 single-point locations. Table 5.3-8 summarizes sampling methods at each station for all Round 2A and 3A sampling events.

Peristaltic surface water samples were analyzed for conventional analytes, metals, and organic compounds (PCB Aroclors, organochlorine pesticides, and SVOCs). High-volume samples were analyzed by high-resolution gas chromatography/high-resolution mass spectrometry (HRGC/HRMS) for PCB congeners, PCDD/Fs, organochlorine pesticides, phthalate esters, and PAHs.

[2] The flow rate values presented here are daily mean stream flow measurements from the USGS National Water Information System, http://waterdata.usgs.gov/nwis. These values were taken from the USGS Web site on June 16, 2008, and are considered to be draft and subject to change by USGS.

The seven Round 2A and 3A surface water sampling events and their corresponding flow rates are presented against the backdrop of the average year (1972–2008) hydrograph measured at Morrison Bridge (RM 12.8) in Figure 5.3-1. Overall, the sampling events were well distributed over the average water year, capturing the range of flow conditions, including base flow, rising limb, peak flow, and falling limb conditions. Additionally, the November 2006 sampling captured a stormwater-influenced event during a low-flow period. Figures 5.3-2 through 5.3-5 present the actual annual hydrograph measured at Morrison Bridge (RM 12.8) and hyetograph during each year of sampling (2004–2007), including daily average and historical average (1978–2008) discharge rates and daily precipitation levels. Several rainfall events occurred during the November 2004 sampling event, and one day of measurable rainfall occurred during each of the March and July 2005 sampling events.

The seasonal cycle of water discharge in the Willamette River is also apparent in Figure 5.3-1. Annual low water levels occur during the summertime regional dry season, and flows increase during the wetter winter months (November to March). Furthermore, a distinct and persistent period of relatively high water levels occurs from late May through June when Willamette River flow into the Columbia is slowed by high-water stage/flow in the Columbia River during the spring freshet in the much larger Columbia River Basin. As described in Section 3.1.3, water levels and currents in the LWR can be influenced by the Columbia River in several ways. The most apparent influence occurs during spring and summer when high flows from the Columbia River act as a hydraulic dam to the Willamette River, resulting in rises in the Willamette River stage. The Columbia River flow drops as the summer progresses and this effect is diminished. During the winter, high seasonal flows on the Willamette River pass through to the Columbia River, which may have diminished flows due to retention at dams.

Hydrodynamic model estimates of flow dynamics at the lower end of the Study Area show that the relative stages of the Columbia and Willamette rivers determines the fraction of the Willamette River flow which flows down Multnomah Channel (WEST 2006a). Figure 5.3-6 presents the average annual hydrograph, based on modeled discharge rates for 2003 through 2007, for RM 4, RM 2, and Multnomah Channel. The Morrison Bridge 25-yr average hydrograph is also shown for comparison. Figure 5.3-7 presents the modeled daily average flows for 2003 through 2007, and highlights the time periods when surface water samples were collected at RM 4, RM 2, and Multnomah Channel. A few key observations are apparent in these figures. First, for a significant portion of each year, generally May through September, the relatively higher Columbia River stage drives a reversal in flow direction at RM 2. During these periods, the Multnomah Channel flow increases and includes the entire Willamette River flow plus some flows from the Columbia River. Second, Figure 5.3-7 shows that surface water sampling events at the RM 2 and Multnomah Channel sample transects did not occur during these flow reversal periods; rather, sampling was conducted when the Willamette River flow was in the downstream direction, and flows split between Multnomah Channel and the main stem. This indicates that surface water samples

collected at RM 2 and Multnomah Channel are representative of Willamette River water and are not strongly influenced by mixing with Columbia River water.

Tidal action also compounds the hydrology and interplay of the two rivers, and affects the Willamette River upstream as far as Portland Harbor and beyond. The high (i.e., flood) tide can influence Willamette River levels by up to 3 ft in Portland Harbor when the river is at a low stage. These tidal fluctuations can result in short-term flow reversals (i.e., upstream flow) in Portland Harbor during times of low river stage combined with large flood tides. Tidal changes were observed at multiple stations during the surface water sampling events. At this time, there is not adequate high-resolution discharge information to determine the potential influence of tidal fluctuations and water mixing on surface water sampling results; however, the overall tidal impact is not expected to be significant.

5.3.1.2 Sampling Methods

Sampling stations included both river-wide transects and single-point sampling stations at specific locations. Transect samples involved vertically integrated (VI) sample composites collected from multiple lateral substations across the width of the river channel. Transect sampling is designed to estimate integrated water concentration and flux through a cross-section of the river or fraction of a cross-section at a point in time. Single-point samples are stationary samples or sample pairs at a constant depth.

Round 2A surface water samples were collected at three transect stations (RM 4, 6.3, and 11) and 20 single-point stations. Round 2A transect samples were collected as equal discharge increment (EDI)[3] composite samples across the channel. Round 2A single-point samples were collected as either NB or VI samples. Single-point NB samples were collected between 1 and 3 ft off the river bottom in water less than 20 ft deep. Single-point VI samples were vertically integrated from 1 ft below the water surface to within 1 to 3 ft of the river bottom at BHHRA stations.

Round 3A surface water samples were collected at six transect stations (RM 2, 2.9 [Multnomah Channel], 4, 6.3, 11, and 16) and 12 single-point stations. Round 3A transect samples were collected as either a single EDI composite sample, a pair (not composited) of NB/NS[4] samples horizontally integrated across the channel, or three VI composite samples at three points across the channel (located at east, middle, and west channel; VI-EMW). Round 3A single-point samples were collected as a pair (not composited) of NB/NS samples at a stationary point in the channel. The Round 3A single-point sampling stations were selected to provide water chemistry data in areas in

[3] At each Round 2A transect station, samples were spatially integrated across the entire width and depth of the channel based on a flow-weighted method (USGS 2000). At each transect, the river was divided into equal flow subareas (i.e., EDIs) using existing bathymetry and river flow data. VI samples were collected from 1 ft below the water surface to within 1–3 ft of the river bottom in each subarea; these samples were combined to produce integrated river cross-section composite samples.

[4] Surface water was collected from two points in the water column. The NB sample was collected at a depth of 1 ft off the river bottom. The NS sample was collected 3 ft below the surface.

higher flow conditions than were sampled in Round 2 to support source identification and the FS. Selected Round 3A single-point stations were relocated into deeper water to accommodate the NB/NS sampling.

XAD water samples were passed through a 0.5-µm filter and an XAD-2 resin column. Chemical mass captured on the filter provides the particulate fraction concentration, while chemical mass on the resin column provides the "dissolved fraction" concentration. Note that this approach to definition of the dissolved and particulate fractions of the whole sample constitutes an operational definition of these fractions. In reality, it is likely that colloidal-sized particles less than 0.5 µm in diameter are present in the "dissolved" fraction. This is a critical consideration in review of the data set.

A summary of the sampling methods employed at each station for all sampling events is presented in Table 5.3-8. Sampling techniques at each station included both standard (peristaltic) and high-volume (XAD) methods as described in detail in the surface water sampling FSPs (Integral 2004c, 2006m).

Sample locations and samples are labeled on the figures and tables according to the following scheme:

- High-volume XAD samples were collected as two analytical samples. The surface water particulates associated with the XAD filter are labeled in tables and figures as XAD-F or XAD filter. The sample from the XAD column represents the dissolved phase and is labeled as XAD-C or XAD column.
- In Round 3A, near-bottom and near-surface samples were collected and are labeled as NB and NS in the tables and figures.
- Also in Round 3A, three VI composite samples were collected at three points across the channel and these stations and samples are labeled as -E, -M, or -W for the samples collected on the east, mid-channel, and west sides of the river.

5.3.1.3 Data Processing and Calculations

For all analyte group total calculations in this report (total PCB congener, total PCDD/Fs, total DDx, and total PAHs), the RI method for summing parameters was applied.[5] All particulate and dissolved total PCB congener and total PCDD/F concentrations were above detection limits. One total DDx particulate concentration was non-detect; in this case, a value of one-half the detection limit was used in calculations and plots.

To support the data analysis presented in this section, several terms were calculated from the surface water data set: the fraction of organic carbon (f_{oc}), POC, average

[5] For all totals, zero is always assigned for individual non-detect values included in the group total. If any of the values included in a total are estimated (J-qualified), then the total value is estimated (J-qualified). If all analytes in a total are non-detects, then the highest detection limit is used for the total and the total is U-qualified. For additional information, see Section 2.1.4.

particle IC concentration (C_s), and apparent organic carbon partitioning coefficient (K_{oc}). The f_{oc} describes the percent of mass of the suspended solids that is made up of organic material, which is truly a TSS-normalized version of the POC value. The f_{oc} was calculated from the peristaltic TOC, dissolved organic carbon (DOC), and TSS values, using the following equation:

$$f_{oc} = \frac{TOC - DOC}{TSS} = \frac{POC}{TSS}$$

For the few cases where the sample DOC was greater than TOC, the POC and f_{oc} values were set equal to zero.

The POC was calculated in this study to represent how much adsorbed (to suspended particulate material) organic carbon was present in the surface water in each sample. This value was simply estimated by subtracting the peristaltic sample DOC results from the corresponding peristaltic sample TOC results.

$$POC = TOC - DOC$$

5.3.1.4 LWG Member Sampling Events

LWG members Siltronic, NW Natural, and the City of Portland conducted independent surface water sampling, as summarized below:

- Siltronic – In May and June 2005 during low flow river conditions, Siltronic collected 22 near-bottom surface water samples at stations located from RM 6.4 to RM 6.5. Samples were generally collocated with TZW sampling locations. Samples were collected with a peristaltic pump and analyzed for conventional parameters, VOCs, and SVOCs. The Supplemental Investigation Report (MFA 2005b) presents additional information on the Siltronic surface water samples.

- NW Natural – Surface water grab samples were collected at 20 locations near the Gasco site from RM 5.9 to RM 6.7. At each location, near-surface, mid-depth, and near-bottom samples were collected in October 2007, in low-flow conditions, during three tidal periods: ebb, slack high, and slack low. A total of 180 surface water samples were tested for conventional parameters, cyanide, and select metals. The Gasco surface water samples are discussed in more detail in the Offshore Groundwater Field Sampling Approach (Anchor 2006b) and the Offshore Investigation Report (Anchor 2008b).

- City of Portland – The City of Portland collected surface water TSS data between February 5, 1992 and March 15, 2006 (Sanders 2006). The surface water samples were collected at a 10-ft depth from the east, middle, and west locations along transects at RM 1.1, 6.8, 8.8, 12.7, 17.9, and 20, and were composited by transect.

TSS and organic carbon data from these three LWG member data sets were incorporated into the discussions of water chemistry presented in this nature and extent evaluation. No other analytes in the LWG member data sets were included on the surface water indicator chemical list.

5.3.2 Data Uncertainties

Uncertainty associated with the surface water data is related primarily to the representativeness of the analytical data set. The data sets are derived from grab samples instead of time-weighted composites, and a limited number of samples were collected under a limited number of flow conditions. This prohibits a quantitative statistical evaluation of temporal and flow variability in surface water quality and is an important source of uncertainty of unknown magnitude. As discussed in Section 2, only high quality Category 1 QA2 data were used in these nature and extent discussions.

5.3.3 Indicator Chemicals in Surface Water

The IC list for surface water is presented in Table 5.0-2. The selection of ICs was guided by the considerations provided in Section 5.0, in coordination with EPA. A total of 22 individual analytes and calculated chemical sums were identified as ICs for surface water. Low frequencies of detection in surface water samples resulted in fewer ICs than for sediment. Indicator chemicals for surface water are organized as follows:

- PCBs
  - Total PCBs*
  - PCB TEQ (ND=0)
- PCDD/Fs
  - Total PCDD/Fs*
  - TCDD TEQ (ND=0)*
- DDx
  - Total DDx (total of 2,4’ and 4,4’-DDD, -DDE, and -DDT)*
  - Total of 2,4’- and 4,4’-DDT
  - Total of 2,4’- and 4,4’-DDE
  - Total of 2,4’- and 4,4’-DDD
- Pesticides (non-DDx)
  - Total chlordanes*
  - gamma-HCH
  - Aldrin*

  - Dieldrin*
- PAHs
  - Total PAHs*
  - Total LPAHs
  - Total HPAHs
  - Total cPAHs BaPEq values
  - BAP
- SVOCs
  - Hexachlorobenzene
- Metals
  - Arsenic*
  - Copper*
  - Lead
  - Zinc.*

This section focuses on the distribution of a subset of 11 ICs, which are indicated with an asterisk (*) in the above list, in surface water in the Study Area, as well as upstream (i.e., RM 16) and at Multnomah Channel (approximately RM 2.9). The 11 ICs in surface water were selected from the chemicals for presentation and discussion in the site-wide CSM (Section 10). Four of these chemicals—total PCBs, total PCDD/Fs, total DDx, and total PAHs—were selected for in depth discussion. The discussion focuses primarily on a description of the data set for each analyte, including concentration ranges for the various sampling event types (low-flow, high-flow, and stormwater-influenced) and frequency of detection. For these four bounding ICs, the narrative also includes the relationship of chemical concentration to location, flow rate, association with suspended solids, and associated organic carbon.

The discussion of the other six ICs in the subset is abbreviated and includes a description of the data set, concentration ranges, and frequency of detection, with references to figures and tables to provide a complete picture of the distribution of those chemicals. Figures for the remaining 11 chemicals are included in Appendix D3.

The nature and composition of the four bounding ICs are discussed to provide background on their environmental chemistry and to provide a site-wide perspective on their distribution. Chemical distributions across abiotic and biotic media in relation to specific potential sources are discussed in more detail in Section 10. Finally, surface water sampling results are compared to various water screening values in Appendix D3.3. Contaminants that exceed one or more screening values are identified and will be

included in the contaminant mobility evaluation in the draft FS. Contaminants that pose potential risk to humans are discussed in the BHHRA (Appendix F).

5.3.4 Description of Surface Water Presentation Tools

The surface water chemistry distributions and supporting information are depicted in several graphical formats: hydrographs and hyetographs of sampling events, discharge rates, and precipitation events (Figures 5.3-1 through 5.3-7), and histograms of sample concentrations for all sampling events for the ICs, along with line plots, stacked bar charts, and scatter plots for the bounding ICs (Figures 5.3-8 through 5.3-120). The stacked bar charts and scatter plots present the data by river mile. Station W027 (RM 2.9) was located in Multnomah Channel, and Stations W018 (RM 8.3), W020 (RM 9.1), W021 (RM 8.7), and W035 (RM 8.5) were located in Swan Island Lagoon. Multnomah Channel and Swan Island Lagoon are indicated by "MC" and "SIL", respectively, on the histograms.

Hydrographs and Hyetographs: The hydrographs show the average discharge rates during the Round 2A and 3A surface water sampling events and the hyetographs show precipitation events and amounts to provide perspective on the timing of the Round 2A and 3A sampling events and the specific conditions prior to, during, and after each event.

Histograms: The histograms provide a graphical summary of the distribution of chemicals for all the surface water sampling events. For each analyte, data are presented in two types of histograms: data sorted by flow event type (high-flow, low-flow, and stormwater-influenced) and data sorted by location in the river channel (west and east channel and transect locations). For the XAD samples (units in pg/L), the individual data points are composed of shaded stacked bars to distinguish between the XAD column and XAD filter samples. The bars are color-coded to distinguish between the single-point samples (blue) and the transect samples (orange). The non-detected samples are displayed with a hatch pattern or open bar. The same scheme is used for the peristaltic samples (units in µg/L), with shaded stacked bars for total and dissolved fractions and blue and orange colors for the single-point and transect samples. In Round 2, stations were analyzed for PAHs in both the XAD and peristaltic samples; the histograms for the peristaltic samples only display samples where PAHs were not analyzed in the XAD sample.

Line Plots: The line plots present the concentrations of the bounding ICs for each flow type (high-flow, low-flow, stormwater-influenced) at the transect stations for all surface water sampling events. The squares, diamonds, and triangles represent the data points. Prior to generating the plots, data were averaged so that only one value per transect per sampling event was used. NB and NS total (dissolved plus particulate) concentrations were averaged for samples from stations W027 (Multnomah Channel), W005 (RM 4), W011 (RM 6.3), and W024 (RM 16) and east, west, and mid-channel total concentrations were averaged for stations W025 (RM 2) and W023 (RM 11), where applicable. The data for the 2007 high-flow event is displayed in two colors because

this event was completed in two phases with a stand-down period between high-flow conditions.

Scatter Plots: Scatter-plot presentations of the surface water data show concentrations of the four bounding ICs by river mile, flow, TSS, and organic carbon, and particulate versus dissolved concentrations are displayed for detailed evaluation of the results. The symbols on the scatter plots distinguish between flow types (high-flow, low-flow, stormwater-influenced) and single-point and transect samples.

Stacked Bar Charts: Bar charts are designed to reveal distinctive patterns in the relative abundance of bounding IC components. These graphs provide a line showing concentrations of the IC (i.e., totals) on a logarithmic scale, but do not display concentrations of the analyte components (e.g., homologs, isomers). The analyte components are shown as a percent of the total concentration. The samples are sorted by flow type and also by river mile within each flow type. Non-detect values were set equal to zero for these charts.

5.3.5 Suspended Solids

Suspended sediment loads and dynamics are potentially an important component of the LWR physical system, and TSS data have been collected as part of the surface water data collection effort to understand distributions and patterns of chemical concentrations. Peristaltic samples were collected and analyzed for TSS (reported in mg/L) during Round 2A and 3A. TSS concentrations in surface water are presented as a histogram arranged by flow conditions (low-flow, high-flow, and stormwater-influenced flow) and river mile in Figure 5.3-8. The same data are presented as a histogram by channel position (west and east channel and transect locations) on Figure 5.3-9. TSS data collected by LWG members are presented in Figures 5.3-10a, 5.3-10b, and 5.3-11.

TSS measured in single-point samples collected during low-flow conditions ranged from 3 J to 15 J mg/L at Station W013 (RM 6.7) in March 2005. Concentrations in transect samples during low flow ranged from undetected (at a detection limit of 1 U mg/L) to 25 J mg/L at Station W011 (RM 6.3) in July 2005.

TSS measured in single-point samples collected during high-flow conditions ranged from 13 to 60 mg/L at Station W031 (RM 6.1) in January 2007. Transect concentrations during high flow ranged from 9 to 62 mg/L at Station W023 (RM 11) in January 2006.

TSS concentrations in single-point samples collected during the November 2006 stormwater-influenced event ranged from 2 to 7 mg/L at Station W026 (RM 2.1). Concentrations in transect samples during the stormwater-influenced event ranged from undetected (at a detection limit of 1 UJ mg/L) to 6 mg/L at Station W005 (RM 4).

LWG members NW Natural and the City of Portland also collected TSS data from the Willamette River. The NW Natural TSS data ranged from 5 U to 14 mg/L. The City's TSS data corresponding to the timing of the Round 2A events and the Round 3A January 2006 high-flow event (extent of TSS data collected) are presented in Figures 5.3-10b and 5.3-11. TSS measurements during this time period ranged from undetected (2 mg/L) to 42 mg/L. The NW Natural and City of Portland TSS measurements collected during the Round 2A and 3A sampling events were comparable to the range of LWG TSS (undetected at 1 mg/L to 62 mg/L). Looking at the complete City of Portland data set collected from 1992–2007, the range of TSS concentrations (0.4 to 243 mg/L) measured in the City of Portland composite samples was wider than that measured by LWG or the City of Portland during the time period of the Round 2A and 3A sample collection events.

The TSS concentrations as a function of flow rate for all samples in the surface water data set are presented in Figure 5.3-12. A general trend of increasing TSS concentration with increasing flow rate is more readily apparent in Figure 5.3-13, which presents TSS concentration as a function of river mile. Figure 5.3-14 presents a line plot of TSS data collected at surface water transect stations, presented by event. Both of these figures show that TSS concentrations varied over a large concentration range (single-point and transect samples) upstream from RM 16 to 4. By RM 2, there is a decrease in the TSS concentration range.

Figures 5.3-15 and 5.3-16 present the f_{oc} on the TSS in each surface water sample as a function of flow rate and river mile, respectively. The surface water transect particulate and dissolved organic carbon data are presented by event on Figures 5.3-17 and 5.3-18. The f_{oc} values on the TSS range from 0 to 20 percent in the low-flow samples and 0 to 50 percent in the stormwater-influenced samples. Conversely, the f_{oc} on the TSS in high-flow samples is distinctly lower, ranging from 0 to less than 4 percent, suggesting the introduction of suspended particles with low organic carbon content during high-flow events. Generally low f_{oc} values may be a function of larger particles (lower surface area per volume and therefore fewer organic carbon binding sites) introduced during high-flow conditions.

Figure 5.3-19 presents a scatter plot of f_{oc} and TSS that summarizes the overall trend of solids concentrations and f_{oc} in the data set. High-flow samples tend to exhibit lower f_{oc} associated with TSS. The shape of the curve is largely driven by the fact that f_{oc} is a function of TSS. The suspended solids associated with the stormwater-influenced samples appear to have the highest levels of organic carbon content. The TSS concentrations and corresponding f_{oc} values vary somewhat between flow types, and the low-flow samples appear to fall between the high-flow and stormwater-influenced samples based on the level of organic carbon. There is the possibility that there may be local nearshore effects at the point of discharge that were not captured in the surface water sampling data set.

Some portion of the organic carbon content of suspended solids may be attributable to phytoplankton living in the Willamette River. In riverine systems, plankton are generally considered a minor component of the biota (Allan 1995; Horne and Goldman 1994). Plankton have limited powers of locomotion and therefore cannot maintain their presence in rapidly flowing waters (Wetzel 2001). Some species are conspicuously absent from running waters, whereas other taxa found in riverine systems tend to be associated with sheltered areas such as shallow littoral zones and backwater eddies and in pools in streams with well-developed pool-riffle hydrology (Pennak 1989). The high flow surface water sampling events were conducted in January and February, when plankton populations would be expected to be small due to seasonal conditions. This factor, in addition to the increase in large, low organic carbon particles, may contribute to the relatively low f_{oc} values measured in the high flow events.

5.3.6 Total PCBs in Surface Water

High-volume surface water samples were analyzed for PCB congeners (reported in units of pg/L) by HRGC/HRMS, and peristaltic samples were analyzed for PCB Aroclors (reported in units of µg/L) by routine methodology (i.e., EPA method 8081). There are 1,000,000 pg in 1 µg. PCB Aroclors were not detected in the majority of the peristaltic samples (approximately 90 percent undetected) at an average detection limit of 0.00125 µg/L.

Peristaltic samples (single-point samples) were collected and analyzed for PCB Aroclors during Round 2A low-flow conditions only (i.e., November 2004, March 2005, and July 2005). Total Aroclor concentrations ranged from below detection limits (0.0025–0.0027 U µg/L) to 0.015 µg/L. Detections were limited to six samples with concentrations ranging from 0.0047 J µg/L to 0.015 µg/L. Because of the low frequency of detection, these data are not summarized in the figures. The discussion of the extent of PCBs in the Study Area within this section is limited to PCB congeners (XAD samples) because of the high frequency of non-detects for the PCB Aroclor analysis.

Dissolved and particulate PCB congener concentrations in surface water XAD columns and filters are depicted in histograms in Figures 5.3-20 through 5.3-23. The figures show the data arranged by river mile and sampling conditions (high-flow, low-flow, and stormwater-influenced; Figures 5.3-20 and 5.3-21) or channel position (west and east channel and transect locations, Figures 5.3-22 and 5.3-23). Figures 5.3-21 and 5.3-23 are scaled versions of Figure 5.3-20 and 5.3-22, respectively, showing details of lower concentrations. Each of the histograms presents the particulate (0.5-µm filter) and dissolved (XAD column) fractions of the total concentration as stacked bar graphs.

Total PCB congener concentrations (the sum of the dissolved and particulate concentrations) measured in single-point samples during low-flow conditions ranged from 375 J pg/L to 12,000 J pg/L (Station W013 at RM 6.7 collected in March 2005). Concentrations measured in transect samples during low-flow ranged from 159 J pg/L to 950 J pg/L (Station W005 at RM 4 in September 2006).

Total PCB congener concentrations measured in the single-point samples during high-flow conditions ranged from 111 J pg/L to 749 J pg/L at Station W035 (RM 8.5) in January 2007. Concentrations measured in transect samples during high-flow ranged from 41.9 J pg/L to 391 J pg/L at Station W005 (RM 4) in January 2007.

A single stormwater-influenced flow event was observed and sampled in November 2006. Total PCB congener concentrations measured in single-point samples during this stormwater-influenced event ranged from 112 J pg/L to 2,590 J pg/L at Station W030 (RM 5.5). Concentrations measured in transect samples ranged from 121 J pg/L to 1,290 J pg/L at Station W025E (RM 2).

The following subsections describe observations of total PCB congener concentrations (PCBs) in the complete Round 2A and 3A data set. The spatial distribution of dissolved and particulate PCB concentrations and relationships to flow rate, TSS, and f_{oc} are described. The composition of this multi-component IC is also described. Total PCB congeners were detected in all XAD filter and column samples collected during Round 2A and 3A sampling events.

5.3.6.1 PCB Spatial Distribution

The distribution of total PCB concentrations by river mile throughout the Study Area (and upstream to RM 16 and downstream to Multnomah Channel) is presented in Figures 5.3-24 and 5.3-25. (Note the symbols and colors indicate the sample type—point vs. transect—and the general flow conditions of the sampling event—low-flow, stormwater-influenced, or high-flow.) The majority of the highest total PCB concentrations (six out of seven >2,000 pg/L) were associated with single-point samples collected at RM 6.7 (Station W013) within Willamette Cove during low-flow conditions. The fourth highest single-point concentration was collected at RM 5.5 (W030 during the stormwater-influenced event). These data suggest that local PCB sources may exist in this region of the Study Area. The range of total PCB concentrations within the complete data set across the Study Area was fairly consistent between RM 11 and 2, excluding the highest single-point concentrations, and elevated concentrations near the east side of the river at RM 6.7.

Upstream of the Study Area, the total PCB concentrations at RM 11 were consistently higher than at RM 16, suggesting the existence of a source or sources between these two locations. Two of the three highest total PCB concentrations at RM 11 were from the sampling stations on the east side of the channel (Figure 5.3-21), indicating a possible source in this area. The second highest result at RM 11 was from a Round 2A vertically- and horizontally-integrated transect, and the field crew noted stormwater runoff entering the east side of the channel during collection of this sample (Jones 2007, pers. comm.). Within the Study Area, total PCB concentrations continued to increase between RM 11 and RM 4 in six of seven transect-based sampling events (the sole exception is the November 2004 low-flow sampling event). Total PCB concentrations at both RM 2 and in Multnomah Channel transects generally decreased from those at RM 4 but remained higher than those at RM 16. An exception to this was the RM 2

total PCB concentration from the November 2006 stormwater-influenced event, which was higher than other transect concentrations measured in that event.

The dissolved and particulate fractions of total PCBs are presented by river mile on Figures 5.3-26 and 5.3-27, respectively. Figure 5.3-28 presents the particulate data with a reduced y-axis scale to better show lower-concentration data. For both dissolved and particulate data, the highest concentrations (>1,000 pg/L) were measured at RM 6.7 (Station W013) within Willamette Cove during low-flow conditions. In addition to the Willamette Cove samples, a high concentration (2,370 pg/L) dissolved PCB sample was measured during stormwater conditions at the near-surface Station W030 at RM 5.5, and a high concentration (1,120 pg/L) particulate PCB sample was measured at RM 8.3 (W018) during low flow. With the exception of these individual points, particulate PCBs were fairly consistent across the Study Area between RM 11 and 2. Dissolved PCBs appear to increase somewhat from RM 11 downstream to RM 2 during the stormwater sampling event; low-flow and high-flow dissolved samples cover constant ranges across the Study Area. Finally, upstream of the Study Area, the particulate and dissolved concentrations in RM 11 EDI and east-channel samples were higher than concentrations measured at RM 16, again suggesting the existence of a source or sources between these two locations.

Line plots of total, particulate, and dissolved PCBs measured at transect stations are presented on Figures 5.3-29 through 5.3-31. These figures support the suggestion of a possible source driving increased concentrations from RM 16 to 11. Total and particulate concentrations also show a sharp decline at the RM 2 downstream transect; dissolved concentrations remained fairly constant across the site and did not decrease at RM 2. These observations are consistent with expected settling of suspended particles at the downstream end of the Study Area where the river velocity decreases.

5.3.6.2 PCB Relationships to Flow Rate

Total PCB concentrations as a function of flow rate are presented in Figures 5.3-32 and 5.3-33. Twelve concentrations above 1,000 pg/L were associated with samples collected during low-flow events. PCB concentrations were consistently lower in high-flow samples as compared to the low-flow and stormwater-influenced samples, suggesting that inflow concentrations at high flow rates overwhelm local effects and control PCB concentrations until flow rates subside. No concentrations above 1,000 pg/L were observed at flows of 60,000 cfs and concentrations were below 210 pg/L for the 160,000 cfs event.

Figures 5.3-34 and 5.3-35 show the dissolved and particulate fractions of total PCBs plotted against flow rate. Figure 5.3-36 presents the particulate data with a reduced y-axis scale to better show lower-concentration data. Similar to total PCBs, all of the particulate and dissolved samples with concentrations >1,000 pg/L were collected during low-flow conditions, with the exception of a single dissolved sample collected during the stormwater-influenced sampling event. For the particulate fraction, low-flow samples cover a greater concentration range as compared to the high-flow and

stormwater-influenced samples. For the dissolved fraction of total PCBs, low-flow and stormwater-influenced samples cover similar concentration ranges, while high-flow exhibit generally lower concentrations.

5.3.6.3 Distribution between PCB Dissolved and Particulate Fractions

The following subsections describe the observed trending of dissolved and particulate fractions by river mile, event type, sample type, TSS, and f_{oc} of the TSS. This analysis was specific to total PCBs and, therefore, does not extend to individual congener analysis. However, Section 5.3.6.4 provides a discussion of PCB composition (i.e., homologs) and observed patterns, which may assist with the identification of observed PCBs in the Study Area.

5.3.6.3.1 PCB Particulate and Dissolved Concentrations

Figures 5.3-37 and 5.3-38 present a scatter plot of total PCB dissolved and particulate concentration pairs. Linear regression lines are shown for the high-flow, low-flow, and stormwater-influenced sets of transect and point data. The PCB concentrations of the particulate (filter) and dissolved (column) fractions of each sample are shown on histogram Figures 5.3-20 through 5.3-23.

As seen on the scatter plots, samples with concentrations less than 500 pg/L particulate and less than 500 pg/L dissolved showed a fairly consistent trend in particulate/dissolved ratios, whereas the particulate/dissolved ratio varies substantially for high-concentration samples. In particular, low-flow point samples collected at the upper end of the dissolved concentration range (>500 pg/L) had a higher particulate component. Stormwater-influenced point samples collected at the upper end of the dissolved concentration range had the lowest particulate component.

The high-flow samples (both point and transect) tend to show lower dissolved concentrations for the given particulate concentration (or conversely, higher particulate concentrations for the given dissolved concentrations), relative to the stormwater and low-flow samples. This suggests a different character/source of PCB contaminated sediment and/or suspended solids concentration and character during high-flow events. The following section further investigates differences in the suspended solids concentration and character.

The transect sample collected at RM 11 during the low-flow event in November 2004, exhibited a high particulate to dissolved ratio. As noted previously, during collection of this sample, the field crews observed runoff from a nearby storm drain, which may have contributed to this result.

5.3.6.3.2 PCB Associations with Suspended Solids

The associations of PCBs with suspended solids and apparent partitioning were evaluated from the surface water data set. Total PCB concentrations as a function of TSS are presented in Figures 5.3-39 and 5.3-40. High-flow samples (single-point and transect) exhibited the widest range and highest concentrations of TSS but the lowest

PCB concentrations. Conversely, the remaining samples exhibited a wide range in concentration over a small range in TSS. As stated above, the high-flow samples also exhibited a lower dissolved/particulate concentration ratio relative to the stormwater-influenced and low-flow samples.

Particulate total PCB concentrations and POC (TOC–DOC) concentrations are compared in Figures 5.3-41 and 5.3-42. The high-flow samples (single-point and transect) exhibited relatively low PCB concentrations for the corresponding POC associated with the solids. The low POC values are consistent with the lower f_{oc} associated with TSS observed in high-flow samples, as shown on Figure 5.3-19. As discussed in Section 5.3.6.2, this observation may suggest the introduction of suspended particles with low organic carbon content during high-flow events. Further, the solids that become suspended in the water column during high-flow events may have a different character (low f_{oc} and low PCB concentrations) than those introduced during low-flow or stormwater-influenced events. The single-point samples characterized by high concentrations of PCBs (RM 5.5, 6.9, 8.3, and 11) did not have corresponding high TSS concentrations from the same events (Figures 5.3-39 and 5.3-40). However, these samples did exhibit high particulate-phase PCB concentrations as a function of POC (Figures 5.3-41 and 5.3-42), reflecting a higher f_{oc}.

5.3.6.4 PCB Patterns and Trends

The patterns and trends of PCB homologs in surface water XAD column and filter samples are discussed below. PCB homologs are described in Section 5.1.5. Stacked bar graphs depicting the PCB homolog distributions in the surface water dissolved (XAD column) and particulate (XAD filter) samples by flow condition and river mile are presented in Figures 5.3-43 and 5.3-44. The discussion of patterns and trends in the PCB homologs is qualitative based on visual examinations of the stacked bar graphs.

The PCB composition of the particulate fraction generally exhibited a more highly chlorinated pattern than the PCBs in the dissolved phase. Several November 2006 stormwater-influenced samples, which exhibited greater contributions from monoCBs and/or diCBs than other particulate samples and their dissolved counterparts, were exceptions to this pattern. This PCB distribution reflects the greater solubility and lower partitioning coefficients of the less-chlorinated congeners, as described in Section 6.

The PCB composition was generally similar in the dissolved fractions of the four Study Area transect locations and the downstream transect (Station W027, Multnomah Channel) during all sampling events. The dissolved PCBs at the transect locations generally consisted of a fairly equal mixture of diCBs, triCBs, tetraCBs, and pentaCBs, with contributions from hexaCBs. The sample from transect W005 (RM 4) collected during the January 2006 high-flow event had a high percentage of tetraCBs and no diCBs. However, this sample had a relatively low total PCB congener concentration (73 pg/L), and therefore, this homolog distribution may be reflective of this low concentration.

Generally, the PCB compositions in the dissolved fraction of the upstream transect (W024, RM 16) exhibited a less chlorinated pattern than the PCBs in the Study Area transect samples. The samples from the low-flow events collected at Station W024 had high proportions of diCBs. Conversely, the sample from this station collected during the January 2007 high-flow event was characterized by a high proportion of monoCBs. The PCB homolog patterns in stormwater-influenced samples collected from W024 were similar to stormwater-influenced samples from the Study Area.

The more highly chlorinated PCB composition of the particulate fractions of the transect locations is reflected in the predominance of tetraCBs, pentaCBs, hexaCBs, and heptaCBs, with spatial variations in the proportions of each homolog present. The PCBs at transects W023 (RM 11) in November 2004 (low-flow), from W023E in September 2006, and the NB sample from W005 (RM 4) in January 2007 (high-flow) contained higher proportions of hexaCBs and heptaCBs than samples collected during other events. The sample from transect W025E (RM 2) collected during the November 2006 stormwater-influenced event contained a higher proportion of triCBs than samples from other events. The W023 (RM 11) sample collected during the November 2004 low-flow event may have been influenced by stormwater discharges from a nearby outfall (OF-45) due to a relatively heavy rainfall event (0.37 inches) on the day the sample was collected (Integral 2006a); however, the September 2006 low-flow sample from the east side of this transect had a higher total PCB concentration and a similar PCB composition to the November 2004 sample. No stormwater discharges were noted during the collection of the September 2006 sample. The interpretation of these surface water samples may be confounded by maintenance dredging conducted at Glacier's docking facility located at RM 11.3. Glacier apparently conducted dredging following approval in June 2004 by the USACE. Attempts by the LWG to obtain post-dredging information have not been successful.

The PCB composition of particulate fractions of the upstream transect Station W024 (RM 16) generally exhibited a less chlorinated pattern than the PCBs in the Study Area transect samples upstream of RM 2 for the low- and high-flow sampling events. The stormwater-influenced (September 2006) NB sample at this station had a similar PCB composition to the Study Area transect profiles, while the NS transect from this event exhibited the same less-chlorinated pattern observed at this station during the other sampling events.

The particulate fractions of the Multnomah Channel transect, Station W027, exhibited PCB profiles similar to that of the Study Area transect stations during the high- and low-flow sampling events. The PCB profile at Station W027 during the September 2006 stormwater-influenced event was similar to the profiles of the west and mid-channel samples from transect W025 (RM 2) collected during the same event, exhibiting a more highly chlorinated PCB profile. The east sample from transect W025 exhibited a less chlorinated PCB profile than other transect stations during the stormwater-influenced event. PCB patterns in several other samples from location W025 were also less chlorinated than other Study Area transect samples.

Several of the single-point particulate fraction samples from the November 2006 stormwater-influenced sampling contained unusually high proportions of the less chlorinated homolog groups in comparison to the other sampling events. At Station W037 (RM 9.6, west bank), the NB sample had a higher proportion of diCBs and the NS sample had higher proportions of monoCBs and diCBs. Similarly, the Station W038 (RM 9.9, east bank) NB sample had higher proportions of monoCBs and diCBs. Higher proportions of monoCBs and diCBs were observed in the NS sample at Station W031 (RM 6.1) and in the NB sample at Station W028 (RM 3.6). The dissolved fractions of these samples had a similar homolog distribution to the other samples from these locations. MonoCBs accounted for 91 percent of PCBs in the dissolved fraction of the NS sample at Station W030 (RM 5.5). This sample had a high total PCB congener concentration (2,370 J pg/L). With the exception of Station W037 (RM 9.6), the corresponding NS or NB samples did not exhibit similar distributions. These findings would point to a stormwater-related source of monoCBs and diCBs near these locations.

The PCB homolog composition in samples from the NS and NB locations differed from that of the transect samples to varying degrees, likely reflecting local sediment conditions or upland sources. With the exceptions mentioned above, the NS and NB samples exhibited similar PCB distributions at both the NB/NS transect and single-point stations. The PCBs in Swan Island Lagoon, Stations W018 (RM 8.3) and W035 (RM 8.5), were slightly more chlorinated overall than PCBs in the transect samples and the sampling locations in the main river channel, with the exception of particulate fractions from Willamette Cove (Station W013, RM 6.7) and Station W029 (RM 4.4), both of which are discussed below. The predominant homolog groups in this area are pentaCBs, hexaCB, and heptaCBs.

At Station W015 (RM 6.9) on the west bank, the abundance of tetraCBs was generally slightly less than at Station W016, located less than half a mile upstream. However, in November 2004, the triCBs were unusually abundant at this station in both the particulate and dissolved fractions. The total PCB congener concentration in the November 2004 sample was also unusually high (1,290 pg/L in the particulate fraction and 639 J pg/L in the dissolved fraction). The PCB concentration and homolog profiles imply contributions from an additional source or sources in November 2004. Because rainfall of 0.24 inches was recorded on this sampling date (Integral 2006a), a source or sources related to stormwater discharge near this sampling location is possible.

In Willamette Cove, Station W013 (RM 6.7) east bank, the total PCB concentration was high relative to other areas (maximum of 9,560 J pg/L for particulate PCBs and 2,420 J pg/L for dissolved PCBs). The PCB profile in the particulate fraction was more chlorinated than other areas, dominated by hexaCBs and heptaCBs. The dissolved PCB profile was similar to that of the transect samples.

5.3.7 Total PCDD/Fs in Surface Water

High-volume surface water samples were analyzed for PCDD/Fs by HRGC/HRMS. Concentrations of PCDD/Fs in surface water XAD column and filter samples are depicted in Figure 5.3-45. Dissolved and particulate PCDD/F congener concentrations in surface water XAD columns and filters are depicted as a histogram with data arranged by river mile and sampling conditions (high-flow, low-flow, and stormwater-influenced). Figure 5.3-46 presents the same data as a histogram sorted by channel position (west and east channel and transect locations). Each of the histograms presents the particulate (0.5-µm filter) and dissolved (XAD column) fractions of the total concentration as stacked bar graphs.

Total PCDD/Fs (dissolved plus particulate concentrations) measured in single-point samples collected during low-flow conditions ranged from 30.7 pg/L to 162 pg/L at Station W013 (RM 6.7) collected in July 2005. Concentrations measured in transect samples collected during low-flow conditions ranged from 5.9 J pg/L to 51.6 J pg/L at Station W005 (RM 4) in September 2006.

Total PCDD/Fs measured in single-point samples collected during high-flow conditions ranged from 24.7 pg/L to 74.9 pg/L at Station W035 (RM 8.5) in January 2007. Concentrations measured in transect samples collected during high-flow conditions ranged from 5.3 J pg/L to 44.0 pg/L (Station W005 at RM 4 in January 2006).

Total PCDD/Fs measured during the stormwater-influenced event (November 2006) ranged from 36.0 pg/L to 55.2 J pg/L (Station W032 at RM 6.7) in single-point samples and 5.5 J pg/L to 118 pg/L (Station W023E) at RM 11 in transect samples.

The following subsections describe observations of total PCDD/Fs congener concentrations. The spatial distribution of dissolved and particulate concentrations and relationships to flow rate, TSS, and f_{oc}, as well as the composition of this multi-component IC are described. Total PCDD/Fs were detected in all XAD filter and column samples collected during Round 2A and 3A sampling events.

5.3.7.1 PCDD/F Spatial Distribution

The distribution of total PCDD/F concentrations (dissolved plus particulate concentrations) by river mile throughout the Study Area (and upstream to RM 16 and downstream to Multnomah Channel) is presented in Figure 5.3-47. The highest concentration (162 pg/L) was measured in the single-point sample collected at RM 6.7 (Station W013) during the July 2005 low-flow sampling event. The concentration measured in the field replicate from this location (30.8 pg/L) collected the following day was much lower. The second highest PCDD/F concentration (118 pg/L) was measured in the transect sample from RM 11 on the east side of the river (Station W023E), collected during the stormwater-influenced event (November 2006). This value is more than twice the concentration of the other samples collected at RM 11, which were all less than 40 pg/L. This observation may indicate the presence of a localized PCDD/F source or sources within this area.

The next highest concentrations were measured at RM 6.9 (third highest concentration; Station W015) and RM 8.5 (fourth and fifth highest concentrations; Station W035). The distribution of PCDD/Fs throughout the Study Area and upstream to RM 16 did not vary greatly, with the exception of the five highest concentrations. The lowest concentrations were generally measured at RM 2 (Figures 5.3-45 and 5.3-46).

At the downstream end of the Study Area total PCDD/F concentrations in Multnomah Channel were generally higher than those at RM 16 while total PCDD/F concentrations at RM 2 were generally lower than those at RM 16 (Figure 5.3-48). From sampling event to sampling event, total PCDD/F concentrations within the Study Area were variable and did not display consistent trends from upstream to downstream.

5.3.7.2 PCDD/F Relationships to Flow Rate

Total PCDD/F concentrations as a function of flow rate are presented in Figure 5.3-49. Two of the three highest concentrations were measured in single-point samples during low-flow conditions. The second highest concentration was measured in a transect sample on the east side of the river at RM 11 during the stormwater-influenced event. However, excluding these high-concentration samples, there does not appear to be a trend towards high total PCDD/F values associated with low-flow conditions. At the downstream end of the Study Area, the low-flow sample concentration ranges decreased relative to ranges observed within the Study Area.

5.3.7.3 Distribution between PCDD/F Dissolved and Particulate Fractions

The following subsections describe the observed trending of dissolved and particulate fractions by river mile, event type, sample type, TSS, and f_{oc} of the TSS. This analysis was specific to total PCDD/Fs and therefore does not extend to individual dioxins and furans. However, Section 5.3.7.4 provides a discussion of PCDD/F composition and patterns, which may assist with the identification of observed concentrations in the Study Area.

5.3.7.3.1 PCDD/F Dissolved and Particulate Concentrations

The dissolved (XAD column) and particulate (XAD filter) fractions of total PCDD/F concentrations for each surface water sample are presented as histograms by flow event type on Figure 5.3-45 and by channel position on Figure 5.3-46. These figures clearly demonstrate that PCDD/Fs tend to partition to the particulate fraction in surface water within the Study Area. The extent of this tendency is presented in Figure 5.3-50, which shows a scatter plot of the dissolved vs. particulate concentrations of total PCDD/Fs. The two highest concentrations measured at RM 6.7 and 11 during low-flow and stormwater-influenced conditions, respectively, exhibit extremely high particulate to dissolved ratios (greater than an order of magnitude difference between the two phases). If these samples are excluded from the analysis, as presented in Figure 5.3-51, where the x-axis (particulate concentration) is zoomed, the tendency for partitioning into the solid phase is also apparent for low-flow and high-flow samples. However, the

stormwater-influenced sample concentrations appear to be more evenly distributed within the data set.

5.3.7.3.2 PCDD/F Associations with Suspended Solids

The associations of total PCDD/Fs with suspended solids and apparent partitioning were evaluated from the surface water data set. Total concentrations as a function of TSS are presented in Figures 5.3-52 and 5.3-53. PCDD/F concentrations increased slightly with higher suspended solids for high-flow transect samples. However, concentrations appear to increase more rapidly with increased suspended solids for the high-flow single-point samples. The stormwater-influenced samples increased independently of suspended solids in surface water. The transect and single-point samples collected during this event were all characterized by TSS values less than 10 mg/L. The low-flow transect samples appear to have a non-linear distribution with varying TSS and PCDD/F concentrations and, therefore, do not exhibit a clear trend (Figures 5.3-52 and 5.3-53).

Particulate total PCDD/F concentrations and POC concentrations are compared in Figures 5.3-54 and 5.3-55. As with PCBs, the high-flow samples (single-point and transect) exhibited relatively low PCDD/F concentrations for the corresponding POC associated with the solids. Conversely, the stormwater-influenced samples tended to exhibit higher POC values in relation to total PCDD/F concentrations and were associated with lower TSS (less than 10 mg/L) values. This may indicate that solids that become suspended during stormwater-influenced events may have a unique character of high f_{oc} and varying loads of PCDD/Fs. Samples characterized by high concentrations of PCDD/Fs did not have corresponding high TSS concentrations compared to other samples from the same events (Figures 5.3-52 and 5.3-53). However, these samples did exhibit a high particulate-phase PCDD/F concentration as a function of POC.

5.3.7.4 PCDD/F Patterns and Trends

Stacked bar graphs depicting the PCDD/F homolog distributions in the surface water dissolved (XAD column) and particulate (XAD filter) samples by flow condition and river mile are presented in Figures 5.3-56 and 5.3-57. The discussion of patterns and trends in the PCDD/F homologs is qualitative based on visual examinations of the stacked bar graphs.

Overall, the PCDD/F homolog profiles were dominated by OCDD and HpCDDs in both the dissolved and particulate fractions. At most transect stations and in most sampling events, OCDD constituted more than half of the PCDD/Fs in the particulate fraction, and from 30 to 95 percent of the PCDD/Fs in the dissolved fraction. HpCDDs accounted for another 15 to 30 percent of the total in both the dissolved and particulate fractions. As would be expected, the dissolved fraction generally contained a greater abundance of the less chlorinated homolog groups than the particulate fraction. The homolog pattern for the particulate fraction PCDD/Fs varied little from transect to transect and from upstream to downstream of the Study Area.

In the dissolved fraction of the transect samples, only a few samples differed from the trend of OCDD and HpCDD dominance. The NS sample from the eastern bank at transect W024 (RM 16) and the eastern bank sample from transect W025 (RM 2) from the January 2007 sampling event, and the November 2006 sample from the west bank at transect W025 exhibited high abundance of TCDDs. Two samples from the November 2006 sampling event had very low (<0.01 pg/L) total PCDD/F concentrations, and only TCDDs were detected in these samples. With these exceptions, only minor differences between the transects were observed; the low-flow sampling events exhibited higher proportions of the less chlorinated homolog groups in the dissolved fraction, and the upriver stations exhibited a less chlorinated profile than those in the Study Area.

The NS and NB transects had very similar particulate PCDD/F homolog profiles during all sampling events, with similar dissolved profiles during low-flow events. Homolog patterns in the NS samples were different than their corresponding NB samples at Stations W011 (RM 6.3) and W005 (RM 4) in the November 2006 stormwater-influenced sampling event and at Station W024 (RM 16) in the January 2007 high-flow sampling event. The NS and NB single-point samples also generally exhibited similar particulate and dissolved PCDD/F homolog profiles during all events.

The single-point samples from Stations W035 (RM 8.5) and W033 (RM 7) exhibited similar PCDD/F homolog profiles to the transect stations in both the dissolved and particulate fractions under high-flow and stormwater-influenced conditions. These stations were not sampled under low-flow conditions.

However, at Station W015, on the western riverbank at RM 6.9, furans were somewhat more abundant than at other locations during all sampling events, in both the dissolved and particulate fractions. The pattern of the dioxins was similar to that of the transect locations. This area may be influenced by a different PCDD/F source compared to the river as a whole.

In Willamette Cove, Stations W013 and W032 (both at RM 6.7), the profile for PCDD/Fs in the dissolved and particulate fractions generally resembled the profiles at the transect locations during each sampling event, but the particulate PCDD/F concentrations found at this location were consistently higher than at nearby transect Station W011 (RM 6.3). The PCDD/F patterns in field replicate samples LW2-W3013-2 F (particulate phase, July 2005) and LW2-W2013-2 C (dissolved phase, March 2005) included a distinctly larger fraction of dioxins and a smaller furan fraction than the respective "parent" samples and were not similar to transect Station W011. These replicate samples were collected sequentially, a day apart, and may reflect temporal differences in the PCDD/Fs in Willamette Cove.

5.3.8 Total DDx in Surface Water

High-volume samples were analyzed for total DDx by HRGC/HRMS. Concentrations of total DDx in surface water XAD column and filter samples are depicted as histograms with data arranged by river mile and sampling conditions (high-flow, low-

flow, and stormwater-influenced) in Figures 5.3-58 and 5.3-59. Peristaltic total DDx data are presented in Figure 5.3-60. Figure 5.3-61 through 5.3-63 present the same data as histograms sorted by channel position (west and east channel and transect locations). Each of the XAD histograms presents the particulate (0.5-µm filter) and dissolved (XAD column) fractions of the total concentration as stacked bar graphs. The peristaltic data histograms present the total (filtered plus unfiltered) fraction.

Total DDx was detected in virtually all of the XAD samples (99 percent of XAD filters and 100 percent of XAD columns) collected during Round 2A and 3A sampling events. In contrast, total DDx was detected in only 34 percent of the peristaltic samples, due to elevated detection limits based on smaller sample volumes than those collected for XAD samples.

Total DDx measured in single-point XAD samples collected during low-flow conditions ranged from 49.2 J pg/L to 9,760 pg/L (Station W016 at RM 7.2 in July 2005). Concentrations measured in transect samples collected during low-flow ranged from 42.8 J pg/L to 546 J pg/L (Station W005 at RM 4 in September 2006).

Total DDx measured in single-point XAD samples collected during high-flow conditions ranged from 266 J pg/L to 857 J pg/L (Station W031 at RM 6.1) in January 2007. Concentrations measured in transect samples collected during high-flow ranged from 162 J pg/L to 618 J pg/L (Station W023E) at RM 11 also in January 2007.

Total DDx measured in single-point XAD samples collected during the stormwater-influenced event (November 2006) ranged from 76.7 J pg/L to 675 J pg/L at Station W033 (RM 7). Concentrations measured in transect samples ranged from 33.2 J pg/L to 201 J pg/L at Station W011 (RM 6.3).

Total DDx was detected in only four peristaltic samples collected from single-point stations during low-flow conditions. Concentrations ranged from 0.000693 J µg/L to 0.0187 NJ µg/L at Station W001 (RM 2 in March 2005).

Total DDx was detected in 12 of 14 single-point peristaltic samples collected during high-flow conditions. Concentrations ranged from 0.00017 J µg/L to 0.00205 NJ µg/L at Station W037 at RM 9.6 in January 2007.

Concentrations of total DDx measured in peristaltic samples during the stormwater-influenced event ranged from 0.0015 µg/L to 0.0047 NJ µg/L at Station W037 (RM 9.6).

The following subsections present observations of total DDx concentrations in the combined Round 2A and Round 3A surface water data set.

5.3.8.1 DDx Spatial Distribution

The distribution of total DDx concentrations (dissolved plus particulate concentrations) by river mile throughout the Study Area (and upstream to RM 16 and downstream to

Multnomah Channel) is presented in Figures 5.3-64 and 5.3-65 (scatter plots), 5.3-66 (line plot), as well as 5.3-58 and 5.3-59 (histograms). The highest concentrations were measured in single-point samples collected during low-flow conditions near the middle of the Study Area at RM 6.9 (Station W015; 3,590 to 7,660 pg/L) and RM 7.2 (Station W016; 1,240 J to 9,760 pg/L). Excluding these higher concentrations, the overall range of observed concentrations across the Study Area and upstream to RM 16 was fairly consistent. Further, concentrations measured in the high-flow samples were generally higher than those associated with the low-flow and stormwater-influenced samples. This was especially evident at RM 11 and 16, where the high-flow concentrations were much greater than the low-flow and stormwater-influenced samples. These high-flow sample concentrations also covered a fairly consistent range across the Study Area, including RM 2.9 (Multnomah Channel) and RM 16 (Figures 5.3-58 and 5.3-59). This observation may suggest that DDx concentrations in the majority of the Study Area during high-flow are primarily controlled by upstream loading (see Section 6.1.1). Finally, the stormwater and low-flow sample concentrations covered a low range through RM 16 and 11, and increased around RM 7.5, suggesting a potential source or sources in this portion of the Study Area.

Total DDx concentrations increased from upstream (RM 16 and 11) to downstream (RM 4) in six of seven transect sampling events (Figure 5.3-66); the sole exception was the February 2007 high-flow event, during which concentrations dropped slightly from RM 11 to RM 4. Further downstream at RM 2.9 (Multnomah Channel) and RM 2, transect total DDx concentrations were also generally higher than upstream transect concentrations in low-flow and stormwater-influenced conditions, but were generally lower than upstream concentrations in high-flow conditions.

5.3.8.2 DDx Relationships to Flow Rate

Total DDx concentrations as a function of flow rate are presented in Figures 5.3-67 and 5.3-68. With the exception of the highest total DDx concentrations that were measured at RM 6.9 and 7.2 (as discussed above in Section 5.3.8.1), the range of concentrations was fairly consistent across all flow regimes (Figure 5.3-59). Again, this indicates that total DDx is transported from upstream sources into the Study Area during high-flow conditions. This figure does not demonstrate a clear relationship between flow rate and total DDx concentrations; however, the highest concentrations were measured during low-flow conditions (Figure 5.3-67). This may suggest the potential for localized sources within the mid-Study Area vicinity, especially near RM 6.9 and 7.2.

5.3.8.3 Distribution between DDx Dissolved and Particulate Fractions

The following subsections describe the observed trending of total DDx dissolved and particulate fractions by river mile, event type, sample type, TSS, and f_{oc} of the TSS. Section 5.3.8.4 provides a discussion of total DDx composition and patterns and trends, which may assist with the identification of observed concentrations in the Study Area.

5.3.8.3.1 DDx Particulate and Dissolved Concentrations

The distribution of total DDx (i.e., the sum of the dissolved and particulate concentrations) by river mile is presented on histograms by flow event type in Figures 5.3-58 and 5.3-59 and histograms by channel position in Figures 5.3-61and 5.3-62. Although these figures show distinctly high total DDx concentrations at RM 6.9 and 7.2, there does not appear to be any clear patterns in the ratio of dissolved and particulate fractions or the distribution between single-point and transect samples.

The dissolved and particulate fractions of total DDx are plotted on scatter plots in Figures 5.3-69 and 5.3-70. Linear regression lines are shown for the high-flow, low-flow, and stormwater-influenced sets of transect and point data. The single-point and transect samples collected during the high-flow events followed the general trend of the other flow types. The samples characterized by the highest dissolved concentration tended to have a higher particulate concentration.

Three samples collected at RM 2 (Station W025) had higher dissolved to particulate ratios. This may be due to the lower suspended solids load in the downstream portion of the Study Area (at RM 2) rather than an actual shift in partitioning behavior. There does not appear to be a clear distinction between low-flow and stormwater-influenced or single-point and transect samples in these figures.

5.3.8.3.2 DDx Associations with Suspended Solids

The associations of total DDx with suspended solids and apparent partitioning were evaluated from the surface water data set. Total concentrations as a function of TSS are presented in Figures 5.3-71 and 5.3-72. The high-concentration, low-flow samples exhibited the highest ratios of total DDx to TSS. The single-point and transect high-flow samples exhibited a much lower ratio of total DDx concentration to TSS. The low-flow and stormwater-influenced samples had low suspended solids loads (25 J mg/L or lower) compared to high-flow samples (up to 62 mg/L).

Particulate total DDx concentrations and POC concentrations are compared in Figures 5.3-73 and 5.3-74. The high-flow transect and single-point samples tended to exhibit higher total particulate DDx concentrations and lower POC concentrations in the water column. As discussed in Section 5.3.4 and shown on Figure 5.3-19, high-flow samples exhibited higher TSS concentrations and lower f_{oc} on TSS percentages. Therefore, the higher concentrations in the surface water during high-flow events (Figures 5.3-67 and 5.3-68) were present in spite of lower POC in the water column. Again, this may suggest a different source or sources of particles, possibly upstream of the Study Area, given the high inflow concentrations at RM 16 and 11 during high-flow events. Higher POC concentrations were found in transect and single-point stormwater-influenced and low-flow samples with lower total particulate DDx concentrations.

5.3.8.4 DDx Patterns and Trends

Stacked bar graphs depicting the DDx patterns for the surface water dissolved (XAD column) and particulate (XAD filter) samplings by flow condition and river mile are

presented in Figures 5.3-75 and 5.3-76. The discussion of patterns and trends in the DDx distributions is qualitative based on visual examinations of the stacked bar graphs.

The DDx distributions for the dissolved fractions varied little from transect to transect within each sampling event, including the stations upstream and downstream of the Study Area. Patterns differed between flow conditions, however. The high-flow transect samples were dominated by the 4,4'-isomers of DDE and DDT, with generally decreasing contributions from DDT as the transects move downriver. The low-flow and stormwater-influenced transect samples had much higher proportions of DDD, primarily the 4,4'-isomer, with a substantial contribution by 2,4'-DDD. For the transects, the contribution of 4,4'-DDD varied with sampling event, whereas the contribution of 2,4'-DDD increased overall moving downriver.

Patterns for particulate DDx were similar to dissolved DDx for high-flow transects, dominated by the 4,4'-isomers of DDE and DDT. The proportion of 4,4'-DDT was more consistent upstream to downstream in the particulate fraction than in the dissolved fraction. 2,4'- and 4,4'-DDD dominated the particulate DDx under low-flow conditions at most transects, with generally decreasing contributions from 4,4'-DDT and increasing contributions from 2,4'-DDD as the transects move downriver. The March and July 2005 transect samples at stations W023 (RM 11) and W005 (RM 4) had higher fractions of 4,4'-DDE than other transect samples, while the 4,4'-DDE contribution in the dissolved fractions of these samples was similar to other transects. The particulate DDx patterns for the transect samples taken during stormwater-influenced flow conditions were also generally dominated by the DDD isomers, although 4,4'-DDE and 4,4'-DDT were also prevalent in these samples. The DDD isomers were not detected in stormwater-influenced samples from the upriver transects (W023 and W024), or from the downriver transect at RM 2 (W025) east bank sample. The total DDx concentrations at RM 16 and RM 2 transects were low during the stormwater-influenced event relative to other transects.

The NS and NB samples were generally similar in most flow conditions for both dissolved and particulate DDx in both transect and single-point samples. Patterns for the NB and NS samples were different at Station W011 (RM 6.3; November 2006 stormwater-influenced sampling event), Station W031 (RM 6.1; January 2007 high-flow sampling event), and for Station W024 (RM 16; November 2006 stormwater-influenced sampling event), where the patterns may be obscured by method sensitivity limits.

The DDx compound distribution for the dissolved and particulate fractions of the single-point stations were generally similar to those of the transects in high-flow conditions, with the exception of particulate DDx in the NB sample and dissolved DDx in both samples for location W031 (RM 6.1). The particulate W031 NB sample shows higher fractions of the DDD isomers and less 4,4'-DDT than the paired NS sample and the other single-point or transect stations.

During low-flow conditions, stations W016 (RM 7.2) and W015 (RM 6.9) total DDx concentrations were much higher than the other samples (5 to 60 times higher in the dissolved fraction and 14 to 80 times higher in the particulate fraction). These stations are both on the west bank of the river, located a short distance apart (0.3 mile). The particulate DDx patterns varied fairly widely between the two stations and between sampling events at each location, and were distinct from patterns in surrounding areas and at the transect locations at RM 6.3 and 11. Dissolved DDx patterns at these two locations were more consistent and included a larger fraction of DDD than is typical in the Study Area. Under stormwater-influenced conditions, the DDx concentration at nearby Station W033 (RM 7) was higher than surrounding areas, but the DDx composition did not vary as distinctly.

5.3.9 Total PAHs in Surface Water

High-volume (XAD) samples were analyzed for PAHs by HRGC/HRMS (reported in pg/L). Peristaltic samples were also collected and analyzed for PAHs (reported in µg/L) by HRGC/low-resolution mass spectrometry (LRMS). Concentrations of total PAHs in surface water XAD column and filter samples are depicted as histograms with data arranged by river mile and sampling conditions (high-flow, low-flow, and stormwater-influenced) in Figures 5.3-77 and 5.3-78. Peristaltic data are presented in Figures 5.3-79 and 5.3-80. Figures 5.3-81 through 5.3-84 present the same data as histograms sorted by channel position (west and east channel and transect locations). Each of the XAD histograms presents the particulate (0.5-µm filter for XAD) and dissolved (XAD column) fractions of the total concentration as stacked bar graphs. The peristaltic data histograms present the total (filtered plus unfiltered) fraction.

Total PAHs were detected in all surface water samples (XAD filters and columns) collected during Round 2A and 3A sampling events. In contrast, total PAHs were detected in only 58 percent of peristaltic samples.

Total PAHs measured in single-point XAD samples collected during low-flow conditions ranged from 12,500 J pg/L to 231,000 J pg/L at Station W015 (RM 6.9 in July 2005). Concentrations in transect samples measured during low-flow ranged from 3,970 J pg/L to 66,000 J pg/L at Station W005 (RM 4) in September 2006.

Total PAHs measured in single-point XAD samples collected during high-flow conditions ranged from 10,400 J pg/L to 107,000 J pg/L at Station W035 (RM 8.5) in January 2007. Concentrations measured in transect samples during high-flow ranged from 5,680 J pg/L to 59,400 J pg/L at Station W005 (RM 4 in January 2006).

Total PAHs measured in single-point XAD samples collected during the November 2006 stormwater-influenced event ranged from 5,860 J pg/L to 50,700 J pg/L at Station W033 (RM 7). Concentrations measured in transect samples during this event ranged from 2,790 J pg/L to 38,900 J pg/L at Station W005 (RM 4).

Concentrations of total PAHs measured in single-point peristaltic samples during low-flow conditions ranged from 0.00260 J µg/L to 2.46 J µg/L at Station W012 (RM 6.3 in July 2005).

Total PAHs measured in single-point peristaltic samples during high-flow conditions ranged from 0.0047 J µg/L to 7.4 µg/L at Station W031 (RM 6.1) in January 2007.

Concentrations of total PAHs measured in peristaltic single-point samples in November 2006 ranged from 0.005 J µg/L to 0.12 J µg/L at Station W033 (RM 7).

The following subsections describe observations of PAHs in the complete data set, as well as identified trends.

5.3.9.1 PAH Spatial Distribution

The distribution of total PAH concentrations (dissolved plus particulate concentrations) by river mile throughout the Study Area (and upstream to RM 16 and downstream to Multnomah Channel) is presented in Figures 5.3-85 and 5.3-86.

Three of the five highest total PAH concentrations (including the highest measured value) were measured in single-point samples at RM 6.9 (Station W015) near the west end of Railroad Bridge during low-flow conditions. The second and third highest concentrations were measured in samples collected at RM 8.5 (Station W035 during the January 2007 high-flow event; Swan Island) and RM 7.2 (Station W016 during the July 2005 low-flow event), respectively. The range of total PAH concentrations within the complete data set across the Study Area was fairly consistent between RM 11 and 2, excluding the five highest single-point values. However, total PAH concentrations increased from RM 11 and 16 downstream to RM 4 in six of seven transect sampling events (Figure 5.3-87), suggesting some influence on surface water PAH concentrations from sources within the Study Area. The sole exception to this pattern occurred in the March 2005 low-flow event. Further downstream, at RM 2.9 (Multnomah Channel) and RM 2, concentrations decreased from those at RM 4 in high-flow, low-flow, and stormwater-influenced conditions.

5.3.9.2 PAH Relationships to Flow Rate

Total PAH concentrations as a function of flow rate are presented in Figures 5.3-88 and 5.3-89. As previously noted, four of the five highest concentrations of total PAHs were measured in single-point samples collected during low-flow conditions. Total PAH concentrations tended to vary within a flow condition rather than over the range of flow conditions sampled, although elevated PAHs were evident in low-flow samples from RM 7 to 2 compared to the high-flow and stormwater-influenced sampling events. Downstream near RM 2, the low-flow sample concentration ranges decreased relative to ranges observed further upstream within the Study Area (Figure 5.3-77).

5.3.9.3 Distribution between PAH Dissolved and Particulate Fractions

The following subsections describe the observed trending of the dissolved and total fractions by river mile, event type, sample type, TSS, and f_{oc} of the TSS. Section 5.3.9.4 provides a discussion of PAH patterns and trends (i.e., LPAHs, HPAHs), which may assist with the identification of observed concentrations in the Study Area.

5.3.9.3.1 PAH Particulate and Dissolved Concentrations

The spatial distribution of dissolved and particulate total PAH concentrations is presented on histograms for each surface water sample by flow event type and river mile on Figure 5.3-77; Figure 5.3-78 presents the same data with a reduced y-axis scale to better show lower concentrations. Figures 5.3-81 through 5.3-84 present the same data arranged by channel position. The majority of total PAH concentrations were found in the dissolved rather than the particulate phase. However, there was one notable exception to this trend: Station W035 at RM 8.5 collected during the January 2007 high-flow event. Both the NB and NS samples collected in January 2007 exhibited a much greater particulate to dissolved ratio. Also, at Stations W011 (RM 6.3) and W005 (RM 4) the NB samples had noticeably higher particulate total PAH concentrations in the low-flow and stormwater-influenced sampling events. In the January 2007 high-flow sampling event, this pattern was reversed at Station W035 (RM 8.5), and the NS sample had the highest particulate total PAH concentration.

The dominance of the dissolved fraction is also illustrated in scatter plots of particulate and dissolved data on Figures 5.3-90 and 5.3-91. Linear regression lines are shown for the high-flow, low-flow, and stormwater-influenced sets of transect and point data. The apparently unique ratio at RM 8.5 is well illustrated on these figures (data points in the lower right quadrant) and does not appear to apply to other high-flow samples. The unique ratio may suggest a different character/source of PAHs and/or suspended solids concentration and character for this sample location.

5.3.9.3.2 PAH Associations with Suspended Solids

Total PAH concentrations as a function of TSS are presented in Figures 5.3-92 and 5.3-93. High-flow samples (single-point and transect) exhibited the widest range and highest concentrations of TSS but generally lower total PAH concentrations compared to the majority of the data set. However, there does appear to be a trend of gradually increasing total PAH concentrations with higher TSS values (somewhat linear) for the high-flow samples. Otherwise, the low-flow and stormwater-influenced samples tended to cluster together exhibiting low TSS but varying PAH concentrations. The stormwater-influenced (single-point and transect) and low-flow single-point samples tended to exhibit this pattern most prevalently, with TSS values below 12 mg/L.

Samples collected from RM 8.5 during high-flow in January 2007 did not adhere to the pattern of other samples collected during this particular flow regime as they exhibited high concentrations of total PAHs relative to TSS (Figures 5.3-92 and 5.3-93). Although total PAHs measured in these samples were high relative to TSS, the

particulate concentrations were much greater than the dissolved fractions (Figures 5.3-77, 5.3-78, 5.3-90, and 5.3-91).

Particulate total PAH concentrations and POC concentrations are compared in Figure 5.3-94. The high-flow samples (single-point and transect) exhibited relatively low total PAH concentrations and POC. The low POC values are consistent with the lower observed f_{oc} of the suspended solids during this flow condition (Figure 5.3-19). Several high-flow samples (e.g., the sample with the highest particulate total PAH concentration at Station W035, NS) exhibited POC values equal to zero (Figure 5.3-94). The explanation for this is that the calculated POC was set to equal zero if the DOC was greater than the TOC. This scenario provides further confirmation that the high-flow events are characterized by suspended solids with low f_{oc}.

5.3.9.4 PAH Patterns and Trends

Stacked bar graphs depicting the PAH distribution by individual PAHs and summed by number of rings in the surface water dissolved (XAD column) and particulate (XAD filter) samples by flow condition and river mile are presented in Figures 5.3-95 and 5.3-96. A summary of PAHs included in each sum is provided in Section 5.1.8.4.

The dissolved PAH profiles varied between locations and flow events. During all sampling events, LPAHs (i.e., two-ring and three-ring PAHs) accounted for at least 50 percent of the dissolved PAHs at most transect, as well as single-point, stations. The two-ring PAHs (i.e., naphthalene and 2-methylnaphthalene) were dominant in many samples. In several samples, these results were restated as undetected during data validation, as discussed below, confounding the interpretation. The principal three-ring PAHs in many of the samples were acenaphthene and phenanthrene, although fluorene and anthracene were also present in many of the samples. Fluorene is dominant in two stormwater-influenced samples, collected from W023 (RM 11) and W024 (RM 16).

Generally, the remainder of the dissolved PAH profile consisted of four-ring HPAHs, primarily fluoranthene and pyrene in the high- and low-flow events. With several exceptions, the four-ring PAHs in the November 2006 stormwater-influenced sampling event were chiefly chrysene upstream of RM 8.5 and fluoranthene downstream of this point.

PAH patterns are affected by the presence of elevated detection limits, which result when individual PAHs are restated as undetected in the samples, since non-detects are treated as zeros in the evaluation of PAH patterns. Elevated detection limits may obscure the presence of a PAH at a concentration above the MDL. The concentrations reported by the laboratory for each of the PAHs in one or more samples were restated as undetected during data validation. Reasons for restating data as undetected included PAHs detected in laboratory and field blanks and details related to identification of the PAHs (i.e., ion abundance criteria were not met on the mass spectrometer). Data for XAD filter and column samples were affected. Results for LPAHs were restated as

undetected more frequently than HPAHs. The evaluation of PAHs in this section takes elevated detection limits into consideration.

The PAH profiles in the particulate fractions also varied between sampling locations and flow events. Among transect locations, the low-flow and stormwater-influenced samples included primarily HPAHs with four and five rings, with fluoranthene, pyrene, and the benzofluoranthene isomers prevalent but many other HPAHs present as well. Samples from the high-flow sampling events were still predominantly four- and five-ring HPAHs but had more contribution of LPAHs, primarily the three-ring phenanthrene. The six-ring PAHs in the high- and low-flow events consisted of both benzo(g,h,i)perylene and indeno(1,2,3-cd)pyrene, while the stormwater event consisted almost entirely of benzo(g,h,i)perylene. Particulate PAHs in single-point samples varied from these patterns in many locations, particularly during low-flow conditions.

There was generally little difference in composition between the NS and NB samples for both transect and single-point samples. In the November 2006 stormwater-influenced sampling event, the dissolved NS samples at stations W024 (RM 16), W035 (RM 8.5), and W005 (RM 4) all had high contributions from 2-methylnaphthalene (a two-ring PAH) that was not present in the NB samples. The NS samples also had total dissolved PAH concentrations more than double the NB samples. A similar pattern is not seen in the particulate fractions of these samples. The higher NS concentrations may be related to a hydrocarbon sheen observed during sample collection for the stormwater event that was not thoroughly mixed within the water column, resulting in higher concentrations in the NS samples than the NB samples.

5.3.10 Additional Indicator Chemicals in Surface Water

This section discusses the occurrence and distribution of seven additional ICs in surface water within the Study Area. These additional ICs are included because they figure prominently in the CSM for the Site. The narrative in this section is abbreviated compared to the evaluation of ICs above and includes references to figures and tables to provide a complete picture of the distribution.

5.3.10.1 TCDD TEQ

High-volume surface water sampling results for PCDD/Fs (analyzed by HRGC/HRMS) were used to calculate TCDD TEQ values. TCDD TEQs were calculated using the 2005 WHO consensus TEF values for mammals (Van den Berg et al. 2006) as the sum of each detected congener concentration multiplied by the corresponding TEF value. Concentrations of TCDD TEQ in surface water XAD column and filter samples are depicted in Figure 5.3-97.

TCDD TEQ measured in single-point samples collected during low-flow conditions ranged from 0.110 J pg/L to 0.917 J pg/L at Station W013 (RM 6.7) in July 2005. Concentrations in transect samples measured during low-flow ranged from 0.0181 J pg/L to 0.327 J pg/L at Station W005 (RM 4 in July 2005).

TCDD TEQ measured in single-point samples collected during high-flow conditions ranged from 0.0491 J pg/L to 0.168 J pg/L at Station W035 (NS; RM 8.5) in January 2007. Concentrations measured in transect samples during high-flow ranged from 0.0113 J pg/L to 0.0912 J pg/L at Station W023 (RM 11 in January 2006).

TCDD TEQ measured in single-point samples collected during the November 2006 stormwater-influenced event ranged from 0.0777 J pg/L to 0.212 J pg/L at Station W033 (NB; RM 7). Concentrations measured in transect samples during this event ranged from 0.0133 J pg/L to 0.278 J pg/L at RM 11, Station W023E.

The TCDD TEQ particulate concentrations were greater than the dissolved values, consistent with the total PCDD/F data. TCDD TEQ concentrations were lowest upstream of the Study Area at RM 16 and at RM 2, the downstream extent (Figures 5.3-97 and 5.3-98).

5.3.10.2 Total Chlordanes in Surface Water

High-volume (XAD) samples were analyzed for total chlordanes (reported in pg/L) by HRGC/HRMS. Peristaltic samples were also collected and analyzed for total chlordanes (reported in µg/L) by routine methodology (i.e., EPA 8081A). Dissolved and particulate total chlordanes concentrations in surface water XAD column and filter samples are depicted in histograms by flow event type in Figures 5.3-99 and 5.3-100. Peristaltic data are presented in Figure 5.3-101. The same data are plotted by channel position in Figures 5.3-102 through 5.3-104. Total chlordanes were detected in 85 percent of particulate samples (XAD filters), 100 percent of dissolved samples (XAD columns), and 34 percent of peristaltic samples.

Total chlordanes measured in single-point XAD samples collected during low-flow conditions ranged from 17.3 J pg/L to 241 J pg/L at Station W015 (RM 6.9) in November 2004. Concentrations in transect samples measured during low-flow ranged from 13.4 J pg/L to 58.8 J pg/L (Station W005 [NB] at RM 4 in September 2006).

Total chlordanes measured in single-point XAD samples collected during high-flow conditions ranged from 46.6 J pg/L to 85.9 J pg/L (Station W033 [NB] at RM 7) in January 2007. Concentrations measured in transect samples during high-flow ranged from 33.6 J pg/L to 94.3 J pg/L (Station W005 [NB] at RM 4) in January 2007.

Total chlordanes measured in single-point XAD samples collected during the November 2006 stormwater-influenced event ranged from 7.3 J pg/L to 36.8 J pg/L at Station W033 (NS; RM 7). Concentrations measured in transect samples during this event ranged from 13.4 J pg/L to 37.6 J pg/L at Station W027 (NS; RM 2.9).

Total chlordanes were detected in only one single-point peristaltic sample during low-flow conditions: 0.0021 J µg/L at Station W002 (RM 2.2) in July 2005.

Total chlordanes were detected in three single-point peristaltic samples during high-flow conditions: 0.00029 J µg/L at Station W029 (NB; RM 4.4), 0.00051 µg/L at W030 (NS; RM 5.5), and 0.0006 NJ µg/L at W030 (NB; RM 5.5) in January 2007.

Total chlordanes were detected in two single-point peristaltic samples during the stormwater-influenced event: 0.00055 NJ µg/L at Station W038 (NS; RM 9.9) and 0.0016 J µg/L at Station W036 (NS; RM 8.6).

The total chlordanes dissolved concentrations were greater than the particulate values, with a major exception at RM 6.9 (W015). Total chlordanes concentrations at RM 16 and at RM 2 were similar except the high-flow event, in which upstream sample concentrations were generally higher.

5.3.10.3 Aldrin in Surface Water

High-volume samples were analyzed for aldrin (reported in pg/L) by HRGC/HRMS. Peristaltic samples were also collected and analyzed for aldrin (reported in µg/L). Dissolved and particulate aldrin concentrations in surface water XAD column and filter samples are depicted in a histogram by flow event type in Figure 5.3-105. Peristaltic data are presented in Figure 5.3-106. The same data are plotted by channel position in Figures 5.3-107 and 5.3-108. Aldrin was detected in 69 percent of particulate (XAD filter) and 73 percent of dissolved (XAD column) samples and only 1 percent of the peristaltic samples during Round 2A and 3A sampling activities.

Total aldrin measured in single-point samples collected during low-flow conditions ranged from 0.31 J pg/L to 16.3 J pg/L at Station W015 (RM 6.9) in November 2004. Concentrations in transect samples measured during low-flow conditions ranged from undetected to 6.62 J pg/L at Station W005 (RM 4) in September 2006.

Total aldrin was not detected in peristaltic samples during the Round 2A and 3A low-flow events.

Total aldrin measured in single-point samples collected during high-flow conditions in January 2007 ranged from undetected to 4.1 J pg/L at Station W033 (NB; RM 7) in January 2007. Concentrations measured in transect samples during high-flow conditions ranged from 1.2 J pg/L to 6.0 J pg/L at Station W025M (RM 2) in January 2007.

Total aldrin was detected in only one peristaltic sample (single-point) during high-flow conditions: 0.0052 µg/L at Station W030 (NB; RM 5.5) in January 2007.

Total aldrin measured in single-point samples collected during the November 2006 stormwater-influenced event ranged from undetected to 4.8 J pg/L at Station W033 (NS; RM 7). Concentrations measured in transect samples during this event ranged from undetected to 5.8 J pg/L at Station W027 (NB; RM 2.9).

Total aldrin was not detected in peristaltic samples during the stormwater-influenced event.

The aldrin dissolved concentrations were greater than the particulate values, with a major exception at RM 6.9 (W015). With several exceptions, total aldrin concentrations were fairly consistent within the flow conditions, with downstream samples typically somewhat higher than upstream samples.

5.3.10.4 Dieldrin in Surface Water

High-volume samples were analyzed for dieldrin (reported in pg/L) by HRGC/HRMS. Peristaltic samples were also collected and analyzed for dieldrin (reported in µg/L) by routine methodology (i.e., EPA 8081A). Dissolved and particulate dieldrin concentrations in surface water XAD column and filter samples are depicted in a histogram by flow event type in Figure 5.3-109. Peristaltic data are presented in Figure 5.3-110. The same data are plotted by channel position in Figures 5.3-111 and 5.3-112. Dieldrin was detected in all dissolved (XAD columns) and 70 percent of particulate (XAD filters) surface water samples and 3 percent of peristaltic samples collected during Round 2A and 3A sampling activities.

Total dieldrin measured in single-point XAD samples collected during low-flow conditions ranged from 22.7 to 62.5 pg/L at Station W015 (RM 6.9 in November 2004). Concentrations in transect samples measured during low-flow ranged from 16.7 J pg/L to 48.7 J pg/L at Station W005 (RM 4 in September 2006). Total dieldrin was not detected in peristaltic samples during the low-flow events.

Total dieldrin measured in single-point XAD samples collected during high-flow conditions in January 2007 ranged from 108 J pg/L to 190 pg/L at Station W033 (NB; RM 7). Concentrations measured in transect samples during high flow ranged from 70.5 J pg/L to 384 pg/L (Station W005 at RM 4 in January 2006).

Total dieldrin measured in single-point XAD samples collected during the November 2006 stormwater-influenced event ranged from 31.9 J pg/L to 50.1 J pg/L at Station W031 (NB; RM 6.1). Concentrations measured in transect samples during this event ranged from 25.1 J pg/L at Station W025W (RM 2) to 53.7 J pg/L at Station W024 (NS; RM 16).

Total dieldrin was detected in three single-point peristaltic samples during the high-flow sampling event at concentrations of 0.001 NJ µg/L at Station W029 (NB; RM 4.4), 0.0012 J µg/L at Station W028 (NB; RM 3.6), and 0.0012 NJ µg/L at Station W036 (NB; RM 8.6) in January 2007.

Total dieldrin was not detected in peristaltic samples during the stormwater-influenced event.

The dieldrin dissolved concentrations were much greater than the particulate values (typically on the order of 90 percent of the total concentration). Total dieldrin

concentrations had a small range in the low-flow and stormwater-influenced samples, and were higher with the high-flow event. The high-flow upstream concentrations were higher than all Study Area concentrations during the low-flow and stormwater-influenced events, whereas concentrations in the Study Area were comparable during the low-flow and stormwater-influenced events, making upstream to downstream patterns difficult to assess with the existing range of sampling events.

5.3.10.5 Arsenic in Surface Water

Peristaltic samples were collected and analyzed for arsenic during Round 2A and 3A. Dissolved and particulate arsenic concentrations in surface water are depicted in histograms by flow event type in Figure 5.3-113, and by channel position in Figure 5.3-114. Arsenic was detected in 78 percent of dissolved peristaltic samples and 90 percent of total peristaltic samples during the Round 2A and 3A sampling events.

Total arsenic measured in single-point samples collected during low-flow conditions ranged from 0.33 µg/L to 0.75 µg/L at Station W001 (RM 2) July 2005. Concentrations in transect samples measured during low-flow ranged from 0.35 to 0.64 µg/L (Station W025E at RM 2 in September 2006).

Total arsenic measured in single-point samples collected during high-flow conditions in January 2007 ranged from 0.3 µg/L to 0.63 J µg/L at Station W034 (NS; RM 7.5). Concentrations measured in transect samples during high-flow ranged from 0.25 to 0.54 µg/L at Station W005 (RM 4) and Station W023 (RM 6.3), in January 2006.

Total arsenic measured in single-point samples collected during the November 2006 stormwater-influenced event ranged from 0.43 J µg/L to 0.53 J µg/L at Station W038 (NB; RM 9.9). Concentrations measured in transect samples during this event ranged from undetected to 0.48 J µg/L at Station W005 (NB; RM 4).

Total arsenic concentrations were generally consistent across the entire Study Area during the Round 2A and 3A sampling events. Concentrations were generally higher in low-flow sampling events.

5.3.10.6 Copper in Surface Water

Peristaltic samples were collected and analyzed for copper during Round 2A and 3A. Dissolved and particulate copper concentrations in surface water are depicted in histograms by flow event type in Figure 5.3-115, and by channel position in Figure 5.3-116. Copper was detected in 99 percent of dissolved and 100 percent of total copper samples during Round 2A and 3A sampling events.

Total copper measured in single-point samples collected during low-flow conditions ranged from 0.68 µg/L to 2.09 µg/L at Station W004 (RM 3.7) in March 2005. Concentrations in transect samples measured during low-flow ranged from 0.68 µg/L to 1.55 µg/L at Station W005 (NB; RM 4) in September 2006.

Total copper measured in single-point samples collected during high-flow conditions in January 2007 ranged from 1.47 µg/L to 3.49 µg/L at Station W031 (NB; RM 6.1). Concentrations measured in transect samples during high-flow ranged from 1.1 µg/L to 3.68 J µg/L at Station W023 (RM 11) in January 2006.

Total copper measured in single-point samples collected during the November 2006 stormwater-influenced event ranged from 0.79 to 1.14 µg/L at Station W035 (NS; RM 8.5). Concentrations measured in transect samples during this event ranged from 0.65 µg/L to 1.1 µg/L at Station W024 (NS; RM 16).

Total copper concentrations were generally consistent across the entire Study Area during the Round 2A and 3A sampling events. Concentrations were generally higher in the high-flow sampling events.

5.3.10.7 Zinc in Surface Water

Peristaltic samples were collected and analyzed for zinc during Round 2A and 3A. Dissolved and particulate zinc concentrations in surface water are depicted in histograms by flow event type in Figures 5.3-117 and 5.3-118, and by channel position in Figures 5.3-119 and 5.3-120. Zinc was detected in 42 percent of dissolved peristaltic samples and 76 percent of total peristaltic samples collected during Round 2A and 3A sampling events.

Total zinc measured in single-point samples collected during low-flow conditions ranged from 1.7 µg/L to 57.9 µg/L; the highest concentration was found at Station W022 (RM 9.7) in November 2004. Concentrations in transect samples measured during low-flow ranged from 2.1 µg/L to 6.1 µg/L; the highest concentration was found at Station W023W (RM 16) in September 2006.

Total zinc measured in single-point samples collected during high-flow conditions ranged from 3.0 µg/L to 8.4 µg/L; the maximum concentration was found at Station W031 (NB; RM 6.1) in January 2007. Concentrations measured in transect samples during high-flow ranged from 1.9 µg/L to 6.4 µg/L; maximum concentrations were found at Station W023 (RM 11) and Station W024 (RM 16) in January 2006.

Zinc in samples collected during stormwater-influenced conditions was found in the dissolved fraction only. Single-point concentrations ranged from 4.8 µg/L to 6.6 µg/L; the maximum concentration was found at Station W034 (NS; RM 7.5) in November 2006. Zinc was detected in only one transect sample (5.1 µg/L) at Station W025M (RM 2) in November 2006.

The concentrations of total zinc were generally consistent throughout the Study Area during the Round 2A and 3A sampling events.

5.3.11 Summary of Nature and Extent of Indicator Chemicals in Surface Water

The four ICs selected for discussion for surface water exhibited distinct maxima at one or more of the Study Area locations. This discussion includes concentrations of bounding ICs that are notably high relative to other locations in the surface water data set. Trends relative to flow conditions are also discussed.

Upstream between RM 11 and 16, there were elevated PCB and PCDD/F concentrations (compared to other concentrations in this reach) at RM 11. Two of the three highest PCB concentrations and the second highest PCDD/F concentration at RM 11 were measured on the east side of the river, suggesting the existence of localized sources in this area. For DDx, the high-flow concentrations at RM 11 and 16 were much greater than the low-flow and stormwater-influenced samples, indicating an upstream source or sources of DDx mobilized during high-flow conditions. PAH concentrations between RM 11 and 16 tended to be lower than those within the Study Area.

There is a trend of decreasing PCB concentrations with increasing flow rates. Five of the six highest PCB concentrations were associated with low-flow sampling events, and concentrations were consistently lower in high-flow samples as compared to the two other flow regimes. Excluding the samples at RM 6.7, the, stormwater-influenced samples appear to be lower than the low-flow samples. High-flow conditions may overwhelm the local effects and control PCB concentrations in the Study Area until flow rates subside.

PCDD/F concentrations don't appear to be as dependent upon flow rate as PCBs. At the downstream end of the Study Area, the low-flow sample concentration ranges for both total PCBs and total PCDD/Fs decreased relative to ranges observed within the Study Area.

Within the Study Area, the highest concentrations of PCBs (highest including six of seven) and PCDD/Fs (highest) were found in Willamette Cove (RM 6.7) during low-flow conditions, which suggests a localized source or sources of these ICs in this vicinity.

Total DDx concentrations in surface water across the Study Area were dominated by five high-concentration samples (3,000 pg/L to 10,000 pg/L) collected at RM 6.9 and 7.2, adjacent to known pesticide sources to the river. The highest PAH concentrations (100,000 J pg/L to 230,000 J pg/L) were found at RM 6.9, as well as RM 7.2 and 8.5, adjacent to known PAH sources in the Study Area.

The highest total DDx (RM 6.9 and 7.2) and total PAH (three out of the five highest; RM 6.9) concentrations were measured in single-point samples during low-flow conditions. With the exception of these high-concentration samples, the range of PAH concentrations were fairly consistent across the Study Area. However, total DDx

concentrations were elevated in high-flow samples from RM 11 and 16, as compared to the two other flow regimes, indicating an upstream source or sources of total DDx that becomes mobilized during high-flow conditions.

At the downstream end of the Study Area, total PCDD/Fs, DDx, and PAHs had consistently low concentrations with less variability among sampling event types than seen in Study Area samples. In contrast, total PCB congener concentrations in the downstream end of the Study Area were approximately 900 to 1,300 pg/L in single-point samples near RM 2 for the stormwater-influenced event, and total PCB congener concentrations downstream were higher than the upstream (RM 16) concentrations for all event types (low-flow, high-flow, stormwater-influenced).

5.4 INDICATOR CHEMICALS IN TRANSITION ZONE WATER AND GROUNDWATER SEEPS

This section summarizes the Study Area data for TZW and groundwater seeps. As described in Section 3, the transition zone is defined as the interval where both groundwater and surface water comprise some percentage of the water occupying pore space in the sediments. The primary focus area of the transition zone for this investigation is the surface sediment mixed layer, which is considered to be the upper 30 cm of the sediment and includes the biologically active zone.[1] The mixed layer is characterized by TZW samples collected in the shallow (0 to 38 cm bml) sediments. Deeper (>90 cm bml) TZW samples are also discussed here to lend insight into observed chemical distribution patterns and to support an assessment of potential TZW loading impacts to surface water and surface sediment that is provided in Appendix E and Section 6.1.6.

The following subsections present tables, plan view maps, histograms, scatter plots, and stacked bar charts to support brief discussions of nature and extent for the select IC list (Table 5.0-2). The full RI data sets for TZW and groundwater seeps for all sampled chemicals (those data of adequate quality for use in decision making for the Study Area per the Portland Harbor RI/FS Programmatic Work Plan [Integral et al. 2004]) are presented in the project SCRA database (Appendix A3) and summarized in Appendix D4, Tables D4-1 and D4-2.

The TZW presentation provided in Section 5.4 supports the detailed site-by-site presentation and analysis of groundwater pathways presented in Appendix C2. The Appendix C2 presentation of TZW provides data analysis focused on identification of complete groundwater pathways from upland plumes to the transition zone, including some cross-media analysis. This section (Section 5.4) focuses on presentation of the distribution of ICs observed in the transition zone. As such, this section does not attempt to present cross-media analysis or to relate observations to sources. Due to the spatially focused nature of the TZW and seep sampling programs, this discussion of nature and extent includes use of facility names for location reference; mention of facility names does not necessarily indicate source origin.[2] The findings presented in Appendix C2 and in this nature and extent discussion are incorporated into the detailed cross-media analysis and sources discussions presented in the Study Area-wide CSM discussion (Section 10).

[1] The surface sediment mixed layer depth is based on analysis of bathymetric data, as discussed in Section 2, which indicates that processes disturb or mix sediments from >20 to 30 cm bml in some areas of the Study Area. The biologically active zone is defined by the depth of biological processes. The depth of the true biologically active zone varies widely throughout the Study Area, based on factors that control benthic community structure, such as sediment texture, sediment-water interface dynamics, and organic loading.

[2] The TZW sampling effort was a focused investigation offshore of nine upland sites and was not a harbor-wide study. The approach taken for selection of TZW study sites is presented in Appendix C2. It is possible that other sites will be identified that have a complete pathway for upland groundwater plumes to the Study Area.

This section is organized somewhat differently than the *In-River Distribution* discussions for other media. Specifically, this TZW section includes presentation of all TZW ICs in the main report, which is unlike the discussion format used for other media such as sediment (Section 5.1) and surface water (Section 5.3). This difference reflects the lists of chemicals that were the focus of the various upland site-specific TZW investigations. In particular, PCBs were not sampled in TZW,[3] and only two TZW sampling locations were analyzed for PCDD/Fs.[4] Neither PCBs nor PCDD/Fs are expected to be migrating to the river via groundwater given their high hydrophobicity and the lack of evidence of facilitated transport at upland sites with these COIs. Because of the unique TZW IC list and the complexities of the groundwater pathway, discussion of all TZW ICs is needed for a meaningful presentation of the in-river distribution of TZW results. Because all TZW ICs are discussed in the main text, a paired summary statistics table of a subset of ICs is not provided in the main text, as is done for other media. Instead, all summary statistics are presented in Appendix D4 (Tables D4-1 and D4-2).

5.4.1 TZW Data Set

This subsection describes the TZW chemistry data set presented in this RI Report. The TZW chemistry data were generated during the following field events (sampling locations are shown on Map 2.2-6):

2004 Pilot Study – In 2004, a pilot study was conducted offshore of the Arkema and ARCO sites to evaluate various TZW sampling methods, including Trident® push probe, small-volume peepers, large-volume peepers, bulk sediment centrifugation, Geoprobe®, and vapor diffusion samplers. The findings of the 2004 Pilot Study are discussed in detail in Appendix B of the GWPA Sampling and Analysis Plan (SAP; Integral 2005c), the Discharge Mapping FSP (Integral 2005b), and the TZW FSP (Integral 2006a). The pilot study found that TZW samples collected by Trident push probe and small-volume peepers provided the most accurate and reproducible results for the targeted zones and sediment conditions encountered.[5] Based on those findings, only those tools were applied to subsequent LWG investigations. The 2004 Pilot Study results included in the RI TZW data set consist of results for only the 10 Trident push probe and 9 small-volume peeper locations sampled offshore of the ARCO and Arkema sites (data collected by other evaluated methods are not included). Included Pilot Study

[3] PCBs were not identified as COIs for any of the nine TZW sites. PCB data were not collected in TZW and are not discussed here.

[4] PCDD/Fs were detected in only one TZW sample: Trident filtered (0.865 pg/L) and unfiltered (29 pg/L) at station RP-07-B.

[5] The Round 2 GWPA TZW SCSR (Integral 2006e) presents detailed information regarding the quality of the Round 2 TZW data set developed and analyzed as part of the Round 2 GWPA. These data meet project-specific data quality objectives specified in the SAP (Integral 2005c) and QAPP (Integral 2005a), and statistical analysis of sample replicates shows excellent overall reproducibility of sample results, with both small-volume peepers and Trident samples comparing well. These findings support a high level of confidence in both the analytical data sets and the methods and equipment used for sample collection.

samples were collected in the shallow sediment interval at 30 cm bml (Trident) and from 0 to 38 cm bml (peeper).

2005 Round 2 GWPA – Between August and December 2005, the LWG conducted an investigation of TZW offshore of nine upland sites with known or likely pathways for groundwater plume transport of chemicals to the river. Site selection is discussed further in Section 5.4.1.1 (below) and in Appendix C2. Fifty-six Trident and 27 small-volume peeper locations were sampled as part of this investigation, and all of those results are included here. Trident samples were collected in the shallow sediment interval at 30 cm bml at all 56 sample locations, as well as in deeper sediment (90, 120, or 150 cm bml) at 23 of these locations. Peeper samples were collected in the shallow sediment interval, from 0 to 38 cm bml, at all 27 peeper sample locations. The sampling program is described in detail in the TZW FSR (Integral 2006d), and the findings related to groundwater pathways are discussed in detail in Appendix C2.

2005 Siltronic Investigation – In 2004 and 2005, on behalf of Siltronic, Maul Foster & Alongi (MFA) conducted an investigation of the groundwater pathway offshore of the Siltronic site. These data are presented in detail by MFA (HAI 2005b; MFA 2005b), and discussed in Appendix C2. The results of the 2004 and 2005 investigations indicate areas of groundwater discharge in a subset of the area investigated (MFA 2007). Additional results of these studies are presented in Appendix C2. The TZW samples were collected with a Geoprobe sampler, which can be used as a push probe type of sampling tool for TZW. Forty-one TZW samples collected in the shallow sediment interval at 31 cm bml, as well as 24 samples collected in a deeper interval at 91 cm bml, are included in this presentation of TZW nature and extent.

2007 Gasco Investigation – In 2007, on behalf of Gasco, Anchor Environmental conducted an investigation of the groundwater pathway offshore of the Gasco site. These data are presented in detail in Anchor (2008b) and discussed in Appendix C2. TZW samples were collected with a Geoprobe sampler. Twenty-two TZW samples collected between 0 and 90 cm bml are included in this presentation of TZW nature and extent. Note that very few samples were collected in the shallow (0 to 38 cm bml) sediment interval for this Gasco investigation. Therefore, the sample collected at the uppermost depth at each location is used in this nature and extent discussion as the best available representation of the TZW concentrations in the shallow layer. No deeper data collected as part of the Gasco Investigation was included in this nature and extent discussion.[6]

5.4.1.1 Study Sites

These sampling activities described above focused on the offshore area of nine sites along the west bank of the river (see Map 2.2-6):

- Kinder Morgan Linnton Terminal (RM 4.1 to RM 4.2)

[6] Only one sample (GS-C2, 73 to 103 cm bml) in the 2007 Gasco Investigation was collected in the deeper (90 to 150 cm bml) sample interval; this sample is not included in this nature and extent discussion.

- ARCO Terminal 22T (RM 4.7 to RM 4.9)
- ExxonMobil Oil Terminal (RM 4.8 to RM 5.1)
- Gasco (RM 6.1 to RM 6.5)
- Siltronic (RM 6.3 to RM 6.5)
- Rhone Poulenc (RM 6.7 to RM 6.9)
- Arkema (Acid Plant and Chlorate Plant areas; RM 7.2 to RM 7.5)
- Willbridge Terminal (RM 7.6 to RM 7.8)
- Gunderson (RM 8.3 to RM 8.5).

These nine sites were identified for investigation as part of the Round 2 GWPA as high-priority Category A upland groundwater sites, selected in agreement with EPA as sites with a confirmed or reasonable likelihood for discharge of COIs to Portland Harbor.[7] The approach to site selection is discussed in greater detail in Appendix C2. TZW sampling offshore of each site was focused largely on the in-river areas adjacent to the site shoreline, often extending to, and occasionally just beyond, the navigation channel boundary. As such, the TZW sampling effort was not a harbor-wide study of TZW, but instead was a focused investigation offshore of the nine Round 2 study sites. (Note: non-LWG TZW investigations offshore of the Siltronic and Gasco sites are considered with the LWG Round 2 GWPA results, as discussed in Section 5.4.1 above.) It is possible that there are other areas of groundwater plume discharge to the river not captured in this data set.

The TZW investigations performed for the RI focused solely on areas of confirmed or likely groundwater plume discharge to the river and did not seek to characterize pore water chemistry elsewhere in the Study Area. Accordingly, this discussion does not address TZW/pore water chemistry in areas with no upland groundwater discharge, or areas of clean groundwater flowing through contaminated sediments. Additionally, this study does not distinguish the relative contribution of upland groundwater plumes and chemicals in sediment to the concentrations measured in TZW. Consideration of pore water chemistry affected by in-river sediment contamination is evaluated in Section 6 loading calculations through equilibrium partitioning calculations based on the large data set of sediment concentrations. These calculated estimates of pore water concentrations are not presented here as part of the TZW nature and extent presentation because they are not actual field measurements.

[7] Site selection criteria included: (1) Existing offshore groundwater sampling data indicate that a potentially complete transport pathway exists for groundwater COIs to reach the transition zone; (2) Existing shoreline sampling data from groundwater wells or seeps indicate a reasonable likelihood of a complete transport pathway for groundwater COIs to reach in-river exposure points; (3) Existing observations of NAPL seepage to the river indicate that a complete transport pathway may exist for groundwater COIs to reach in-river exposure points; and (4) Shoreline groundwater seeps containing COIs are known to be present and represent a potentially complete exposure pathway for human receptors.

5.4.1.2 Sampling Techniques

Two general types of sampling techniques were used to collect the TZW samples: small-volume peepers and push probe samplers (Trident and Geoprobe[8] tools were used as push probe samplers). These are described in detail in the Pilot Study FSP (Integral 2004a). The Round 2 TZW investigation used both Trident and small-volume peepers, and the non-LWG investigations at Gasco and Siltronic used Geoprobe to collect TZW samples.

For the Round 2 TZW investigation, paired unfiltered and filtered samples were collected with the Trident tool, where possible. At these locations, collection of unfiltered samples was given priority, and volume limitations prevented collection of filtered samples in some cases. Filtered samples were not collected for small-volume peeper samples from the Round 2 TZW investigation due to volume limitations and the across-membrane-equilibration nature of the sampling technique,[9] which was expected to introduce fewer solids than peristaltic pumping for push probe samples. Filtered Geoprobe samples were collected as part of the Anchor (2008b) investigation at Gasco for metals and PAHs only. No filtered Geoprobe samples were collected by MFA as part of the Siltronic investigation (MFA 2005b).

All peeper samples were collected over the depth interval of 0 to 38 cm bml. Trident samples were collected at 30 cm bml, with a few deeper samples collected between 90 and 150 cm bml.[10] Geoprobe samples were collected at depths ranging from 30 to 6,300 cm bml, though only Geoprobe samples from 0 to 91 cm bml are presented in this discussion of TZW nature and extent. The nature and extent presentation focus on the shallow (0 to 38 cm bml) TZW data corresponds to the surface sediment mixed layer (≤30 cm bml). Deeper Trident and Geoprobe TZW sample data (90 to 150 cm bml) are presented here to advance the understanding of the completeness of specific groundwater pathways, and are discussed in more detail in Appendix C2.

5.4.1.3 Spatial and Temporal Representativeness of the TZW Data Set

TZW sampling was limited to the offshore areas of the nine study sites as described above. As noted above, there may be additional upland plumes for which delineation was not adequate enough to have been categorized as having a confirmed or likely pathway to the river (upland site plume delineation is outside of the LWG investigation responsibility). Further, the sampling investigation of TZW did not seek to delineate

[8] Geoprobe sampling of TZW was performed by Siltronic and Gasco. The data are included in the SCRA project database as non-LWG collected data.

[9] Note that the peeper is a diffusion-based sampling device, and water samples captured in the peeper device must pass through a ~5-µm Teflon® membrane. Therefore, peeper samples are not whole water samples, yet these samples do include particles larger than the 0.45-µm diameter filter used for Trident filtered samples.

[10] One Trident sample was collected at 60 cm bml at location CP-07-B. This sample is included with the 90 to 150 cm bml data set.

TZW chemistry in areas unimpacted by upland plumes but possibly impacted by river sediments.[11]

Because TZW samples were collected at a single point in time (for Trident and Geoprobe sampling) or over a 3-week equilibration period (for peeper sampling), LWG field sampling events were carefully timed to maximize the expected upland groundwater signal (i.e., the time of greatest groundwater discharge rate). For the Pilot Study and Round 2 TZW investigations, TZW analytical samples were collected from November 2004 to January 2005 and October to December 2005, respectively, before river water levels increased to the higher levels that typically occur from mid-winter through spring. The non-LWG TZW samples collected at Gasco that are included in this nature and extent discussion were collected between July and September 2007. The non-LWG TZW samples collected at Siltronic that are discussed here were collected in May and June of 2005.

Daily tidal fluctuations are not expected to have affected the representativeness of the analytical chemistry results collected by either small-volume peepers or the push probe samplers. First, small-volume peepers were left in place over 3 weeks, allowing equilibration over many tidal cycles. Second, seepage meter results (presented and discussed in Appendix C2) suggest that Trident samples from depths of 30 cm bml or deeper are unlikely to be affected by tidal changes in the areas sampled. The typical discharge rate measured by seepage meters was on the order of a few cm/day (average = 1.40 cm/day; minimum daily average = –18.9 cm/day; maximum daily average = 14.2 cm/day). The largest net negative recharge rate[12] among seepage meter locations showing an average positive discharge (i.e., locations where the tidal influence could potentially have a significant timing impact on TZW chemistry) was observed offshore of the Siltronic site. At this location, the negative recharge period covered roughly 9.5 hours, with an average seepage rate of –6.7 cm/day during this period. This corresponds to a net seepage flux of 2.65 cm over the 9.5 hour tidal recharge period. Assuming sediment porosity of 25 percent, the maximum depth of influence for this period of negative seepage would be roughly 10.6 cm before the direction reverses to positive discharge with the tidal change. At this location with the greatest period and magnitude of negative flux (among all locations with net positive flux), this estimated periodic depth of influence of surface water is still well above the minimum sampling depth of 30 cm bml. Therefore, the timing of tidal conditions during Trident sample collection is not expected to have had any effect on chemistry results.[13]

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[11] In areas not directly affected by transport of chemicals originating in upland groundwater, chemicals may be present in TZW as a result of desorption from contaminated sediments and/or geochemical processes within the sediments and associated TZW.

[12] The negative seepage rate values are the focus here because they correspond to observed recharge to the TZW from surface water, which is the concern related to tidal influence on the timing of TZW sampling.

[13] These calculations are further supported by a temporal tidal study conducted offshore of the Gasco and Siltronic sites as part of the Anchor 2007 investigation. That investigation involved collection of 30 TZW samples from three mini-piezometers (~45 cm bml) over multiple tidal cycles and concluded that there were no correlations with river stage for any of the COIs analyzed.

5.4.2 Indicator Chemical List for TZW

The IC list for TZW is presented in Table 5.0-2. A total of 39 individual analytes and calculated sums (total cPAHs, total HPAHs, and total PAHs; total DDE, total DDD, total DDT, and total DDx; total xylenes) were identified as ICs for TZW in consultation with EPA (see Section 5.0). These chemicals were selected from the complete list of TZW sampled analytes, taking into consideration the subset of chemicals found to be relevant through the human health and ecological risk screening process, the chemicals identified for detailed assessment in the CSM discussion in Section 10, as well as any additional chemicals which were the focus of the Round 2 TZW investigation. The resulting list of chemicals can be placed into seven groups, as follows (chemicals included in the Section 10 CSM discussion are denoted with an asterisk [*]):

- PAHs
  - Total PAHs*
  - Total LPAHs[14]
  - Total HPAHs
  - Total cPAHs[14]
  - Total cPAHs BaPEq values
  - BAP
  - Naphthalene
- TPH[15]
  - TPH - DRH
  - TPH - RRH
  - TPH- Gasoline-range hydrocarbons (GRH)
- Metals
  - Arsenic*
  - Barium
  - Cadmium
  - Copper*
  - Lead
  - Manganese

[14] LPAHs and cPAHs are discussed here to provide for complete discussion of total PAHs and HPAHs though they are not included in the indicator chemical list for TZW.
[15] TPH is not on the TZW indicator chemical list; however, it is presented in this section to support discussion of DRH, RRH, and GRH.

  - Nickel
  - Zinc*
- Pesticides
  - Total DDx (2,4'- and 4,4'-DDD, DDE, DDT)*
  - Total of 2,4'- and 4,4'-DDT
  - Total of 2,4'- and 4,4'-DDE
  - Total of 2,4'- and 4,4'-DDD
- Herbicides
  - 2,4,5-TP (Silvex®)
- VOCs
  - Monochlorobenzene (MCB)
  - 1,2-Dichlorobenzene (1,2-DCB)
  - 1,2,4-Trimethylbenzene
  - Chloroethane
  - 1,2-Dichloroethane (1,2-DCA)
  - 1,1,2-Trichloroethane (1,1,2-TCA)
  - Vinyl chloride
  - cis-1,2-Dichloroethene (cis-1,2-DCE)
  - Trichloroethene (TCE)
  - Chloroform
  - Methylene chloride
  - Carbon disulfide
  - Total BTEX (including individual component chemicals)
- Other
  - Perchlorate
  - Cyanide.

Five chemicals (total PAHs, total DDx, arsenic, copper, zinc) included on the TZW IC list are presented in the cross-media analysis of results in Section 10. The following sections present detailed discussions of results for two of these ICs (DDx and PAHs), followed by summary discussions and figures for the remaining ICs on the TZW list.

In addition, TZW sampling results are compared to various water screening values in Appendix D3.3. Contaminants that exceed one or more screening values are identified and will be included in the contaminant mobility evaluation in the draft FS. Contaminants that pose potential risk to humans are discussed in the BHHRA (Appendix F).

5.4.3 Description of TZW Presentation Tools

TZW data for TZW ICs are presented on plan-view maps and/or scatter plots, as well as stacked bar charts for select chemicals to support evaluation of sample composition. These presentations vary by analyte and are summarized in Table 5.4-1. As reflected in Table 5.4-1, the TZW analyte lists varied by study site; therefore, it was often unnecessary to produce maps for each river mile for a given analyte. All maps, scatter plots, and bar charts were generated with consistent approaches as described below.

Maps: Map presentations of TZW data use color-coded symbols and flyout labels to provide the individual concentration values. This presentation includes distinction of shallow TZW Trident samples (0 to 38 cm bml) and deeper Trident samples (90 to 150 cm bml), as well as non-LWG shallow (0 to 90 cm bml)[16] and deeper (91 cm bml) Geoprobe samples. Paired map sets are presented for each river mile to show filtered and unfiltered results, where available. Peeper samples are presented with a unique symbol on both filtered *and* unfiltered images to allow for a detailed evaluation of results. A histogram of detected chemical concentrations is inset on each map to provide context for the results presented on the given river mile relative to the results from the entire Study Area. Histogram bins and concentration color ranges were selected based on professional judgment to best present the complete range of filtered and unfiltered concentration values observed across the Study Area.

Scatter Plots: Scatter-plot presentations of TZW data show sample concentrations plotted according to the river mile of the sample location. Color-coded symbols distinguish sample type and depth. Paired plot sets are presented for each chemical to show filtered and unfiltered results, where available. Peeper samples are presented with a unique symbol on both filtered *and* unfiltered images to allow for a detailed evaluation of results.

Bar Charts: Stacked bar charts of total DDx, total PAHs, and TPH present, for each individual TZW sample, the fractional contribution of each individual constituent of the total concentration (detected sums only). The total sum concentration is also denoted on the figure with a black line (scale on the right-hand y-axis). Samples are organized along the x-axis according to descending river mile order, grouped by understanding of groundwater discharge areas as discussed in detail in Appendix C2. Where available,

[16] For the Gasco study (sample IDs that begin with "GS-"), the sample collected at the uppermost depth in the 0 to 90 cm bml interval at each location is presented on maps as the best available representation of the TZW concentrations in the shallow layer. No deeper data collected as part of the Gasco study is presented. For the Siltronic study (sample IDs that begin with "GP-"), samples collected at 31 cm bml are presented as shallow TZW, and samples collected at 91 cm bml are presented as deeper TZW.

peeper ("PR"), filtered and unfiltered Trident ("TR"), and Geoprobe ("GP") data are shown for each sample location. Sample IDs for filtered results ("-f" in the sample ID) are indicated by highlighting. Deeper Trident samples are denoted by "-90", "-120", or "-150" in the sample ID, referring to depth bml in centimeters. Further, field duplicate samples on these figures are denoted by "dup" in the sample ID.

5.4.4 DDx in TZW

The following subsections present the in-river distribution of DDx results in TZW samples. The TZW IC list includes total DDx, total DDD, total DDE, and total DDT. This presentation also discusses the individual DDx components (2,4'-DDE, 4,4'-DDE, 2,4'-DDD, 4,4'-DDD, 2,4'-DDT, and 4,4'-DDT).

5.4.4.1 Observed Chemical Distribution of DDx in TZW

DDx components were identified as upland groundwater COIs at the former Acid Plant area of the Arkema site in the Round 2 GWPA SAP (Integral et al. 2005). Ten locations offshore of this area were analyzed for DDx pesticides in TZW as part of the Round 2 GWPA sampling effort. Additionally, one sample collected offshore of the adjacent Rhone Poulenc site[17] (RP-03-C) was analyzed for DDx pesticides. There are six peeper samples, 14 shallow (0 to 38 cm bml) Trident samples (with collocated filtered and unfiltered samples collected at six locations), and three deep (90 to 150 cm bml) Trident samples in this data set (with collocated filtered and unfiltered samples collected at one location).

Map 5.4-1 presents filtered (top panel) and unfiltered (bottom panel) total DDx (and constituent sums 2,4'- and 4,4'-DDD, 2,4'- and 4,4'-DDE, 2,4'- and 4,4'-DDT)[18] concentrations measured in shallow (0 to 38 cm bml) Trident and deep (90 to 150 cm bml) Trident samples. Peeper samples are presented with a unique symbol on both filtered *and* unfiltered images to allow for a detailed evaluation of results. Inset histograms on Map 5.4-1 show the distribution of total DDx sample concentrations for detected filtered, unfiltered, and peeper results. All sample results for summed and individual DDx isomers in TZW are presented in the SCRA database (Appendix A3) and are summarized in Appendix D4, Table D4-1.

Total DDx concentrations in the six shallow peeper and six filtered Trident samples ranged from below detection limits (detection limits ranged from 0.0042 UA µg/L to 0.035 UA µg/L) to 0.158 J µg/L. The highest filtered shallow DDx concentration was measured at RP-03-C, at the downstream Rhone Poulenc site. Only one filtered, deep Trident sample was collected (at RP-03-C); this sample had a total DDx concentration of 0.179 J µg/L.

[17] DDx was not included on the COI list for the Rhone Poulenc site for the GWPA sampling because pesticides were not expected to be mobile in groundwater at the site (Integral et al. 2005). Subsequent to completion of the GWPA site categorization process, DDx was detected in groundwater at the site (AMEC 2006).

[18] Note that 2,4'-DDD, 2,4'-DDE, and 2,4'-DDT were not sampled during the 2004 Pilot Study; therefore, the total DDx sum for these samples consists of only the 4,4'-DDx isomers. These results are distinguished with an "A" qualifier on Map 5.4-1.

Total DDx concentrations in the eight shallow unfiltered Trident samples ranged from 0.0075 JA µg/L to 3.05 J µg/L. The highest (>3 µg/L, shown as red symbols on Map 5.4-1) unfiltered shallow DDx concentration was measured at AP-03-A (3.05 J µg/L). The three unfiltered deep DDx sample concentrations ranged from 0.169 J µg/L to 5.73 J µg/L, with the highest deep concentration collocated with the highest shallow concentration at AP-03-A. Patterns and trends in the data set are discussed in the following subsection.

5.4.4.2 Patterns and Trends of DDx in TZW

Patterns and trends of DDx distribution are evaluated based on maps and stacked bar charts, as described in Section 5.4.3. Evaluation of these patterns and trends for DDx is limited due to low sample size and detection frequencies; however, development of basic conclusions regarding chemical composition and the influence of filtration and sample depth is possible, as presented in the following paragraphs.

As shown on the histograms on Map 5.4-1 and described above, the observed ranges of DDx concentrations are generally higher in unfiltered samples as compared to the observed range for filtered and peeper samples. This tendency observed in the histogram ranges is upheld in a point-by-point assessment of the seven collocated filtered and unfiltered Trident pairs, where unfiltered sample concentrations are over 90 percent greater than filtered samples in four pairs collected offshore of the Arkema Acid Plant area. Higher sample concentrations in unfiltered samples as compared to filtered samples is expected for these highly hydrophobic chemicals, indicating the presence of DDx sorbed to solids larger than the filter diameter (>0.45 µm) in the unfiltered Trident samples. These results indicate that the unfiltered samples in the former Acid Plant area are likely affected by intake of sediment in the unfiltered sampling process. Further, the highest filtered sample result is observed offshore of the Rhone Poulenc site, whereas the collocated unfiltered result offshore of the Rhone Poulenc site (0.21 J µg/L) is well below the average (1.78 µg/L) of all measured unfiltered concentrations; this result suggests that uptake of solids did not influence the unfiltered concentrations measured at RP-03-C.

Comparison of total DDx concentrations in unfiltered samples at the three sample locations (RP-03-C, AP-03-A, R2-AP-02) where both shallow (0 to 38 cm bml) and deep (90 to 150 cm bml) TZW samples were collected shows that the deeper samples consistently have higher concentrations than the shallow samples at AP-03-A and R2-AP-02. However, at RP-03-C the deeper sample is generally comparable to the corresponding shallow samples in both filtered and unfiltered data sets.

Figure 5.4-1 presents bar charts showing percent composition of the six DDx congeners in the 15 samples with a detected total DDx result. The chart only presents samples where all six DDx congeners were analyzed.[19] Samples are organized along the x-axis

[19] The 2004 Pilot Study samples were analyzed for 4,4'-DDx congeners only, and are therefore not shown on the stacked bar chart in Figure 5.4-1.

in groups referring to groundwater discharge zones. (These zones are indicative of areas of similar groundwater discharge conditions, as described in Appendix C2.) On these figures, sample IDs indicate sample location, sample method, sample depth, and field duplicates, as described above in Section 5.4.3. Evaluation of the stacked bar chart in Figure 5.4-1 yields the following observations:

- The two duplicate sample pairs (AP-03-A-TR-f and AP-03-A-TR-f-dup; AP-03-A-TR-uf and AP-03-A-TR-uf-dup) show good reproducibility in composition trending.

- Two of the three shallow and deep sample pairs show similar compositions at both depths. The shallow and deep pair collected at R2-AP-02-TR-uf has an extreme concentration difference and shows different composition between the shallow and deep result.

- The bar chart trends further support the overall observed trends noted above in discussions of spatial distribution and filtration effects. The highest total DDx concentrations (concentration indicated by the black line corresponding to the right y-axis) are mainly associated with unfiltered samples (filtered sample IDs are highlighted). These total concentration peaks (designated by the black line) correspond to DDx compositions dominated by 4,4'-DDT plus 4,4'-DDD.

In summary, for the limited data set available, the highest DDx concentrations were observed in unfiltered deep (90 to 150 cm bml) samples collected offshore of the former Acid Plant area. Filtration greatly reduced DDx concentrations measured offshore of the Acid Plant, indicating that DDx is present on solids. The highest filtered sample result is observed offshore of the Rhone Poulenc site. Filtration did not significantly reduce DDx concentrations measured at the single Rhone Poulenc sample location, and the unfiltered results offshore of the Rhone Poulenc site are below the average of unfiltered concentrations observed offshore of the former Acid Plant site. Further, offshore of the former Acid Plant area, TZW concentrations are generally greater at depth (90 to 150 cm bml) as compared those in shallow TZW (0 to 38 cm bml). Conclusions about completeness of the groundwater pathway are presented in detail with additional lines of evidence in Appendix C2 and summarized in Section 4.

5.4.5 Total PAHs in TZW

The following subsections present the in-river distribution of PAHs in TZW samples and note patterns and trends in this data set. The TZW IC list includes total PAHs, HPAHs, cPAHs, BaPEq, as well as individual PAHs BAP and naphthalene. LPAHs are also discussed here to provide for complete discussion of total PAHs and HPAHs, though they are not on the IC list for TZW. The physical properties and behavior patterns of individual PAHs vary with molecular weight, with the larger PAHs exhibiting more hydrophobicity and a stronger tendency to adsorb to sediments.

5.4.5.1 Observed Chemical Distribution of PAHs in TZW

In the Round 2 GWPA SAP (Integral et al. 2005), PAHs were identified as an upland groundwater COI for six of the nine TZW study sites: Kinder Morgan, ARCO, ExxonMobil, Gasco, Siltronic, and Willbridge Terminal. TZW samples for individual PAHs, total PAHs, HPAHs, and cPAHs were collected at a total of 53 individual sample locations offshore of these sites during the 2004 Pilot Study and 2005 GWPA sampling events. Twenty-two locations were sampled for individual PAHs and PAH sums by Gasco in 2007 (Anchor 2008b), and 13 locations were sampled by Siltronic in 2005 (HAI 2005a; MFA 2005a). An additional 24 deep (90 cm bml) naphthalene samples were also collected by Siltronic. All sample results for summed and individual PAH parameters in TZW are presented in the SCRA database (Appendix A3) and are summarized in Appendix D4, Table D4-1.

These sample results are presented on Maps 5.4-2a–e, 5.4-3a–d, 5.4-4a–d, and 5.4-5a–e, for PAHs (including LPAHs and HPAHs), cPAHs (including BaPEq values), BAP, and naphthalene, respectively. Each map set presents filtered (top panel) and unfiltered (bottom panel) TZW results, where available, with inset histograms summarizing the distribution of samples shown on each map relative to the distribution across the TZW data set. There are 34 collocated filtered and unfiltered Trident sample pairs, and there are seven deep (90 to 150 cm bml) Trident samples in this data set. For naphthalene, there are an additional 24 deep (90 cm bml) samples collected from RM 6 to 7. For total PAHs, cPAHs, and BAP, sample results collected between RM 6 and 7 are presented on two maps to allow for presentation of all sample concentration results in this densely sampled area (the first map shows concentration labels for LWG-collected data, and the second map shows concentration labels for non-LWG collected data).

Naphthalene sample results are presented on Maps 5.4-5a–e. Naphthalene is a two-ring LPAH, and has the lowest molecular weight of all the PAHs. Naphthalene data were collected as part of VOC sample analysis (EPA method SW8260), as well as PAH sample analysis (EPA method 8270), and therefore naphthalene was sampled at more locations (160) than other individual PAHs or total PAH sums (88 locations). Where naphthalene results were generated using both analysis techniques, the method 8260 results are presented.[20] Due to its high volatility, only unfiltered sample results are presented on Maps 5.4-5a–e. For naphthalene, sample results collected between RM 6 and 7 are presented on three maps (Map 5.4-5b presents all shallow sample results; Map 5.4-5c is a zoom-in to show shallow sample results in the most densely sampled area; Map 5.4-5d presents all deep sample results).

5.4.5.1.1 Total PAHs

PAHs were detected in TZW samples offshore of all six sites. Total PAH values observed in TZW cover a large concentration range. In shallow filtered Trident and

[20] Seventy-eight naphthalene samples were analyzed with both 8260 and 8270 methods. Of these, 49 samples were below detection limits for both methods. For the 19 samples which were detected with both methods, higher concentrations were found with method 8260 in 73% of the samples.

peeper samples, concentrations ranged from detection limits (0.042 U µg/L to 0.073 U µg/L) to 1,200 J µg/L; in unfiltered Trident and Geoprobe samples, concentrations ranged from detection limits (0.036 U µg/L to 0.043 U µg/L) to 15,100 JA µg/L. The highest PAH sample results were observed offshore of the Gasco and Siltronic sites, with the maximum filtered result measured at GS-02-A, at the downstream end of the Gasco site. The highest (>1,000 µg/L; shown as red symbols on Maps 5.4-2a-e) unfiltered concentrations of total PAHs in TZW were observed at stations GP-73 (15,100 JA µg/L), GP-68 (13,100 JA µg/L), GP-64 (12,600 JA µg/L), and GP-69 (12,100 JA µg/L), all located roughly offshore of the property line between the Gasco and Siltronic sites. Unfiltered, deep Trident (90 to 150 cm bml) total PAH sample concentrations ranged from 0.67 to 430 µg/L, with the highest deep concentration at GS-08-D, also located offshore of the property line between the Gasco and Siltronic sites. No filtered deep Trident samples were collected.

Observed PAH concentration ranges varied substantially among the offshore study areas, with the highest total PAH concentrations consistently being observed offshore of the Gasco (up to 15,100 JA µg/L) and Siltronic (up to 13,100 JA µg/L) sites. The lowest range of TZW PAH concentrations was observed offshore of the Willbridge Terminal site. These relative concentration ranges are apparent on the inset histograms on Maps 5.4-2a–e.

5.4.5.1.2 LPAHs and HPAHs

LPAH and HPAH components of the total PAH sum are shown in concentration labels on Maps 5.4-2a–e. These maps indicate that LPAH concentrations are consistently higher than HPAHs in TZW sampled across the Study Area. Further, LPAH concentrations show similar spatial patterns to the total PAH concentrations.

In peeper and filtered Trident samples, HPAH concentrations range from detection limits (0.0071 U µg/L to 0.38 U µg/L) to 7.6 J µg/L, while LPAH concentrations range from detection limits (0.042 U µg/L to 0.073 U µg/L) to 1,190 µg/L. In unfiltered Trident and Geoprobe samples, HPAH concentrations covered a higher range, from detection limits (0.036 U µg/L to 0.66 U µg/L) to 880 µg/L, as did LPAH concentrations, ranging from detection limits (0.016 U µg/L to 0.069 U µg/L) to 14,600 JA µg/L.

For both HPAH and LPAH sums, the highest results were recorded offshore of the Gasco and Siltronic sites. The highest unfiltered HPAH concentration was recorded at GS-B2 near the north end of the Gasco site. The highest filtered and peeper HPAH concentrations were also measured in this area at GS-01-B (7.6 J µg/L) and GS-02-A (5.46 J µg/L). The highest unfiltered LPAH concentration was recorded at GP-73, roughly offshore of the property line between the Gasco and Siltronic sites. The highest filtered LPAH results were located at the north end of the Gasco site at GS-02-A (1,190 µg/L) and GS-01-B (290 µg/L).

5.4.5.1.3 Total Carcinogenic PAHs

cPAH represents the sum of a subset of HPAHs and, as expected, the cPAH distribution follows that of HPAHs as shown on Maps 5.4-3a–d. In peeper and filtered Trident samples, cPAH concentrations range from detection limits (0.0024 U µg/L to 0.017 U µg/L) to 0.64 J µg/L. In unfiltered Trident and Geoprobe samples, cPAH concentrations range from detection limits (0.0077 U µg/L to 0.086 U µg/L) to 300 µg/L. The maximum filtered, peeper, and unfiltered cPAH concentrations measured were collocated with the maximum HPAH results at GS-02-A, GS-01-B, and GS-B2, respectively. Toxicity-weighted BaPEq concentrations range from detection limits (0.0018 U µg/L to 0.028 U µg/L) to 70.3 µg/L. cPAH BaPEq concentrations are positively correlated with cPAH concentrations and maximum results were collocated with HPAH and cPAH maximum results.

5.4.5.1.4 Benzo(a)pyrene

BAP is a five-ring HPAH. Maps 5.4-4a–d show that the BAP spatial distribution follows that of HPAHs. In peeper and filtered Trident samples, BAP concentrations range from below detection limits (0.0018 U µg/L to 0.012 U µg/L) to 0.082 µg/L. In unfiltered Trident and Geoprobe samples, BAP concentrations range from detection limits (0.0021 U µg/L to 0.062 U µg/L) to 50 µg/L. The maximum filtered, peeper, and unfiltered BAP concentrations measured were collocated with the maximum HPAH results at GS-02-A, GS-01-B, and GS-B2, respectively.

5.4.5.1.5 Naphthalene

Naphthalene is a two-ring LPAH, and has the lowest molecular weight of all the PAHs. Maps 5.4-5a–e present naphthalene concentrations for peeper, unfiltered Trident, and unfiltered Geoprobe samples. In unfiltered shallow Trident and Geoprobe samples, naphthalene concentrations range from detection limits (0.0063 U µg/L to 15 U µg/L[21]) to 13,700 µg/L. The highest (>10,000 µg/L; shown as red symbols on Maps 5.4-5a–e) concentrations were observed offshore of the Gasco and Siltronic sites, with maximum shallow naphthalene concentrations collocated with or proximal to the maximum total PAH and LPAH locations at GP-73 (13,700 µg/L), GP-71 (12,400 µg/L), GP-68 (12,300 µg/L), and GP-64 (12,200 µg/L). Deep Trident (90 to 150 cm bml) and Geoprobe (91 cm bml) naphthalene sample concentrations ranged from detection limits (0.041 U µg/L to 150 U µg/L[22]) to 11,200 µg/L, with the highest deep concentration at GP-76, also located offshore of the property line between the Gasco and Siltronic sites.

Patterns and trends in the data sets for total PAHs, cPAHs, BAP, and naphthalene are discussed in the following subsection.

[21] High detection limits were reported for six samples, all located offshore of the former Arkema Acid Plant site and the Rhone Poulenc site. The median detection limit for naphthalene for the entire TZW data set is 0.29 µg/L.
[22] See previous note above.

5.4.5.2 Patterns and Trends of PAHs in TZW

As shown on the river-mile-specific portions of the histograms of Maps 5.4-2a–e and described above, the observed ranges of unfiltered total PAH concentrations for each river mile sampled are higher than filtered and peeper-sampled concentrations. This trend is upheld in a point-by-point assessment of the 34 collocated filtered and unfiltered Trident pairs for total PAHs. Looking at the filtered and unfiltered Trident pairs, filtration decreased the average total PAH concentration by 24 percent, with a maximum decrease of up to 99 percent (AR-02-B). Note that the apparent large decrease in the total PAH concentration range with filtration seen on the harbor-wide portion of the histograms is largely attributable to the lack of filtered PAH samples from the non-LWG investigations offshore of the Siltronic and Gasco sites. The Siltronic and Gasco unfiltered results account for most of the >1,000 µg/L values.

By examining the effects of filtration, the effects of individual PAH hydrophobicity can be seen in the data set. As shown on inset histograms on Maps 5.4-3a–d and 5.4-4a–d, the large, hydrophobic cPAHs and BAP show extreme differences between filtered and unfiltered results (with an average of over 95 percent decrease in a paired sample comparison for both cPAHs and BAP), suggesting that these chemicals are present on solids >0.45 µm in diameter in unfiltered TZW samples. In contrast, the smaller, less hydrophobic individual PAHs have a lesser tendency to bind to sediment and would be expected to be less affected by filtration, contributing the lower net effect of filtration on total PAH concentrations. This filtration effect on total PAHs is therefore a function of the composition of each sample.

Comparison of total PAH concentrations in the seven collocated deep and shallow (0 to 38 cm bml) TZW samples shows that the unfiltered deep samples consistently exhibit higher concentrations than the corresponding shallow samples. In some cases, the unfiltered, deep sample concentrations are up to 2 orders of magnitude greater than the collocated shallow samples. The three filtered deep/shallow sample pair results, however, show no clear relative trend. As with total PAHs, analysis of collocated deep/shallow sample pair results shows that unfiltered concentrations of HPAHs, cPAHs, and BAP are generally much higher in deep samples. For naphthalene, 17 collocated deep (90 to 150 cm bml) and shallow (0 to 38 cm bml) Trident samples and 24 collocated deep (91 cm bml) and shallow (31 cm bml) Geoprobe samples were collected. In general, for the high (>1,000 µg/L) concentration samples collected from RM 6 to 7, shallow samples showed somewhat higher concentrations than collocated deep samples. For other parts of the Study Area, no clear trends with depth were observed for naphthalene.

Stacked bar charts showing percent composition of detected total PAH sums are shown on Figures 5.4-2a–f. The charts present total PAH concentration (indicated by the black line corresponding to the right y-axis), as well as the fraction of the total contributed by each of the 17 PAHs. Charts are presented for each relevant TZW study site, with samples organized along the x-axis in groups referring to groundwater discharge zones. These zones are indicative of areas of similarly mapped groundwater discharge

conditions offshore of the given study site and are presented and discussed in detail in Appendix C2.

There are a several patterns apparent in these stacked bar charts. First, duplicate sample pairs show good reproducibility in composition trending. Second, the shallow and deep pairs frequently show variable compositions, particularly in cases where the deeper concentration is greater than the shallow concentration (e.g., EM04C-TR-uf-150, which has an extreme concentration difference between the shallow and deep result). This may reflect weathering in the biologically active zone and/or differences in PAH composition in sediment with depth. Next, a distinct chemical composition is generally present at sample locations with very high total PAH concentrations. Acenaphthene is the dominant component of total PAH sums in most samples; however, at sample locations with total PAHs greater than ~1,000 µg/L, naphthalene concentrations clearly dominate the composition. This result is interesting to evaluation of TZW because naphthalene is the most mobile chemical of the PAHs. This composition pattern is apparent in the bar charts for TZW data offshore of the Gasco and Siltronic sites (Figures 5.4-2d and 5.4-2e). These areas correspond to suspected discharge zones offshore of the Siltronic and Gasco sites (discussed in detail in Appendix C2). Composition trends with concentration and location are less apparent at the other TZW study sites, where total PAH concentrations cover a much lower concentration range.

In summary, of the sites sampled, total PAH concentrations were found to be highest offshore of the Gasco and Siltronic sites. Total HPAH, total cPAH, and BAP results showed similar distribution and filtration patterns. Because LPAHs tend to compose the majority of the total PAH concentrations, LPAH and naphthalene results generally followed the distribution patterns apparent for total PAHs. Filtration was observed to decrease the total PAH concentration slightly, with greater effects on the more hydrophobic PAHs, as expected. For total PAHs, total HPAH, total cPAH, and BAP, the unfiltered deeper (90 to 150 cm bml) Trident samples consistently showed higher concentrations than corresponding unfiltered shallow (0 to 38 cm bml) samples. Review of the fractional composition of the 17 individual chemicals that comprise total PAH shows a clear pattern of high naphthalene concentrations associated with high total PAH concentrations offshore of the Gasco and Siltronic sites. For these high-concentration naphthalene locations, shallow (31 cm bml) Geoprobe samples generally had slightly higher concentrations than the collocated deeper (91 cm bml) samples. Additional evaluation of PAHs in TZW is provided in the detailed, site-specific discussions of groundwater pathways in Appendix C2.

5.4.6 TPH in TZW

The following subsections present the in-river distribution of TPH results in TZW samples, as well as a discussion of patterns and trends in this data set. The TZW IC list includes DRH, RRH, and GRH. TPH is not a TZW IC, but it is presented and discussed here to support the discussion of the distribution and patterns of the components. TPH, as analyzed for the Round 2 investigation, is the measure of all hydrocarbons and non-hydrocarbons that can be quantified in the carbon range from n-C_6 to n-C_{38}.

Likewise, the components (DRH, RRH, and GRH) are simply descriptive laboratory terms for the fractions of TPH, and not source assignments or indications of toxicity. Further, the TPH data contain hydrocarbons of both natural and anthropogenic origin.

5.4.6.1 Observed Chemical Distribution of TPH in TZW

In the Round 2 GWPA SAP (Integral et al. 2005), TPH was identified as an upland groundwater COI for six of the nine TZW study sites: Kinder Morgan, ARCO, ExxonMobil, Gasco, Siltronic, and Willbridge Terminal. Shallow TZW samples for TPH were collected at a total of 47 individual sample locations offshore of these sites during the 2004 Pilot Study and 2005 GWPA sampling events. Paired filtered and unfiltered samples were collected at 30 of these locations, and deep samples (90 to 150 cm bml) were collected at eight of these locations. TPH was not sampled during the non-LWG TZW investigations at Siltronic in 2005 (MFA 2005a,b) and Gasco in 2007 (Anchor 2008b).

Sample results are presented on Maps 5.4-6a–d, including distinction in the flyout boxes of diesel-range, residual-range, and gasoline-range fractions of TPH. These maps present filtered (top panel) and unfiltered (bottom panel) TZW results, where available, with inset histograms summarizing the distribution of samples shown on each map relative to the distribution across the entire Study Area. Note that gasoline-range fractions of TPH are volatile, and were therefore not evaluated for filtered samples. All summed and individual TPH sample results for TZW are presented in the SCRA database (Appendix A3) and are summarized in Appendix D4, Table D4-1.

TPH was detected in TZW samples offshore of all six of the studied sites. TPH values observed in TZW cover a fairly large concentration range, varying from undetected (detection limits ranged from 0.058 U mg/L to 0.26 U mg/L) to 4.1 J mg/L in filtered Trident and peeper samples. In unfiltered Trident samples, observed concentrations ranged from undetected (detection limits ranged from 0.038 U mg/L to 0.18 U mg/L) to 11.3 J mg/L. The highest concentrations of TPH in TZW were observed offshore of the Gasco and Siltronic sites, with the highest unfiltered Trident concentrations of TPH observed at Gasco locations GS-07-B (11.3 J mg/L) and GS-02-A (6.01 J mg/L) in unfiltered samples, and the highest filtered or peeper TPH results observed at Gasco location GS-02-A (4.1 J mg/L) and Siltronic location SL-04-F (2.35 J mg/L). The lowest range of TZW TPH concentrations were observed offshore of the Willbridge Terminal site.

5.4.6.2 Patterns and Trends of TPH in TZW

As shown on the histograms of the TPH map series, the distribution of unfiltered samples covers a larger and higher concentration range than the distribution of filtered samples; however, it should be noted that filtered TPH values do not include GRH (per sampling protocols for VOCs, filtered samples of GRH were not collected). Focusing on DRH and RRH (Table D4-1), it is clear, however, that filtration consistently decreases the concentration of these components. Further interpretation of this apparent effect of filtration is complicated by the expected variability in TPH composition (and

corresponding hydrophobicity) from site to site (and likely sample to sample). As noted above, TPH, as analyzed for the Round 2 investigation, is simply the measure of all hydrocarbons and non-hydrocarbons that can be quantified in the carbon range from *n*-C_6 to *n*-C_{38}, with no distinction/identification of specific component chemicals.

Comparison of TPH concentrations in the eight deep sample locations (90 to 150 cm bml) with the corresponding shallow results (0 to 38 cm bml) shows deep unfiltered results to be generally greater than shallow unfiltered results. Filtered deep results, however, show no clear trend. As discussed in Appendix C2, this information was evaluated in greater detail for each TZW study site in the consideration of groundwater pathway along with other lines of evidence.

To support consideration of patterns in the fractional composition of TPH, stacked bar charts are presented in Figures 5.4-3a–f, with one figure for each relevant TZW study site. Each figure presents results grouped by interpreted groundwater discharge zones. These zones are indicative of areas of similarly mapped groundwater discharge conditions offshore of the given study site and are discussed in detail in Appendix C2. These figures present both filtered and unfiltered results, with filtered sample IDs highlighted. As noted above, the filtered results for TPH do not contain GRH fractions. A general review of these figures reveals that, with few exceptions, duplicate results (indicated by "dup" in the sample ID) show similar composition and total concentration to the original sample.

Looking at the bar chart results site-by-site, there are few notable patterns in these plots relative to the interpreted zoning of groundwater discharge. Offshore of the Siltronic site, the "offshore discharge zone," where there is a complete pathway for groundwater discharge of select VOCs (discussed in Appendix C2), shows a fairly clear TPH composition shift toward GRHs, as compared to the samples from the other Siltronic zones. (Note again that the highlighted filtered samples do not contain GRH in the TPH composition presented.) Similar increases in GRH fractional composition of TPH can be seen in samples offshore of the Gasco, ARCO, and ExxonMobil sites; however, dominance of the TPH concentration in these unfiltered samples offshore of these sites is also generally accompanied by a sharp decrease in TPH concentration, making the pattern more difficult to interpret (possibly more reflective of the lack of DRH and RRH than any increase in GRH). Further, these concentration ranges are close to detection limits, and the GRH results typically have lower detection limits (the average gasoline-range detection limit was 0.06 mg/L compared to 0.14 and 0.16 mg/L for RRH and DRH samples).

In summary, the general nature of the analyte TPH (unknown mixture of anthropogenic and natural hydrocarbons and non-hydrocarbons that can be quantified in the carbon range from *n*-C_6 to *n*-C_{38}) significantly confounds detailed, source-related interpretation of results. Overall, filtration was observed to decreases the DRH and RRH fractions. Review of the fractional components (DRH, RRH, and GRH) showed a composition shift toward GRH in the offshore discharge zone at the Siltronic site; however, patterns

elsewhere were generally weak and more difficult to interpret when TPH concentration changes were also considered. Site-specific conclusions about groundwater pathway are presented in detail with additional lines of evidence in Appendix C2 and summarized in Section 4.

5.4.7 Additional Indicator Chemicals in TZW

The following subsections present brief descriptions of the in-river distribution of TZW ICs not on the Study Area-wide bounding chemical list (total PCBs, total PCDD/Fs, total DDx, total PAHs). Note that the bounding ICs were not the focus chemicals at all TZW study sites and were not sampled at many of the sites. As described in the introduction to Section 5.4, the distribution of additional ICs in TZW is briefly discussed here. These additional ICs are grouped by chemical type, where possible, in the subsequent discussions. Supporting maps are presented for all chemicals. All sample results for TZW are presented in the SCRA database (Appendix A3) and are summarized in Appendix D4, Table D4-1.

5.4.7.1 Silvex in TZW

Samples were analyzed for the herbicide Silvex in TZW in the vicinity of the former Rhone Poulenc plant. The sampling results for Silvex in TZW are shown on Map 5.4-7. Seven locations were sampled in the shallow TZW (0 to 38 cm bml), and 2 collocated deeper samples were collected (90 to 150 cm bml). Additionally, six collocated filtered samples were collected.

Silvex was detected in 27 percent of the samples. Silvex concentrations in TZW samples ranged from undetected (detection limits ranged from 0.06 U µg/L to 0.37 U µg/L) to 22 µg/L, with the maximum concentration measured at RP-03-E. All Silvex detections were located within a zone identified as an offshore groundwater discharge zone for the Rhone Poulenc site. This zone is presented in Section 4 and discussed in detail in Appendix C2, along with the lines of evidence used to make this determination.

5.4.7.2 Cyanide in TZW

Samples were analyzed for cyanide in TZW offshore of only the Gasco and Siltronic sites during the Round 2 GWPA sampling effort and during the two non-LWG sampling efforts (Anchor 2008b; MFA 2005a,b). The sampling results for cyanide in TZW are shown on Map 5.4-8. Fifty-two locations were sampled offshore of these two sites for cyanide, and three paired deep (90 to 150 cm bml) samples were collected. For cyanide, only unfiltered samples were collected. Total cyanide, very soluble and chemically stable, is not expected to adsorb significantly to solids. Measurements of the most toxic form of cyanide, free cyanide, were made offshore of the Gasco and Siltronic sites as part of the non-LWG 2007 investigation (Anchor 2008b); however, free cyanide was only detected in one of those samples (GS-B2). Free cyanide is discussed in detail in the *Offshore Investigation Report, NW Natural "Gasco" Site* (Anchor 2008b).

Total cyanide was detected in 95 percent of the TZW samples. Cyanide concentrations ranged from undetected (detection limits ranged from 0.003 U mg/L to 1.4 U mg/L) to 23.1 J mg/L, with the maximum concentration observed at GS-02-A. Cyanide concentrations in the three deeper TZW samples (90 to 150 cm bml) were comparable to the concentrations observed in the shallow (0 to 38 cm bml) samples. Site-specific conclusions about groundwater pathway are presented in detail with additional lines of evidence in Appendix C2.

5.4.7.3 Perchlorate in TZW

Samples were analyzed for perchlorate in TZW only offshore of the former Chlorate Plant and former Acid Plant areas of the Arkema site. The sampling results for perchlorate in TZW are shown on Map 5.4-9. Twenty locations were sampled in the shallow TZW (0 to 38 cm bml), and seven samples were collected in deeper TZW (90 to 150 cm bml). Only unfiltered samples were collected, since perchlorate is ionic and not expected to adsorb significantly to solids.

Perchlorate was detected in 52 percent of the samples. Perchlorate concentrations in shallow TZW samples ranged from undetected (detection limits ranged from 0.4 U µg/L to 20,000 U µg/L[23]) to 177,000 µg/L; and the maximum deeper (90 to 150 cm bml) sample concentrations ranged from undetected (detection limits ranged from 40 U µg/L to 40,000 U µg/L[24]) to 210,000 µg/L. These values are comparable to or higher than those observed in the upland nearshore groundwater. Perchlorate concentrations in the deeper TZW samples (90 to 150 cm bml) collected offshore of the Arkema site were often higher than in the collocated shallower samples (0 to 38 cm bml). The highest (>100,000 µg/L; shown as red symbols on Map 5.4-9) perchlorate concentrations were found along transect 7 (sampling locations CP-07-A, CP-07-B, and CP-07-D).

5.4.7.4 Metals in TZW

The TZW IC list includes eight metals: arsenic, barium, cadmium, copper, lead, manganese, nickel, and zinc. As part of the 2004 Pilot Study, Round 2 GWPA, and non-LWG Gasco and Siltronic sampling efforts, samples were analyzed for metals at all nine TZW study sites. Where sample volumes permitted in the Round 2 GWPA sampling effort, filtered samples were also collected; however, only three filtered metals samples were collected as part of the non-LWG investigation offshore of the Gasco and Siltronic sites. This distinction is important to note when reviewing the paired figures showing filtered and unfiltered results for each metal. In all, 126 locations were sampled for metals. Of these, filtered pairs were collected at 62 locations and paired deep samples (90 to 150 cm bml with Trident) were collected at 22 locations.

[23] High detection limits (4,000 U µg/L, 20,000 U µg/L, and 40,000 U µg/L) were reported for 3 samples, all located offshore of the former Arkema Acid Plant site. The median detection limit for perchlorate for the entire TZW data set is 10 µg/L.
[24] See note 23 above.

Sampling results for metals are also presented on scatter plots in Figures 5.4-4a–h. These figures show sample concentrations along an x-axis noting the river mile of each sample location. Color-coded symbols distinguish sample type and depth. Paired plot sets are presented for each chemical to show filtered and unfiltered results, where available. Additionally, arsenic, copper, and zinc results are presented on Maps 5.4-10a–e, 5.4-11a–e, and 5.4-12a–e, respectively, to support cross-media consideration in Section 10 (these are the TZW IC metals that are also included on the CSM IC list, Table 5.0-2).

5.4.7.4.1 Arsenic Distribution

Arsenic TZW results are shown on Figure 5.4-4a and Maps 5.4-10a–e. Arsenic was detected in 88 percent of TZW samples, and detection limits ranged from 0.2 U µg/L to 20 U µg/L. Detected arsenic concentrations ranged from 0.3 J µg/L to 77.3 µg/L, with an average concentration of 11.1 µg/L. Both filtered and unfiltered arsenic concentrations spanned generally consistent ranges across the Study Area. Evaluation of paired filtered and unfiltered detected samples shows that filtration decreases the total arsenic concentration by 16 percent on average. Deep (90 to 150 cm bml) Trident samples showed higher concentrations than collocated shallow (0 to 38 cm bml) samples in 17 of 24 collocated deep/shallow sample pairs The tendency of arsenic compounds to associate with sediment particles can be highly variable and is strongly influenced by redox chemistry and microbial processes. A more detailed biogeochemical discussion of arsenic in TZW, sediments, and upland groundwater is presented in Appendix C2.

5.4.7.4.2 Barium Distribution

Barium TZW results are shown on Figure 5.4-4b. Barium was detected in 99 percent of TZW samples. Detected barium concentrations ranged from 4.06 to 4,630 µg/L, with an average concentration of 334 µg/L. The highest (>1,000 µg/L) peeper, unfiltered Trident, and unfiltered push probe barium concentrations were observed offshore of the Gasco, Arkema, and Rhone Poulenc sites, while notably high filtered barium measurements were observed offshore of the Arkema site. Evaluation of paired filtered and unfiltered detected samples shows that filtration decreases the total barium concentration by 38 percent on average. Deep (90 to 150 cm bml) Trident samples showed higher concentrations than collocated shallow (0 to 38 cm bml) samples in 14 of 24 collocated deep/shallow sample pairs. The tendency of barium compounds to associate with sediment particles can be highly variable and is strongly influenced by redox chemistry and microbial processes. A more detailed biogeochemical discussion of barium in TZW, sediments, and upland groundwater is presented in Appendix C2.

5.4.7.4.3 Cadmium Distribution

Cadmium TZW results are shown on Figure 5.4-4c. Cadmium was detected in 68 percent of TZW samples, with detection limits ranging from 0.002 U µg/L to 0.2 U µg/L and detected concentrations ranging from 0.004 J µg/L to 36 µg/L. The average cadmium concentration is 0.45 µg/L. The highest (>1.5 µg/L) peeper, unfiltered Trident, and unfiltered push probe cadmium concentrations were observed

offshore of the Arkema (former Chlorate Plant area), Gasco, and Siltronic sites, while filtered Trident and push probe cadmium concentrations were consistently low across the Study Area. Evaluation of paired filtered and unfiltered detected samples shows that filtration decreases the total cadmium concentration by 53 percent on average. Deep (90 to 150 cm bml) Trident samples showed higher concentrations than collocated shallow (0 to 38 cm bml) samples in 9 of 24 collocated deep/shallow sample pairs. Neither the filtered nor unfiltered Trident samples exhibited consistent trends with depth.

5.4.7.4.4 Copper Distribution

Copper TZW results are shown on Figure 5.4-4d and Maps 5.4-11a–e. Overall, copper was detected in 51 percent of TZW samples, and detection limits ranged from 0.08 U µg/L to 32.9 U µg/L. Detected peeper, Trident, and push probe copper concentrations ranged from 0.28 to 555 µg/L, with an average concentration of 27.9 µg/L. The highest (>100 µg/L) concentrations were measured offshore of the Gasco and Siltronic sites in push probe samples. The low (30 percent) detection frequency for filtered copper samples is evident in the lower plot on Figure 5.4-4d, with few detected filtered copper samples observed across the Study Area. Evaluation of paired filtered and unfiltered detected samples shows that filtration decreases the total copper concentration by 92 percent on average, suggesting that most copper in the transition zone is associated with >0.45 µm particles. Deep (90 to 150 cm bml) Trident samples showed higher concentrations than collocated shallow (0 to 38 cm bml) samples in 5 of 24 collocated deep/shallow sample pairs. Neither the filtered nor unfiltered Trident samples exhibited consistent trends with depth.

5.4.7.4.5 Lead Distribution

Lead TZW results are shown on Figure 5.4-4e. Lead was detected in 55 percent of TZW samples, with detection limits ranging from 0.01 U µg/L to 20 U µg/L. Detected lead concentrations ranged from 0.01 J µg/L to 382 J µg/L, with an average concentration of 17.4 µg/L. The highest (>50 µg/L) total lead concentrations (peeper, unfiltered Trident, and unfiltered push probe) were observed offshore of the Gasco, Siltronic, Arkema, and ARCO sites, while filtered Trident and push probe lead concentrations were consistently low across the Study Area. Evaluation of paired filtered and unfiltered detected samples shows that filtration decreases the total lead concentration by 97 percent on average, suggesting that most lead in the transition zone is associated with solids. Deep (90 to 150 cm bml) Trident samples showed higher concentrations than collocated shallow (0 to 38 cm bml) samples in 11 of 24 collocated deep/shallow sample pairs. Neither the filtered nor unfiltered Trident samples exhibited consistent trends with depth.

5.4.7.4.6 Manganese Distribution

Manganese TZW results are shown on Figure 5.4-4f. Manganese was detected in all of the TZW samples collected, with a concentration range from 23 J µg/L to 66,200 µg/L and an average concentration of 4,930 µg/L. Filtered, peeper, and unfiltered manganese concentrations spanned generally consistent ranges across the Study Area. Filtration

reduced manganese sample concentrations somewhat, with an average reduction of 22 percent for detected unfiltered and filtered sample pairs. Deep (90 to 150 cm bml) Trident samples showed higher concentrations than collocated shallow (0 to 38 cm bml) samples in 11 of 24 collocated deep/shallow sample pairs. Neither the filtered nor unfiltered Trident samples exhibited consistent trends with depth. The tendency of manganese compounds to associate with sediment particles can be highly variable and is strongly influenced by redox chemistry and microbial processes. A more detailed biogeochemical discussion of manganese in TZW, sediments, and upland groundwater is presented in Appendix C2.

5.4.7.4.7 Nickel Distribution

Nickel TZW results are shown on Figure 5.4-4g. Nickel was detected in 87 percent of TZW samples, and detection limits ranged from 0.2 UJ µg/L to 20 U µg/L. Detected nickel concentrations ranged from 0.2 J µg/L to 367 µg/L, with an average concentration of 18.5 µg/L. The highest (>100 µg/L) total nickel concentrations (peeper, unfiltered Trident, and unfiltered push probe) were observed offshore of the Gasco, Siltronic, and Arkema sites, while filtered Trident and push probe nickel concentrations were consistently low across the Study Area. Evaluation of paired filtered and unfiltered detected samples shows that filtration decreases the total nickel concentration by 49 percent on average. Deep (90 to 150 cm bml) Trident samples showed higher concentrations than collocated shallow (0 to 38 cm bml) samples in 8 of 24 collocated deep/shallow sample pairs. Neither the filtered nor unfiltered Trident samples exhibited consistent trends with depth.

5.4.7.4.8 Zinc Distribution

Zinc TZW results are shown on Figure 5.4-4h and Maps 5.4-12a–e. Zinc was detected in 66 percent of TZW samples, with detection limits ranging from 0.78 UJ µg/L to 16.4 U µg/L. Detected zinc concentrations ranged from 0.95 to 3,590 µg/L, with an average concentration of 113 µg/L. The highest (>1,000 µg/L) total zinc concentrations were observed offshore of the Gasco and Siltronic sites in unfiltered push probe samples, while the highest (>20 µg/L) filtered Trident and push probe zinc concentrations were measured offshore of the ARCO and Gasco sites. Evaluation of paired filtered and unfiltered detected samples shows that filtration decreases the total zinc concentration by 81 percent on average, suggesting that most zinc in the transition zone is associated with >0.45 µm particles. Deep (90 to 150 cm bml) Trident samples showed higher concentrations than collocated shallow (0 to 38 cm bml) samples in 9 of 24 collocated deep/shallow sample pairs.

5.4.7.4.9 Metals Patterns and Trends

In general, cadmium, copper, lead, nickel, and zinc in TZW showed a relatively consistent baseline range of concentrations offshore of all the nine TZW study sites, with relatively elevated levels offshore of a few of the study sites as shown on Figures 5.4-4a–h and Maps 5.4-10a–e, 5.4-11a–e, and 5.4-12a–e. Filtration reduced TZW concentrations by more than 50 percent for cadmium, copper, lead, nickel, and zinc. Both filtered and unfiltered arsenic and manganese concentrations spanned

generally consistent ranges across the Study Area. The spatial distribution of metals likely reflects some combination of upland groundwater transport, local sediment redox conditions, and sediment contamination, varying by site and metal.

Filtration significantly and consistently reduced TZW sample concentrations for cadmium, copper, lead, nickel, and zinc, suggesting that much of the observed total concentration for these metals was associated with solid particles. Differences with filtration were not as pronounced for arsenic, barium, and manganese. (Note in review of Figures 5.4-4a–h, paired filtered results are not available for most samples collected offshore of the Gasco and Siltronic sites as part of the non-LWG investigations.) Analysis of collocated shallow (0 to 38 cm bml) and deep (90 to 150 cm bml) TZW results did not show consistent trends with depth for any of the metals.

Site-specific conclusions about groundwater pathways are presented in detail with additional lines of evidence in Appendix C2 and summarized in Section 4. Additionally, a geochemical analysis of arsenic, barium, and manganese is presented in Appendix C2 to further investigate the origin of the observed TZW concentrations. The geochemical analysis includes a statistical evaluation of the spatial distribution of these metals in TZW across the nine TZW study sites and a comparison of TZW concentrations with available upland groundwater concentrations. Geochemical controls on arsenic, barium, and manganese in TZW were evaluated by exploring correlations between metal concentrations and measured variables (e.g., pH, oxidation-reduction potential, alkalinity, and TOC) that could be expected to exert an influence upon their geochemical behavior. Geochemical modeling was performed to identify stable mineral and aqueous phases as a function of pH and Eh, mineral saturation indices, and mineral phases controlling the aqueous solubility of arsenic, barium, and manganese.

5.4.7.5 Chlorinated VOCs and SVOCs in TZW

The TZW IC list includes nine chlorinated VOCs (MCB; chloroethane; 1,2-DCA; 1,1,2-TCA; vinyl chloride; cis-1,2-DCE; TCE; chloroform; and methylene chloride) and one chlorinated SVOC (1,2-DCB). These VOCs were sampled offshore of all nine study sites as part of the 2004 Pilot Study, Round 2 GWPA, and non-LWG Gasco and Siltronic sampling efforts. In all, 150 locations were sampled for VOCs. Of these, paired deep (90 to 150 cm bml) Trident samples were collected at 18 locations throughout the Study Area, and 24 paired deep (91 cm bml) Geoprobe samples were collected offshore of Gasco and Siltronic. No filtered VOC or 1,2-DCB samples were collected, per sampling protocols for volatile compounds. Sampling results for these chemicals are presented spatially on Maps 5.4-13 through 5.4-18. Results are presented graphically for only those river miles where the chemicals were detected (Table 5.4-1 summarizes the mapped areas for each chemical). All sample results for VOCs and SVOCs in TZW are also presented in the SCRA database (Appendix A) and are summarized in Appendix D4, Table D4-1. Detailed site-by-site assessments of VOC concentrations with depth, along with other lines of evidence, are provided in Appendix C2 to support the assessment of groundwater pathway.

5.4.7.5.1 MCB and 1,2-DCB

Maps 5.4-13a–e present observed MCB and 1,2-DCB concentrations in TZW offshore of the sites where these analytes were detected (Gasco, Siltronic, Rhone Poulenc, and Arkema; RM 6–7 and 7–8). Detected MCB concentrations ranged from 0.15 J µg/L to 30,000[25] µg/L, with the highest MCB concentrations (>1,000 µg/L; indicated by red symbols in the upper panels on Maps 5.4-13a–e) observed offshore of the Arkema site. Detected 1,2-DCB concentrations ranged from 0.14 J µg/L to 640 µg/L. The highest 1,2-DCB concentrations (>250 µg/L; indicated by red symbols in the lower panels on Maps 5.4-13 a–e) were observed near Rhone Poulenc. There were no clear trends in sample concentration with depth for these chemicals across these study sites. The presence of both MCB and DCB offshore may be indicative of the chemicals' release and/or the degree of degradation that has occurred from 1,2-DCB to MCB.

5.4.7.5.2 1,2-DCA and 1,1,2-TCA

Maps 5.4-14a–b present observed 1,2-DCA and 1,1,2-TCA concentrations in TZW offshore of the sites where these analytes were detected[26] (Arkema and Gunderson; RM 7–8 and 8–9). For 1,2-DCA, there were seven detected observations offshore of the Arkema site and four detected observations offshore of the Gunderson site. Detected 1,2-DCA concentrations ranged from 0.13 J µg/L to 770 µg/L. The highest 1,2-DCA concentrations (>100 µg/L; indicated by red symbols in the upper panels on Maps 5.4-14a–b) were observed offshore of the Arkema area. 1,1,2-TCA was detected in three samples measured offshore of the Arkema site and four samples offshore of the Gunderson site. Detected 1,1,2-TCA concentrations ranged from 0.36 J µg/L to 400 µg/L. 27 The highest 1,1,2-TCA concentrations (>100 µg/L; indicated by red symbols in the lower panels on Maps 5.4-14a–b) were also observed offshore of the Arkema area. Detected concentrations in deeper Trident (90 to 150 cm bml) and Geoprobe (91 cm bml) were higher than concentrations observed in shallow samples (0 to 38 cm bml). The presence of both 1,1,2-TCA and 1,2-DCA offshore may be indicative of the chemicals' release and/or the degree of degradation that has occurred from 1,1,2-TCA to 1,2-DCA.

5.4.7.5.3 Chloroethane and Vinyl Chloride

Maps 5.4-15a–f present observed chloroethane and vinyl chloride concentrations in TZW offshore of the sites where these analytes were detected (Siltronic, Rhone Poulenc, Arkema, and Gunderson; RM 6–7, 7–8, and 8–9). The highest concentrations of chloroethane (>100 µg/L; indicated by red symbols in the upper panels on Maps 5.4-15a–f) were observed offshore of the Gunderson site, and chloroethane was only

[25] The maximum value of 30,000 µg/L was measured in a deep (90 to 150 cm bml) sample. The maximum shallow (0 to 38 cm bml) concentration was 12,000 µg/L. Only shallow samples were used in the loading analysis presented in Section 6.

[26] There were two or fewer detections for 1,2-DCA at RM 4–5 and at RM 6–7 for 1,1,2-TCA; therefore, these areas are not presented on maps.

[27] The maximum value of 400 µg/L was measured in a deep (90 to 150 cm bml) sample. The maximum shallow (0 to 38 cm bml) concentration was 360 µg/L. Only shallow samples were used in the loading analysis presented in Section 6.

sporadically detected offshore of the other sites. Detected chloroethane concentrations ranged from 0.23 J µg/L to 160 µg/L. In general, chloroethane detections were located in the same areas as TCA and particularly DCA detections, likely related to this degradation chain. Vinyl chloride was more widely detected, with the highest concentrations (>100 µg/L; indicated by red symbols in the lower panels on Maps 5.4-15a–e) observed offshore of the Siltronic and Arkema sites. Detected vinyl chloride concentrations ranged from 0.05 J µg/L to 1,640 µg/L (GP-65; Siltronic), with the exception of higher concentrations measured in shallow (4,300 µg/L) and deep (28,900 µg/L) samples collected offshore of Siltronic at station GP-67.[28] In general, vinyl chloride detections were located in the same areas as TCE and particularly DCE detections, likely related to known degradation patterns for chlorinated solvents. There were no clear trends in sample concentration with depth for these chemicals offshore of these study sites.

5.4.7.5.4 Chloroform and Methylene Chloride

Maps 5.4-16a–d present chloroform and methylene chloride concentrations in TZW offshore of the sites where these analytes were detected (Gasco, Siltronic, Rhone Poulenc, Arkema, and Gunderson; RM 6–7, 7–8, and 8–9). The highest concentrations of chloroform (>1,000 µg/L; indicated by red symbols in the upper panels on Maps 5.4-16a–d) were observed offshore of the Arkema site. Chloroform concentrations ranged from undetected (detection limits ranged from 0.056 U µg/L to 130 U µg/L) to 820,000 µg/L[29] (AP-03-A). The highest concentrations of methylene chloride (>100,000 µg/L; indicated by red symbols in the lower panels on Maps 5.4-16a–d) were also observed offshore of the Arkema site, with only sporadic detections offshore of other sites. Detected methylene chloride concentrations ranged from 0.23 J µg/L to 520,000 µg/L (AP-03-B). For both of these chemicals, the deeper Trident (90 to 150 cm bml) and Geoprobe (91 cm) samples generally exhibited higher concentrations than the corresponding shallow TZW samples.

5.4.7.5.5 cis-1,2-DCE and TCE

Maps 5.4-17a–f present cis-1,2-DCE and TCE concentrations in TZW offshore of the sites where these analytes were detected[30] (Siltronic, Rhone Poulenc, Arkema, and Gunderson; RM 6–7, 7–8, and 8–9). The highest concentrations of cis-1,2-DCE (>1,000 µg/L; indicated by red symbols in the upper panels on Maps 5.4-17a–f) were observed offshore of the Siltronic site and the former Acid Plant area of the Arkema site. Detected cis-1,2-DCE concentrations ranged from 0.12 J µg/L to 19,200 µg/L[31]

[28] GP-67 is located in an area which is understood to be impacted by historical direct discharges of TCE from an outfall and may not be representative of upland groundwater, as discussed in Appendix C2 and Section 10.

[29] The maximum value of 820,000 µg/L was measured in a deep (90 to 150 cm bml) sample. The maximum shallow (0 to 38 cm bml) concentration was 770,000 µg/L, measured at AP-03-B. Only shallow samples were used in the loading analysis presented in Section 6.

[30] Note three low-level detections of TCE (0.15 J µg/L to 0.46 J µg/L) offshore of the ARCO site, and one low-level detection of cis1,2-DCE (0.59 µg/L) offshore of the ExxonMobil site are not shown on maps.

[31] The maximum value of 19,200 µg/L was measured in a deep (91 cm bml) sample. The maximum shallow (30 cm bml) concentration was 14,400 µg/L, also measured at GP-65. Only shallow samples were used in the loading analysis presented in Section 6.

(GP-65), with the exception of higher concentrations measured in shallow (67,000 µg/L) and deep (574,000 µg/L) samples collected offshore of the Siltronic site at GP-67.[32] The highest concentrations of TCE (>1,000 µg/L; indicated by red symbols in the lower panels on Maps 5.4-17a–f) were also observed offshore of the Siltronic site and the former Acid Plant area of the Arkema site. Detected TCE concentrations ranged from 0.14 J µg/L to 7,100 µg/L[33] (AP-03-A), with the exception of higher concentrations measured in shallow (88,500 µg/L) and deep (585,000 µg/L) samples collected at GP-67.[34] For the higher-concentration (>1,000 µg/L) cis-1,2-DCE samples, the deeper (91 cm bml) Geoprobe samples had higher concentrations than collocated shallow (31 cm bml) Geoprobe samples. For TCE, the deeper Trident (90 to 150 cm bml) and Geoprobe (91 cm bml) samples generally exhibited higher concentrations than the corresponding shallow (0 to 38 cm bml) TZW samples.

5.4.7.6 Non-chlorinated VOCs in TZW

The TZW IC list includes two non-chlorinated VOCs, 1,2,4-trimethylbenzene and carbon disulfide. Maps 5.4-18a–e present 1,2,4-trimethylbenzene and carbon disulfide concentrations in TZW offshore of sites where these VOCs were present at concentrations above detection limits[35] (Gasco and Siltronic from RM 6–7 for 1,2,4-trimethylbenzene; Gasco, Siltronic, and Arkema from RM 6–8 for carbon disulfide).

Carbon disulfide was sampled at 150 locations offshore of all nine study sites. Of these, paired deep (90 to 150 cm bml) Trident samples were collected at 18 locations throughout the Study Area, and 24 paired deep (91 cm bml) Geoprobe samples were collected offshore of Gasco and Siltronic. Detected concentrations of carbon disulfide in shallow peeper, Trident, and non-LWG push probe samples ranged from 0.15 J µg/L to 800 µg/L, with the two highest concentrations (>10 µg/L; indicated by red and orange symbols in the upper panels on Maps 5.4-18a–e) observed offshore of the northern end of the Gasco site at locations GS-01-B (800 µg/L) and GS-02-A (53 µg/L). Due to low detection frequencies (only 1 detection in collocated shallow pairs), trends between the shallow (0 to 38 cm bml) and deeper (90 to 150 cm bml) Trident samples were not evaluated for carbon disulfide.

1,2,4-Trimethylbenzene samples were collected only during the Siltronic-led sampling event (MFA 2005a; HAI 2005a). In all, 41 locations were sampled with Geoprobe for 1,2,4-trimethylbenzene offshore of the Siltronic and Gasco sites, and 24 collocated deep (91 cm bml) Geoprobe samples were also collected. No filtered samples were

[32] GP-67 is located in an area which is understood to be impacted by historical direct discharges of TCE from an outfall and may not be representative of upland groundwater, as discussed in Appendix C2 and Section 10.
[33] The maximum value of 7,100 µg/L was measured in a deep (90 to 150 cm bml) sample. The maximum shallow (0 to 38 cm bml) concentration was 48.7 µg/L, measured at GP-65. Only shallow samples were used in the loading analysis presented in Section 6. 48.7
[34] See note above.
[35] Note one low-level detection of carbon disulfide (0.23 J µg/L) offshore of the Kinder Morgan site is not shown on maps.

collected, per sampling protocols for VOCs. 1,2,4-Trimethylbenzene detected concentrations ranged from 1.04 µg/L to 69.9 µg/L, with the maximum concentrations (>40 µg/L; indicated by red symbols in the lower panels on Maps 5.4-18a–e) measured offshore of the property line between the Gasco and Siltronic sites. There were no clear trends in sample concentration with depth for 1,2,4-trimethylbenzene.

5.4.7.7 BTEX in TZW

Maps 5.4-19a–h present BTEX concentrations measured in TZW offshore of all nine study sites. The upper panel of each map presents the total BTEX concentration, and the lower half presents the relative contribution of each BTEX constituent (benzene, toluene, ethylbenzene, and xylenes) in the form of a pie chart.[36] To accommodate the large numbers of measurements taken between RM 6 and 7, this set of results is presented on four maps (Maps 5.4-19c–f). In all, BTEX was sampled in TZW at 150 locations, with 18 deeper (90 to 150 cm bml) paired Trident samples collected across the Study Area and 24 deep (91 cm bml) Geoprobe samples collected offshore of the Siltronic and Gasco sites. No filtered samples for BTEX constituent chemicals were collected, per sampling protocols for VOCs.

Across the study sites, total BTEX was detected in 75 percent of samples, with detected sample concentrations covering a large range, from 0.11 J µg/L to 4,700 µg/L (GP-68); detection limits for undetected samples ranged from 0.22 U µg/L to 11 U µg/L. The highest BTEX concentrations (>1,000 µg/L; indicated by red symbols in the upper panels on Maps 5.4–19a-h) were observed offshore of the Gasco and Siltronic sites, with one high value also observed offshore of Arkema (AP-04-D; 1,600 µg/L). In general, there were no consistent patterns with depth in the paired shallow and deep samples.

The pie charts presented in the lower panels on Maps 5.4-19a–h present the relative concentration of BTEX constituent chemicals for each detected sample. No consistent patterns of BTEX composition related to spatial distribution or magnitude of total BTEX concentration were apparent across all nine TZW study sites. More localized patterns may be evident in the data, such as a general tendency for toluene and xylenes to dominate BTEX composition offshore of sites where generally lower concentrations (detection limits to 1.79 J µg/L) of BTEX were observed in TZW (Kinder Morgan Linnton, ARCO, ExxonMobil, and Willbridge Terminal). These results are considered and discussed in detail on a site-by-site and sample-by-sample basis in Appendix C2 to support the assessment of groundwater pathway at each study site.

5.4.8 Groundwater Seeps

This section summarizes the location, available chemical data, and data quality assessment for upland groundwater seeps. Groundwater seeps were assessed for the BHHRA because of the potential for humans to come into contact with seep water. The

[36] The pie charts show only those constituent chemical concentrations that are above detection limits. If all components of the total BTEX sum are non-detect for a given sample, no pie chart is shown for that sample.

potential effects of human exposure to groundwater discharge in surface seeps are presented in the initial BHHRA summary in Section 8 of this report. Seep data are not appropriate for assessing ecological risks and therefore were excluded from the TZW BERA.

The groundwater seep data set is limited and does not lend itself to the general organization of the nature and extent discussion applied to TZW and other media in this report (specifically, discussion of select analytes followed by remaining ICs and a summary). Instead, this section will present all available seep locations and data of adequate quality for use in BHHRA, presenting simple comparisons of concentration to upland groundwater and TZW, to provide context and support understanding of the groundwater seep data.

5.4.8.1 Groundwater Seep Locations

A seep reconnaissance survey was conducted during Round 1 of the Portland Harbor RI/FS (GSI 2003b) to support the BHHRA and CSM. This survey documented readily identifiable groundwater seeps based on visual observations along approximately 17 miles of riverbank from RM 2 to 10.5. For the purposes of this survey, a seep was defined as groundwater discharge above the Willamette River water line as observed during the seep reconnaissance survey. This groundwater may be discharged from local shallow groundwater systems, perched groundwater, water seeping through utility backfill, or return flow from tidally influenced bank storage. Observed seeps were classified into one or more of five types:

- Seepage line at the base of embankments (nine seeps)
- Linear and point seeps at the foot of beaches (six seeps)
- Seepage through backfill surrounding outfalls (four seeps)
- Seepage of NAPL (two seeps)
- Potential seep locations identified by observation of extensive ferric hydroxide staining of bank materials (eight potential seeps).

Additionally, eight seeps were categorized as combinations of the above seep types.

Twelve seeps were observed at or near potential human-use areas (GSI 2003b). No additional seeps or other surface expressions of groundwater have been observed on or near a human-use beach since the seep reconnaissance survey.

5.4.8.2 Groundwater Seep Water Quality Data

Water quality data have been collected at six seeps in four general areas (Figure 5.4-5). Groundwater seep discharge rates have not been empirically quantified. The water quality sampling efforts to date for upland groundwater seeps include:

- City of Portland stormwater Outfalls 22B and 22C, located directly north and south of the Railroad Bridge at RM 6.89 and 6.82, respectively, are type 3 (backfill surrounding outfalls) seeps. Both Rhone Poulenc and NW Natural have collected water quality samples in Outfalls 22B and 22C to evaluate potential groundwater infiltration to the conveyance systems.
- Rhone Poulenc sampled Outfall 22B on five occasions between October 1, 1993 and September 23, 2004 and Outfall 22C four times between August 13, 2002 and September 23, 2004. Samples were collected at the end of the pipe and were analyzed for 231 individual parameters, including conventionals, PCDD/Fs, herbicides, metals, PAHs, PCB Aroclors, pesticides, petroleum hydrocarbons, phenols, phthalates, SVOCs, and VOCs. The results are Category 1 data validated to the QA2 level.
- NW Natural sampled Outfall 22C on February 24, 2005 for 89 individual parameters, including conventionals, metals, PAHs, phenols, phthalates, SVOCs, and VOCs. Data were validated to Category 2, QA1 level.
- Seeps-01, -02, and -03 are located at the Gunderson site near RM 8.5. These type 3 seeps are associated with cracked stormwater drain pipes. Each seep was sampled once in November 2004 and again in April 2005, with samples analyzed for 31 individual parameters, including conventionals, metals, PCB Aroclors, petroleum hydrocarbons, and phthalates. Data were validated to Category 1, QA1.
- ExxonMobil sampled areas with visible sheen on sand and in pooled water along the riverbank at the ExxonMobil site under the direction of DEQ on August 13, 2004 (Kleinfelder 2004). Two composite samples were analyzed as soils for DRH, GRH, and RRH. Data were validated to the QA1 level.

The sampling events described above summarize all groundwater seep data collected to date; the associated data table presented in Appendix D4 (Table D4-2) focuses only on the City of Portland Outfall 22B seep location, which is the only seep relevant for use in the BHHRA based on data quality and location. Outfall 22B is located on a human use beach (Figure 5.4-5) and has Category 1 data validated to the QA2 level.

For this nature and extent discussion, a subset of the detected Category 1, QA2 Outfall 22B seep data was compared to nearshore upland groundwater data and TZW data from the Rhone Poulenc site (Figure 5.4-6). Because of the small number of seep, upland groundwater, and TZW samples, these data are presented as a simple comparison and should not be considered statistically significant. At Outfall 22B, detected seep concentrations of Silvex (0.14 µg/L), benzene (0.19 µg/L), and TCE (0.34 µg/L) were below the minimum concentrations found in upland groundwater and in TZW. The measured concentration of 1,2-DCB at Outfall 22B (0.864 µg/L) falls within the lower end of the range for upland groundwater and TZW.

5.4.9 Summary of Indicator Chemicals in TZW and Seeps

The following subsections present a brief summary of the findings for TZW and groundwater seeps.

5.4.9.1 Transition Zone Water Summary

The preceding discussions present the observed distribution and patterns of ICs in TZW. The TZW data set was generated for the purposes of assessing TZW offshore of upland groundwater plumes with likely or known complete pathways to the river; therefore, the data set does not cover all areas where TZW may be affected by unknown plumes or TZW quality impacted by contaminated sediments. Specifically, the data set is focused on the offshore area of nine sites, identified as high-priority, Category A upland groundwater sites due to a confirmed or reasonable likelihood for discharge of upland groundwater COIs to Portland Harbor. Consideration of pore water chemistry affected by in-river sediment contamination is evaluated in Section 6 through equilibrium partitioning calculations based on the large data set of sediment concentrations.

The observed distribution of TZW chemicals could not be adequately described focusing exclusively on the bounding ICs (total PCBs, total PCDD/Fs, total DDx, and total PAHs) because these chemicals were often not part of the focus of the upland site plume-defined analyte lists. Therefore, the TZW nature and extent presentation briefly addresses all TZW ICs, as opposed to focusing on chemicals also included as CSM ICs. Note that PCBs were not sampled in TZW, and only two TZW sampling locations were analyzed for PCDD/Fs; these chemicals are not included in the TZW nature and extent discussion.

DDx was measured in TZW at ten locations offshore of the former Acid Plant area of the Arkema site, as well as at one location offshore of the adjacent Rhone Poulenc site. The highest concentrations were observed in unfiltered samples collected in both shallow (0 to 38 cm bml) and deep (90 to 150 cm bml) TZW samples offshore of the former Acid Plant area. Review of the filtered and unfiltered results indicates that DDx is present on solids. Further, concentrations in unfiltered samples are generally greater in deep than shallow TZW samples. These results with filtration and depth were not apparent in the limited data collected at the single sample location offshore of the Rhone Poulenc site.

PAHs were sampled at six of the TZW sites (Kinder Morgan, ARCO, ExxonMobil, Gasco, Siltronic, and Willbridge Terminal). Total PAH concentrations were found to be highest offshore of the Gasco and Siltronic sites. Total HPAH, total cPAH, and BAP results showed similar distribution and filtration patterns. Because LPAHs tend to comprise the majority of the total PAH concentrations, LPAH and naphthalene results generally followed the distribution patterns apparent for total PAHs. Filtration was observed to decrease the total PAH concentration slightly, with greater effects on the more hydrophobic PAHs, as expected. Review of the fractional composition of the 17 individual chemicals that compose total PAHs shows a clear pattern of high

naphthalene concentrations associated with high total PAH concentrations offshore of the Gasco and Siltronic sites.

TPH was sampled at six of the TZW sites (Kinder Morgan, ARCO, ExxonMobil, Gasco, Siltronic, and Willbridge Terminal). The highest concentrations of TPH in TZW were observed offshore of the Gasco and Siltronic sites, with the highest individual result measured offshore of the Gasco site. The complex-mixture nature of the analyte TPH, however, confounded detailed interpretation of results. Overall, filtration was observed to decrease TPH concentration; however, it is unclear whether this effect is related to loss of volatiles during filtration or loss of more hydrophobic portions of the TPH mixture, with composition varying in inconsistent ways across filtration pairs. Review of the fractional components (DRH, RRH, and GRH) showed a pattern of TPH composition shift toward GRHs offshore of a portion of the Siltronic site, but did not provide useful insights elsewhere.

Cyanide, Silvex, and perchlorate were detected offshore of the sites where they were sampled in areas that also showed detections of other upland COIs on the TZW analyte list for the sites. The highest detections of Silvex were observed offshore of the Rhone Poulenc site in an area thought to be an offshore discharge area for upland groundwater. Total cyanide was detected offshore of the Gasco and Siltronic sites. Perchlorate was observed offshore of the Arkema site former Chlorate Plant area

The TZW IC list includes eight metals: arsenic, barium, cadmium, copper, lead, manganese, nickel, and zinc. These were sampled offshore of all nine TZW study sites. Filtration significantly and consistently reduced TZW sample concentrations for cadmium, copper, lead, nickel, and zinc, suggesting that much of the observed total concentration for these metals was associated with >0.45 µm particles. Differences with filtration were not as pronounced for arsenic, barium, and manganese. Shallow versus deep TZW results did not show consistent trends for any of the metals. The spatial distribution of metals likely reflects some combination of upland groundwater transport, local sediment redox conditions, and sediment contamination, varying by site and metal. Detailed geochemical analysis of select metals concentrations in TZW is provided in Appendix C2.

The TZW IC list includes nine chlorinated VOCs (MCB; chloroethane; 1,2-DCA; 1,1,2-TCA; vinyl chloride; cis-1,2-DCE; TCE; chloroform; and methylene chloride) and one chlorinated SVOC (1,2-DCB). These chemicals were sampled at all nine TZW study sites. In summary, the chlorinated VOCs and 1,2-DCB show spatial distribution patterns that appear to reflect common groundwater pathways and degradation chains. Specifically, the highest concentrations of TCE, cis-1,2-DCE, and vinyl chloride are located offshore of the Siltronic site and the former Acid Plant area of the Arkema site. Likewise, the highest concentrations of 1,1,2-TCA, 1,2-DCA, and chloroethane are generally located offshore of the former Acid Plant area of the Arkema site and the Gunderson site. Further, the majority of detected results for chloroform and methylene chloride are also generally located offshore of the Arkema site. Finally, the highest

concentrations of 1,2-DCB and MCB are generally located offshore of the former Acid Plant area of the Arkema site and the Rhone Poulenc site.

The TZW IC list includes two non-chlorinated VOCs, 1,2,4-trimethylbenzene and carbon disulfide. Carbon disulfide was sampled at all nine TZW study sites, but 1,2,4-trimethylbenzene was sampled offshore of the Siltronic site only. The highest concentrations of carbon disulfide were observed offshore of the Gasco site. These high concentrations were generally observed in areas of other elevated VOCs offshore of this site.

BTEX was analyzed in TZW samples from all nine study sites. The highest values were consistently observed offshore of the Gasco and Siltronic sites, with one high value also observed offshore of Arkema.

5.4.9.2 Groundwater Seeps Summary

The groundwater seep data are limited and do not allow for definitive conclusions. Only one seep, Outfall 22B (at the Rhone Poulenc site), is relevant for use in the BHHRA. At this location, groundwater infiltrates into the outfall pipe, which subsequently discharges to a beach. The beach where Outfall 22B discharges was identified as a potential transient use area, so exposure to the groundwater seep in that beach by transients is considered a potentially complete pathway. For most analytes evaluated, Outfall 22B concentrations were near or below detection limits and were well below nearby upland groundwater and TZW concentrations.

5.5 INDICATOR CHEMICALS IN BIOTA

This section summarizes the analytical tissue data collected from the LWR. Fish and invertebrates used to evaluate the nature and extent of chemical concentrations in biota were collected from the Study Area and from adjacent areas, including those immediately upstream and downstream of the Study Area and Multnomah Channel near its divergence from the LWR. Data are also available for biota from the upriver reach (RM 15.3–28.4) and above Willamette Falls. The sample set includes fish and invertebrates collected by the LWG as part of Rounds 1, 2, and 3 of the Portland Harbor RI/FS, as well as recent data collected by other parties as described in Section 2. Eleven fish species, four benthic invertebrate species, epibenthic communities, and fish stomach contents are represented.

The biota sample set and sampling locations are summarized below in Section 5.5.1. The ICs and the rationale for their selection are provided in Section 5.5.2. Data presentation tools are described in Section 5.5.3. The distribution of ICs in biota samples in the Study Area and immediately adjacent areas is provided in Sections 5.5.4 through 5.5.8. Chemical distribution in samples collected in the upriver reach and above Willamette Falls is described in Section 5.5.9. A summary is provided in Section 5.5.10. Tables, sample location maps, and a more detailed description of the samples and data set are provided in Section 2.2.8.

5.5.1 Biota Sample Set

Fish species collected in the Study Area and adjacent areas included brown bullhead (*Ameiurus nebulosus*), black crappie (*Pomoxis nigromaculatus)*, common carp (*Cyprinus carpio carpio*), juvenile Chinook (*Oncorhynchus tshawytscha*), largescale sucker (*Catostomus macrocheilus*), northern pikeminnow (*Ptychocheilus oregonensis)*, juvenile lamprey (i.e., ammocoetes and macropthalmia; *Lampetra* sp.), peamouth (*Mylocheilus caurinus*), sculpin (*Cottus* spp.), smallmouth bass (*Micropterus dolomieui*), and prebreeding white sturgeon (*Acipenser transmontanus*). With the exception of sturgeon, fish tissue samples were composites of more than one individual fish. The types of tissue examined were whole-body fish (all species) and fillets (limited to black crappie, brown bullhead, carp, smallmouth bass, and sturgeon). For Round 1 and non-LWG data, separate fish were caught for fillet and whole-body composite samples, but for Round 3, fillet and whole-body data were obtained from the same composite group of fish. Fillets were removed from Round 3 carp and smallmouth bass, and fillets and remaining bodies without fillets (i.e., carcass) were composited and analyzed separately. In the maps that accompany Section 5.5, those samples described as “fillet and body without fillet fractions” represent calculated whole-body concentrations. Calculated whole-body concentrations are provided in the SCRA database (Appendix A3). The calculations used to determine whole-body concentrations are described in Appendix A4.

Tissues from four invertebrate groups were analyzed: the Asiatic clam (*Corbicula fluminea*), an oligochaete worm (*Lumbriculus variegatus*), two species of mussels (tentatively identified as *Margaritifera falcata* and *Anodonta nuttalliana*) were composited together and analyzed as one group of mussels, and crayfish (*Pacifasticus spp.*). In addition to field-collected clams from the Study Area, other clams of the same species were collected from the Clinch River in Virginia by the Virginia Marine Institute for bioaccumulation tests. Clams from the Clinch River were exposed to Site sediment samples under controlled laboratory conditions. The oligochaete worms included only culture-grown *Lumbriculus variegatus* exposed to Site sediment samples for bioaccumulation tests under controlled laboratory conditions. During Rounds 1 and 2, field-collected clams were not depurated. During Round 3, field-collected clam composite samples were generated with and without depuration at sampling locations where sufficient sample mass was collected. Clams were also depurated as part of other non-LWG laboratory bioaccumulation studies. Epibenthic communities from multiplate samplers were composited for analysis. Epibenthic communities composed mostly of invertebrates belonged to four major taxonomic groups: Annelida (worms), Chironomidae (midges/blood worms), Crustacea (amphipods), and Anthozoa (freshwater *Hydra* sp.). Two zooplankton daphnid species were also identified on multiplates, with *Sida crystallina* being the most abundant. Stomach contents (primarily aquatic organisms and terrestrial insects) of sturgeon and juvenile Chinook salmon were also analyzed and are discussed as part of the invertebrate data.

Additional details regarding the number and type of tissues collected from the Study Area are provided in Section 2.2.8 of this report and in the BHHRA (Appendix F) and the BERA (Appendix G). Table 2.2-9 provides the study name, sample count, and a summary of analyses for each species and tissue type, and Table 2.2-10 details the sample count for each individual chemical analyzed for each species and tissue type for LWG and non-LWG samples. Table 2.2-11 provides the number of fish and invertebrates in each sample composite.

Fish and invertebrate sampling locations (Maps 2.2-7 through 2.2-14) represent either discrete locations or larger fishing areas as described below:

- For LWG Rounds 1, 2, and 3, the sampling design included collection of species over areas similar in size to likely home ranges, except for species that are assumed to have home ranges larger than the Study Area itself.
- Epibenthic community collection methods (artificial substrates) reflect essentially single-point samples.
- Clams, sculpin, crayfish, and collocated sediment (used in clam and worm bioaccumulation tests) were composite samples collected at multiple points, transects, or tows within discrete nearshore areas.
- Smallmouth bass, largescale sucker, northern pikeminnow, and peamouth may utilize fairly localized areas and were collected and composited over 1-mile

river segments. In Round 3 smallmouth bass were composited from each bank of the river at 1-mile river segments.[1]

- Black crappie, brown bullhead, and carp may utilize home ranges on the scale of the entire Study Area (or greater) and were collected and composited over 3-mile river segments (4-mile segments for Round 3 carp).
- Sturgeon and juvenile Chinook also have large home ranges; however, the samples were collected within discrete set line areas (for sturgeon) or beach seine areas (for juvenile Chinook).

Map symbols of the same color represent individual fish in each composite collected within each of the sampling areas or fishing zones.

Sample locations are labeled on the maps according to the following scheme:

- Round 1 field-collected clams, crayfish, sculpin, and juvenile Chinook are shown as points that represent composites of several individuals collected within a narrow shoreline or nearshore transects and are referenced as single centroid points within a river mile reach (03Rnnn through 09Rnnn).
- Round 1 smallmouth bass, largescale sucker, northern pikeminnow, and peamouth were sampled over 1-mile increments of the Study Area, referenced as river mile areas (03Rnnn through 09Rnnn).
- The remaining Round 1 fish species (brown bullhead, black crappie, carp) are represented by fishing zones (FZ0306 and FZ0609 for RM 3–6 and 6–9).
- Round 2 invertebrate tissue sampling locations refer to FCnnn for field-collected clam tissue samples and BTnnn for collocated sediments collected for laboratory-exposed clams and worms.
- Epibenthic community tissue composites were collected from multiplate samplers deployed at locations referenced as MITnn.
- Juvenile Chinook whole-body and stomach content samples collected at beach transects during Round 2 are referenced as T01 through T04; stomach content samples end with the letters SC.
- Round 3 lamprey ammocoete and macropthalmia sampling locations are referenced as LTAnnn and LTMnnn.
- Round 3 sturgeon were collected at five reaches (e.g. ST001 refers to RM 2–3) and were split into three types of samples: sturgeon whole body minus liver (e.g., ST001-01), sturgeon liver (e.g. ST001-01 [liver]), and sturgeon gut content (SG001-01).

[1] Due to a Round 3 labeling error in the field, one smallmouth bass (LW3-SB11W-11) collected from the west side of the river at RM 11 was inadvertently included in composite samples from the east side of the river at RM 11. The affected composites are LW3-SB11E-C00B and LW3-SB11E-C00F (see Integral 2008c).

- Round 3 fish and invertebrates (except carp) use a sample nomenclature that includes species name, river mile, and river side designation (SSRRB or SSRRRB; SB011W refers to smallmouth bass from RM 10.5 to 11.5, western nearshore zone).
- Round 3 carp sampling locations use a 6-digit nomenclature to depict the species and river segment fished (e.g., CP0408 refers to RM 4–8).
- Samples collected by parties other than the LWG retain the label assigned by the original authors and represent single-point data.

Throughout this section, the river mile and river zone (east, west, navigation channel, Multnomah Channel) are provided with the sample location code when the location code does not include this information. River miles are not provided for composited biota samples that include individuals from different locations.

Summary statistics (sample size, frequency of detection, minimum, maximum, mean, median, and 95th percentile) for ICs in tissue samples are provided in Tables 5.5-1 through 5.5-5. Tables 5.5-1 (Study Area and adjacent locations), 5.5-3 (upriver reach, below falls), and 5.5-5 (above falls) contain summary statistics for fish, and Tables 5.5-2 (Study Area and adjacent locations) and 5.5-4 (upriver reach, below falls) for invertebrates. Two sets of summary information are provided in the summary statistics tables: one set for detected values only and a second set for detected and non-detected values combined. The discussion of the nature and extent of ICs in tissue provided in the remainder of this section is based on statistics calculated for detected concentrations only. Summary statistics for all analytes measured in tissue samples are provided in Tables D5.1-1 through D5.1-5 in Appendix D5.

5.5.2 Indicator Chemicals for Biota

ICs (Table 5.0-2) were selected in a multistage screening process as described in the introduction to Section 5. A total of 26 individual analytes and calculated contaminant sums were identified as ICs for biota; however, as noted previously, though these contaminants have been determined to potentially pose the most widespread risks for biota, it should be kept in mind that other contaminants that may pose risk to human health and the environment are also present and comingled throughout the Portland Harbor Study Area. Low frequencies of detection in tissue samples resulted in fewer ICs than for sediment. ICs for biota are organized as follows:

- PCBs
  - Total PCBs*
  - PCB TEQ (ND=0)
- PCDD/Fs
  - Total PCDD/Fs*

  - TCDD TEQ (ND=0)*
- DDx
  - Total DDx (sum of 2,4'- and 4,4'-DDD, DDE, DDT)*
  - Total of 2,4'- and 4,4'-DDT
  - Total of 2,4'- and 4,4'-DDE
  - Total of 2,4'- and 4,4'-DDD
- PAHs
  - Total PAHs*
  - Total cPAH BaPEq values
  - Total LPAHs
  - Total HPAHs
- SVOCs
  - BEHP*
  - Hexachlorobenzene
- Pesticides
  - Total chlordanes*
  - gamma-HCH
  - Aldrin*
  - Dieldrin*
- Metals
  - Arsenic*
  - Cadmium
  - Chromium*
  - Copper*
  - Lead
  - Mercury
  - Zinc*
- Organometallic Compounds
  - TBT*.

A subset of indicator chemicals, which are indicated with an asterisk (*) in the above list, are described in this section. The 13 ICs (total PCDD/Fs and TCDD TEQ are grouped as one chemical in this count) for biota were selected to match the chemicals for presentation and discussion in the site-wide CSM (Section 10).

Data for ICs are presented in the following sections for the Study Area and immediately adjacent areas, followed by discussion of the upriver reach and above falls areas. In addition, lipid content of samples is discussed in Sections 5.5.8.10 (Study Area) and 5.5.9.14 (upriver reach and above falls) to provide a context for understanding bioaccumulative substances. Data are not available for every IC in every tissue because study designs varied and because insufficient material was available in some cases to complete all planned analyses. Tables 2.2-9 and 2.2-10 provide summaries of samples and analyses available for the Study Area as provided in the SCRA database.

Four bounding ICs—total PCBs, total PCDD/Fs (as both total dioxin/furans and TCDD TEQ), total DDx, and total PAHs—are described first and in greater detail. The nature and composition of these four chemicals are discussed to provide background on their environmental chemistry and to provide a site-wide perspective on their distribution. The discussion of the remaining ICs is abbreviated and includes a description of the data set, concentration ranges, and frequency of detection, with references to figures and tables to provide a complete picture of the distribution of those chemicals. Graphic representations of distributions of the remaining ICs are provided in Appendix D5.

5.5.3 Description of Biota Presentation Tools

Three types of figures are provided to summarize the biota data: box-whisker plots, scatter plots, and bar graphs. Because the number of tissue samples collected from areas immediately adjacent to the Study Area (i.e., RM 0–1.9, RM 11.8–12.2, and Multnomah Channel) was small, these biota data are combined with the Study Area data set in the graphical displays and tissue summary statistics presented in this section. However, it is important to note that these data are combined for presentation purposes only and that these samples (tabulated below) are not part of the Study Area data set and so were not included in the Study Area BHHRA (Appendix F) and BERA (Appendix G) data sets.

River Mile	Species	Tissue	Sample ID
RM 1–2	clam	body without shell	LW3-CA01E-C01
RM 1–2	clam	depurated w/o shell	LW3-CA01E-C00D
RM 1–2	crayfish	whole body	LW3-CR01E-ALT-C00
RM 1–2	crayfish	whole body	LW3-CR01W-C00
RM 1–2	sculpin	whole body	LW3-SP01E-C00
RM 1–2	sculpin	whole body	LW3-SP01W-C00
RM 11.8–12.2	clam	body without shell	LW3-CA12W-C00
RM 11.8–12.2	clam	body without shell	LW3-CA12E-C00
RM 11.8–12.2	clam	depurated w/o shell	LW3-CA12E-C00D
RM 11.8–12.2	crayfish	whole body	LW3-CR12W-C00

River Mile	Species	Tissue	Sample ID
RM 11.8–12.2	crayfish	whole body	LW3-CR12E-C00
RM 11.8–12.2	sculpin	whole body	LW3-SP12W-ALT-C01
RM 11.8–12.2	sculpin	whole body	LW3-SP12E-C00
Multnomah Channel	clam	body without shell	LW2-BTFC003
Multnomah Channel	lab clam	body without shell	LW2-BTLC003
Multnomah Channel	*Lumbriculus variegatus*	whole body	LW2-BTLW003

Box-Whisker Plots: Figures 5.5-1 through 5.5-8 are a series of box-whisker plots that present the concentrations of ICs for the various fish and invertebrate species and tissue types. These plots were developed using R for Windows v. 2.7.0. (R Development Core Team 2008). The horizontal center line in each box represents the median concentration, and the top and bottom of the box represent the upper and lower quartiles, respectively. The upper whisker represents the highest concentration that is less than the upper quartile plus 1.5 times the interquartile range, and the lower whisker represents the lowest concentration that is greater than the lower quartile minus 1.5 times the interquartile range. Outliers are represented individually by small circles above and below the boxes.

Scatter Plots: A series of scatter plots (Figures 5.5-9 through 5.5-23) for each IC provides chemical concentrations by river mile for each species and tissue type. A number of species were caught within target fishing zones, including smallmouth bass, largescale sucker, northern pikeminnow, and peamouth (1-mile fishing zones); brown bullhead and black crappie (3-mile fishing zones); and carp (3-mile fishing zones for Round 1 and 4-mile fishing zones for Round 3). Individual fish caught in different areas of each fishing zone were composited to create the sample for analysis. The centroid of the fishing zone is used to represent these samples on the scatter plots.

Bar Graphs: Bar graphs (Figures 5.5-24 through 5.5-30) show the distribution of PCB homologs, PCDD/F homologs, and individual components of total DDx in each sample, and the distribution of PAHs in clam samples (see Section 5.5.7). Patterns displayed in the bar charts must be interpreted with caution. The changes in contaminant composition and apparent trends shown by the bar charts may be indicative of significant patterns, or they may be within the normal range of data variability. Further detailed forensic analyses outside the scope of this RI/FS would be required to determine statistically significant variations among the data and contributions from various potential sources.

Concentration Maps: Concentration maps (Maps 5.5-1 through 5.5-16) plot IC concentrations (and qualifiers) for each tissue sample used in the BHHRA or BERA. All forms of tissue (i.e., whole body, fillet, fillet without skin, stomach contents, etc.) are shown. Individual fish collection locations from each sample composite are color coded to match the appropriate sample concentration table. Units are indicated for each contaminant in the individual sample concentration tables.

Box-whisker plots and scatter plots for additional ICs identified in Table 5.0-2 are provided as part of Appendix D5.

The range of average concentrations for the various species is provided in the following sections for each IC for a general indication of the distribution of ICs across species. The maximum concentration is also provided for context and comparison to the average concentrations. For fish tissue samples, statistics for whole-body and fillet samples are summarized separately. All tissue concentrations are reported on a wet-weight basis in the following discussion.

5.5.4 Total PCBs in Biota

PCBs in tissue samples were analyzed as Aroclors or congeners. In most Round 1 samples, both analyses were completed; however, Round 1 whole-body largescale sucker, northern pikeminnow, peamouth, all fillets, and crayfish samples were only analyzed for Aroclors. In Rounds 2 and 3, only PCB congener analyses were completed for LWG biota samples and all 209 PCB congeners were analyzed. Biota samples collected by other parties were sometimes analyzed for a limited number of congeners. In accordance with the RI data summation rules, samples with fewer than 100 PCB congeners were not summed. This section presents a summary of the distribution of total PCBs using congeners, when available, because congener sums are considered a more reliable estimate of total PCBs than Aroclor sums, based on the greater specificity and accuracy of the PCB congener analysis. Data for total Aroclors are included when PCB congener data are not available. PCB homolog patterns in the biota samples are also summarized in this section.

Fish assimilate and metabolize different PCB congeners at different rates, and therefore the PCB pattern in fish tissue may not closely resemble the pattern of the Aroclor that was released into the river or of the standard used by the laboratory to identify and quantify the PCBs in the sample. The limitations of the Aroclor analysis are described in Appendix D1.4. Because of the uncertainties related to Aroclor identification and because PCB congeners were analyzed in most of the tissue samples used for the Portland Harbor RI, PCB patterns in tissue are described only in terms of homologs (i.e., congeners grouped according to chlorination level).

A box-whisker plot showing the distribution of PCBs for each species and tissue type is provided in Figure 5.5-1. PCB concentrations by river mile for each species and tissue type are shown in Figures 5.5-9a–j.

5.5.4.1 PCBs in Fish

PCBs were detected in all fish samples from the Study Area. Average total PCBs in whole-body fish samples ranged from 111 µg/kg (juvenile Chinook) to 2,760 µg/kg (carp; Table 5.5-1). The maximum total PCB concentration in whole-body fish occurred in a carp composite sample from the RM 4 to 8 fishing zone (CP0408; 25,100 J µg/kg; Map 5.5-3b). In fillets, average total PCB concentrations ranged from

24.1 µg/kg (black crappie) to 1,840 µg/kg (carp), with the maximum concentration also occurring in a carp fillet composite sample from CP0408 (19,700 J µg/kg; Map 5.5-3b). PCB concentrations are shown on Figures 5.5-9a–j.

Although PCB homolog composition varied throughout the river, tetraCBs, pentaCBs, hexaCBs, and heptaCBs were typically present in proportions greater than 10 percent (Figure 5.5-24). HexaCBs and heptaCBs were generally dominant in fish tissue samples from throughout the Study Area. MonoCBs, diCBs, and nonaCBs were very rare, while triCBs and octaCBs were rare in comparison to the other homologs.

For many fish species, more highly chlorinated PCB homologs appear to be seen more frequently in the upper reaches of the Study Area than in the lower third.

5.5.4.2 PCBs in Invertebrates

PCBs were detected in all invertebrate samples from the Study Area that were analyzed for PCB congeners. PCBs were not detected in 10 crayfish tissue composites that were only analyzed for Aroclors, which have a higher detection limit than PCB congeners. Average total PCB concentrations ranged from 10.6 µg/kg (sturgeon stomach contents) to 612 µg/kg (laboratory-exposed worms) (Table 5.5-2). The maximum total PCB value (4,310 J µg/kg) was measured in worms exposed to sediment from location BT028 (RM 8.8W; Map 5.5-14e). Aroclors were only measured in Round 1 and non-LWG samples.

Among PCB homologs in invertebrate samples, tetraCBs, pentaCBs, hexaCBs, and heptaCBs were most abundant overall, appearing similar to the fish tissue samples (Figure 5.5-25). Laboratory-exposed clams were distinctive in appearing to contain a relatively large fraction of diCBs, which were present in relatively lower percentages in field-collected clams from the same locations, with the exception of location FC027 Rep 1 (RM 8.7E). PCB patterns in crayfish varied widely. In particular, the PCB pattern in crayfish sample 06R031 (RM 6.8E) is notable for its apparently high percentage of monoCBs and diCBs and relatively low content of triCBs, tetraCBs, and pentaCBs. The PCB pattern in clams and worms exposed at the laboratory to sediments from three locations (BT002 at RM 2.3E, BT017 at RM 6.9W, and BT028 at RM 8.8W) appeared to contain distinctly higher levels of tetraCBs and lower levels of hexaCBs and heptaCBs than the remaining samples. This pattern was reflected only slightly in field-collected clam samples from these locations.

5.5.5 Total PCDD/F Homologs and TCDD TEQ in Biota

PCDD/Fs were analyzed for in all invertebrate tissue types (although not all clam or crayfish samples were analyzed) and in all fish species except largescale sucker, northern pikeminnow, and peamouth. Total PCDD/Fs are the sum of the various homologs (i.e., TCDD, PeCDD, HxCDD, HpCDD, and OCDD; and TCDF, PeCDF, HxCDF, HpCDF, and OCDF). PCDD/F results for individual congeners are expressed in terms of 2,3,7,8-TCDD toxicity by applying mammalian TEFs (Van den Berg et al.

2006) to the individual isomers and calculating a final TCDD TEQ concentration (see Section 2.0 for group summing rules).

Box-whisker plots showing the distribution of total PCDD/Fs and TCDD TEQ for each species and tissue type are provided in Figure 5.5-2. Total PCDD/F and TCDD TEQ concentrations by river mile for each species and tissue type are shown in Figures 5.5-10a–j and Figures 5.5-11a–j, respectively.

5.5.5.1 Total PCDD/F Homologs and TCDD TEQ in Fish

PCDD/Fs were detected in all fish tissue samples from the Study Area. The average total PCDD/F homolog concentrations in whole-body fish tissue samples ranged from 8.62 pg/g (sturgeon) to 78.9 pg/g (juvenile lamprey) (Table 5.5-1). The maximum whole-body concentration (388 pg/g) was measured in a sculpin composite sample from 07R006 (RM 7.3W; Map 5.5-11d). Average fillet concentrations ranged from 4.76 pg/g (smallmouth bass) to 29.6 pg/g (carp) (Table 5.5-1). The maximum total PCDD/F concentration in a fillet composite sample was measured in smallmouth bass from SB07W (56.9 J pg/g; Map 5.5-12c2).

The homolog distribution in fish tissues was quite variable with differences evident between species as well as between locations and tissue types within species (Figure 5.5-26). In carp, juvenile lamprey, and sculpin tissues with total PCDD/F concentrations above 50 pg/g, OCDD tended to dominate many of the samples, with secondary dominance shown variously by HxCDDs, HpCDDs, TCDFs, PeCDFs, and HxCDFs. In smallmouth bass samples with total PCDD/F concentrations above 50 pg/g, PCDD/F patterns were dominated by PeCDFs and TCDFs. Overall, smallmouth bass were notable for the apparent dominance of TCDFs and PeCDFs and relatively low levels of HpCDDs and OCDD at many locations compared to other fish species from the Study Area. Round 3 sturgeon, which were individual whole-body samples, were dominated by TCDFs. The PCDD/F patterns for sturgeon samples from the ODHS/EPA/ATSDR Fish Contaminant Study, which were fillet samples, appear distinctly different from Round 3 whole-body sturgeon.

The average TCDD TEQs for whole-body samples ranged from 0.849 pg/g (sturgeon) to 4.77 pg/g (smallmouth bass) (Table 5.5-1). The maximum TCDD TEQ (51.9 J pg/g) among whole-body samples occurred in a smallmouth bass composite sample collected from SB07W. Average TEQs for fillet samples ranged from 0.564 pg/g (sturgeon) to 2.83 pg/g (carp) (Table 5.5-1). The maximum TEQ (8.74 J pg/g) in a fillet composite sample occurred in smallmouth bass from SB07W, the same location, and a composite of the same fish that yielded the maximum TEQ for whole-body composite samples.

5.5.5.2 Total PCDD/F Homologs and TCDD TEQ in Invertebrates

PCDD/Fs were detected in all invertebrate tissue samples from the Study Area. Average total PCDD/F homolog concentrations in individual invertebrate tissue composite samples ranged from 30.6 pg/g (field-exposed depurated clam tissue) to

360 pg/g (laboratory-exposed worm) (Table 5.5-2). The maximum concentration (6,440 pg/g) was measured in worms exposed in the laboratory to sediment from BT017 (RM 6.9W; Map 5.5-14d). All worm and epibenthic community samples, many field-collected clam samples, and several laboratory-exposed clam, crayfish, and mussel samples had total PCDD/F homolog concentrations greater than 50 pg/g. Homolog concentrations appear strongly associated with species in that concentrations in worms were typically higher than in field-collected clams, which in turn were higher than laboratory-exposed clams. Concentrations in epibenthic tissues and crayfish fell between worms and field-collected clams from similar locales (Figure 5.5-2).

The PCDD/F homolog distribution was more consistent across invertebrate species than fish tissue species in that OCDD was often the most abundant homolog, usually followed by TCDFs and HpCDDs. PeCDDs, HxCDDs, HpCDFs, and OCDF tended to be the least abundant homolog groups (Figure 5.5-27). Distinct homolog patterns seemed to be present in many samples that contained higher total PCDD/F concentrations than typically found for each invertebrate sample type. Apparently unique signatures were seen in every species and in several locations. For example, various invertebrate samples collected from the vicinity of RM 7W exhibited a visually unique PCDD/F pattern, including field-collected and laboratory-exposed clam and worm samples from BT017 and BT018 and crayfish from location 07R006. TCDFs were the most abundant group in these samples, followed by PeCDFs. Among epibenthic samples, MIT003/005/006 (multiplate samples from RM 6.5–7.4) appeared to have an unusual pattern, with TCDFs most abundant, followed by OCDD. The PCDD/F pattern in crayfish from location 08R003 (RM 8W, near the mouth of Swan Island Lagoon) was also unusual, with PeCDFs dominant, followed by OCDD and TCDFs. Additional apparently unique patterns were found in various samples from other locations (Figure 5.5-27).

The average TCDD TEQs ranged from 0.229 pg/g (field clams, depurated) to 16.5 pg/g (laboratory-exposed worms) (Table 5.5-2). The maximum TCDD TEQ occurred in a worm sample (448 J pg/g) exposed in the laboratory to sediments from Station BT017 (Map 5.5-14d).

5.5.6 DDx Compounds in Biota

The distributions of DDx compounds—the sum of ortho (2,4'-) and para (4,4'-) isomers of DDD, DDE, and DDT—are described in this section, including concentration trends and DDx analyte patterns in tissue samples from the Study Area. A box-whisker plot showing the distribution of total DDx for each species and tissue type is provided in Figure 5.5-1. Total DDx concentrations by river mile for each species and tissue type are shown in Figures 5.5-12a–j.

5.5.6.1 DDx Compounds in Fish

DDx compounds were detected in all fish tissue samples from the Study Area. Average concentrations of total DDx ranged from 77.4 µg/kg (juvenile Chinook) to 322 µg/kg

(northern pikeminnow) in whole-body tissues (Table 5.5-1). Maximum total DDx concentrations in whole-body samples occurred in sculpin composites from 07R006 (RM 7.3W; 3,060 µg/kg total DDx; Map 5.5-11d). Average fillet concentrations ranged from 11.4 µg/kg (black crappie) to 151 µg/kg (carp) (Table 5.5-1). The maximum concentration in fillet samples was measured in a carp composite sample from CP0408 (RM 4–8; 494 µg/kg total DDx; Map 5.5-3b).

4,4'-DDE accounted for more that 50 percent of the total DDx concentration in fish tissues, with some exceptions (Figure 5.5-28). DDDs constituted 25 percent or more of the total DDx in various samples of sculpin, carp, largescale sucker, juvenile Chinook, juvenile lamprey, and smallmouth bass, particularly in fishing areas from RM 3 to 9. In many of the sculpin samples and in juvenile Chinook samples from T01 (RM 3.4E) and T03 (RM 9.7W), 4,4'-DDT dominated; and in juvenile Chinook samples from T02 (RM 6.9W), 4,4'-DDD dominated the three whole-body composite samples.

5.5.6.2 DDx Compounds in Invertebrates

DDx compounds were detected in all invertebrate tissue samples from the Study Area. Average total DDx concentrations ranged from 2.47 µg/kg (mussel) to 114 µg/kg (laboratory-exposed worm) (Table 5.5-2). The maximum concentration (1,490 µg/kg total DDx) was measured in worms exposed in the laboratory to sediments from BT017 (RM 6.9W; Map 5.5-14d), with similar concentrations in laboratory-exposed worm tissues from Station BT018 (RM 7.2W; Map 5.5-14d).

In general, 4,4'-DDD and 4,4'-DDE were the dominant isomers in invertebrate tissues (Figure 5.5-29). Crayfish DDx profiles were dominated by 4,4'-DDE or various other DDx compounds, depending on location. Laboratory-exposed clam samples from the USACE investigation (Tetra Tech 2006) also had a high percentage of 4,4'-DDT; however, the 2,4'-isomers of the DDx compounds were generally not detected in these samples, resulting in the absence of data for 2,4'-DDD and a shift in the apparent DDx pattern.

5.5.7 PAHs in Clam Tissue

PAHs are a biota IC for clam tissue only (Table 5.0-2). PAHs are generally metabolized in vertebrates and do not bioaccumulate. However, they do bioaccumulate in invertebrates, and total PAHs are discussed in this section with respect to clam tissue.

PAHs were detected in all clam tissue samples collected from the Study Area or exposed to sediment from the Study Area. Average total PAH concentrations for clam tissue ranged from 76 µg/kg (depurated clam tissue) to 478 µg/kg (undepurated clam tissue; Table 5.5-2), with the highest concentration found in undepurated clam tissue from FC015 at RM 6.4W (4,980 µg/kg; Map 5.5-5d). A box-whisker plot showing the distribution of PAHs for clams is provided in Figure 5.5-3. PAH concentrations by river mile for each clam sample type are shown in Figure 5.5-13.

Clam tissue from the Study Area generally contained relatively higher proportions of phenanthrene, fluoranthene, pyrene, and chrysene, with lesser proportions of benzo(a)anthracene. LPAHs other than phenanthrene and the five- and six-ring HPAHs (see Table 5.1-7) were generally present but not prevalent (Figure 5.5-30). Naphthalene and dibenzo(a,h)anthracene were each detected in only a few samples, but naphthalene was the dominant PAH in the field-collected, undepurated clam from location CA12E (RM 12.1E; sample LW3-CA12E-C00). Naphthalene was not detected in the depurated clam sample from this location. Additional samples with PAH patterns that appeared to differ from surrounding areas included field-collected clams from locations FC014 (RM 5.9W), FC015 (RM 6.4W), and FC025 (RM 8.5W), and laboratory-exposed clams for locations WR-VC-29 (RM 7.7, just downstream of Swan Island Lagoon in the navigation channel), BT026 (RM 8.5E, in Swan Island Lagoon), and BT031 (RM 9.5E). PAH patterns in three field-collected clam samples from Round 1 are obscured by non-detects; only fluoranthene and pyrene were detected in two of these samples.

5.5.8 Additional Indicator Chemicals in Biota

5.5.8.1 Bis(2-ethylhexyl)phthalate in Biota

A box-whisker plot showing the distribution of BEHP for each species and tissue type is provided in Figure 5.5-3. BEHP concentrations by river mile for each species and tissue type are shown in Figures 5.5-14a–j.

5.5.8.1.1 Fish

BEHP, the most frequently detected phthalate, was detected in 25 of the 160 fish samples analyzed from the Study Area, and was not detected in juvenile Chinook or carp. Black crappie, northern pikeminnow, and peamouth were not analyzed for phthalates. Average detected BEHP concentrations in whole-body tissues ranged from 150 µg/kg (sturgeon) to 21,000 µg/kg (smallmouth bass) (Table 5.5-1). The maximum concentration (87,000 J µg/kg) was measured in a smallmouth bass whole-body composite sample from 04R023 (fishing zone from RM 3.5 to 4.5; Map 5.5-12b). A second smallmouth bass composite sample from this location contained 32,000 J µg/kg BEHP. The BEHP concentration in a sculpin composite sample from Station 08R003 (RM 8.2E; Map 5.5-11e) was reported at 28,000 J µg/kg. All other concentrations were 1 to 2 orders of magnitude lower than the maximum concentration.

Phthalates were analyzed in brown bullhead skinless fillet, carp fillet, and smallmouth bass fillet samples. BEHP was not detected in any of the carp fillet samples and in only one brown bullhead fillet sample (100 µg/kg). BEHP was detected in three smallmouth bass fillet samples with an average concentration of 97 µg/kg and a maximum concentration of 130 J µg/kg (Table 5.5-1).

5.5.8.1.2 Invertebrates

BEHP was most frequently detected in laboratory-exposed clams (82.2 percent), followed by laboratory-exposed worms (60 percent) and field mussels (57.1 percent).

BEHP was not detected in crayfish. Epibenthic community composites were not analyzed for phthalates.

Average BEHP concentrations ranged from 91 µg/kg (mussels) to 360 µg/kg (laboratory-exposed clams) (Table 5.5-2). The maximum concentration (8,600 µg/kg) was measured in laboratory-exposed clams to sediments from BT028 (RM 8.8W; Map 5.5-7e). All other concentrations were below 350 µg/kg, and most were below 200 µg/kg (Figure 5.5-14f). Paired clam and worm samples exposed in the laboratory tended to have concentrations within the same order of magnitude (typically within a factor of 5 or less).

5.5.8.2 Total Chlordanes in Biota

A box-whisker plot showing the distribution of total chlordanes for each species and tissue type is provided in Figure 5.5-4. Total chlordane concentrations by river mile for each species and tissue type are shown in Figures 5.5-15a–j.

5.5.8.2.1 Fish

Chlordanes were detected with varying frequency in all species except northern pikeminnow. The frequency of detection ranged from 25 percent (black crappie fillets) to 100 percent (whole-body black crappie and sturgeon). Average whole-body concentrations ranged from 3.2 µg/kg (juvenile Chinook) to 20.1 µg/kg (juvenile lamprey) (Table 5.5-1); the maximum concentration (67 µg/kg) was measured in a brown bullhead whole-body composite sample at FZ0306 (fishing zone between RM 3 and 6; Map 5.5-2a). Average fillet concentrations ranged from 1.1 µg/kg (black crappie) to 9.15 µg/kg (carp). The maximum detected concentration in any fillet sample (12 NJ µg/kg) occurred in carp from Station CP0408 (fishing zone from RM 4 to 8; Map 5.5-3b).

5.5.8.2.2 Invertebrates

Chlordanes were detected in all invertebrate samples except 22 crayfish tissues. Species mean concentrations ranged from 0.45 µg/kg (mussels) to 6.51 µg/kg (laboratory-exposed worms) (Table 5.5-2). Maximum concentrations were below 9 µg/kg in juvenile Chinook stomach contents, laboratory-exposed clams, crayfish, and epibenthic community samples. For field-collected clams, the maximum was 16 J µg/kg (Map 5.5-5e); for laboratory-exposed worms, the maximum was 71.9 µg/kg (Map 5.5-14e). These highest concentrations were measured in samples exposed in the laboratory to sediments from BT028 (RM 8.8W; Map 5.5-14e).

5.5.8.3 Aldrin in Biota

A box-whisker plot showing the distribution of aldrin for each species and tissue type is provided in Figure 5.5-4. Aldrin concentrations by river mile for each species and tissue type are shown in Figures 5.5-16a–j.

5.5.8.3.1 Fish

Aldrin was only detected in Round 2 and Round 3 fish samples, including juvenile lamprey, carp, smallmouth bass, sturgeon and sculpin. Laboratory detection limits were lower for these samples than for Round 1 samples. The average whole-body concentrations for these species ranged from 0.014 µg/kg (sculpin) to 1.60 µg/kg (juvenile lamprey) (Table 5.5-1), with the maximum concentration found in juvenile lamprey from RM 2.4 (1.82 µg/kg; Map 5.5-8). Aldrin was detected in the fillet composite samples of smallmouth bass and carp only (averages are 0.084 µg/kg and 0.0066 µg/kg, respectively; Table 5.5-1).

5.5.8.3.2 Invertebrates

Aldrin was detected in all invertebrate species. The frequency of detection ranged from 2.8 percent (crayfish) to 100 percent. Average concentrations ranged from 0.004 µg/kg in the contents of sturgeon stomachs to 1.49 µg/kg in laboratory-exposed worms (Table 5.5-2). The maximum concentration (37 µg/kg) occurred in worms exposed in the laboratory to sediment from BT028 on the west bank near RM 8.8W (Map 5.5-14e). Field-collected and laboratory-exposed clams also contained their maximum concentrations at Station BT028 (5.07 µg/kg and 2.14 µg/kg; Maps 5.5-5e and 5.5-7e, respectively). The maximum aldrin concentrations in Chinook stomach contents and epibenthic community composites were below 0.1 µg/kg.

5.5.8.4 Dieldrin in Biota

A box-whisker plot showing the distribution of dieldrin for each species and tissue type is provided in Figure 5.5-5. Dieldrin concentrations by river mile for each species and tissue type are shown in Figures 5.5-17a–j.

5.5.8.4.1 Fish

Dieldrin was detected in all fish species except largescale sucker, northern pikeminnow, and peamouth. Laboratory detection limits for dieldrin were also higher for Round 1 samples than for subsequent LWG samples, and these species were only collected in Round 1. Among species with dieldrin detections, the frequency of detection ranged from 16.7 percent (brown bullhead fillets) to 100 percent (Table 5.5-1). The average whole-body concentrations for these species ranged from 1.6 µg/kg (Chinook) to 4.56 µg/kg (sculpin) (Table 5.5-1). The maximum concentration was found in sculpin from 02R001 (24 J µg/kg at RM 2.4E; Map 5.5-11b). Similar concentrations were also found in sculpin from 03R004 and 03R005 (15 NJ µg/kg at RM 4.1E and 19 NJ µg/kg at RM 3.7E, respectively; Map 5.5-11b). Average fillet concentrations ranged from 0.555 µg/kg (smallmouth bass) to 2.1 µg/kg (brown bullhead) (Table 5.5-1). The maximum concentration in fillet occurred in a smallmouth bass fillet composite sample from 03R014 (3.3 NJ µg/kg; fishing zone from RM 2.4 to 3.5; Map 5.5-12a).

5.5.8.4.2 Invertebrates

Dieldrin was detected in all invertebrate species with frequencies from 22 percent (crayfish) to 100 percent (several species). Average concentrations ranged from

0.018 µg/kg (crayfish) to 1.48 µg/kg (Chinook stomach contents) (Table 5.5-2). The maximum concentration was from a laboratory-exposed worm composite sample exposed to sediment from BT028 (26.7 µg/kg at RM 8.8W; Map 5.5-14e). All other concentrations were less than 4.5 µg/kg.

5.5.8.5 Arsenic in Biota

A box-whisker plot showing the distribution of arsenic for each species and tissue type is provided in Figure 5.5-5. Arsenic concentrations by river mile for each species and tissue type are shown in Figures 5.5-18a–j.

5.5.8.5.1 Fish

Arsenic was detected in all fish species and tissues analyzed within the Study Area, with average whole-body concentrations consistently higher than those of fillets. Average whole-body arsenic concentrations ranged from 0.056 mg/kg (brown bullhead) to 0.594 mg/kg (sturgeon) (Table 5.5-1). The highest whole-body concentrations were associated with all sturgeon samples (0.46–1.06 mg/kg), peamouth from Swan Island Lagoon (0.48 mg/kg; Map 5.5-10b), black crappie from FZ0609 (0.42 mg/kg; fishing zone from RM 6 to 9; Map 5.5-1b), and smallmouth bass and northern pikeminnow from 03R014 (0.39 mg/kg and 0.36 mg/kg, respectively; Maps 5.5-12a and 5.5-10b). Average fillet concentrations ranged from 0.02 mg/kg (brown bullhead) to 0.34 mg/kg (sturgeon) (Table 5.5-1), with the maximum concentration in sturgeon from RM 3.5 to 9.2 (0.54 mg/kg; Maps 5.5-13a-b).

5.5.8.5.2 Invertebrates

Arsenic was detected in all invertebrate samples, with the exception of one crayfish composite sample. Average concentrations ranged from 0.34 mg/kg (crayfish) to 1.03 mg/kg (laboratory-exposed worms) (Table 5.5-2). The maximum concentrations (3.04 mg/kg) occurred in worms exposed in a laboratory to sediment collected from BT019 (RM 7.4E; Map 5.5-14d). Most of the field-collected and laboratory-exposed clam samples were below 1.0 mg/kg; most of the worm sample concentrations were higher, but below 2.0 mg/kg.

5.5.8.6 Chromium in Biota

A box-whisker plot showing the distribution of chromium for each species and tissue type is provided in Figure 5.5-6. Chromium concentrations by river mile for each species and tissue type are shown in Figures 5.5-19a–j.

5.5.8.6.1 Fish

Chromium was detected in all fish species with varying detection rates. Average whole-body concentrations were higher than those of fillets, except for black crappie, where chromium was only detected in fillets. Average whole-body chromium concentrations ranged from 0.13 mg/kg (Chinook) to 20.2 mg/kg (sturgeon) (Table 5.5-1). The maximum concentration in an individual sample occurred in a sturgeon from STWB01 near RM 2.3 (40.2 mg/kg; Map 5.5-13a). Maxima for all other species

were less than one-tenth that of the sturgeon. Average fillet concentrations ranged from 0.12 mg/kg (brown bullhead) to 1.55 mg/kg (sturgeon) (Table 5.5-1), with the maximum concentration in sturgeon from RM 3.5 to 9.2 (3.25 mg/kg; Map 5.5-13a-b). All other fillet concentrations were less than 1.50 mg/kg.

5.5.8.6.2 Invertebrates

Chromium was detected in all invertebrate tissue samples, with detection rates ranging from 42.9 to 100 percent. Average concentrations ranged from 0.20 mg/kg (laboratory clams) to 1.6 mg/kg (sturgeon stomach contents) (Table 5.5-2). The highest concentration (4.1 mg/kg) occurred in sturgeon stomach content sample STWB03 (RM 6.6; Map 5.5-13a).

5.5.8.7 Copper in Biota

A box-whisker plot showing the distribution of copper for each species and tissue type is provided in Figure 5.5-6. Copper concentrations by river mile for each species and tissue type are shown in Figures 5.5-20a–j.

5.5.8.7.1 Fish

Copper was detected in all fish species and tissues analyzed within the Study Area, with whole-body concentrations consistently higher than those of fillets. Average whole-body copper concentrations ranged from 0.661 mg/kg (northern pikeminnow) to 4.27 mg/kg (juvenile lamprey). The maximum concentration occurred in a sculpin composite sample from location SP10W at RM 10.3W (7.16 mg/kg; Map 5.5-11f). Average fillet concentrations ranged from 0.178 mg/kg (black crappie) to 0.459 mg/kg (carp) (Table 5.5-1), with the maximum concentration in smallmouth bass from 05R006 (1.12 mg/kg; fishing zone from RM 4.5 to 5.5; Map 5.5-12b).

5.5.8.7.2 Invertebrates

Copper was detected in all invertebrate tissue samples. Average concentrations ranged from 1.39 mg/kg (mussel) to 14.8 mg/kg (crayfish) (Table 5.5-2). The maximum concentration (20.2 mg/kg) was found in laboratory-exposed worms, which were exposed to sediment collected from location BT023 (RM 8.1E; Map 5.5-14e), and crayfish from location CR11E (RM 11.3E; Map 5.5-6d). Copper concentrations were below 4 mg/kg in the remaining worm samples, while all crayfish samples contained more than 10 mg/kg of copper.

5.5.8.8 Zinc in Biota

A box-whisker plot showing the distribution of zinc for each species and tissue type is provided in Figure 5.5-7. Zinc concentrations by river mile for each species and tissue type are shown in Figures 5.5-21a–j.

5.5.8.8.1 Fish

Zinc was detected in tissues of all fish species analyzed within the Study Area, with whole-body concentrations consistently higher than fillet concentrations. Average

whole-body zinc concentrations ranged from 9.53 mg/kg (sturgeon) to 91.9 mg/kg (carp) (Table 5.5-1). The maximum whole-body concentration was measured in a carp composite sample from fishing zone CP0408 (113 mg/kg; fishing zone from RM 4 to 8; Map 5.5-3b); sample maxima for all other species were less than 35 mg/kg. Average fillet concentrations ranged from 2.56 mg/kg (sturgeon) to 25.7 mg/kg (carp) (Table 5.5-1). The maximum fillet sample concentration occurred in a carp composite from CP0408 (31 J mg/kg; Map 5.5-3b); average zinc concentrations in fillet samples of the remaining fish species were below 11 mg/kg.

5.5.8.8.2 Invertebrates

Zinc was detected in all invertebrate tissues analyzed. Average concentrations ranged from 13 mg/kg (laboratory-exposed clams) to 33.1 mg/kg (field-collected clams) (Table 5.5-2). The maximum concentration (54 mg/kg) in an invertebrate sample was measured in field-collected clams from FC023 (RM 8.1E; Map 5.5-5e). All epibenthic community sample composites contained less than 25 mg/kg zinc. Zinc concentrations in crayfish tissue did not exceed 21 mg/kg; in laboratory-exposed worms and clam samples, concentrations were below 55 mg/kg.

5.5.8.9 Tributyltin Ion in Biota

A box-whisker plot showing the distribution of TBT for each species and tissue type is provided in Figure 5.5-7. TBT concentrations by river mile for each species and tissue type are shown in Figures 5.5-22a–i.

5.5.8.9.1 Fish

TBT was analyzed in samples of juvenile Chinook and all Round 3 samples, including carp, juvenile lamprey, sculpin, smallmouth bass, and sturgeon (see Table 2.2-8). Average whole-body concentrations ranged from 0.78 µg/kg (sturgeon) to 5.6 µg/kg (carp) (Table 5.5-1). The maximum concentration occurred in a carp whole-body composite sample at CP0812 (8.6 µg/kg; fishing zone from RM 8 to 12; Map 5.5-3c). Average fillet concentrations were 0.68 µg/kg (smallmouth bass) and 5.4 µg/kg (carp) (Table 5.5-1). The maximum fillet sample concentration also occurred in a carp composite from CP0812 (11 J µg/kg; fishing zone from RM 8 to 12; Map 5.5-3c).

5.5.8.9.2 Invertebrates

TBT was detected in just over half of the field-collected clams (59.5 percent), laboratory-exposed worms (55.6 percent), and crayfish (55.6 percent), and slightly less frequently in the laboratory-exposed clams (42.2 percent). TBT was detected in all seven mussel tissue composite samples. It was not detected in the four field-collected clam samples that had been depurated prior to analysis and was not measured in epibenthic community samples.

Average TBT concentrations in invertebrate species ranged from 1.3 µg/kg (crayfish) to 77 µg/kg (laboratory-exposed worms) (Table 5.5-1). The maximum concentration in an individual composite sample (1,700 µg/kg) was measured in worms exposed in the

laboratory to sediment from BT023 near the entrance to Swan Island Lagoon (RM 8.1E; Map 5.5-14e). The next-highest concentrations were also found at this location in the field-collected and laboratory-exposed clams (680 µg/kg and 530 µg/kg, respectively).

5.5.8.10 Lipids

A box-whisker plot showing the distribution of lipids for each species and tissue type is provided in Figure 5.5-8. Lipids concentrations by river mile for each species and tissue type are shown in Figures 5.5-23a–j.

5.5.8.10.1 Fish

The lipid content was measured in most fish tissue samples (Table 2.2-8). The average lipid content in fish whole-body tissues ranged from 1.8 percent (Chinook) to 8.93 percent (peamouth) (Table 5.5-1). The maximum lipid content (13 percent) in an individual composite was found in a carp whole-body composite from FZ0609 (fishing zone from RM 6 to 9; Map 5.5-3c). As expected, fillets typically had lower lipid content, with averages ranging from 1.04 percent (smallmouth bass) to 5.49 percent (carp).

5.5.8.10.2 Invertebrates

The lipid content was measured in most tissue samples with sufficient volume (e.g., no lipid was measured in epibenthic community samples). Average lipid content in invertebrates ranged from 0.40 percent (mussel) to 2.32 percent (laboratory-exposed worms) (Tables 5.5-2). The maximum lipid content occurred in field-collected clam tissue from Station BT028, RM 8.8W (4.63 percent; Map 5.5-5e).

5.5.9 Indicator Chemicals in Biota Collected Upriver from the Study Area and Above Willamette Falls

Seven upriver locations, between Ross Island (RM 16) and Willamette Falls (RM 26) and above Willamette Falls, were sampled in connection with studies completed in Portland Harbor (see Table 5.5-6, Maps 5.5-15 and 5.5-16):

- Station 20R001 (sampling locations between RM 21 and 24) and Station 28R001 (sampling locations between the Tualatin River mouth and approximately RM 32, above Willamette Falls), where whole-body brown bullhead and smallmouth bass were collected during Round 1
- Stations LT023b (Sellwood Bridge, RM 16.5–17) and LT023c (Eld Rock Island, RM 18.7), where lamprey ammocoetes and macropthalmia were collected during Round 3
- Station 26R111 (just above Willamette Falls) and Station T04 (on the east bank of the river below RM 18), where juvenile Chinook whole-body tissue (and stomach content tissues in the case of T04) were collected during Round 1 (26R111) and Round 2 (T04)

- Station WR-PG-Ref03 below RM 19, which was used as a reference station for sediment sampled as part of the USACE investigation (Tetra Tech 2006) for bioaccumulation testing using worms and clams.

These data are discussed here to assist in establishing typical tissue concentrations of ICs outside of the Study Area. The results are summarized in this section and in Tables 5.5-3, 5.5-4, and 5.5-5. Adult Pacific lamprey from the Willamette Falls area and adult spring Chinook from the Clackamas River hatchery were not included in the evaluation of ICs in upriver areas because they are anadromous species and do not represent conditions upriver of the Study Area; their tissue burdens are representative of exposures throughout the Lower Willamette River and beyond. Data from these investigations are more fully described in the BHHRA (Appendix F).

5.5.9.1 Total PCBs in Biota

5.5.9.1.1 Fish

Average total PCB concentrations in fish from the upriver reach and above falls areas ranged from 14.5 µg/kg in juvenile Chinook to 238 µg/kg in smallmouth bass. The maximum concentration of 317 J µg/kg was measured in a smallmouth bass composite collected near RM 24, one of the samples from location 20R001.

As in the Study Area, PCB congeners in samples from the upriver reach and above-falls locations included predominantly tetraCBs, pentaCBs, hexaCBs, and heptaCBs. Chlorination levels were lower for juvenile Chinook and juvenile lamprey than for brown bullhead and smallmouth bass: juvenile Chinook and juvenile lamprey contained higher proportions of triCBs and tetraCBs, whereas brown bullhead and smallmouth bass contained more heptaCBs and octaCBs.

5.5.9.1.2 Invertebrates

Total PCBs (as congeners) were detected in one juvenile Chinook stomach content sample, at a concentration of 10.6 J µg/kg. Aroclors were not detected in laboratory-exposed clams and worms, and these samples were analyzed for too few PCB congeners to calculate a total PCB congener value.

5.5.9.2 Total PCDD/F Homologs and TCDD TEQ in Biota

5.5.9.2.1 Fish

Total PCDD/F averages for the various fish species ranged from 3.03 pg/g in brown bullhead to 53 pg/g in juvenile lamprey. The maximum concentration (63 pg/g) was measured in a juvenile lamprey composite collected near RM 18.9. As they did in the Study Area, dioxin and furan homolog patterns varied among species, locations and individual samples. For example, TCDFs dominated the PCDD/F pattern for juvenile lamprey from above the falls, whereas OCDD was the predominant homolog for juvenile lamprey from the upriver reach area (below the falls); OCDD was also clearly the dominant homolog in juvenile Chinook. Smallmouth bass contained a high proportion of PeCDF in many cases, whereas HxCDD was the most abundant homolog

in two of the three brown bullhead samples. One brown bullhead sample was notable for its large proportion of TCDD (30 percent of total PCDD/Fs as the sum of homologs); this homolog did not account for more than 10 percent of the PCDD/Fs in any other sample from the upriver reach or above the falls. Additional differences were noted among samples from the same species and similar locations.

Average concentrations of TCDD TEQ ranged from 0.45 pg/g (brown bullhead) to 2 pg/g (juvenile lamprey). The maximum concentration was found in a juvenile lamprey composite collected near RM 18.5 (3.1 J pg/g).

5.5.9.2.2 Invertebrates

PCDD/F analyses were not conducted for juvenile Chinook stomach contents or bioaccumulation test organisms.

5.5.9.3 Total DDx in Biota

5.5.9.3.1 Fish

Average total DDx concentrations ranged from 8.5 µg/kg in juvenile Chinook whole-body tissues to 94 µg/kg in smallmouth bass. The highest concentrations, 120.4 NJ µg/kg and 104.5 J µg/kg, were measured in smallmouth bass (20R001 replicates collected near RM 23 and 24, respectively).

The most abundant isomer in all tissues was 4,4'-DDE, which accounted for at least 50 percent and up to 85 percent of the total DDx. Typically 4,4'-DDT was the next most abundant; all other DDx compounds were also present at varying levels.

5.5.9.3.2 Invertebrates

Three invertebrate samples from the upriver reaches were analyzed for DDx isomers. The total DDx concentration in the juvenile Chinook stomach content sample was 6.61 J µg/kg. The most abundant isomer was 4,4'-DDE, which contributed over 80 percent to the total DDx concentration. Total DDx concentrations were 11.3 J µg/kg and 5.89 J µg/kg in laboratory-exposed worm and clam samples, respectively. In worms, 4,4'-DDE and 2,4'-DDT were the two most abundant isomers; in the laboratory-exposed clams, 4,4'-DDE, 4,4'-DDT, and 2,4'-DDT were most abundant.

5.5.9.4 Total PAHs in Clam Tissue

PAH data are available for only one clam sample that represents exposures upriver of the Study Area. The total PAH concentration in this laboratory-exposed clam tissue was 29.7 J µg/kg. HPAHs accounted for almost two-thirds of this total.

5.5.9.5 Bis(2-ethylhexyl)phthalate in Biota

5.5.9.5.1 Fish

BEHP was detected in eight of the 12 fish tissue samples from the upriver reach and none of the five fish from above the falls. The detected concentrations ranged from

140 J µg/kg (in two juvenile Chinook composites) to 4,800 µg/kg in smallmouth bass from RM 21 to 24.

5.5.9.5.2 Invertebrates

BEHP was reported at 85 J µg/kg in clams and 66 J µg/kg in worms used in the bioaccumulation tests from the USACE investigation upriver area (Tetra Tech 2006). Contents of juvenile Chinook stomachs were not analyzed for phthalates.

5.5.9.6 Total Chlordanes in Biota

5.5.9.6.1 Fish

Chlordanes were detected in all fish tissue samples, except juvenile Chinook, from above Willamette Falls. Average total chlordanes ranged from 1.1 µg/kg (brown bullhead) to 15.4 µg/kg (juvenile lamprey). The maximum concentration (25.2 µg/kg) was reported in a juvenile lamprey composite sample collected near RM 18.9.

5.5.9.6.2 Invertebrates

Total chlordanes were reported at 2.26 J µg/kg in the juvenile Chinook stomach content sample and 4.6 J µg/kg in clam tissues analyzed from the reference area bioaccumulation tests. This pesticide group was not detected in the worms.

5.5.9.7 Aldrin in Biota

5.5.9.7.1 Fish

Aldrin was only detected in the four juvenile lamprey composite samples from RM 16.9 and 18.3, with an average concentration of 1.86 µg/kg. The maximum concentration was 2.72 µg/kg. The detection limits for aldrin in fish tissue were higher than for the Round 2 and 3 Study Area fish samples, which were the only Study Area fish tissue samples in which aldrin was detected.

5.5.9.7.2 Invertebrates

Aldrin was not detected in the juvenile Chinook stomach composite from fish collected at T04 or in the clams exposed to reference area sediments. The worm sample concentration was reported as 0.36 J µg/kg.

5.5.9.8 Dieldrin in Biota

5.5.9.8.1 Fish

Dieldrin was detected in 15 of 17 fish tissue samples from the upriver reach and above the falls. The average concentration of dieldrin in upriver fish tissue composites ranged from 0.74 µg/kg (juvenile Chinook) to 3.5 µg/kg (smallmouth bass) with a maximum concentration (5.36 µg/kg) reported in a juvenile lamprey composite sample from RM 18.9.

5.5.9.8.2 Invertebrates

Dieldrin concentrations in upriver invertebrates ranged from 0.22 J µg/kg (laboratory-exposed worms) to 0.905 µg/kg (juvenile Chinook stomach contents).

5.5.9.9 Arsenic in Biota

5.5.9.9.1 Fish

Arsenic was detected in all tissue samples from the upriver reach and above the falls. Average arsenic concentrations in the upriver fish tissues ranged from 0.03 mg/kg (juvenile Chinook) to 0.32 mg/kg (smallmouth bass). A maximum of 0.36 mg/kg occurred in a smallmouth bass composite collected near RM 23 (part of 20R001).

5.5.9.9.2 Invertebrates

The arsenic concentration in the laboratory-exposed clam composite was reported as 0.392 mg/kg; the laboratory-exposed worm composite sample had a higher concentration, at 1.52 mg/kg. Arsenic was not measured in juvenile Chinook stomach contents.

5.5.9.10 Chromium in Biota

5.5.9.10.1 Fish

Chromium was detected in 12 of 17 fish tissue samples from the upriver reach and above the falls. Average concentrations of chromium in fish tissues ranged from 0.28 mg/kg (juvenile lamprey) to 2.04 mg/kg (brown bullhead). A maximum concentration of 2.79 mg/kg occurred in smallmouth bass at 20R001, RM 21 to 24. Chromium was not detected in juvenile Chinook.

5.5.9.10.2 Invertebrates

The chromium concentration was 0.0936 mg/kg in the laboratory-exposed clam composite sample and 0.197 mg/kg in the laboratory-exposed worm composite sample. Chromium was not measured in juvenile Chinook stomach contents.

5.5.9.11 Copper in Biota

5.5.9.11.1 Fish

Copper was detected in all fish tissue samples from the upriver reach and above the falls. Average concentrations of copper in fish tissue ranged from 0.44 mg/kg (smallmouth bass) to 4.43 mg/kg (juvenile lamprey). A maximum concentration of 4.8 mg/kg occurred in juvenile lamprey (LT023c at RM 18.7).

5.5.9.11.2 Invertebrates

The copper concentration was 3.19 mg/kg in the laboratory-exposed clam composite sample and 1.86 mg/kg in the laboratory-exposed worm composite sample. Chromium was not measured in juvenile Chinook stomach contents.

5.5.9.12 Zinc in Biota

5.5.9.12.1 Fish

Zinc was detected in all fish tissue samples from the upriver reach and above the falls. The average concentrations of zinc in upriver fish tissue composites ranged from 13.8 mg/kg (smallmouth bass) to 34 mg/kg (juvenile Chinook) with a maximum concentration of 34 mg/kg in juvenile Chinook from Willamette Falls.

5.5.9.12.2 Invertebrates

The zinc concentration in the laboratory-exposed clam composite sample was reported as 10.6 mg/kg; the laboratory-exposed worm composite sample concentration was 22 mg/kg. Zinc was not measured in juvenile Chinook stomach contents.

5.5.9.13 Tributyltin Ion in Biota

5.5.9.13.1 Fish

TBT concentrations ranged from 0.37 J to 0.45 J µg/kg (average of 0.41 µg/kg) in the three juvenile Chinook samples from T04. TBT was not measured in other upriver fish tissues.

5.5.9.13.2 Invertebrates

The TBT concentration was reported as 0.091 NJ µg/kg in laboratory-exposed clams (although qualified as tentatively identified) and was not detected in laboratory-exposed worms from the USACE (Tetra Tech 2006) reference area. TBT was not measured in juvenile Chinook stomach contents.

5.5.9.14 Lipids

5.5.9.14.1 Fish

Average percent lipids ranged from 1.8 percent (juvenile Chinook composites) to 6.2 percent (smallmouth bass composites). These values were very similar to the range reported for the same species from the Study Area. The maximum concentration occurred in a juvenile lamprey composite from RM 18.9 (10 percent).

5.5.9.14.2 Invertebrates

Lipid levels in the single clam and single worm composite sample from the bioaccumulation tests were reported as 1.1 percent and 1.6 percent, respectively. These values fall within the range reported for clams and worms within the Study Area, although the upriver worm lipid value is lower (1.6 percent) than the average reported for worms used in Study Area bioaccumulation tests (2.3 percent). No lipid data are available for the upriver juvenile Chinook stomach contents.

5.5.10 Summary of Nature and Extent of Indicator Chemicals in Biota

The majority of the ICs discussed in this section are distributed across the entire Study Area at concentrations greater than those seen in the samples from the upriver reaches

and above Willamette Falls. Concentrations generally varied by orders of magnitude within and between species, with localized areas of concentrations elevated in comparison to the rest of the Study Area for some species and ICs. Concentrations in fish tissues were greater than those in invertebrate samples, with the exceptions of PCDD/Fs, arsenic, copper, and TBT.

The average maximum total PCB concentrations within the Study Area were elevated in comparison to the upriver reaches by at least an order of magnitude. The distribution of PCB homologs varied within and between species, but was generally dominated by tetraCB, pentaCBs, hexaCBs, and heptaCBs.

The average maximum whole-body fish tissue concentrations of PCDD/Fs in the Study Area and the upriver reaches were similar; however the maximum Study Area concentration was greater than that of the upriver reach by more than a factor of five. Invertebrate concentrations were greater than fish tissue concentrations within the Study Area; no upriver invertebrate data is available for comparison. PCDD/F homolog distributions were variable between fish species but were more consistent between invertebrate species.

Average maximum total DDx concentrations within the Study Area were also elevated in comparison to the upriver reaches. The distribution of DDx isomers varied between and within species, with the 4,4'-isomer of DDE being dominant in fish tissues. Invertebrates from the upriver reaches were also dominated by 4,4'-DDE, while invertebrates (except crayfish) from the Study Area were dominated by a mixture of 4,4'-DDD and 4,4'-DDE.

PAHs were an IC for clam tissues only, and only one sample from the upriver reach was available for comparison. The clam tissues from the Study Area were elevated in comparison to the upriver sample and were generally dominated by phenanthrene, fluoranthene, pyrene, and chrysene.

BEHP and TBT were the only ICs not found across the entire Study Area. These chemicals were found river-wide in clam and laboratory-exposed worm samples but only in localized areas in fish tissues.

Concentrations of total chlordanes, aldrin, dieldrin, arsenic, and copper within the Study Area were generally higher than those found in the upriver reaches, but were not elevated to the same degree relative to upriver reaches as were the other ICs.

Note to Reviewers:

Section 5.6 of the Draft RI Report, Distribution of Indicator Chemicals in Upstream and Downstream Areas and in Upper Study Area Sediment Depositional Areas, has been removed from the main text and placed in Appendix H. Data sets collected or received between June 2, 2008 (the Draft RI report data lockdown date) and July 19, 2010 (the revised RI report data lockdown date) are stored in an updated RI data set, provided as an attachment (on CD) to Appendix H. Section 5.6 text, tables, and figures have been updated to include data from this new data set. In addition, this appendix updates the presentation of the Downtown Reach and the area off RM 11 using this newly collected data[1].

[1] Sections 2 through 11 of this RI report present data using the draft RI report's lockdown date of June 2, 2008. The SCRA database contained on CD in Appendix A also reflects the June 2, 2008 lockdown date.

6.0 LOADING, FATE, AND TRANSPORT FOR SELECT INDICATOR CHEMICALS

This section presents an assessment of contaminant loading mechanisms to the Study Area from external sources as well as in-river processes affecting the concentration, transport, and fate of select ICs within the Study Area. The evaluation provided in this section is presented in two main parts. First, Section 6.1 assesses current and historical chemical inputs (i.e., external loading) to the Study Area. The loading terms/pathways discussed include upstream loading (via surface water and sediment bedload), stormwater runoff, permitted discharges (point-source, non-stormwater), upland groundwater (plume transport to river), atmospheric deposition (to the river surface), direct upland soil and riverbank erosion, groundwater advection through subsurface sediments (chemical partitioning from subsurface sediment to pore water and advection to the surface sediment interval), and overwater releases. Loading from surface sediment to the surface water via sediment erosion and resuspension (internal fate and transport processes) is not quantitatively evaluated in this RI report. Second, Section 6.2 describes chemical fate and transport processes that act on contaminants in abiotic and biotic media within the Study Area. The discussion of fate and transport processes is grouped into sediment and pore water processes, surface water processes, and biotic processes.

The primary purpose of this loading assessment and discussion of relevant fate and transport processes is to support development of the CSM (Section 10). The CSM will, in turn, support the FS in consideration of recontamination issues, monitored natural recovery, and development and evaluation of cleanup alternatives. The loading assessment will also support the fate and transport modeling effort, which is being conducted in parallel with development of the RI to support the FS and will be documented in separate project deliverables.

The information presented in this section is organized by individual loading, fate, and transport mechanism. A comparative, contaminant-by-contaminant assessment of these processes is presented in the CSM (Section 10), along with the other informational components considered in development of the CSM (i.e., the observed cross-media contaminant distribution patterns, the baseline risk assessment findings, and the understanding of current and historical sources).

The IC lists for this assessment are presented in Table 6.0-1. This table includes separate lists for surface water, stormwater, upland groundwater plumes, atmospheric deposition, and equilibrium partitioning (advective loading from subsurface sediment to surface sediment and from surface sediment to surface water). These lists were generated from the overall list of ICs for loading, fate, and transport developed in consultation with EPA, and reflect data availability by media and relevance of the contaminant to the loading mechanism (e.g., equilibrium partitioning ICs primarily focus on hydrophobic chemicals and metals; stormwater and atmospheric deposition ICs reflect the limited available data sets; upland plume loading ICs reflect individual

upland plumes, etc.). Development of the loading IC lists considered the chemical lists for in-river nature and extent (presented in Section 5), as well as the lists for CSM presentation (Section 10) and fate and transport modeling currently under development for the FS.

6.1 EXTERNAL LOADING

This section evaluates current and historical contaminant loading to the surface sediment and surface water of the Study Area by external loading mechanisms. The boundaries of the Study Area includes the upstream and downstream river mile designations (RM 1.9 and 11.8), the surface of the river, the river bank sediment/riparian soil boundary at an elevation of +13 ft NAVD88, and the surface sediment/subsurface sediment boundary at 30 cm bml. Contaminant masses passing through these boundaries into the Study Area are external loads.

These loading mechanisms are differentiated from the "sources" described in Section 4 in that they represent the combined estimated load from all Study Area sources for the corresponding pathway. A simplified conceptualization of the external loading pathways (loading terms) to the Study Area and internal transport processes within the Study Area is presented in Figure 6.1-1. The loading terms presented on that figure were determined in consultation with EPA (Integral et al. 2006, pers. comm.), and include the following:

- Upstream loading (via surface water, including suspended sediment load and sediment bedload)
- Stormwater runoff
- Permitted point source discharges (non-stormwater)
- Upland groundwater (plume transport to the river)
- Atmospheric deposition (to the river surface)
- Upland riverbank erosion
- Groundwater advection through sediments (partitioning from deep sediment, advecting to shallow sediments)
- Overwater releases.

Numerical loading estimates were generated for all of the above terms except for upland riverbank erosion (per agreement with EPA), sediment bedload (per agreement with EPA), and overwater releases. Contaminant releases from current and/or historical overwater activities (e.g., sandblasting, painting, material transfer, maintenance, repair, and operations at riverside docks, wharfs, or piers), discharges from vessels (e.g., gray, bilge, or ballast water), fuel releases, and spills are not considered quantifiable and are not addressed in this section. Releases of this nature are expected to have been more significant historically, prior to improved BMPs. While improved BMPs are likely to

have reduced the occurrence of overwater releases significantly, it is acknowledged that current and future releases could occur. Due to insufficient available information, no attempt is made in this report to predict and quantify such releases as a current loading term.

This section also presents loading estimates for groundwater advection through surface sediments (partitioning from surface sediment to surface water), which is an internal loading process within the Study Area. Internal transfer mechanisms are those that involve the transport of chemical mass from one media to another within the Study Area, as defined above, but which do not add new chemical mass to the Study Area. Internal fate and transport mechanisms include sediment resuspension/transport/ deposition, solid/aqueous-phase portioning, abiotic/biotic transformation and degradation, biological uptake, and groundwater advection through surface sediments (partitioning from surface sediment to surface water). Groundwater advection is the only internal process that is quantified here. The other internal processes may be very significant in the transfer of chemicals from abiotic media and to biota, and many of these terms will be quantified as part of the FS fate and transport modeling effort.

For all loading terms for which numerical estimates were generated, a range of estimates (central estimate and upper- and lower-bounds) is provided to give perspective on the uncertainty associated with a given pathway and chemical. The estimation approach for each term varies in nature and approach from qualitative to quantitative,[1] depending on the degree to which loading associated with a given transport pathway could be evaluated using available information. The assessments of external loading terms in Section 6.1 are intended to illustrate the estimated magnitude and variability in contaminant loads to the Study Area under "typical" conditions, i.e., in an average water year.[2] Assessment of year-to-year temporal variability was not the intent of this analysis.[3] The target scale of assessment of current loading rates is mass per year to the entire Study Area; however, in many cases, the data set supports (and calls for) calculation of loading estimates at higher temporal and/or spatial resolution. Where possible and relevant to understanding the system for the purposes of the RI,

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[1] The following definitions of qualitative, semi-quantitative, and quantitative assessments are applied to this discussion:

Qualitative – The attributes of the analysis are not measured/measurable. Discussions consider available information and examine general trends and relative effects.

Semi-quantitative – Some of the attributes of the analysis are measurable while others are not. General (or literature) non-Study-Area-specific data may be used in these assessments.

Quantitative – Most, if not all, of the attributes are measured and can be applied to the analysis. Some assumptions and modeling or calculation techniques may be used in these assessments.

[2] Because every water year is slightly different from the theoretical "average water year," the analyses include data collected during a range of environmental conditions. This variability is taken into account in the analyses to the extent possible. The approach to assessment of each term is discussed in detail in Appendix E.

[3] Extreme events are discussed for bedload in Section 6.1.1.2 to present additional information from the numerical modeling work used to generate the estimates for a typical water year. Such numerical modeling information is not available at this time to estimate extreme event loading for other loading terms.

these more refined (e.g., by river mile) loading estimates were generated and are presented and discussed.

Table 6.1-1 summarizes the load estimate quantification level and calculation approach for the estimated loading terms, including the approach for generating the range of estimates. Because of data limitations for some mechanisms and chemicals, the range of estimates does not always represent the full understanding of uncertainty in the estimate; therefore, additional discussion of variability and uncertainty in the estimates is provided after discussion of approach for each loading mechanism in the following subsections.

The loading rates presented in this section are *estimates.* They are subject to varying and sometimes considerable degrees of uncertainty that necessarily influence their utility as lines of evidence for the CSM (Section 10). In Section 10, this information will be considered along with nature and extent information for all media, understanding of the physical system, understanding of contaminant behavior, risk information from the baseline RAs, and the understanding of current and historical sources. As such, the present section focuses on simple presentation of the loading analyses and general, mechanism-specific findings and patterns.

Subsections 6.1.1 through 6.1.7 define each current external loading term,[4] identify and discuss the term's target IC list, note its targeted level of estimation (qualitative, semi-qualitative, or quantitative), describe the approach to assessment of the term's current loading rates, discuss the understanding of uncertainty in the estimates, and summarize and discuss any numerical estimates. Additional details of data sources, calculation approaches, and numerical results are presented in Appendix E. It is recognized that each loading term has a historical component that may be responsible for much of the observed sediment conditions in the Study Area, particularly in the subsurface. Very limited quantitative data are available to support estimates of these historical terms. Therefore, historical loading is discussed qualitatively in Section 6.1.8, and further analysis is deferred to Section 10 (CSM), where the locations and pathways of historical sources are assessed relative to the surface and subsurface sediment record. Finally, a detailed comparative assessment of loading terms is provided on a chemical-specific basis in Section 10 (CSM).

6.1.1 Upstream Loading

Upstream loading is defined as the mass rate of transport of a given contaminant into the Study Area at RM 11.8 via dissolved and particulate transport mechanisms. Upstream loading is subdivided into loading via the surface water column (dissolved and suspended solids fractions) and sediment bedload (rolling, sliding, and saltating of sediment grains). Per discussions with EPA during scoping of the Fate and Transport Model being developed for the FS, quantification of sediment bedloads into the Study

[4] Due to insufficient available information, no attempt is made in this report to predict and quantify overwater releases as a current loading term.

Area and associated contaminant transport was deemed unnecessary because the sediment bed of the LWR is mainly cohesive; hence, bedload transport processes are not significant components of upstream loading. Surface water loading is assessed in this section for typical flow conditions, as well as extreme observed and modeled high flow conditions.

While upstream loading terms are presented here simply as dissolved surface water and suspended particulate loads, it is important to recognize that these loads represent the combined input to the Study Area from a variety of loading processes in the upstream watershed. These inputs include upstream point sources, upstream stormwater runoff, upstream CSOs, upstream atmospheric deposition, upstream in-river sources, etc. Distinguishing these individual contributions to the combined upstream load is beyond the scope of this document.

6.1.1.1 Empirical Surface Water Loading Estimates

Upstream surface water loading for a given contaminant is defined as the mass rate of transport of that contaminant into the Study Area at RM 11.8 via surface water flow. This includes both dissolved and particulate (associated with suspended solids) fractions of the chemical mass. Empirical concentration and flow rate data from the site were used to generate a range of quantitative estimates of annual upstream surface water loading rates. Estimates for dissolved, particulate, and total loading rates are presented.

The surface water loading ICs listed in Table 6.0-1 were the focus of the calculations. This IC list was based on a combined list of the sediment, biota, and surface water ICs designated for discussion in Section 5 regarding in-river distribution, as well as contaminants to be assessed in the Fate and Transport Model under development for the FS and the CSM (Section 10). Chemicals from this combined set of ICs that were not analyzed in the RI surface water sampling program were removed from the final surface water IC list.

Upstream surface water loading at RM 11.8 was estimated based on the combined analytical data collected from the two nearest sampling transects: RM 16 and 11. In addition to upstream surface water loading at RM 11.8, loading rates were generated at the other RI surface water sample transect locations in the LWR: RM 16, 11, 6.3, 4, and 2. Estimated loads are also presented for the transect located in Multnomah Channel, near its upstream connection with the Willamette River at RM 3. These loading estimates are not representative of upstream loading, but were generated to provide additional insight into the nature of the contaminant load transported by surface water as it moves through and out of the Study Area. The sum of the estimated loads at Multnomah Channel and RM 2 represents the estimated load of contaminants leaving the Study Area.[5]

[5] Due to the complicating influence of water from the Columbia River on Willamette River flow volume and direction at the RM 2 and Multnomah Channel transect locations (described in Section 5.3), modeled flow data

The following subsections provide a summary of the data sets and approach used in the calculations, as well as a presentation and discussion of the findings. Detailed presentations of the data sets, data treatment, calculations, assumptions, and results can be found in the supporting Appendix E2.0.

6.1.1.1.1 Data Sets and Approach

Surface water loading rates were estimated based on Round 2A and 3A surface water chemical concentration data from transect sampling locations and USGS flow information from RM 12.8 (Morrison Bridge Station 14211720). Three surface water sampling events from the Round 2A sampling effort and four surface water sampling events from the Round 3A sampling effort provided the analytical data for the surface water loading calculations. Of these seven sampling events, four occurred during low-flow conditions (<50,000 cfs), two were during high-flow conditions (>50,000 cfs), and one was during a low-flow stormwater event (active runoff to the Study Area with river flow rate <50,000 cfs). Average discharge rates (recorded as cfs) for each event are based on 30-minute measurements collected by the USGS at the stream flow station located upstream of the Morrison Bridge at RM 12.8 (Station 14211720).[6] Total flow volumes and high-flow:low-flow volume fractions for the individual years and the 28-year average are presented in Table E2-1. Because the data sets compared well, the 28-year hydrograph was considered adequately representative for use as the basis for defining the high-flow:low-flow volume ratio for a typical year. Fifty-two percent of the total annual volume occurred during high-flow conditions and 48 percent during low-flow conditions, for a volume ratio of 1.07. The average annual duration of the low-flow period is 268 days, while high flows occur for a much shorter period of 98 days. The Round 2A and 3A surface water sampling events and daily hydrograph data for the years characterized by those events (2004 through 2007) are presented in Figures 5.3-2 through 5.3-5. Surface water sampling events superimposed on the 36-year average annual hydrograph are presented in Figure 5.3-1.

The distribution of Willamette River flows between Multnomah Channel and RM 2 were based on the results from the EFDC physical transport modeling effort (WEST 2006a). The results of this model provided the average daily flow rates at Multnomah Channel, RM 2, and upstream transects (modeled upstream transect flows were used for comparison purposes only). A detailed description of the use of these model flow results is presented in Appendix E2.2.1. The surface water data include particulate and

were used to estimate loads for these two transects. The load calculation approach is discussed in detail in Appendix E2.

[6] Note: The flow rate values presented here are daily mean stream flow measurements from the USGS National Water Information System, www.waterdata.usgs.gov. These values were taken from the USGS Web site on June 16, 2008, and are considered to be draft and subject to change by USGS, which may refine ratings and calculations as needed.

dissolved chemical concentrations.[7] Loading estimates for these fractions were combined to generate the total surface water loading estimates.

To differentiate surface water loads associated with high-flow and low-flow conditions during a typical flow year,[8] the first step in the analysis was to determine the fraction of a typical water year that is described by each flow regime. To do this, the average annual hydrograph (based on the 28-year period of record from 1975 to 2003) for the Study Area was evaluated relative to a 50,000 cfs threshold for low-flow versus high-flow conditions. Based on this, the average annual duration of the low-flow period is 268 days, while high flows occur for a much shorter period of 98 days. In terms of total annual flow volume, 52 percent is associated with the high-flow periods and 48 percent with the low-flow periods. Accordingly, a ratio of 52:48 was used to apportion the fractions of the typical annual hydrograph assigned as high flow to that assigned as low flow. This step is described in detail in Appendix E2.2.1.

Next, the surface water analytical data set was used to estimate representative concentration ranges for high-flow and low-flow conditions for each transect. This calculation step, described in detail in Appendix E2.2.2, involved "subaveraging" individual data points associated with a transect (e.g., NB, NS, east, middle, west, etc.) to resolve the data set to one concentration estimate per transect per sampling event. Using the subaveraged data, a set of minimum, mean, and maximum measured concentrations at each transect was generated for both high-flow and low-flow conditions.

Loading rates at each transect were then estimated as the product of the chemical concentrations and the flow volumes associated with the high-flow and low-flow portions of the hydrograph. For each transect, lower, central, and upper estimates of high-flow loading were estimated by multiplying the minimum, mean, and maximum concentrations, respectively, by the total annual flow volume estimated for high-flow conditions. Likewise, a range of low-flow rate loading estimates for each transect was estimated using the minimum, mean, and maximum of subaveraged concentrations and the estimated flow volume for low-flow conditions. The range of annual mass loading rate estimates were, in turn, generated by summing the fractional loading contributions estimated for high-flow and low-flow conditions at the given transect (i.e., the total annual central flow estimate was calculated as the sum of the central high-flow estimate and the central-low flow estimate).

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[7] A comparison of surface water particulate concentrations and physical characteristics of suspended sediment to sediment trap results was conducted to assess the adequacy of the surface water information for describing an entire water year. This assessment is presented in Appendix E2. The assessment indicated that the ranges in surface water suspended sediment concentration generally capture the range observed in the sediment trap data set.

[8] The analysis of the surface water data presented in Section 5.3, which broke down the data set by high- and low-flow conditions, yielded generally consistent concentration patterns and supported interpretation of results. Therefore, the loading analysis perpetuated this distinction to maintain this additional level of detail (as opposed to simply presenting annual loads) in the assessment of the surface water loading estimates.

There are no surface water sample results available from RM 11.8, which defines the upstream boundary of the Study Area. Therefore, high-flow and low-flow concentrations at RM 11.8 were estimated by combining data from RM 16 with selected data from RM 11. Because some of the surface water samples collected at RM 11 appear to have been influenced by one or more source areas of ICs between RM 11.8 and RM 11, some of the RM 11 results are not representative of water quality at RM 11.8. Prior to combining the data, the RM 11 data set was assessed for each IC to determine whether the data represented the same population of upstream data as that sampled at RM 16. This approach assumes that the surface water concentrations at RM 11.8 would be more similar to those at RM 16 than those at RM 11, recognizing that although there are additional sources between RM 11.8 and 16, the proximity of likely sources to the RM 11 transect are expected to have the larger effect. A comprehensive graphical and statistical comparison of the chemical data from RM 16 and 11 was conducted for each IC.[9] This involved plotting and evaluating data, running Mann-Whitney U statistical tests, and removing RM 11 sample results from the combined set where the evidence indicated the potential influence of a local source area. This analysis and all figures and statistical results are presented in detail in Appendix E2.2.2.1.

6.1.1.1.2 Uncertainty

Uncertainty associated with the surface water loading estimates is related primarily to the adequacy and representativeness of the analytical data set. The data sets are derived from grab samples, not time-weighted composites. Further, a limited number of samples were collected under a limited number of flow conditions. This prohibits a thorough understanding of temporal and flow variability in surface water quality and is an important source of uncertainty. The magnitude and direction (i.e., potential positive or negative bias on loading estimates) is unknown. Finally, additional uncertainty is associated with the RM 2 and Multnomah Channel loading results stemming from the use of modeled flow rates and the variable influence of the Columbia River in the lower reaches of the Study Area.

Regarding uncertainty in the suspended solids component of the surface water loading estimates, sediment trap data provide some additional perspective because they reflect longer sampling periods (four quarterly samples); however, for direct comparison, these samples are spatially limited (not representative of an entire transect) and mechanism-limited (not necessarily likely to equally capture particles of all sizes). The sediment trap results are discussed in comparison to surface water TSS in Appendix E2.4.1. This comparison suggests that the surface water TSS data reasonably approximate the longer-term sediment concentrations provided by the sediment traps, in spite of the snapshot nature of the surface water samples, lending some confidence to the use of the TSS data in the loading estimates.

[9] Data were available for RM 11 and RM 16 for all ICs except BEHP, which was only sampled at RM 11. For this chemical, upstream loads were calculated based on the RM 11 sampling results.

6.1.1.1.3 Findings

Upstream (RM 11.8) surface water loading estimates are presented on an annualized basis for all surface water loading ICs and are further differentiated for high-flow and low-flow periods of the annual hydrograph, as well as for the particulate and dissolved fractions. Surface water loading estimates at RM 16, 11, 6.3, 4, Multnomah Channel (RM 3), and RM 2 are also presented for select indicator chemicals—PCBs, PCDD/Fs, total DDx, and PAHs—to provide insight into patterns of mass transport of these chemicals in surface water. Loading estimates for all transects for the surface water ICs are presented in Appendix E2.3.

Annual Upstream Loading

Table 6.1-2 presents the range of total (dissolved plus particulate) annual upstream loading estimates (RM 11.8) for each surface water loading IC. The following paragraphs discuss the findings for each chemical (or chemical group) regarding patterns in total loading rate estimates, loads as a function of surface water flow regime (high flow versus low flow), and particulate-dissolved components of the loads.

Figure 6.1-2 presents total surface water loading estimates for total PCB congeners, selected individual PCB congeners, and PCB TEQ. Both the total PCBs and PCB TEQ loading estimates show higher aggregate loads during the 268-day low-flow period of the year (based on the 28-year average annual hydrograph), as compared to the 98-day high-flow period. On a daily basis, total PCB loads are higher during the high flow period than during the low-flow period. Total PCBs and PCB TEQ show significant contributions of particulate-associated concentrations to the total surface water PCB loads for most flow conditions (Figure 6.1-3), which is expected given the highly hydrophobic nature of PCBs.[10] These patterns in flow conditions and particulate/dissolved ratios are also generally apparent in the individual congener data sets. Total PCBs 81, 126, and 169 exhibited low detection frequency in the upstream surface water loading data set (19 percent, 40 percent, and 14 percent, respectively; see Table E2-5 in Appendix E); therefore, the loading estimates presented for these analytes are considered more uncertain than those for other ICs with higher detection frequency.

Total PCDD/Fs and TCDD TEQ (Figures 6.1-4 and 6.1-5) are primarily associated with particulate matter, which is also expected given the highly hydrophobic nature of these chemicals. For both total PCDD/Fs and TCDD TEQ, neither the aggregate 268-day low-flow load estimates nor the aggregate 98-day high-flow load estimates dominate the total annual loading; instead, the relative contributions to the annual load from high-flow and low-flow periods are comparable (Figure 6.1-4). On a daily basis, total PCDD/Fs and TCDD TEQ loads are higher during the high-flow period than during the low-flow period.

[10] In surface water, partitioning theory (often discussed in the context of groundwater or pore water) still holds; however, the total chemical mass in the dissolved state in surface water can be comparable to or greater than the total chemical mass in the sorbed state, even for very hydrophobic chemicals. This is because the mass ratio of water to solids (suspended) in surface water is much higher than in groundwater or sediment.

The results for DDx compounds (Figures 6.1-6 and 6.1-7) indicate a different pattern than those noted for PCBs and PCDD/Fs. While the loads are generally higher in the particulate fraction, as typically seen for the other strongly hydrophobic ICs, the patterns with surface water flow regime differ. For DDx compounds, the annual aggregate upstream load associated with the 98-day high-flow period is consistently higher than that associated with 268-day low-flow period of the year (based on the 28-year average annual hydrograph). On a daily basis, total DDx loads are higher during the high-flow period than during the low-flow period. Further, the 4,4'-isomers of the DDx components compose the majority of the total DDx upstream load, with DDT isomers being the greatest fraction, and DDD isomers being the smallest fraction of the total DDx.

Upstream surface water loads of total PAHs are significantly higher (approximately an order of magnitude) than total cPAHs (Figure 6.1-8). This indicates that PAHs in the water column are dominated by LPAHs rather than HPAHs, as the total cPAH sum includes the majority of HPAH compounds. This pattern is also reflected in loads for naphthalene (an LPAH) as compared to BAP (an HPAH). The cPAH BaPEq trends parallel total cPAHs and BAP trends. For both LPAHs and HPAHs, the annual aggregate load associated with the 98-day high-flow period is higher than that associated with the 268-day low-flow period of the year (based on the 28-year average annual hydrograph). On a daily basis, total PAH loads are higher during the high-flow period than during the low-flow period. The LPAHs (naphthalene), which have higher solubility than HPAHs, show higher fractions of dissolved as compared to particulate load (Figure 6.1-9). In contrast, the HPAHs (cPAHs, cPAH BaPEq, and BAP), which are more hydrophobic, show higher fractions of particulate as compared to dissolved load. It is important to note that total naphthalene exhibited low (10 percent; see Table E2-5 in Appendix E) detection frequency in the upstream surface water loading data set; therefore, the loading estimates presented for this analyte are considered more uncertain than those for other ICs with higher detection frequency.

Upstream loading rate ranges for BEHP and hexachlorobenzene are presented on Figures 6.1-10 and 6.1-11. The total annual BEHP load is almost exclusively associated with high-flow periods of the hydrograph, with three high-flow samples measured above detection limits and one detected low-flow sample. Total BEHP exhibited low (19 percent; see Table E2-5 in Appendix E) detection frequency in the upstream surface water loading data set; therefore, the loading estimates presented for this analyte are considered more uncertain than those for other ICs with higher detection frequency. For hexachlorobenzene, the high-flow contribution is also higher than the low-flow contribution, but only by approximately 15 percent for the central estimate. Hexachlorobenzene shows consistent fractions of particulate and dissolved contributions to the total load under all flow conditions, with the particulate fraction making up roughly 15 to 20 percent of the total load.

Upstream surface water loading rate estimates for the indicator non-DDx pesticides are presented on Figures 6.1-12 and 6.1-13. These figures show that, of these compounds,

dieldrin exhibits the highest annual upstream loads, whereas aldrin loads are comparatively very low. This difference may reflect the fact that aldrin, in comparison to dieldrin, degrades relatively rapidly in surface water by photochemical or microbial processes (discussed further in Section 6.2). Similar to the DDx pesticides discussed previously, all of the non-DDx pesticides show higher loads during high-flow conditions, with the exception of gamma-HCH, which shows approximately 60 percent of the annual load during low-flow conditions. In contrast to the DDx pesticides, however, the total surface water loads for the non-DDx pesticides are dominated by the dissolved fraction (Figure 6.1-13).

Figures 6.1-14 and 6.1-15 present the upstream surface water loading rate estimates for the indicator metals. The highest overall loading rates are observed for zinc and copper, two common elements. Nickel and chromium loads are the next highest and show similar loading rate estimates. Lead and arsenic show the next highest loading estimates, with mercury loads being the lowest and the result of only a few detections. Total mercury exhibited low (23 percent; see Table E2-5 in Appendix E) detection frequency; therefore, the loading estimates presented for this analyte are considered more uncertain than those for other ICs with higher detection frequency. For all of these metals, loading rates during high-flow conditions are greater than loading rates during low-flow conditions. Further, for the majority of the metals, the particulate fraction contributes more than the dissolved fraction to the total loading estimates, especially under high-flow conditions (Figure 6.1-15).

Estimated upstream total surface water loads for TBT are presented on Figure 6.1-16. (There is no presentation of dissolved versus particulate fractions for TBT because the surface water data set includes measurements of total concentrations only.) TBT in upstream surface water was detected only once, and that was during a low-flow sampling event (high-flow results were below detection limits); therefore, the loading estimates for low flow represent 100 percent of the annual load estimates (the load associated with the high-flow period was estimated to be zero per loading calculation data rules; i.e., non-detect results were set to zero). Generally, due to the low detection frequency (5 percent; see Table E2-5 in Appendix E) in the upstream surface water loading data set, the loading estimates presented for total TBT are considered more uncertain than those for other ICs with higher detection frequency.

In summary, regarding the relationships of loading to flow conditions, all of the surface water ICs, with the exception of PCBs and gamma-HCH, show higher upstream loading rates during high-flow conditions than during low-flow conditions, when aggregated over the 98-day high-flow period and the 268-day low-flow period (based on the 28-year average annual hydrograph). On a daily basis, loads for all of the ICs are generally higher during high flows than during low flows. Regarding overall patterns in dissolved and particulate fractions of total upstream loads, the particulate fraction represents the larger component for PCBs, PCDD/Fs, DDx pesticides, and metals. The dissolved fraction is the larger component for LPAHs, non-DDx pesticides, and hexachlorobenzene. In general, the ratios of particulate to dissolved mass loading for

all surface water loading ICs do not show large or consistent variations under different flow conditions (high flow versus low flow), possibly suggesting conditions of equilibrium or near equilibrium (discussed further in Section 6.2.2).

Surface Water Mass Transport within the Study Area

To evaluate spatial trends in surface water loading estimates moving downstream through the Study Area, loading estimates were plotted for each available transect for total PCBs, total PCDD/Fs, TCDD TEQ, total DDx, and total PAHs. These plots are presented in Figures 6.1-17 through 6.1-19. As mentioned above in the approach discussion, upstream loads at the RM 11.8 Study Area boundary were estimated based on combining RM 11 and RM 16 data. Further, the fraction of the Willamette River load leaving the Study Area at RM 2 and Multnomah Channel was estimated based on measured concentrations and modeled river flows, due to the hydrologically complex influence of water from the Columbia River on flows in the lower Willamette River. These transect estimates are distinguished with blue (RM 11.8) and orange (RM 2 and Multnomah Channel) symbols on Figures 6.1-17 through 6.1-19. The sum of the estimated loads at Multnomah Channel and RM 2 represents the estimated load of chemicals leaving the Study Area.

The total PCB plots (Figure 6.1-17) show that central estimates of surface water loading increase continually from RM 16 to 4. The upper and lower estimates show the same increasing trend. Comparison of the central estimate results from the upstream estimate (RM 11.8) and the from the Multnomah Channel and RM 2 transects indicate that the total PCB load leaving the Study Area is estimated to be greater than the upstream end of the Study Area. This observation of a loading increase from upstream to downstream is supported by the observed concentration increase between RM 11 and RM 4 in six of seven surface water sampling events, as discussed in Section 5.3.6.

Total PCDD/Fs and TCDD TEQ loads show patterns similar to PCBs (Figure 6.1-18), with increasing loads moving downstream through the Study Area; however the increase is not as steady, with comparable central loading estimates at RM 11 and 6.3. Comparison of the central estimate results from for RM 11.8 with the Multnomah Channel and RM 2 transects indicates that the total PCDD/F and TCDD TEQ loads leaving the Study Area are estimated to be greater than the upstream end of the Study Area. However, the RM 11 transect loads are comparable to the summed RM 2 and Multnomah Channel loads, suggesting that much of this loading increase occurs between RM 11.8 and 11. Consistent with the loading observations, total PCDD/F concentrations in surface water do not display consistent trends from upstream to downstream within the Study Area, as discussed in Section 5.3.7.

Total DDx pesticides (Figure 6.1-19, top plot) show fairly consistent loads between RM 16 and 11, with loads increasing in the Study Area between RM 11 and 4. Finally, total PAHs (Figure 6.1-19, bottom plot) show a continually increasing load between RM 16 and 4. As with PCBs and PCDD/Fs, the central estimates of DDx and PAH loads leaving the Study Area at Multnomah Channel and RM 2 are somewhat higher

than the loads estimated at the upstream end of the Study Area (RM 11.8). For both DDx and PAHs, this observation of a loading increase from upstream to downstream is supported by the observed concentration increase between RM 11 and 4 in six of seven surface water sampling events, as discussed in Section 5.3.8 (DDx) and 5.3.9 (PAHs).

A graphical summary comparison of loading terms is presented in Section 10.2 for the 13 CSM chemicals.

6.1.1.2 Modeled Sediment Fluxes

The 2009 FS HST model was used to evaluate the relative scale of the sediment fluxes to the Study Area over a range of flow conditions. As discussed previously, bedload is not quantified in the FS HST model because the physical CSM for the LWR assumes that bedload represents a relatively small fraction of the total sediment load entering the Study Area at RM 11.8. This is because of the LWR's morphology and the fact that its flows are regulated by upstream control structures. As noted in Section 3, the Study Area occupies the lower portion of the LWR where the river widens and has been deepened by dredging. The reach upstream of the Study Area, from Willamette Falls through downtown Portland, is generally narrower and faster moving, so suspended loads tend to be transported into the Study Area before settling out. In addition, dams at Willamette Falls (RM 26) and further upstream trap bedload moving downstream from the middle Willamette River to the LWR. The only significant tributary to the LWR below RM 26, the Clackamas River at RM 24.7, is a gravel-bed stream. Much of the bedload from the Clackamas River is likely deposited in the Willamette River well upstream of the Study Area—e.g., the main stem of the LWR just upstream and at Ross Island has several large depressions in the channel (see Map 3.1-6) that trap bedload materials moving downstream.[11] From a contaminant loading perspective, the significance of bedload is further reduced because this process transports relatively coarse-grained, non-cohesive, clastic sediments, which have a generally low organic carbon content and low surface-to-volume ratio. Therefore, associated contaminant concentrations are presumably much lower than concentrations associated with finer grained suspended sediment.

6.1.1.2.1 Modeled Sediment Flux

The FS HST model was used to predict sediment loads entering the Study Area from upstream at RM 11.8 under five different flow regimes. The flow regimes range from the 5th to 95th percentiles of flow conditions based on the LWR flow data recorded since 1972. The specific flow years modeled (starting on October 1 and running through September 30 of the year indicated) and the mean annual flows for those years are listed below:

- 5th Percentile Flow Year 2001 – mean flow 454 m^3/sec
- 25th Percentile Flow Year 1981 – mean flow 787 m^3/sec

[11] As discussed in Section 5.6, there are also several man-made borrow pits spread across the channel in the upper Study Area between RM 10 and 11, which also likely entrain bedload entering the Study Area.

- 50th Percentile Flow Year 1986 – mean flow 878 m^3/sec
- 75th Percentile Flow Year 1995 – mean flow 1,078 m^3/sec
- 95th Percentile Flow Year 1997 – mean flow 1,522 m^3/sec.

These flow year percentiles were developed for use in the fate and transport modeling effort. The Fate and Transport Model is being developed to support evaluations of contaminant fate, transport, and loading for the RI and FS. As noted above, the fate and transport modeling effort is ongoing and the results will be evaluated and reported separately.

The FS HST model estimates the daily-averaged flux of water and suspended sediment (cohesive and non-cohesive). Table 6.1-3 and Figure 6.1-20 (Petroni 2011, pers. comm.) summarize the modeled suspended sediment fluxes for each flow regime entering the Study Area at RM 11.8 and exiting the Study Area in the main stem of the LWR at RM 1.2[12] and in Multnomah Channel. Negative values in Table 6.1-3 indicate material leaving the Study Area. The averaged annual modeled flow estimates illustrate the relative percent increase is discharge out the Multnomah Channel at lower LWR flow levels. The lower portion of Table 6.1-3 shows the combined sediment fluxes exiting the harbor at RM 1.2 and in the Multnomah Channel and then tallies the modeled suspended and total sediment loads entering and leaving Portland Harbor for each flow regime.

The net flux of suspended sediment for all flow regimes modeled is positive, indicating that the harbor is generally a trap for material entering from upstream. Across all flow regimes, an average of 82 percent of the total modeled sediment flux entering the harbor at RM 11.8 passes through and exits at RM 1.2 and the Multnomah Channel. The 18 percent retained within the Study Area represents an estimated average annual net sediment accumulation of 277 million kg/year and reflects the net depositional character of the harbor.

6.1.2 Stormwater Runoff

This section presents the results of the calculated stormwater loading to the Study Area using stormwater and outfall sediment trap data collected as part of the RI/FS.

The chemicals listed in Table 6.0-1 as stormwater loading ICs were the focus of the calculations described in this section. The stormwater ICs were derived from the sediment, biota, and surface water ICs designated for fate and transport evaluation using the Fate and Transport Model and for discussion in the RI CSM. The stormwater IC list was further reduced based on analyte availability in the stormwater data set.

[12] The Study Area boundary is RM 1.9 but the FS HST model grid extends downstream to RM 1.2. This model boundary, as well as the downstream end of the Multnomah Channel model cell, were used as the downstream boundaries for these calculations.

Estimated stormwater loads were generated for each IC and Fate and Transport Model cell and for the entire Study Area. The following subsections present a summary of the data sets and approach used in the calculations, as well as a presentation and discussion of the findings.

Appendix E3.0 describes the detailed steps taken to calculate stormwater loading estimates, from data sources/data treatment to calculation approach. Appendix E3.0 also presents the complete results, as well as a brief discussion of the associated uncertainty. The results are summarized here and also discussed in Section 10.

6.1.2.1 Data Sources and Calculation Approach

The stormwater composite water and sediment trap data were collected in accordance with the Round 3A Stormwater FSP and Addendum (Anchor and Integral 2007a,b) and its companion document, the Round 3A Stormwater Sampling Rationale (Anchor and Integral 2007c) and analyzed in accordance with the QAPP Addendum 8 (Integral 2007b).

The stormwater sampling location rationale was developed in accordance with a commonly used approach of applying "representative" estimates of stormwater chemical concentrations for various land use types (Scheuler 1987). A land-use-based chemical load modeling approach was used to estimate loads across the entire Study Area. Chemical loading models use site characteristics (e.g., land use and percent impervious area) and land-use-specific loading rates to estimate overall loading into the receiving waters. This approach has been modified to better fit the unique data needs and land use characteristics of the Study Area, as well as the practical constraints for this sampling effort.

Loads to the Study Area are calculated based on composite water and sediment trap data collected from heavy industrial, light industrial, residential, parks/open space, major transportation, and non-representative locations. Twenty-seven stormwater outfalls were sampled within the Study Area to estimate stormwater loads. In general, three to five composite water samples and one sediment trap sample were collected at each stormwater sample site. As discussed in Appendix E3.5.1, pesticides were analyzed at a small subset of locations (8 stations) in composite water samples, but they were analyzed at nearly all locations (22 stations) in sediment trap samples. Due to the lack of representative composite water samples for pesticides, sediment trap data and the resulting statistics (e.g., central tendencies) used in loading estimates were substituted for composite water statistics for light industrial, parks/open space, residential, and transportation land uses, as well as for 1 of 3 non-representative locations that did not have composite water data (WR-147). Additionally, composite water data and the resulting statistics used in loading estimates were substituted for sediment trap statistics for 2 of 3 non-representative locations that did not have sediment trap data (OF-22B and WR-96). For most non-pesticide chemicals, load estimates for composite water are based on roughly 100 stormwater samples across all land uses and sites. For these non-pesticide chemicals, the ranges of numbers of composite water samples available

for loading estimates were 27 to 72 for heavy industrial, 10 to 16 for light industrial, 9 to 10 for residential, 2 to 3 for parks/open space, 7 to 9 for major transportation, and 3 to 5 for each non-representative location. For pesticides, load estimates are based on approximately 26 composite water samples (from 8 stations) and about 19 sediment trap samples (from 19 stations) across all land uses and sites. For the pesticides, the ranges of numbers of composite water samples available for loading estimates were 12 for heavy industrial, 4 for light industrial (from one station), 3 for residential (from 1 station), zero for parks/open space, zero for major transportation, and 3 to 4 for each non-representative location. (Note that composite water data were only used for loading estimates for the heavy industrial and non-representative locations.) The numbers of sediment trap samples available for pesticide loading estimates were 11 for heavy industrial, 3 for light industrial, 2 for residential, 1 for parks/open space, and 1 for major transportation, and 1 for non-representative locations.

As detailed in Section 4.4.1.2 and Appendix E3.0, "representative" samples from five general categories of land use (heavy industrial, light industrial, residential, major transportation, and parks/open space), as well as samples from "non-representative" locations, were included to obtain a practicable and sufficient data set to estimate stormwater loading to the Study Area. Non-representative sites are those sites with non-representative chemical sources that cannot be easily extrapolated from generalized land use measurements. Samples were collected from a subset of drainage basins/outfalls within each land use category in the Study Area. These locations were sampled by LWG and Port of Portland (Terminal 4) during two sampling efforts in the spring/summer of 2007 (Round 3A) and the fall/winter of 2007-2008 (Round 3B). One additional site (GE Decommissioning) was sampled by GE during the same time frame. Results from the GE investigation are also included in the overall LWG stormwater data set. In early 2008, the City of Portland collected three additional samples to supplement the residential data set, and these samples are included as well.

As a first step, the stormwater analytical data set was used to generate concentration ranges for each land use and non-representative site. Both stormwater composite water chemistry samples and sediment trap chemistry data were used to provide two independent means of estimating stormwater chemical loads.

Next, the stormwater runoff volumes draining to each Fate and Transport Model cell were calculated for each land use and non-representative location using the City of Portland's GRID model as summarized in Appendix E3.5.3. It was not possible to develop runoff volumes and stormwater load estimates for individual outfalls due to uncertainty of stormwater basin boundaries for many outfalls.

As explained in Appendix E3.5.1 and E3.5.2, loads were then estimated as a product of the calculated concentration estimates and the flow rate from the 50th percentile flow year to represent the central tendency (CT) of flow conditions. The annual mass loads were generated by adding the loading contributions from each land use and non-representative site for each Fate and Transport Model segment:

Annual stormwater chemical load (kg/yr) = heavy industrial stormwater chemical load (kg/yr) + light industrial stormwater chemical load (kg/yr) + residential stormwater chemical load (kg/yr) + parks/open space stormwater chemical load (kg/yr) + major transportation stormwater chemical load (kg/yr) + non-representative site stormwater chemical load (kg/yr)

The process for estimating stormwater loads, as well as all figures, maps, and statistical results, is presented in detail in Appendix E3.0.

6.1.2.2 Uncertainty

The primary sources of uncertainty in the stormwater loading estimates are the sample size and sampling period extrapolated to represent the composite conditions of a typical water year over the entire LWR runoff area. Specifically, data used to estimate the stormwater loads were collected during a total of 15 storm events, with each outfall sampled an average of three times. Sediment traps were left in place for three to seven months during two separate sampling periods. Due to the limited time span of sampling and the known variability of stormwater, these data should be considered to represent a "snapshot" of stormwater entering the Study Area during the sampling period. Additionally, particular records were peremptorily excluded from the working database due to various factors that have been identified by the Stormwater Technical Team. These specific exclusions and associated uncertainty are discussed in Appendix E3.0.

The methodology for calculating stormwater loading assumes that concentrations measured in individual sampled outfalls at non-representative sites are indicative of concentrations for all stormwater discharging from the site. This methodology has inherent uncertainty associated with it, as concentrations can vary significantly based on the physical characteristics of the drainage basins associated with the stormwater discharges. For example, if a drainage basin that was sampled drains a known upland source area, the concentrations measured in this discharge will be significantly higher than stormwater discharges at the remainder of the site. Thus, this example could overestimate stormwater loading for this site. Overall, however, the direction of any bias in the estimates created by these uncertainties is unknown.

Additionally, other more specific factors within this particular study's methods that may contribute to the uncertainty of the stormwater loading estimates are discussed in Appendix E3.0.

6.1.2.3 Summary of Findings

This section presents the findings of the stormwater loading analysis. Stormwater loading to the Study Area is presented and discussed for stormwater ICs using both the stormwater composite water and sediment trap data sets, where available, as two independent ways of estimating loads.

In addition to Study Area stormwater loads, loading estimates by Fate and Transport Model cell are presented for PCBs, PAHs, and total DDx to provide insight into patterns

of loading throughout the Study Area. Loading estimates for each IC and Fate and Transport Model cell for the entire Study Area are presented in both graphic and tabular format in Appendix E3.6.

Table 6.1-4 presents the range of annual stormwater loads to the Study Area for each stormwater IC for both composite water and sediment trap data. These ranges are also presented graphically on Figures 6.1-21 through 6.1-35 for each chemical group, including ranges of the annual load estimated using both composite water and sediment trap data. The findings regarding patterns of total loads estimates are discussed for each chemical group in the following paragraphs. Refer to the maps in Appendix E3.6 (Maps E3-1a–b through E3-32a–b) for a graphical representation of loads for each IC throughout the Study Area. Tables 6.1-5a–b present a percentage comparison of loads to the Study Area by land use and non-representative location for both composite water and sediment trap data.

For total PCBs, the loads estimated using composite water data are slightly higher than the sediment trap estimated loads (Figure 6.1-21). The estimated loading rate for total PCBs is highest for the heavy industrial land use category as compared to other land uses, although one non-representative location contributes the highest estimated load. However, uncertainty in the calculated loading rates from non-representative sites due to sampling only one outfall from the site and then extrapolating these concentration results to stormwater runoff from the entire site may bias these results (see Section 6.1.2.2). A comparison of loads of individual PCB congeners and total PCB TEQ are shown in Figures 6.1-22 and 6.1-23. Generally, composite water estimated loads for the various PCB components are slightly higher than the sediment trap estimated loads.

Stormwater loads for DDx pesticides are presented in Figure 6.1-24. The results for these compounds show that the composite water estimated loads are generally within the range of loads calculated from the sediment trap data. The estimated annual loads for total DDX is highest for the heavy industrial land use category as compared to other land uses; however, the highest estimated loading rates are from non-representative sites in Basin 20 (RM 6.8 to 7.4W). It should be noted that uncertainty exists in the calculated loading rates from non-representative sites because only one outfall was sampled (from the former DDT process area) and these concentration results were then extrapolated to stormwater runoff from the entire site, thereby biasing the loading calculation. The estimated loads for non-DDx pesticides are highest for the heavy industrial land use category. Stormwater loads for non-DDx pesticides are presented on Figure 6.1-28. Generally, composite water estimated loads for non-DDx pesticides are typically higher than the sediment trap estimated loads.

For total PAHs, annual load estimates using composite water data compared well with estimates using sediment trap data (Figure 6.1-25). The estimated load for total PAHs is highest for the heavy industrial land use as compared to other land uses, with four non-representative locations contributing a substantial portion to the total stormwater load. However, uncertainty in the calculated loading rates from non-representative sites

due to sampling only one outfall from the site and then extrapolating these concentration results to stormwater runoff from the entire site may bias these results (see Section 6.1.2.2).

Stormwater loads for BEHP are presented on Figure 6.1-26. BEHP annual loads estimated using composite water data are higher than those generated using sediment trap data. The highest estimated loading rates for BEHP are from the heavy industrial land use areas.

Stormwater loads for hexachlorobenzene are presented on Figure 6.1-27. The results for hexachlorobenzene indicate that the sediment trap estimates are within the range of the composite water estimates. The highest estimated annual loads for hexachlorobenzene are from the heavy industrial land use areas.

Stormwater loads for non-DDx pesticides (aldrin, dieldrin, gamma-HCH, and total chlordanes) are presented on Figure 6.1-28. For each of these chemicals, the composite water annual load estimates were higher than estimates developed using sediment trap data.

Stormwater loads for metals are presented on Figure 6.1-29. Typically, for metals, the composite water load estimates were slightly higher than estimates developed using sediment trap data. The highest overall estimated loads are observed for zinc, copper, and lead. Chromium, arsenic, and nickel have the next highest loads, and of the metals evaluated, mercury has the lowest. The highest estimated annual loads for metals are from the heavy industrial land use areas.

Stormwater loads to the Study Area are presented by river mile for total PCBs, total PAHs, and total DDx pesticides in Figures 6.1-30 through 6.1-35.

A graphical summary comparison of stormwater discharges to other loading terms is presented in Section 10.2 for the 13 CSM chemicals.

6.1.3 Permitted Point Source Discharges

Point source permitted non-stormwater discharges to the Study Area include NPDES-permitted discharges from commercial, industrial, private, and municipal outfalls or operations. This section presents the results of semi-quantitative estimation of the current annual mass load of chemicals from these outfalls to the Study Area. The details of data compilation and loading estimation are presented in Appendix E4.0.

Both Oregon DEQ general and individual NPDES permits were considered in this evaluation. Active NPDES permits inside the Study Area were located using Oregon DEQ's Facility Profiler 2.0[13] and the DEQ Wastewater Permits Database[14] was used to query the permit file numbers. As of this writing, there are 14 NPDES wastewater

[13] Oregon DEQ's Facility Profiler 2.0: http://deq12.deq.state.or.us/fp20/
[14] DEQ Wastewater Permits Database: http://www.deq.state.or.us/wq/sisdata/sisdata.asp

permitted discharges in the Study Area listed as either Individual or GEN 15A Permits. Map 6.1-1 shows the facility locations for these 14 permits. Note that this analysis is specifically limited to permitted wastewater discharges to the Study Area and does not represent stormwater discharges (included in stormwater loading term analysis; see Section 6.1.2) or other types of point sources.

Permitted direct discharge loading analyses were based on water chemical concentration data and discharge/flow data in discharge monitoring reports (DMRs), where available. These data were available for the following 10 of the 14 NPDES wastewater permitted discharges:

- EOSM
- Kinder Morgan/Portland Bulk Terminal 4
- Koppers Inc.
- Starlink Logistics, Inc.
- Siltronic Corporation
- ARCO Products Company
- Kinder Morgan Liquid Terminals
- Equilon Enterprises
- Pinnacle Condominium Complex
- Univar USA.

The remaining four NPDES wastewater permitted discharges listed below were not included in the loading calculations due to insufficient data for calculations and are shaded orange in Map 6.1-1:

The facilities and the reasons they were not included are:

- Ash Grove – No flow or chemical data reported
- Columbia River Sand and Gravel – No flow data reported and no chemical analysis required (only TSS and turbidity monitored)
- Vigor (Cascade General) – No flow data reported on DMRs
- Hoyt Street Properties – No flow or concentration data reported.

The discharge information from these sites would be expected to increase the upper and lower end estimates of total loading to the Study Area for the chemicals included in their permits. The lack of data for these facilities is not expected to represent a significant loading data gap for any parameters.

Ranges of loading estimates were generated by considering the DMR discharge flow rates and chemical concentration data for all IC-listed chemicals. Because of limited analyte lists in the DMRs and the permits, data for some IC-list parameters were not available for all facilities. Additionally, several IC-list chemicals were never monitored at any of the facilities. The results are summarized in Table 6.1-6 for the subset of IC chemicals for which data were available.

While there is uncertainty associated with the annual estimates for this loading term, the findings are expected to be reasonably representative of the relative significance of this pathway (as defined above) for current loading of ICs to the Study Area. The primary source of uncertainty in these estimates is the limited monitoring records available for many sites. It should be noted that there are four sites that could not be included in this assessment due to lack of information. If there is flow related to these permits, then discharge information from these sites would be expected to increase the upper and lower end estimates of total loading to the Study Area for the chemicals included in their permits. It should also be noted that this analysis is specifically limited to permitted wastewater discharges to the Study Area and does not represent stormwater discharges (included in stormwater loading term analysis; Section 6.1.2) or other types of point sources.

Review of these results indicates first that only a few of the analytes on the combined loading IC list are presented in the DMRs (for one or more permit, results are presented for DDT, select PAHs, TPH, select metals, select VOCs, and cyanide). For all of the parameters analyzed, the estimated range of results is narrow—ranging over a factor of 5. While flow volumes are relatively large for some dischargers (total permitted discharge volume is estimated to be only slightly less than stormwater runoff), the concentrations ranges are low, and the resulting loads are generally low. Because of limited volume and low chemical concentrations, permitted point source discharges were not found to be a primary source of ICs to the Study Area for those facilities and parameters for which data was available. Overall, it is expected that this loading term, as defined and assessed here, is not currently a primary source of ICs to the Study Area since permitted discharges are regulated and monitored. A graphical summary comparison of permitted point source discharges to other external loading terms is presented in Section 10.2 for the 13 CSM chemicals.

6.1.4 Atmospheric Deposition

Chemicals present in the atmosphere as a result of emissions from stationary sources (e.g., industrial smokestacks), mobile sources (e.g., vehicle emissions) and non-point sources (e.g., fugitive dust) produce a load to the Study Area through the processes of dry and wet deposition. Further, persistent chemicals can travel long distances through the atmosphere from other parts of the world. Dry deposition refers to the deposition of air pollutants from atmospheric suspension in the absence of precipitation. Wet deposition refers to deposition of air pollutants from atmospheric suspension via liquid and/or frozen precipitation.

The following subsections present the approach and data sources applied to generate semi-quantitative estimates of the annual loading of selected analytes to the Study Area via dry and wet atmospheric deposition. Air deposition loading estimates presented here focus on dry and wet deposition directly onto the water surface of the LWR within the Study Area. Atmospheric deposition to land in the Study Area watershed, which could subsequently be transported to the Study Area via stormwater runoff, is captured, though not distinguishable from other sources, in the stormwater loading assessment (Section 6.1.2). A qualitative discussion of atmospheric deposition to land in the Study Area watershed is provided in subsection 6.1.4.3.2.

The IC list for atmospheric deposition loading is presented in Table 6.0-1. This list was limited to those chemicals on the combined IC loading list for which data were available to support the atmospheric loading estimates. The detailed data sets, methodologies, and results for dry and wet deposition loading to the Study Area water surface are presented in Appendix E5.0. As with other loading term estimates, the atmospheric deposition estimates are presented as a range representing the relative uncertainty, as discussed further below.

6.1.4.1 Data Sets and Approach

Atmospheric deposition is the sum of both dry and wet deposition loads. Under conditions of no precipitation, gases and particles are deposited to the ground or river surface in a process known as dry deposition. Dry deposition is driven by the gravitational force on the particulate matter and the gas aerosol. Numerous studies have been performed to characterize the dry deposition of various chemicals onto a variety of ground surfaces. EPA has conducted a review of many of these studies and concluded that dry depositions of both particulate matter and gases will contribute to the chemical concentrations in soils and surface water bodies (EPA 2005a). During precipitation events, gases and particles can be scavenged by rain droplets, frozen precipitation elements (freezing rain or snow), or fog droplets that deposit to the surface. This latter process is known as wet deposition. The following subsections briefly describe the data sets and approach applied to assess dry and wet deposition loading to the Study Area; additional detail on the calculations and data sources is provided in Appendix E5.0.

6.1.4.1.1 Dry Deposition to the River Surface

Atmospheric dry deposition to the Study Area was estimated semi-quantitatively based on an assumed deposition velocity, Study Area-specific and non-local air concentration monitoring data, and the Study Area surface water extent. For a given analyte, dry deposition loading (kg/yr) to the Study Area can be calculated as the product of the air concentration (mass/volume), the deposition velocity (length/time), and the surface area of the Study Area (length^2). The rate of chemical deposition to a surface (deposition velocity) is a function of atmospheric turbulence, properties of the chemical species, and the relative reactivity of the species with the receiving surface (Seinfeld and Pandis 1998). Where available, Study Area-specific or local ambient air concentration data were used. For those chemicals for which local sampling data were not available, concentration values from publicly available data sources, including DEQ and EPA,

were used. The assumptions and data sources applied to determine these factors are presented in detail in Appendix E5.0. In summary, for dry deposition, local information[15] was used in dry deposition calculations for all the metals, BAP, naphthalene, TPH (diesel), total PCBs, hexachlorobenzene, total PAHs, and total cPAHs; exclusively external data sources were used for dry deposition estimates for the rest of the atmospheric deposition ICs. A range of estimates was generated for the dry deposition loading fraction of the total atmospheric load. This range was based on the range of ambient air concentration results compiled.[16] Specific effort was made to analyze the local monitoring and modeling data for BAP and naphthalene to ensure the representativeness of the data values for dry deposition loading over the river surface (see Appendix E for details).

6.1.4.1.2 Wet Deposition to the River Surface

Wet deposition describes the process by which chemicals in particle and gas phases are scavenged from the atmosphere (in cloud and below cloud) by liquid or frozen precipitation and transported to the earth's surface. Wet deposition flux can be modeling numerically; however, the most reliable way to estimate wet deposition is to collect precipitation in suitable samplers, measure the chemical concentrations, and calculate the deposition flux corresponding to the sampling period (Reinfelder et al. 2004). Subsequently, the total annual wet deposition loading is calculated by multiplying the deposition flux by the total area of the Study Area. Unfortunately, such data are limited. From the IC list, Study Area-specific wet deposition monitoring results were only found for total PCBs (MWH 2008) and mercury. In the MWH (2008) study, wet deposition data were collected from three monitoring stations within the Study Area for a two-month sampling period spanning May through June of 2007. The MWH (2008) study reported wet deposition loading rates calculated from the monitoring concentration data (taking into consideration the field blank values). For mercury, findings from Hope (2005) were considered for comparison with estimates based on the New Jersey Atmospheric Deposition Network (NJADN) data (Reinfelder et al. 2004; see next paragraph). Briefly, the Hope (2005) study used precipitation monitoring data from Oregon Mercury Deposition Network sites (one site near Beaverton and one site near the southern end of the basin), and found wet deposition estimates comparable to those generated here by the NJADN ratio approach described in the following paragraph.[17] In summary, for wet deposition calculations, local

[15] Local is defined here as monitoring data or modeling results for Portland, Oregon or Multnomah County, Oregon.

[16] For BAP, a maximum value of 0.32 µg/m^3 was determined to be an outlying value among the values from the LASAR data based on statistical analysis and was excluded from the calculation; an average value of 0.19 µg/m^3 was also excluded for the same reason. For naphthalene, the following values were excluded from calculations based on statistical analysis: 2.16 µg/m^3 as one of the maximum values, 1.87 µg/m^3 as an average value, and 1.55 µg/m^3 as a minimum value. Please see Appendix E for more details.

[17] Hope (2005) calculated dry, wet, and total mercury loading rates to surface water for the entire Willamette River basin (398,000,000 m^2). When scaled down to the sub-area of the basin represented by the Study Area (8,791,735 m^2, 2 percent of the open water area estimated by Hope), Hope estimates a total atmospheric mercury load of 0.08 kg/yr. This result is slight lower than, but comparable to, the lower mercury load (0.11 kg/yr) presented here.

information was used for mercury (Hope 2005) and total PCBs (MWH 2008), and non-local/modeling data sources were used for all other chemicals.

For the ICs other than PCBs, which lack Study Area-specific precipitation monitoring results, the monitoring results from NJADN (Reinfelder et al. 2004) were used, corrected by the ratios of 1) total atmospheric concentrations between Portland, Oregon and Jersey City, New Jersey (where available in both), and 2) total annual precipitation between Portland and Jersey City. This approach of scaling NJADN data sets to develop wet deposition loading estimates generated only one estimate (not a range of estimates) for each IC of annual wet deposition loading to the Study Area because only the average values were reported from the NJADN study. The detailed methodology, data sources, loading results, and associated uncertainties are presented in Appendix E5.0. Uncertainty and results are also discussed in Section 6.1.4.2 and 6.1.4.3, respectively.

6.1.4.1.3 Total Deposition to the River Surface

For each IC, the total deposition loading to the Study Area was estimated simply by summing the dry deposition and wet deposition loading estimates. Since only central estimates could be generated for wet deposition loading, the ratio of the central estimate for wet deposition to the central estimate for dry deposition was assumed to be representative of the ratios across the range of wet deposition loading estimates. From this, upper and lower range estimates were generated for wet deposition for use in estimating the total deposition range. Where wet deposition data were inadequate to allow for estimation of even a central estimate, total loads were assigned based on the dry deposition estimates. Wet deposition estimates were unavailable for PCB TEQ, TCDD TEQ, 4,4'-DDE, 4,4'-DDT, naphthalene, total PAHs, DRH, hexachlorobenzene, aldrin, and dieldrin. The estimates are still considered to be useful, however, based on the relatively low contribution of wet deposition to the total estimates for similar chemicals: total DDx (<2 percent), BAP (~10 percent) and total cPAHs (~21 percent), and total chlordanes (~16 percent). This source of uncertainty is noted in the following subsection.

6.1.4.2 Uncertainty

The lack of the Study Area-specific, analyte-specific, and temporally proximal data inputs for many of the ICs places significant uncertainty on the estimates for the atmospheric deposition loading term. Specifically, local data were available only for metals, BAP, naphthalene, cPAHs (modeled), total PAHs (modeled; based on 16 individual PAHs), hexachlorobenzene, DRH, and total PCBs (modeled) for dry deposition calculations; for wet deposition calculations, local data were available only for mercury and total PCBs (limited sampling period). In the case of the atmospheric deposition loading estimates, the presented range of estimates (lower, central, upper) is not expected to fully capture or represent the uncertainty associated with this term, due to significantly limited local empirical data. The uncertainty varies by IC, and is discussed qualitatively (noting data sources and findings relative to empirically estimated terms) in Section 10.

For dry deposition loading estimates, the major uncertainties are as follows:

- The limited available local atmospheric concentration data
- The necessarily simplified calculation methodology
- The uncertainty associated with selection and uniform application of a deposition velocity.

For wet deposition loading estimates, the major uncertainties are as follows:

- The extremely limited local wet deposition monitoring data (data only found for PCBs [only a partial year of sampling record available] and mercury)
- The uncertainty associated with application of precipitation correction factors to allow for use of NJADN data.

In summary, primarily due to the limited availability of local atmospheric concentration and precipitation concentration monitoring data, atmospheric deposition to the river surface is expected to be one of the most uncertain loading terms. The direction of any bias in the estimates created by these uncertainties is unknown. Fortunately, deposition to the watershed and subsequent runoff to the river is captured in the empirical stormwater runoff data set and stormwater loading estimates (discussed in Section 6.1.2).

6.1.4.3 Findings

This section presents the findings from the semi-quantitative estimation of atmospheric deposition to the river surface described above. A qualitative discussion of atmospheric deposition to the watershed is also provided.

6.1.4.3.1 Atmospheric Deposition to the River Surface

Table 6.1-7 presents the estimated ranges of annual total atmospheric deposition to the river surface for the entire Study Area. Figures 6.1-36 through 6.1-40 graphically present the estimated ranges of annual loads for dry deposition, wet deposition, and total atmospheric deposition to the Study Area for each IC chemical group.

PCBs and TCDD TEQ – The estimated ranges of dry, wet, and total deposition for total PCBs and TCDD TEQ are presented in Figure 6.1-36. For total PCBs, the dry deposition fraction of the annual load represents the majority of the total annual loading estimate, with only less than 0.5 percent of the load attributed to wet deposition. No wet deposition data were available for PCB TEQ or TCDD TEQ estimates.

Pesticides – The estimated ranges of dry, wet, and total deposition for pesticides are presented in Figure 6.1-37. The total annual loads for pesticides are dominated by the dry deposition load estimates; however, it should be noted that wet deposition estimates were only available for DDx and total chlordanes. Of these, wet deposition composed 2 percent and 16 percent of the total, respectively (see Appendix E, Table E5.2). Further,

it should be noted that the total DDx estimates based on NJADN estimates are lower than the 4,4'-DDE and 4,4'-DDT estimates based on ATSDR ambient concentration estimates. This difference reflects the range of uncertainty in the various data sources. In light of this uncertainty, total DDx estimates to be used in the CSM discussion in Section 10 will reflect the combined ranges of the estimates (lower = 0.0068 kg/yr, central = 0.17 kg/yr, and upper = 0.21 kg/yr).

PAHs – The estimated ranges of dry, wet, and total deposition for PAH ICs are presented in Figure 6.1-38. The total annual loading estimates are significantly higher for naphthalene than BAP (10 times for upper value, 3 times for central, and about the same for lower value), suggesting dominance of the LPAH fraction. Further, for all PAHs, the dry deposition fraction of the annual load represents the majority of the total annual loading estimate, with only a very small fraction of the load attributed to wet deposition.[18] (Total PAH atmospheric loads are based on 16 PAHs from Oregon EPA National Air Toxics Assessment data [EPA 1996], which includes all of the Study Area PAHs except for 2-methylnapthalene.) The PAH loading estimates are considered to be highly uncertain based on comparison with other loading term estimates, as discussed in Section 10.2. Furthermore, statistical analysis EPA LASAR data for BAP and naphthalene indicated some data values are out of the statistical ranges that are suitable for atmospheric loading calculations, therefore, the total PAHs values could be affected by the outliers.

Diesel-Range Hydrocarbons and Hexachlorobenzene – The estimated ranges of dry and total deposition for DRH and hexachlorobenzene are presented in Figure 6.1-39. No wet deposition data were available for these chemicals. Furthermore, as discussed in Appendix E, no data to support estimates of dry, wet, or total atmospheric deposition rates were available for other TPH fractions (i.e., gasoline-range and residual-range hydrocarbons).

Metals – The estimated ranges of dry, wet, and total deposition for metals are presented in Figure 6.1-40. Lead, zinc, and copper showed the greatest total annual loading estimates by atmospheric deposition. For the metal ICs assessed, the dry deposition loading contribution to total annual deposition was greater than the wet deposition contribution, with the exception of mercury, which exhibited seven times greater annual deposition by wet deposition. For the other metals, while dry deposition estimates were greater than wet deposition, the dry deposition estimates were all within a factor of 10 of the wet deposition estimates, suggesting both mechanisms are important to the overall load.

[18] Wet deposition data were not available for total PAHs based on Oregon EPA NATA data (EPA 1996) for direct calculation of wet loading estimates; however, a closer look at the NJADN data set suggests that wet deposition is not expected to be a significant fraction of the total deposition for this chemical set. Wet deposition data were available from the NJADN study for a total based on 36 PAHs. Analysis of that New Jersey data shows that wet deposition loads are 3 orders of magnitude lower than dry deposition loads. Similarly, analysis of the 13 Study Area PAHs included in the New Jersey data set of 36 also shows that wet deposition loads are 3 orders of magnitude lower than dry deposition loads.

Comparison between these estimated atmospheric annual loads to the surface water and loads estimated for other loading mechanisms is presented on a chemical-by-chemical basis in Section 10.2.

6.1.4.3.2 Atmospheric Deposition to the Watershed

Chemicals that are deposited via atmospheric deposition to soils and impervious surfaces in the Study Area watershed may subsequently be transported to the Study Area via stormwater runoff. In general, for surface water bodies with relatively smaller watershed areas (e.g., the Great Lakes) compared to water surface area, the total atmospheric deposition loading to the surface water is greater than the deposition loading to the watershed (Steuer 1995). But for a riverine system such as the LWR, with small surface water areas relative to the contributing watershed, atmospheric deposition to the watershed plays a greater role.

The available literature indicates that the relative importance of the atmospheric deposition loading term, relative to other loading terms, varies by site and by chemical. Some studies found atmospheric deposition to the watershed to be a significant source to the surface water bodies. For instance, atmospheric deposition was found to be the dominant source term for total PCBs to the North and Baltic Seas (Struyf and Van Grieken 1993; Wania et al. 2001) and for HCH to the North Sea (Struyf and Van Grieken 1993). A recent study performed by Sun et al. (2007) in the Great Lakes region correlates average gas-phase atmospheric PCB concentrations with local population size, suggesting a strong urban source of atmospheric PCBs. Likewise, Motelay et al. (2006) found atmospheric deposition to impervious surfaces to be the most important source of PAHs to the urbanized Seine River basin near Le Havre, France. Further, one of the most recent systematic monitoring studies (the NJADN) found that direct (dry, wet, and gaseous air-water exchange) and indirect (runoff) atmospheric deposition are of major importance to the accumulation of certain elements (e.g., mercury) and major nutrients in surface water ecosystems (Reinfelder et al. 2004). Findings from a separate, locally relevant study led by Hope (2005) of Oregon DEQ produced loading rate estimates for mercury comparable to those from the NJADN study.

Other studies found atmospheric deposition to the watershed to be less significant as a source of chemicals to surface water. A study of numerous urban U.S. streams (not including the Willamette River) evaluated the relative importance of different non-point sources of VOCs to total loading, finding that atmospheric deposition was of secondary importance for VOCs compared to the loading from urban land sources (Lopes and Bender 1998).

Because of the complexity of the fate and transport of chemicals via stormwater runoff, a simple application of the flux rate estimated for deposition to the water surface is not appropriate for estimating loads to the Study Area from atmospheric deposition to the watershed. Chemicals deposited in the watershed surfaces are subject to a number of loss mechanisms outside of runoff transport, including leaching, degradation (biotic and abiotic), and volatilization (EPA 2005a). Further, it is difficult to appropriately estimate

the amount of deposited chemical mass that would be transported by runoff, and even more difficult to determine how much of that entrained chemical mass would be transported to the Study Area surface water given the complexity of routing and settling, etc. along the pathway. Other studies (Deletic et al. 1997; Grottker 1987) highlight the complexity of quantitatively estimating the relative contribution of atmospheric deposition to surface water bodies. These studies note that such estimates require a detailed understanding of the geochemical process and transport fluxes specific to the urban watersheds.

Keeping in mind the complexity noted above, the only empirical information available to assess the atmospheric contribution to the stormwater load is present in the stormwater data set. While many areas sampled as part of the LWG stormwater program have IC sources other than atmospheric sources, it could be assumed that samples collected from open space areas (and possibly residential areas, depending on the IC) represent primarily atmospheric deposition sources. All target ICs for stormwater loading were detected in stormwater runoff in water and/or sediment trap samples in all sampled open space land-use type locations except 4,4' DDD, total DDD, aldrin, dieldrin, gamma-HCH, hexachlorobenzene, naphthalene, PCB 081, PCB 126, PCB 169, and total chlordanes. PCDD/Fs and TPH were not sampled in stormwater runoff for any land-use type. A more rigorous assessment of the stormwater data set is not warranted, given the complexities/variables of runoff routing, adsorption of chemicals to varying surfaces, stormwater controls, etc. These variables confound the utility of a direct comparison of open-space runoff to other land-use type runoff for the purposes of assessing atmospheric deposition contributions. A discussion of stormwater data by land use type is presented in Section 4, and the stormwater annual load estimates by land use type were presented in Section 6.1.2.

6.1.5 Upland Groundwater Plumes

Upland groundwater plumes flowing toward the river are a possible source of chemicals to the in-river sediments, TZW, and surface water in the Study Area. This section presents the approach, data sources, and findings of a quantitative estimation of the loading of chemicals to the Study Area from upland groundwater plumes. Empirical seepage rate and TZW concentration data information from the nine GWPA study sites (see Appendix C2 for detailed discussion of site selection and GWPA data interpretation) were applied to generate an estimated range of annual loads for the individual study sites.[19] There may be additional sites that lack upland groundwater data but have complete groundwater pathways; however, such sites have not been identified or assessed.

[19] It is acknowledged that these Study Area loading estimates are based on empirical information from only nine study sites, and it is possible that other sites will be identified that have a complete pathway for upland groundwater plumes to the Study Area. The groundwater pathway site selection process was designed to identify all sites with a reasonable likelihood of exhibiting a complete transport pathway for upland groundwater plumes to the Study Area. Detailed discussion of the groundwater pathway site selection process is presented Appendix C2.

The chemicals listed in Table 6.0-1 as upland groundwater plume loading ICs were the focus of this loading assessment. This IC list was based on the TZW ICs designated for discussion in Section 5 regarding in-river distribution (with the exceptions of the localized source chemicals of TPH, cyanide, perchlorate, and Silvex), as well as chemicals to be assessed in the Fate and Transport Model under development for the FS and the RI CSM presented in Section 10.

It should be noted that, in order to generate estimates for this loading term, a simplifying assumption was applied that is recognized to not be reflective of actual conditions in all areas/for all ICs. Specifically, these estimates assume that observed TZW concentrations are entirely attributable to upland groundwater. In areas where there are both upland groundwater plume and sediment sources of chemicals, chemicals detected in TZW samples may be partly or wholly attributable to chemicals originating in sediment solids (partitioning into pore water). For certain chemicals (e.g., redox-sensitive metals, petroleum-related hydrocarbons, etc.), as discussed in Appendix C2, differentiation of the origin of chemicals present in the pore water in areas with groundwater discharge and upland groundwater plumes was often not possible with the available information. In such instances, the estimates of groundwater plume loading are expected to be redundant with advective loading estimates in the specific TZW study areas. (Advective loading estimates based on equilibrium partitioning assumptions and sediment concentrations are presented in Section 6.1.6.) This redundancy was recognized and accepted in this analysis to allow for assessment/approximation of each of the terms and comparison of the relative, Study Area-wide effects. (Note that loading estimates for the Fate and Transport Model will address loading from upland groundwater plumes and advective loading from groundwater discharge through sediments somewhat differently. The model simulates the transport of chemical within, and out of, the sediment bed via processes such as advection due to movement of groundwater, diffusion, and dispersion; this transport includes partitioning. In the specific areas where there are contributions from upland plumes, an upland plume loading term is specified based on available TZW concentrations and flux estimates from filtered trident and peeper data; this additional mass is subject to the same transport processes and partitioning within the bed.)

The following subsections present a summary of the data sets and approach used in the upland groundwater plume loading calculations, as well as a presentation and discussion of the findings. Detailed presentations of the data sets, data treatment, calculations, assumptions, and results are presented in the supporting Appendix E6.1.

6.1.5.1 Data Sets and Approach

Estimates of groundwater plume chemical loading to the Study Area are based on site-specific identification of potential plume discharge zones offshore of the nine TZW study sites, measured concentrations of ICs in TZW, and measured groundwater discharge rates in potential plume discharge zones. The following data sources were used to determine these terms:

- Twenty-eight flow zone areas identified offshore of the nine TZW study sites were used to group data sets for the calculations. These flow zones are presented with discussions supporting the interpretations in Appendix C2. The zones are also presented in Appendix E in support of the detailed approach discussion presented in Appendix E6.1.
- Measured shallow TZW chemical concentrations from 150 sample locations at the nine study sites were applied to the calculations. These samples represent the complete TZW data set for the sample depth interval from 0 to 38 cm bml (see Map 2.2-6). The sampling methods used to produce this data set include small-volume peeper, Trident, and Geoprobe samplers. Both unfiltered and filtered (where available) results were included in the evaluation. These TZW analytical results were presented in detail and discussed in Section 5.4.
- Seventy-seven seepage meter measurements from the 28 flow zone areas were used to estimate groundwater flux for each zone. This seepage rate data is presented in Appendix C2.

As a first step, Thiessen polygons[20] based on the TZW sampling locations were generated within each flow zone based on the TZW sampling locations to assign an area to each sample. This step was necessary to support area-weighting of each TZW analytical result. Loading estimates were prepared for each flow zone area by summing the estimated loads for each of the sample polygons within the flow zone, using the following general equation:

$$Load_{flowzone} = \Sigma(C_{sample} \times A_{sample} \times UnitFluxRate)$$

Where,

$Load_{flowzone}$ = the estimated annual mass load to surface water, µg/yr

C_{sample} = the chemical concentration in the TZW, µg/L

A_{sample} = the area of the Thiessen polygon associated with the given sample, ft^2

UnitFluxRate = groundwater seepage flux rate for the given flow zone, $L/ft^2/yr$.

A range of load estimates for each flow zone was determined by applying both the filtered and unfiltered concentrations to the calculations, per agreement with EPA, as well as the average and the maximum measured seepage flux for the given flow zone. From the resulting four estimates, the highest and lowest values were assigned to designate the range. The estimate based on the average measured flux and the unfiltered concentrations was assigned as the central estimate.

[20] Thiessen polygons are formed as a network of polygons generated around seed points. In this case, the seed points are sampling locations. The polygon around each seed point delineates all areas that are closer to the seed point than to any other seed point.

The range of estimated annual loads for a given study site was determined by summing the estimated ranges for each Thiessen polygon. The ranges of load estimates for the Study Area were, in turn, generated by summing the estimates for each of the nine study sites. Detailed presentation of the steps in this approach and complete tabulated results are provided in Appendix E6.1.

6.1.5.2 Uncertainty

The upland groundwater plume loading estimates are based on empirical, Study Area-specific TZW chemistry and groundwater flux data collected offshore from the nine upland sites included in the GWPA sampling program conducted as part of the RI. The range of results presented for this term is expected to be a reasonable approximation of the uncertainty in the loading estimates, though there are additional potential sources of uncertainty that may not be reflected in these ranges. Specifically, the following sources of uncertainty are acknowledged in the upland groundwater plume loading estimates:

- This assessment does not include loading from sites other than the nine study sites where empirical TZW data were collected. As described in the site selection process (see Section 4.4.3.1 and Appendix C2), these nine sites represent those with a confirmed or reasonable likelihood for discharge of upland groundwater COIs to Portland Harbor. Eighty-three other upland sites reviewed during the site selection process lacked sufficient data to determine the completeness of the groundwater pathway. To the extent that a complete groundwater transport pathway to the LWR could be identified in the future at one or more of these 83 sites or other currently unidentified sites, total groundwater plume loading to the Study Area may be underestimated.
- The spatial resolution of the analysis is limited to the resolution of the sampling data sets, as reflected in the Thiessen polygon approach.
- There is no attempt made in these estimates to distinguish the origin of the chemicals in the TZW, and it is expected that the empirical TZW data set includes chemicals originating from sediment contamination (as assessed in the advective loading analysis in Section 6.1.6 and Appendix E6.2).
- The GWPA study design specifically targeted areas of higher seepage and higher TZW concentrations for sampling in the areas offshore of the study sites.
- The TZW concentration estimates do not account for any additional chemical attenuation to sediments that may occur in the upper 38 cm bml.
- Sampling was conducted during the hydrologic season of highest expected groundwater flow rates to maximize the observed groundwater signal (plume concentration and flow rate). Consequently, the lower end of the groundwater signal in the discharge areas is not captured in the empirical data set.

6.1.5.3 Findings

The estimated ranges of upland groundwater plume annual loads are presented in Table 6.1-8 at the Study Area scale (sum of all nine study sites) and in Table 6.1-9 for the individual study sites. Groundwater plume loads at the Study Area scale are also presented graphically by chemical group in Figures 6.1-41 through 6.1-45. These chemical group plots show the estimated loads based on filtered and unfiltered estimates to allow for comparison of these data sets. Load estimates for the individual study sites are presented graphically in Figures 6.1-46 and 6.1-47 for total DDx and total PAHs only. (Note: PCBs and dioxin/furans were not sampled in TZW.)

Figure 6.1-41 presents load estimates based on filtered and unfiltered TZW sampling data for the DDx components on the upland groundwater plume loading IC list. The unfiltered results are consistently higher than the filtered results for this group of hydrophobic chemicals. As discussed in Section 5.4, unfiltered results are likely biased high due to entrainment of sediments in the TZW samples. Further, the DDD components compose the largest fraction of the total DDx load estimates for both filtered and unfiltered samples, while the DDE components make up the smallest fraction. Figure 6.1-46 presents the total DDx loading estimates for the two study sites where this IC was sampled in TZW.

Figure 6.1-42 presents the upland groundwater plume loading estimates for the PAHs on the IC list. The majority of the total PAH load from upland groundwater plumes is from LPAHs (e.g., naphthalene), as opposed to the HPAHs (e.g., BAP and total cPAHs [cPAH BaPEq estimates follow total cPAH trends]; Figure 6.1-42). This result follows behavior expectations that the LPAHs are more soluble than the HPAHs. Both HPAHs and LPAHs show a pattern of higher unfiltered concentrations and lower filtered concentrations. This pattern is expected of hydrophobic molecules such as PAHs.

Estimates for upland groundwater plume loading of metals at the Study Area scale (sum of all nine study sites) are presented on Figure 6.1-43. These estimates cover a large range of values, from a central estimate of approximately 0.02 kg/yr for mercury to a central estimate of 8,500 kg/yr for manganese. It is interesting to note that the unfiltered/filtered loading ratios vary for different metals. The ratios for two metals, arsenic and manganese, show little difference; for barium, cadmium, nickel, and mercury unfiltered loading estimates are moderately larger than filtered estimates. The remaining metals show large disparities between unfiltered and filtered loading estimates: Zn (31), Cu (86), and Pb (120). Estimated metals loads associated with groundwater discharges at individual study sites are provided in Table 6.1-9.

Figures 6.1-44 and 6.1-45 present the upland groundwater VOC and SVOC loading estimates at the Study Area scale (sum of all nine study sites). These plots are broken into two groups of VOCs: Group 1 includes all of the chlorinated, non-aromatic VOCs on the IC loading list; Group 2 contains the aromatic VOCs, including BTEX chemicals, as well as 1,2-DCB and carbon disulfide. Among the Group 1 VOCs (Figure 6.1-44), it is apparent that chloroform and methylene chloride dominate the

loading scale. On the TCE degradation chain, cis-1,2-DCE shows the highest loads.[21] On the TCA degradation chain, chloroethane shows the highest loads. Among Group 2 (Figure 6.1-45), benzene loads dominate the BTEX chemicals, and chlorobenzene loads are higher than 1,2-DCB. Estimated VOC and SVOC annual loading are provided in Table 6.1-9.

Upland groundwater loading estimates are compared to other loading rate estimates as part of the CSM presentation in Section 10.

6.1.6 Advective Loading

Advection of groundwater through contaminated sediments can transport chemicals that desorb from sediment solids into the aqueous phase and then migrate with the flowing groundwater; in this document, this mechanism is termed "advective loading." Two types of advective loading were evaluated for this RI Report:

- Subsurface advective loading: migration of chemicals associated with subsurface sediments (deeper than 30 cm bml) to surface sediments via desorption and groundwater advection and sorption to surface sediments; this is considered to be an external loading term to the Study Area per verbal agreement with EPA.
- Surface advective loading: the migration of chemicals associated with surface sediment (0 to 30 cm bml) to surface water via desorption and groundwater advection. Per agreement with EPA, surface advective loading to surface water is categorized in this RI as an internal fate and transport process (as opposed to an external loading term). Surface advective loading thus involves chemical mass transfer within the Study Area (from surface sediment to surface water).

Section 6.1 generally focuses on external loading mechanisms, such as subsurface advective loading, which transport mass into the Study Area. However, surface advective loading, which is an internal fate and transport process, is also described in this section. The description is included here (as opposed to presentation of the surface advection term in the fate and transport discussion under Section 6.2) to allow for parallel description of the common calculation approach and for comparison of results for both surface and subsurface sediment advective loading. This section describes the approach and results of the analyses to generate estimates of subsurface and surface advective loads.

The surface and subsurface advective loading terms were assessed for the advective loading IC list presented in Table 6.0-1. These chemicals were selected from the complete list of loading ICs because they are likely to sorb to sediment solids and are subject to the chemical partitioning processes relevant to this loading mechanism. Both

[21] Note that the loading estimates for TCE, cis-1,2-DCE, and vinyl chloride are dominated by results from a single TZW sample offshore of the Siltronic site. The groundwater pathway for TCE is discussed in detail in Appendix C2.

terms were assessed on the scale of the relevant available sediment data set. Loading estimates for each term were generated in units of mass loading per year and presented for the entire Study Area and by river mile.

These advective loading estimates differ from the upland groundwater plume loading estimates (presented previously in Section 6.1.5) in that in-river sediment contamination, as opposed to upland groundwater contamination, is considered to be the chemical source for the advective loading analysis. In areas where ICs in pore water are attributable to both upland groundwater plumes and in-river sediment sources, the plume loading and advective loading assessments may overlap, resulting in some double-counting of loads. The extent of this overlap depends on the relative magnitude of the groundwater plume concentrations versus the sediment-derived pore water concentrations based on equilibrium partitioning. (Note: the TZW plume study areas account for less than 5 percent of the Study Area plan-view area.) As described previously in Section 6.1.5, loading estimates for the Fate and Transport Model address loading from upland groundwater plumes and advective loading from groundwater discharge through sediments somewhat differently than the empirical estimates developed for the RI.

6.1.6.1 Data Sets and Approach

The following subsections briefly describe the data sets and approaches applied to generate estimates of surface and subsurface advective annual loads as well as accumulation rates of chemicals in surface sediment. Detailed presentation of data sources and approaches is provided in Appendix E6.2.

6.1.6.1.1 Loading Estimates

The first step in the analysis to generate annual loading estimates for surface and subsurface advective loading was to produce Thiessen polygon sets for each IC for surface and subsurface sediment. Each polygon represents a sediment concentration for the given IC, as well as a sediment bulk density and a sediment organic carbon value.

For each polygon, advective loading rates were developed by first estimating pore water concentrations under an assumption of equilibrium, then applying an estimated advection rate to generate a mass annual load. This approach required two major assumptions. First, it is assumed that the chemical partitioning between sediment and pore water is at a condition of equilibrium at all times. Second, it is assumed that the groundwater advection rate is constant throughout the system. It is recognized that these assumed conditions do not directly reflect the heterogeneity of conditions throughout the Study Area (limited assessment of proximal pairs of pore water and sediment concentration data are provided in Sections 6.2.1.1.1 [organic chemicals] and 6.2.1.1.2 [inorganic chemicals]); however, both of these assumptions are considered necessary to allow for development of semi-quantitative loading estimates for the entire Study Area.

Pore water concentrations in the surface sediment and subsurface sediment were estimated for each sediment sample, where surface sediment was designated as the data set representative of the top 30 cm bml, and subsurface sediment was defined as the interval descriptive of the concentrations immediately below 30 cm bml (detailed description of the data set definitions are presented in Appendix E6.2). For each sediment sample, pore water concentrations were calculated for a range of equilibrium partitioning values (K_{oc} or K_d), applying the sediment-sample-specific percent solids value and organic carbon content (for non-metals). The formulas applied and the steps to compile ranges of K_{oc} and K_d values, as well as the full compilation, are presented in Appendix E6.2.

The groundwater discharge rate was estimated using upland hydrogeologic data gathered for the Round 2 Report (Integral et al. 2007), applying Darcy's Law to generate an estimated total discharge rate to the river, and converting that discharge to a unit flux through the river sediment surface. The complete data sets and the calculations are presented in detail in Appendix E6.2. This approach generated estimates of total groundwater discharge rates to the Study Area of 4.6 cfs (lower estimate) to 11 cfs (upper estimate), with a central estimate of 7.3 cfs. Dividing by the surface area of sediments in the Study Area, this corresponds to a range of unit groundwater flux rates of 1.5 to 3.6 ft/yr, with a central estimate of 2.4 ft/yr.

The estimated groundwater discharge rate used in the advective loading calculations was compared with the flow rates observed by seepage meter measurements as part of the groundwater plume loading estimates. The Darcy's Law estimate range is roughly 15 percent of the unit discharge rate observed with seepage meters in the nearshore groundwater plume discharge areas. The selective placement of these seepage meters[22] and the measurement exclusively during hydrogeologic times of higher groundwater discharge (by sampling design) explains the disparity between the two ranges. Overall, the order-of-magnitude agreement between the unit flux rates developed using these two information sources offers confidence in the utility of the Darcy's Law-estimated rates.

From the ranges of estimated pore water concentrations, the range of groundwater advection rates, and the areas associated with each sediment sample,[23] ranges of advective loads were estimated for each sediment polygon by the following equation:

$$Load_{polygon} = C_{TZW} \times A_{polygon} \times Q$$

Where,

[22] In the design of the TZW study, seepage meters were purposefully placed at locations where there was an indication (based on pore water temperature measurements, sediment texture, or screening results) of higher flow rates. As such, the seepage meter measurements are expected to be biased high relative to an average unit discharge for the entire Study Area.

[23] Each sediment sample was assigned an area, based on Thiessen polygon sets generated for the surface and subsurface sediment data sets. This step is described in detail in Appendix E6.2.1.

C_{TZW} = the estimated chemical concentration in the pore water

$A_{polygon}$ = the area of the polygon

Q = the estimated annual groundwater flux rate.

Annual loading estimates for surface advection to surface water were arrived at by summing the loading assessed for all surface sediment polygons in the Study Area (or within a river mile). For subsurface advective loading (to the surface sediment interval), estimates were only generated for areas where OC-normalized subsurface sediment concentrations for the given analyte were greater than the corresponding OC-normalized concentration for the surface sediment.[24] This approach is described in greater detail in Appendix E, including graphical presentation of the spatial breakdown of this approach for PCBs, PAHs, DDx, and dioxin/furans, and a table of the spatial breakdown for all other advective loading analytes. In general, the percent of the Study Area with subsurface loading assessed to surface sediment ranges widely, from 6 percent for aldrin to 94 percent for total PAHs (these percentages reflect the percent areas where OC-normalized subsurface sediment concentrations exceed OC-normalized surface sediment concentrations, and are discussed further in the historical loading section below, Section 6.1.8). Table E6-4 in Appendix E summarizes the percent of the Study Area over which subsurface advective loading was assessed for the entire suite of chemicals assessed.

Mass loading estimates to surface sediment (by subsurface advective loading) and to surface water (by surface advective loading) were generated for the advective loading ICs following the approach described above. The range in each estimate reflects the range of equilibrium partitioning values and groundwater discharge rates applied to the calculations. These results are presented and discussed below in Section 6.1.6.3.

6.1.6.2 Uncertainty

There is significant uncertainty associated with the advective annual load estimates related to applied assumptions (i.e., equilibrium behavior of all ICs and uniform groundwater discharge rates), as well as the data sets used in the calculations (i.e., literature equilibrium partitioning coefficients, and roughly estimated groundwater discharge rates).

Related to equilibrium, the primary uncertainty is the assumption of equilibrium in all parts of the complex sediment/pore water environment at all times. This calculation fails to capture reaction kinetics and the sorption-desorption-resorption dynamics that

[24] Upward flux of chemicals from subsurface sediments due to desorption and advection can increase chemical concentrations in surface sediments if surface bulk sediment concentrations (OC-normalized) are initially lower than the underlying subsurface interval. If surface concentrations are greater than or equal to subsurface concentrations, mass flux from the subsurface will not accumulate in surface sediments but will instead discharge to overlying surface water. It is assumed that the existing estimates of advective flux from surface sediments to surface water are inclusive of the latter phenomenon. Therefore, the scope of this analysis is restricted to locations in the Study Area where “cleaner over dirtier” conditions exist.

occur in advective transport through sediment. For example, to the extent that non-equilibrium conditions may exist in the pore water environment as a result of kinetic limitations on desorption from contaminated sediments, the assumption of equilibrium will overstate pore water concentrations and advective loading rates. Beyond the assumption of equilibrium, it should be noted that the Study Area organic carbon associated with sediments may differ in character from that defined by the range of literature K_{oc} values. Likewise, the location-specific chemical and geochemical conditions (redox, pH, ionic strength and composition, sediment matrix composition, etc.) likely differ in character from those associated with the applied specific literature values. Further, this assessment ignores any chemical or biological transformation processes that may occur in the migration process.

Related to the groundwater flux rate estimates, there are a number of significant uncertainties. First, they are based on the limited available upland data and not on groundwater modeling of the area or direct measurement of seepage rates representative of the entire Study Area. Second, the groundwater advection rate estimates rely on a simple and conservatively high cross-sectional area. Third, the advection rate estimates apply a projection of the sediment surface area to represent the actual sediment surface area (thereby increasing the unit discharge estimate). Finally, the assumption of a uniform groundwater discharge rate for the entire Study Area does not capture the spatial variability that is inherent in groundwater discharges to rivers. The discharge rates are assumed to be constant in time and do not account for variability caused by seasonal recharge patterns, changes in river flow rates and stages, and tidal fluctuations (aka, “tidal pumping”). As discussed in Section 5.4.1.3, and presented in more detail in Appendix C2, the largest net negative recharge rate measured in a seepage meter during a rising tide, among all seepage meter locations where net positive average advective groundwater fluxes were measured, was offshore of the Siltronic site. At this location, the negative recharge period covered roughly 9.5 hours, with an average seepage rate of –6.7 cm/day. This corresponds to a net negative seepage flux of 2.65 cm into the sediment bed over the 9.5 hour tidal recharge period. Assuming sediment porosity of 25 percent, the maximum depth of influence for this period of negative seepage would be roughly 10.6 cm before the direction reversed to positive discharge with the tidal change. Although tidal pumping may in some instances lead to increased loading of chemicals from the sediment bed to the water column by introducing relatively clean surface water into the uppermost several centimeters of the sediment bed with each tidal cycle, uncertainty in the loading estimates due to this effect is expected to be minor compared to the other sources of uncertainty inherent in these calculations.

The large range in most of the estimates (presented in the following subsection) reflects the range in literature equilibrium partitioning coefficients. These large-scale estimates of advective annual loads are considered to be highly uncertain, but useful for general comparison to other loading terms for each chemical (Section 10). Any local recontamination concerns identified by the FS may require consideration of specific chemical and localized physical conditions, including focused collection of additional information and/or more complex assessments of advective loading.

6.1.6.3 Findings

This section discusses the results of the calculations described above. This analysis considers the results for surface sediment and subsurface sediment advective loading estimation for the entire Study Area for all advective loading ICs, as well as results by river mile for PCBs, PCDD/Fs, DDx, and PAHs.

6.1.6.3.1 Study Area Annual Loading Estimates

Study Area-wide loading estimates for the subsurface sediment and surface sediment advective loading terms are presented in Table 6.1-10 for all advective loading ICs. These results are also presented graphically on Figures 6.1-48 through 6.1-60, showing both surface and subsurface annual loading estimate ranges. Patterns and other observations for each of the IC chemical groups are discussed in the following paragraphs.

The central estimated ranges of annual loads for total PCBs (Figure 6.1-48) are slightly higher from the surface sediment to the surface water as compared to those from the subsurface sediment to the surface sediment. Note, however, the difference for total PCBs (subsurface sediment advection versus surface sediment advection) is only a factor of approximately 1.5 for the central estimates. This observation of slightly greater upper loading estimate for subsurface advective loading to surface sediments is expected due to the Study Area-wide higher average PCB concentrations in subsurface sediments as compared to surface sediments. For the individual congeners, estimated load ranges were generally higher for subsurface sediment loading to surface sediments, as compared to surface sediment advective loading to surface water (Figure 6.1-49). Of the individual congeners analyzed, PCB 118 and PCB 105 exhibit the highest annual loads, whereas PCB 169 is the smallest contributor.

For PCDD/Fs (Figure 6.1-50), the advective loading estimates show a slightly greater surface advective loading to surface water, as compared to subsurface advective loading to surface sediments. Estimates of surface sediment advective loading to surface water are higher by a factor of 2.5 for PCDD/Fs as compared to rates of subsurface sediment advection partitioning to surface sediments. The OC-normalized PCDD/F concentrations used in the load calculations are generally similar in surface sediment and subsurface sediment, with 54 percent of the subsurface sediments showing greater concentrations than the surface sediments (Table E6-4). The Study Area-wide loading estimates, however, are dominated by individual high surface sediment concentration values, resulting in the greater Study Area-wide total PCDD/F advective loading from surface sediment as compared to subsurface sediment.

For DDx pesticides (Figure 6.1-51), DDD isomers compose the largest share of the central estimate total DDx load for both surface sediment and subsurface sediment advection load estimates, followed by DDT then DDE. The Study Area-wide subsurface loading (to the surface sediment interval) is slightly greater than the advective loading to surface water for each of the DDx pesticides evaluated (by a factor of 1.8 for total DDx). For DDx, approximately 35 percent of the Study Area is not

expected to exhibit advective loading from the subsurface sediment to the surface sediment due to higher OC-normalized sediment concentrations in the surface.

For PAHs (Figure 6.1-52), total PAH annual load is from both the surface and subsurface sediments are dominated by the LPAHs (e.g., naphthalene). The HPAHs (e.g., BAP and cPAHs [cPAH BaPEq estimates show similar trends to total cPAHs]) show slightly higher surface sediment loading to surface water as compared to subsurface sediment loading to surface sediments. In contrast, the naphthalene and total PAHs show higher subsurface sediment loading to surface sediment, with the total PAH estimated subsurface advective loading being higher than the surface advective loading by a factor of almost 3 for the central estimate. Note also that the LPAHs and the HPAHs on the IC list all exhibit higher OC-normalized subsurface sediment concentrations over 60 to 65 percent of the Study Area (Table E6-4).

Figure 6.1-53 presents the range of advective load estimates for BEHP. The large range in the estimated loads—5 orders of magnitude—for BEHP is a direct reflection of the large range in the literature K_{ow} (octanol-water partitioning coefficient) values. BEHP estimates show slightly more surface sediment loading (by a factor of approximately 2 for the central estimates).

Non-DDx pesticide advective loading estimates are presented on Figure 6.1-54. Among these, gamma-HCH shows the highest mass loading, and aldrin shows the lowest (possibly due to a tendency for aldrin to degrade to dieldrin in environmental systems). For aldrin and dieldrin, the subsurface to surface advective loading ratio is close to 1, at 0.9 and 1.15, respectively. The ratio is lower for gamma-HCH (0.6) and higher for total chlordanes (~2.5). All of these non-DDx pesticides show lower OC-normalized subsurface sediment concentrations (as compared to surface sediment) in over 85 percent of the Study Area, except total chlordanes at approximately 60 percent (Table E6-4).

The advective loading rate estimates for arsenic, copper, lead, and mercury are presented on Figure 6.1-55. Arsenic and copper show the highest Study Area-wide loading, followed by lead. Mercury produced the lowest loading estimates, with central estimates 4 orders of magnitude below central estimates for arsenic and copper. For copper, mercury, and lead, the surface and subsurface loading were very close (ratios of 0.8 to 1.1), in agreement with the fairly even distribution of areas with higher or lower surface sediment concentrations[25] (approximately 60 to 70 percent; Table E6-4). For arsenic, the loading estimate for the surface sediment advection to surface water is 2.2 times that for subsurface sediment loading to surface sediment. This is also reflected in the fact that only 40 percent of the Study Area exhibits subsurface concentrations greater than the surface sediment concentrations (Table E6-4).

[25] Note that for metals, the surface sediment and subsurface sediment concentrations comparisons to support the calculations were not OC-normalized, reflecting the use of K_d values as opposed to K_{oc} values.

Estimated TBT advective loads (Figure 6.1-56) vary over 3 orders of magnitude from the lower to upper estimates. The subsurface-to-surface sediment TBT loading estimate is 3.7 times higher than the surface sediment loading to surface water for the central estimates. For TBT, approximately 40 percent of the Study Area was not considered to be loading to the surface sediment due to higher OC-normalized surface sediment concentrations as compared to the subsurface (Table E6-4).

In summary, Study Area-wide advective annual loads from subsurface sediment to surface sediment were higher than advective loading from surface sediment to surface water for PCBs, DDx pesticides, LPAHs (and total PAHs, which are dominated by LPAHs), BEHP, arsenic, total chlordanes, and TBT. The opposite was true only for PCDD/Fs, gamma-HCH, and HPAHs. There was little difference in the surface and subsurface advective loading estimates for aldrin, dieldrin, copper, mercury, and lead. These differences are a direct reflection of the patterns of relative OC-normalized concentration of each chemical in the surface sediment as compared to the subsurface sediment. This pattern is addressed further in the historical loading discussion (Section 6.1.8).

6.1.6.3.2 Annual Loading by River Mile

Figures 6.1-57 through 6.1-60 present annual surface sediment and subsurface sediment advective loading for each river mile in the Study Area for total PCBs, total PCDD/Fs, total DDx, and total PAHs. Given the nature of the analysis (Thiessen-polygon-based scale of assessment), variations in river-mile scale annual load estimates are indicative of variations in sediment chemical concentrations and organic carbon content. Groundwater flux rates and assumptions of equilibrium behavior were held constant over the entire Study Area in advective loading calculations.

For total PCBs, the sediment advective loading pattern is fairly complex. Figure 6.1-57 shows that the highest central estimate advective loading from subsurface sediment to surface sediment for total PCBs are observed at RM 8 to 9, followed by the reaches at the upstream and downstream ends of the Study Area. The highest annual surface sediment loading estimates to surface water are observed at RM 9 to 9.9, followed by the reaches at the upstream and downstream ends of the Study Area.

For total PCDD/Fs (Figure 6.1-58), both subsurface sediment advective annual loading to surface sediment and surface sediment advection to surface water are fairly consistent across the Study Area, with the exception of higher annual loading estimates from RM 7 to 7.9. No clear pattern of higher subsurface advective loading to surface sediments, as compared to surface advective loading to surface water, are apparent through the Study Area.

For DDx (Figure 6.1-59), the highest surface and subsurface sediment advective loads are predicted at RM 7 to 7.9. Further, the DDx subsurface-to-surface sediment advective loading annual estimates are higher or comparable to the estimates for surface sediment advection to surface water in all river miles except RM 9 to 9.9 and RM 11 to

11.9. Finally, for PAHs (Figure 6.1-60), the highest subsurface advective loading to surface sediment is observed at RM 6 to 6.9.[26] The maximum surface sediment advection to surface water load estimates are observed at RM 5 to 5.9. For PAHs, the subsurface-to-surface sediment advective loading estimates are higher or comparable to the load estimates for surface sediment advection to surface water in all river miles except RM 5 to 5.9 and RM 9 to 9.9.

6.1.7 Riverbank Erosion

This section assesses the susceptibility of Study Area riverbanks to erosion and presents the limited data set available for chemically characterizing riverbank materials. No quantitative estimation of the loading of chemicals to the in-water portion of the Study Area from bank erosion is performed in this section, per agreement with EPA.

For the purposes of this analysis, riverbank materials are defined as soil and sediment that are between the mean high water mark (MHWM)[27] and the ordinary high water mark (OHWM).[28] This includes gently sloped upper beach areas as well as steeper bank areas. Erosion of bank materials containing chemicals from above the OHWM, including materials much farther upland, is also possible; however, such upland erosion is primarily caused by overland transport (stormwater runoff), which is discussed in Section 6.1.2, and flood events.

The primary mechanisms for riverbank erosion are river water moving over bank materials, direct overland transport across these materials, and sporadic mass wasting or slumping events when bank slopes become over-steepened or otherwise unstable. Wind erosion, shoreline construction and other human activities, activities of animals, etc. are also possible erosion mechanisms; however, these can reasonably be considered to be minor in comparison to river and stormwater flows. (Note: construction is considered minor because such projects are regulated and permitted to minimize erosion of bank materials into surface waters.)

River water can cause erosion at times when river levels rise and come into contact with the bank. The MHWM (elevation +13.3 ft NAVD88) is based on the monthly average water level for the 16-year period from 1987 to 2002. Thus, during some periods, particularly during winter months, riverbanks above this elevation can become inundated by river water. Erosive mechanisms during these periods include the direct and shear stress forces of currents with sufficient nearshore velocity to suspend soil and

[26] This peak comprises the majority of the Study Area total load assessed for this term, and is attributed largely to LPAHs (see Table E6-7 and E6-8 in Appendix E).

[27] The MHWM is the elevation defining the shoreline boundary of the Portland Harbor Superfund Site, which is +13.3 ft (NAVD88). This elevation is based on a DEQ memorandum dated July 9, 2003 to EPA regarding the upland/in-water boundary for the Superfund Site (DEQ 2003b).

[28] The OHWM refers to the upper edge of the riverbank and is defined as approximately +20 ft (NAVD88; DEQ 2003b). The OHWM defines the elevation beyond which inundation by the river is limited to extreme flow events, which occur approximately every five years.

sediment particles. Nearshore velocities can be affected by a number of factors, including the following:

- Bends in the river, where outer bends tend to be subject to greater velocities
- Other shoreline features that may create eddies
- The presence of nearshore structures, which tend to slow nearshore currents unless localized focusing effects or strong eddies are generated
- The general "roughness" and physical complexity of the bank surface.

Wind-generated wave action or vessel wakes can also cause bank erosion as these waves break on the shoreline and dislodge riverbank materials. Wave action can be diminished or augmented in particular areas due to concentration of reflected waves and/or the length of wind fetch to which the shoreline is exposed.

Erosion of relatively exposed bank material can occur in localized areas where stormwater sheet flow, particularly from nearshore impervious surfaces, flows to small low spots and becomes concentrated into rivulets or small streams. These flows can also cause saturation of bank soil and sediment, which may make them more unstable and susceptible to mass wasting.

For riverbank materials to represent a potential loading term to a river, two conditions must be met: 1) the materials must be in a form that is potentially available for erosion into the river; and 2) chemicals must be present at elevated levels within bank soils and sediment. Because most banks in the Study Area are longstanding, vegetated, and in equilibrium with common currents, boat wakes, and overland transport or runoff, mass wasting during extreme events may be the only significant loading process. For typical years, there is likely no significant load from riverbanks, as indicated below.

6.1.7.1 Bank Materials Available for Erosion

Regardless of the exact force exerted on the bank, the degree of erosion generated is highly dependent on the physical conditions of the bank itself, the type of materials present, and how directly exposed the materials are to these forces. Primary factors affecting the susceptibility of banks to erosion include the following:

- Presence of protective and stabilizing vegetation (natural or planted)
- Presence of stabilizing structures such as bulkheads
- Presence of riprap, concrete, or other materials intended to protect the bank
- Steepness and overall profile of the bank
- Type of soils and sediment (e.g., consolidated, loose, gravel, sand, silt, cohesive clay, fill, or natural materials)
- Degree of saturation

- Presence of debris or artificial bank structures placed for purposes other than bank protection (e.g., boat ramps)
- Presence of docks, piers, dolphins, pilings, breakwaters, groins, and shoreline structures
- Presence of bench or beach areas below the bank, which can act to dissipate wave forces higher on the bank.

Where protective vegetation, structures, or materials are present, the type and condition of the materials underneath are often less important to erosion rates. An ODFW study (Vile and Friesen 2004) reviewed and inventoried the shoreline features from Willamette Falls (RM 26.5) to the confluence with the Columbia River (RM 0) and broke them into twelve general classifications based on nearshore habitat types. Using this data and several other sources of information, the City of Portland updated its natural resource inventory from the Broadway Bridge (RM 11.6) to the Columbia River. The City of Portland (2008d) reported the most common bank types within this area are vegetated riprap, sandy and rocky beach, and unclassified fill. Percentages of shoreline bank classification are as follows:

- Vegetated riprap—25 percent
- Beach—23 percent
- Unclassified fill—21 percent
- Pilings limiting light—13 percent
- Non-vegetated riprap—12 percent
- Bio-technically engineered banks (artificial materials with vegetation aimed at bank stabilization)—3 percent
- Sea wall—2 percent
- Rock—1 percent
- Pilings allowing light—0 percent.

Map 3.1-17, adapted from the ODFW (Vile and Friesen 2004) study, maps these bank classifications. Of these classifications, only beach and unclassified fill, which together represent approximately 44 percent of the shoreline, are likely to be at all susceptible to erosion. Unclassified fill occurs in areas that were filled over time with a variety of unconsolidated materials and debris. No engineered riprap covers the surface, and banks can become unstable from erosive river forces and slump into the river. River beach areas are less steep, are in equilibrium with in-river physical processes, and can often act as a buffer, as noted above. Thus, unclassified fill (at 21 percent) is likely to be the most susceptible to erosion during extreme events. This classification, however, represents a diverse range of physical conditions, and some of these areas have surfaces composed of various-sized rocks, sporadic vegetation, artificial debris of various types,

and natural debris such as logs and wood, all of which may protect some of these areas to some extent.

For loading estimation purposes, approximately 15 to 25 percent of the shoreline is assumed to be potentially susceptible to erosion of bank materials. The low end of the range accounts for the presence of vegetation and natural and artificial debris that can reduce the potential for erosion. The high end of the range overestimates the amount of unclassified fill that is susceptible to erosion, but it allows for the fact that some portions of the other categories may include small areas that are relatively susceptible to erosion.

6.1.7.2 Bank Material Chemistry

The LWG has searched documents available from DEQ through December 2007 for bank soil and sediment chemistry results associated with upland sites. Very little bank soil and sediment chemistry data are available in Portland Harbor. Nearshore soil and sediment chemistry data from other parties were compiled from DEQ documents and uploaded to the LWG's SCRA database. The bank data contained in the SCRA database are summarized in Table E7-1, Appendix E, and are provided in full in the SCRA data file provided on CD in Appendix E7.0 (Attachment E-3). The selection process for summarizing the bank data included the following:

- Data collected between RM 1.9 and 11.8
- Data collected since May 1997
- Surface sediment or soil data (0–40 cm)
- Samples collected between 11 and 20 ft NAVD88
- Category 1 and 2 data.

The locations of the riverbank data points relative to the bank categories described above are shown in Map E7-1, Appendix E. As noted above, the table and map include samples that were taken near but not necessarily within the zone from +13 to +20 ft NAVD to capture as much bank data as possible. It should also be noted that riverbank soil and sediment data are available upstream of the Study Area in the downtown reach (Zidell South Waterfront property and Ross Island) and upriver (Oaks Bottom Landfill area and Willamette Falls). These data are not summarized in Table E7-1, but they are provided with the complete bank SCRA data file provided on CD in Appendix E7.0.

The table and map show that a considerable portion of the harbor riverbank materials that may be generally susceptible to erosion have no existing bank chemistry data. In these cases, it is impossible to quantify the concentrations and extent of chemicals that may be present and available for transport to the river via bank erosion. Among the sites with available bank chemistry data, chemicals on the IC loading lists were detected in bank soils and sediment that are potentially susceptible to erosion (Table E7-1).

6.1.7.3 Riverbank Erosion Loading

Due to the paucity of existing bank condition and chemistry information at multiple shoreline sites, it is not possible at this time to make even semi-quantitative estimates of loading from this source to the river. Although it is estimated that approximately 15 to 25 percent of the banks within the Study Area are potentially susceptible to erosion, it is not possible to estimate typical erosion rates or a range of rates that might apply to these areas given the wide range of conditions present.

Bank loading estimates are not possible given the data limitations described above. However, it is unnecessary to develop such loading estimates for the purposes of the RI/FS. Because bank erosion is an area-specific condition dependent on both the erodibility and chemical concentrations of any given bank area, the potential role of bank erosion may need to be evaluated as a part of the future remedial design process for each sediment management area (SMA). For the purposes of the FS, it will be assumed that potential bank erosion sources will be controlled before, or as part of, remedial action.

6.1.8 Historical Loading

The previous sections focused on current loading to the Study Area; however, historical loading is an important element of the CSM because historical sources affect the current IC distribution. Historical loading is defined as any loading of ICs to the Study Area that occurred prior to collection of the data sets used to assess the current loading terms. Historical loading is reasonably expected to have occurred by all of the loading mechanisms assessed above: upstream surface water, bedload, stormwater, non-stormwater direct permitted discharges (as defined in Section 6.1.3), groundwater plume discharge, atmospheric deposition, riverbank erosion, and advective loading. For each mechanism, the historical load may be significantly greater than current load due to changes in regulations and reduction or elimination of chemical use in the Study Area and Willamette Basin. Additionally, historical records indicate that releases from overwater activities[29] were significant in the Study Area, though they are not specifically quantifiable with available records.

This section presents a qualitative discussion of historical loading to the Study Area, focusing on the expected relative load from historical versus current sources. This discussion complements the presentation of historical sources in Section 4. Information from both of these sections and from the subsurface sediment record are presented together in Section 10 (CSM) and evaluated to assess the current traces of existing contamination to historical sources.

Upstream Surface Water Loading – Historical surface water loading to the Study Area is relevant to the extent that associated suspended solids were deposited and

[29] While improved BMPs are likely to reduce the occurrence of overwater releases significantly, it is acknowledged that current and future releases could occur. No attempt is made in this report to predict and quantify such releases as a current loading term, and no additional analysis of this term is planned for the RI/FS.

remain within the Study Area. It is likely that historical loading via upstream surface water was significantly greater than current loading for many chemicals, due to widespread historical use of pesticides and herbicides, historical wastewater, sanitary and stormwater management practices, and generally more limited regulatory controls on many ICs.

Upstream Bedload – The relative contribution of historical upstream sediment loading, as compared to current upstream sediment loading, is uncertain. Factors, including historical dredging, lack of information regarding historical versus current upstream bedded sediment concentrations, sediment transport during flood events, and complex sediment deposition patterns, make it difficult to predict or generalize about the duration and long-term impact of upstream sediment migration and loading.

Stormwater Loading – Stormwater loading to the Study Area is expected to have been higher historically, prior to implementation of upland stormwater runoff controls in some areas, control of CSOs, changes in chemical use/production/incidental production (PCBs, dioxins), and control of industrial discharges to storm drains and CSOs.

Non-Stormwater Direct Discharge Loading – Historical loading to the Study Area from industrial discharges, prior to adoption and regulation of discharge permits and controls, is likely to have been significantly higher than current loading.

Upland Groundwater Plume Loading – Upland groundwater plumes are generally the result of historical releases. Groundwater plume loading to the Study Area was more significant historically, prior to institution of groundwater controls. However, the transport time for chemicals in upland groundwater to reach the river makes it difficult to predict or generalize about the timing of the peak historical loads.

Atmospheric Deposition – It is likely that overall atmospheric loading to the Study Area has decreased from historical levels due to widespread adoption by many countries of controls on chemical production, usage, and air emissions. As a result, atmospheric concentrations have decreased, but have not been eliminated entirely.

Riverbank Erosion – The historical composition and chemistry of bank materials—and thus the overall impact of historical bank erosion—is highly uncertain. Nevertheless, chemical loading to the Study Area due to bank erosion was likely more significant historically, prior to upland soil cleanups and installation of erosion controls in many areas, including riprap and sea walls.

Overwater Releases – Historical loading from overwater releases is expected to have been more significant historically, prior to improved BMPs. While available records do not support quantification of these historical releases, the current sediment record provides some information in known release areas (see further discussions in Section 10.2).

Advective Loading – The relative historical advective loading rate from subsurface sediments to surface sediments is uncertain; however, assuming that subsurface sediment, as defined for the advective loading analysis, generally reflects historical rather than recent releases to the Study Area, a comparison of surface and subsurface advective loading estimates (discussed in detail in Section 6.1.6.3) offers initial insight into the combined effects of historical loading to sediment from all sources. For PCBs, PCDD/Fs, DDx, and PAHs, subsurface advective loading exceeds surface advective loading at the scale of the entire Study Area. This suggests that loading of these chemicals to the sediment bed, as indicated by the subsurface interval, was higher historically than currently.[30]

6.1.9 External Loading Summary

The project data sets and relevant literature were evaluated to assess current and historical external chemical loading terms to the Study Area. The following external loading terms were considered in this evaluation (list developed in consultation with EPA; Integral et al. 2006, pers. comm.)—upstream loading via surface water (including suspended sediment load and sediment bedload), stormwater runoff, permitted discharges (non-stormwater), upland groundwater (plume transport to the Study Area), atmospheric deposition (to the river surface), upland soil and riverbank erosion, groundwater advection through sediments (partitioning from deep sediment, transported by advecting to shallow sediment interval), and overwater releases. Of these, a range of quantitative to semi-quantitative current annual loads to the Study Area were estimated for upstream surface water loading, stormwater runoff, permitted discharges (non-stormwater), upland groundwater plumes, atmospheric deposition, and groundwater advection through sediments. Historical loading is expected to have occurred by all of these loading mechanisms; however, insufficient data are available to support quantitative estimates. Therefore, historical loading was assessed qualitatively.

Table 6.1-11 provides a summary of the central estimates of external current loading to the Study Area for upstream surface water, stormwater runoff, permitted discharge (non-stormwater), atmospheric deposition, upland groundwater plumes, and advection through subsurface sediments. The estimated annual loads for the internal transport[31] mechanism of advection through surface sediments to surface water is also shown in Table 6.1-11 for comparison. For the 13 select indicator chemicals evaluated in the CSM (total PCBs, TCDD TEQ, total DDx, total PAHs, BEHP, total chlordanes, aldrin, dieldrin, arsenic, chromium, copper, zinc, and TBT), Section 10 provides additional

[30] It should be noted, however, that the total PAH loading rate estimate is dominated by LPAHs, which are much more mobile in groundwater than HPAHs. Based on this, it is possible that the subsurface advective loading of PAHs is influenced by ongoing (current) upland groundwater plume loading, in addition to historical PAH contamination of subsurface sediments.

[31] Groundwater advection is the only internal process that is quantified in this RI. The other internal processes, including sediment resuspension/transport/deposition, solid/aqueous-phase portioning, abiotic/biotic transformation and degradation, and biological uptake, may be significant in the transfer of chemicals from abiotic media to biota, and many of these terms will be quantified as part of the FS fate and transport modeling effort.

summary information regarding the relative loads associated with each loading term, as well as the range of potential variability associated with each term.

6.2 FATE AND TRANSPORT

This section describes the physical, chemical, and biological processes that influence the fate and transport of contaminants within the in-water portion of the Study Area. This discussion of fate and transport processes is organized in three main subsections, corresponding to the major environmental compartments of the Study Area: 1) surface mixed sediment layer and associated pore water, 2) surface water, and 3) biota. Figure 6.1-1 presents a conceptual drawing of these major environmental compartments.

For each of these three compartments, the relevant fate and transport processes are discussed for each of the chemical groups in the combined list of loading ICs (Table 6.0-1). These include PCBs, PCDD/Fs, pesticides (DDx and non-DDx), PAHs, petroleum hydrocarbons, SVOCs, metals, TBT, and VOCs. Insights into fate and transport processes gained from assessment of empirical information and the sediment physical transport modeling (FS HST modeling) are also discussed. Due to project timing and under agreement with EPA, results from the ongoing fate and transport modeling effort will be presented in separate FS-related deliverables.

6.2.1 Sediment and Pore Water Fate and Transport Processes

The following subsections discuss fate and transport processes relevant to ICs in the sediment and pore water[32] environment. General discussion of organic and inorganic chemical behavior in sediment and pore water is presented, followed by discussion of physical transport processes for these media. Insights from empirical data and from FS HST model results, where applicable, are also presented to help assess the relative importance of the various processes for the chemicals on the combined loading IC list.

6.2.1.1 Chemical Distribution between Sediment and Pore Water

In the sediment and pore water environment, the distribution of a chemical between the solid and aqueous phases is among the most important physiochemical processes affecting its migration, bioavailability, and half-life. The equilibrium distribution of a chemical between water (dissolved aqueous phase) and solid (sorbed to sediment or associated organic matter) is often generally described by a solid/water distribution coefficient (K_d):

$$K_d \approx (C_s/C_w)$$

Where,

C_s = the concentration of the chemical associated with solids

[32] The general term "pore water" is used here instead of TZW to acknowledge that the discussion also includes interstitial water in the sediment, which does not contain upland groundwater.

C_w = the aqueous concentration of the chemical[33]

Major processes and environmental factors that control this distribution (which varies in response to environmental conditions such as pH, temperature, and salinity) are discussed below in general terms for organic and inorganic analytes. Observed partitioning ratios in the LWG-collected TZW and paired sediment data sets are compared to published literature values, for relevant analyte groups, for additional perspective. Finally, degradation and transformation mechanisms for chemicals in the sediment/pore water environment are also discussed.

6.2.1.1.1 Organic Chemicals

For organic analytes, the K_d term describes the combined effect of all possible equilibrium partitioning mechanisms affecting distribution between sediment and pore water, including hydrophobic sorption onto organic matter associated with the sediment, electrostatic attractions of oppositely charged ionic functional groups, and covalent bonding or complexation of ionic organic molecules with reactive surface groups. For nonionic organic chemicals, which include most of the organic chemicals on the IC loading list (PCBs, pesticides including DDx, PCDD/Fs, PAHs, SVOCs, and VOCs), the primary mechanism defining the K_d term is hydrophobic sorption onto organic matter. Therefore, for nonionic organic chemicals, K_d describes two-phase partitioning[34] to the organic matter on the solid surfaces and is a function of the tendency of the chemical to sorb to organic carbon (K_{oc}) and the fractional organic matter content of the solids (f_{oc}):

$$K_d = (K_{oc} * f_{oc})$$

Where,

[33] The rigorous definition of K_d includes the ratio of the thermodynamic activity coefficient of the chemical sorbed to the solids to the thermodynamic activity coefficient of the chemical in the aqueous phase. For equilibrium environmental conditions (generally low concentrations relative to the water and solids), activity coefficients approach 1, and the K_d is considered equal to the ratio of the chemical concentrations in sediment to the chemical concentrations in the aqueous phase.

[34] All of the sorption estimates presented in the RI assume two-phase partitioning. Two-phase partitioning of the chemical between water and organic matter neglects a potential third phase that may be present in the sediment environment, namely free or residual hydrocarbons. Based on field observations and a review of the sediment core data, evidence of free or residual hydrocarbons in the sediments is minimal. Out of 1,086 sediment cores collected in the Study Area, 35 core logs (or approximately 3 percent of the logs) made reference to visual observations of TPH or tar, indicating the potential presence of free or residual hydrocarbons in the sediments at these locations. Of these 35 cores, 18 cores were collected between RM 6 and 7, predominantly near the Koppers, Gasco, and Siltronic sites. Because field observations do not indicate widespread presence of free or residual hydrocarbons in the sediments, the assumption of two-phase partitioning was deemed adequate to capture the predominant mechanisms affecting chemical distribution between sediments and pore water across the Study Area.

K_{oc} = the coefficient of partitioning between water and organic matter[35]

f_{oc} = the mass fraction of organic carbon in the solids.

In addition to temperature, several factors can affect equilibrium partitioning behavior for nonionic organic chemicals:

- **Salinity** – High-salinity environments (e.g., seawater) can cause increased adsorption (decreased solubility and higher observed K_d than predicted at lower salinity). This may be relevant in the highly saline sediment and pore water environment local to offshore areas on the west side of the river, between roughly RM 7 and 7.5, where pore-water salinities in excess of typical seawater have been observed. It is unlikely to be a significant factor elsewhere in the river.
- **Cosolvents** – The presence of miscible organic liquids in solution with hydrophobic chemicals can result in increased solubility (and therefore decreased K_d) of the hydrophobic chemical. This effect, however, requires significant amounts of cosolvent chemicals in solution (more than 10 percent by volume [Yalkowsky et al. 1976]).
- **Colloids** – Colloids are organic and/or inorganic particles in the system defined by their behavior (tendency to remain dispersed in water, not settle rapidly, and not filter easily) and size (usually 1 nm to 1 µm in diameter [Lyklema 1991]). Colloids represent a portion of the surface area available for sorption of organic chemicals. Because colloids can be mobile in water within a sediment matrix, they can increase the "apparent" concentration of the hydrophobic chemical in the aqueous phase. Because colloids are <1 µm in diameter, they would be included in both filtered and unfiltered water samples.
- **Characteristics of natural organic matter** – The nature of the organic matter present in the sediment can also affect the extent of partitioning, making partitioning behavior variable across different environments.

TPH, as analyzed for the RI, is the measure of all hydrocarbons and non-hydrocarbons that can be quantified in the carbon range from n-C_{12} to n-C_{40}. Further, the components (DRH, RRH, and GRH) are simply descriptive laboratory terms for the fractions of TPH (and not source assignments or indications of toxicity). TPH therefore represents a vast mixture of hydrocarbon chemicals, of both natural and anthropogenic origin, with an equally vast range of partitioning behaviors. As such, its behavior as a chemical group cannot be accurately characterized by a K_{oc} value. Because the subcomponents of TPH are unknown for all sampling results, the subcomponents also cannot be accurately characterized by K_{oc} values.

[35] K_{oc} is organic-matter-specific; however, it is often generically calculated from empirical expressions based on the octanol-water partitioning coefficient (K_{ow}) for a given chemical, which describes partitioning of the chemical between water and highly non-polar liquid octanol.

The only definitively ionic organic compound on the combined IC loading list is TBT. The partitioning behavior of TBT is strongly affected by pH and the identity of anions in solution that pair with the TBT ion (Arnold et al. 1997). Specifically, for pH 10 to 7 the measured log K_{oc} values are on the order of 4; from pH 7 to pH 3 they drop to roughly 2, corresponding to 2 orders of magnitude of behavioral variability across the pH 7 boundary. The mean surface water pH is 7.38 (10th percentile is 6.98 and 90th percentile is 7.76). The observed pore water pH values measured in the GWPA ranged from 5.6 to 8.1. TBT in its nonionic form (not likely observed in the Study Area) is highly hydrophobic, with K_{ow} values on the order of 7 (K_{ow}WIN).[36]

Ranges of literature equilibrium partitioning values for organic chemicals on the advective loading IC list, representing the subset of chemicals for which equilibrium partitioning assessment is relevant, were compiled for the advective loading analysis (discussed in Section 6.1.6) and are presented in Appendix E, Table E6-5. The average range in the K_{oc} values for organic analytes is 1 order of magnitude, with PCDD/Fs, TBT, and BEHP showing a range of more than 2 orders of magnitude, representing significant variability/uncertainty in partitioning behavior.

Limited site-specific empirical information, consisting of the filtered TZW data set with paired surface sediment samples, is available to assess sediment pore-water partitioning of organic chemicals. This data set is limited as a representation of the Study Area because it focuses on the offshore area of only the TZW study sites (the TZW investigation is described in detail in Section 5.4), and not all chemicals of interest in sediment were analyzed in TZW samples (e.g., PCBs were not analyzed in any TZW samples, and DDx and PCDD/Fs were analyzed in only a small fraction of the samples). However, because it is the only empirical data set, observed partitioning values were plotted against the corresponding literature partitioning values. Observed partitioning for PAHs and DDx pesticides between filtered TZW and sediment are shown in Figures 6.2-1 and 6.2-2, respectively. These figures also show the literature values compiled for use in the advective loading assessment (Section 6.1.6). Results for DDx show a wide range of literature partitioning coefficients for these analytes. Comparisons of the literature-derived partitioning values for DDx pesticides with observed partitioning behavior are limited by the small number of sample pairs (n=4) for which a given isomer was detected in both TZW and sediment. This limited set of observed partitioning values also spans a broad range. PAHs (Figure 6.2-1), in contrast, show a tighter range of literature partitioning coefficients for most individual compounds, while the observed partitioning shows much wider ranges, especially for the LPAH constituents. The wide variability in observed partitioning for PAHs may reflect multiple factors, possibly including disequilibrium conditions between TZW and sediment, small-scale spatial variability (sediment and TZW sample pairs were not perfectly collocated), and/or filtered samples not reflecting truly dissolved concentrations (e.g., effects of colloidal fractions smaller than the 0.45 μm filter size).

[36] KowWIN software available online as part of EPA Estimation Program Interface suite of programs: http://www.epa.gov/oppt/exposure/pubs/episuite.htm.

6.2.1.1.2 Inorganic Chemicals

The fate and transport of inorganic species (e.g., metals) in pore water is defined by the distribution of the species between the aqueous and solid phases. A wide range of mechanisms control the distribution of metals between the aqueous and solid phases. Most commonly these mechanisms include precipitation/dissolution reactions and sorption/ion-exchange processes. Precipitation and dissolution are controlled by the concentration of species present both in solution and as mineral phases. Sorption and ion exchange are controlled by a variety of factors, including electrostatic attraction, covalent bonding, and weak intermolecular attractions such as van der Waals forces.

The distribution of inorganic species between the aqueous and solid phases is controlled by a number of mechanisms that are a function of the physical, chemical, and biological characteristics of the solid-aqueous system. The characteristics most important for the aqueous solution phase include the following:

- pH
- Oxidation-reduction potential (Eh)
- Presence of competing ions
- Aqueous complexation reactions
- Ionic strength and the specific ions in solution.

The solid phase characteristics of importance include the following:

- Grain size
- Composition/mineralogy
- Sorbed organic carbon content and type
- Surface characteristics such as charge, coatings, and area.

In addition, there is a range of factors that cannot easily be assigned to one phase, such as temperature and the fugacity of gases such as oxygen and carbon dioxide.

The aqueous-solid chemistry of the sediment and pore water environment can be strongly influenced by microbial processes. Microbial oxidation of labile organic carbon, both natural and anthropogenic, frequently depletes dissolved oxygen in pore water, resulting in chemically reduced conditions and the production of alkalinity. Further, under anaerobic conditions, microbial processes can induce numerous environmentally relevant changes to the chemical environment, such as dissolution of iron and manganese oxide minerals and production of sulfides.

A basic understanding of the fate and transport of inorganic species requires an understanding of the saturation state of aqueous chemical species in the system with respect to minerals that are composed of those species and that may precipitate or

dissolve, depending on the saturation state. For a given mineral phase, the solubility product, K_{sp}, defines the equilibrium condition between dissolved ions and the corresponding mineral. Generically, for the dissolution of a hypothetical mineral A_bC_d:

$$K_{sp} = [a]^b * [c]^d$$

Where,

[a] and [c] = the concentrations[37] of ions a and c in solution

b and d = the stoichiometric coefficient of each ion that is present in mineral phase (A_bC_d).

An excess of the ion activity products (i.e., right hand side of the equation) in solution relative to K_{sp} favors the precipitation of the mineral from solution, while a deficit of the ion activity products in solution favors the dissolution of the mineral to solution (provided the mineral phase is present). The solubility product defines the aqueous-solid phase condition under thermodynamic equilibrium. However, many reactions are kinetically limited and thus it is not uncommon for non-equilibrium conditions to exist in natural aqueous systems.

Sorption and ion-exchange mechanisms for metals can empirically be described by the K_d relationship described above; however, unlike for organic chemicals, the appropriate K_d term is not a function of f_{oc} (although organic matter can also sequester inorganic chemicals, thereby affecting the K_d value). Literature K_d values were compiled for the metal advective loading ICs: arsenic, copper, lead, and mercury. These values were used in the calculation of pore water concentrations for estimation of loading to surface water from surface sediment via groundwater advection. This analysis and the significant uncertainty associated with the inherent assumptions are presented in Section 6.1.6, and the range of literature K_d values is presented in Appendix E, Table E6-6. These values show ranges of 1 to 3 orders of magnitude. The wide range in literature K_d values for metals reflects the strong, highly variable geochemical factors, described above, that influence partitioning behavior in environmental systems. Considering this, literature K_d values should be considered site-specific estimates resulting from the geochemical conditions particular to individual studies. Limited site-specific empirical information, consisting of the filtered TZW data set with paired surface sediment samples for arsenic, copper, lead, and mercury, is presented for general comparison purposes on Figure 6.2-3.

[37] Rigorously, this refers to the activity of the metal in solution, which refers to the product of the thermodynamic activity coefficient and the concentration. For equilibrium environmental conditions (generally low concentrations relative to the water and solids), activity coefficients approach 1, making the concentration a good approximation of the product.

6.2.1.2 Degradation/Transformation Processes

A variety of abiotic degradation/transformation reactions, including hydrolysis, dehalogenation, oxidation, and reduction, can occur in aqueous systems. Hydrolysis is a well-known chemical reaction by which alkyl halides, esters, or ester analogs are split, and hydrogen and hydroxide (the components of water) are added to produce alcohols or organic acids (or both in the case of esters). Dehalogenation refers to reactions in which chlorine atoms are removed from alkyl halides. Oxidation and reduction are complementary reactions that involve the loss of one or more electrons (oxidation) by one chemical species and the gain of one or more electrons (reduction) by another. Metals in environmental systems are subject to both oxidation and reduction reactions, depending on the particular metal, its speciation in the environment, and other geochemical conditions. Of the organic chemicals on the combined IC loading list, PAHs, PCP, TPHs (aromatic and aliphatic components), DDx, and aldrin are subject to degradation/transformation by abiotic processes in the sediment/pore water environment. PCBs, BEHP, hexachlorobenzene, 1,2-DCB, chlordanes, dieldrin, and dioxins are also subject to abiotic transformation/degradation; however, abiotic processes in sediment and pore water are very slow for these chemicals. Many of the IC chemicals are subject to photo-oxidation in surface water, including PCBs (minor), dioxins (when near the water surface), aldrin, PAHs (especially LPAHs), PCP, and TBT, as discussed in Section 6.2.2.3.

Biodegradation is likely a significant process for some types of organic compounds in Study Area sediments and pore water. Biodegradation involves the metabolic oxidation or reduction of organic compounds and is carried out predominantly by bacteria in aqueous environments, but yeasts and fungi may also contribute to biodegradation. In general, oxidation of organic compounds occurs under aerobic conditions and reduction under anaerobic conditions, although both processes can occur under both conditions. Microbes may either gain chemical energy directly as a result of biodegradation of an organic compound, or during the process of co-metabolism, the concurrent degradation of another substrate with the organic compound. The chemicals on the IC list that can typically be transformed or degraded by microbially mediated processes include DDx, gamma-HCH, TBT, PCP, aldrin, PAHs, and to a lesser extent PCBs, BEHP, and hexachlorobenzene. Of the metals on the IC list, microbially mediated transformations are significant for mercury[38] and lead[39] organocompounds.

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[38] Mobilization of sorbed mercury can be caused by bioreduction to elemental mercury and bioconversion to more volatile and soluble forms, such as methylmercury. Methylmercury is the most hazardous mercury species due to its high stability, its lipid solubility, and its possession of ionic properties that allow it to readily pass through cellular membranes (Eisler 1987). Mercury discharged into rivers, bays, or estuaries can be converted into methylmercury compounds by natural biological (bacterial microorganisms) or chemical processes (Eisler 1987). The mercury methylation process depends on mercury loadings, microbial activity, nutrient content, pH and redox condition, suspended sediment load, sedimentation rates, and other variables; anaerobic conditions favor methylmercury formation more than aerobic conditions (Eisler 1987). Bacterial microbes are also responsible for methylmercury decomposition (demethylation).

[39] Of the organoleads, tetraethyllead and tetramethyllead are the most stable and the most important because of their widespread use as antiknock fuel additives. Both are clear, colorless, volatile liquids, highly soluble in

The biodegradation rate depends on the chemical structure and concentration of the organic compound, the concentration of bacteria responsible for the biodegradation, the availability of organic matter to serve as food and energy sources for bacterial growth, and physical and chemical conditions at the site, such as temperature and oxygen level. The extent to which the organic compound is bound to particles may also affect the biodegradation rate because the bound organic compounds may be biologically less available for microbial uptake. Biodegradation models commonly used include the Michaelis-Menten equation and the Monod equation, which account for chemical concentrations, microbial density, changes in uptake rates with changes in concentration, and changes in microbial density over time. By assuming constant biodegradation rates and bacterial density, the power rate law can be used to describe biodegradation empirically:

$$dC/dt = kC^n$$

Where,

C = chemical concentration

t = time

k = rate constant for biodegradation, determined experimentally in the laboratory or measured in the field

n = 0 for zero-order kinetics and 1 for first-order kinetics.

A wide variety of microbial species that use a large number of different biochemical pathways to metabolize anthropogenic chemicals have been identified. Biodegradation can proceed to full mineralization of the compound with end products of carbon dioxide and water, or an intermediate compound may be formed that is not easily biodegraded further. DDT, for example, is biodegraded to DDE relatively easily, but DDE is more persistent. The susceptibility of organic compounds to biodegradation depends on several factors, such as the presence and type of functional groups (oxygen- and nitrogen-containing groups increase biodegradation rates), the size and chemical structure of the organic compound (small molecules biodegrade more readily than large molecules), and solubility (more soluble organic compounds biodegrade more readily). A literature review has been completed to find appropriate biodegradation rate constants for use in the Fate and Transport Model; these were compiled in cooperation with EPA and will be presented in the final RI and discussed in the context of fate and transport findings.

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many organic solvents; however, solubility in water is only 0.18 mg/L for tetraethyllead, and 18.0 mg/L for tetramethyllead. Both undergo photochemical degradation in the atmosphere to elemental Pb and free organic radicals, although the fate of automotive organoleads has yet to be fully evaluated (Eisler 1988). In general, organolead compounds are more toxic than inorganic Pb compounds, food chain biomagnification of Pb is negligible, and younger organisms are most susceptible (Eisler 1988).

6.2.1.3 Sediment Physical Transport Processes

Many of the Portland Harbor ICs (e.g., PCBs, PCDD/Fs, pesticides including DDx compounds, and PAHs) are hydrophobic chemicals strongly associated with sediment particles, in particular cohesive or fine-grained particles (silts and clays); therefore, the physical transport of sediments, especially silts and clays, may impact the distribution and fate of certain ICs within the Study Area.

Sediment movement into, within, and through the Study Area occurs as suspended load in the water column and as bedload along the riverbed. Suspended particulate transport in the water column refers to the transport of solids suspended in the surface water. Cohesive or fine-grained sediments generally move as suspended load. Bedload transport of sediment refers to sediment transported along or very close to the riverbed (as described in Section 6.1.1.2). Noncohesive sediments (sands and coarser) typically move as bedload; however, a variable fraction of noncohesive sediments move as suspended load as a function of the flow regime (i.e., as flows increase, a larger fraction of noncohesive sediment will move in suspension).

The movement (resuspension and down gradient suspended load and bedload transport) of sediments in the LWR is controlled by the both natural and anthropogenic forces that affect water movement and the bottom shear stresses. As noted in Section 3, natural flow regimes vary dramatically (typically 10–20x annually) between the dry summers and rainy winters in Portland Harbor. Based on empirical, site-specific erosion velocities measured with Sedflume and modeled bottom shear stress, significant natural resuspension and movement of sediments does not generally occur at river flows below approximately 40–50,000 cfs (Section 3.1.4.2.3). Late spring through fall LWR flows are typically well below this level (see Figure 3.1-4), whereas late fall and winter flows exceed this threshold for variable lengths of time depending on the intensity of winter precipitation events in the Willamette Basin. This strong seasonal pattern in dynamics applies to the deep channel environment in the LWR which, on an areal basis, makes up much of the riverbed. Flows in nearshore, off-channel areas are severely dampened by nearshore structures, bottom drag, and shoreline configuration (e.g., sheltered embayments and slips). Map 3.1-11 shows that modeled bottom shear in many off-channel areas remains relatively low even during a river flow event of 160,000 cfs. This pattern is important because most of the areas of relatively high sediment contaminant concentrations in Portland Harbor are located in off-channel areas (Section 5.1).

In contrast to the channel environment, sediment disturbance/resuspension/scour in nearshore areas, particularly around working piers, berths, marine terminals, and others areas with significant boat traffic, may be largely a function of anthropogenic factors, such a prop wash, rather than natural factors. This effect may be accentuated during low flow portions of the year (late summer/fall) when river stage is also low (Figure 3.1-3) reducing vessel drafts. This anthropogenic influence is suggested by the time-series bathymetric measurements (Map 3.1-7), which reveal scour patterns in sheltered areas, such as Swan Island Lagoon and the International Slip, as well as very

close to shore in portions of the main stem that do not experience high bottom shear forces even during high river flows.

As described in Section 3.1 and summarized in Table 3.1-5, the physical character of the LWR transitions rather abruptly near the upstream end of the Study Area (about RM 10) from a relatively narrow, high velocity river characterized by coarse-grained riverbed channel sediments upstream to a broader, slower river dominated by fine-grained sediments downstream. This relatively wide, fine-grained character extends from RM 10 to the lower end of the RI Study Area at RM 1.9, with the exceptions of a distinct, narrow, higher energy reach between RM 5 and 7 and a small area at the head of the Multnomah Channel; both of these areas are dominated by sands (Map 3.1-2). Map 3.1-12 illustrates these overall patterns in surface grain-size gradients, measured areas of sediment scour and deposition from 2002 to 2009 (time-series bathymetry), and modeled predicted bed change during a major flood event. The three panels on Map 3.1-12 show consistent areas of erosion and deposition in the LWR during both typical (observed) and extreme (modeled) flow conditions, and these correspond to sand-dominated (erosion) and silt-dominated (deposition) reaches. Again, this pattern applies to the deeper, in-channel portions of the river and appears to reflect the influences of natural forces. Nearshore areas are subject to a more complex mix of natural forces and smaller-scale, anthropogenic factors, such as vessel traffic, river stage variations, and in-water construction/dredging and fill activities that affect localized sediment texture and resuspension/transport patterns.

Within the general framework of Study Area sediment transport regimes summarized above and detailed in Section 3.1, the major transport and fate processes relevant to sediment-bound chemicals are sediment transport into the Study Area from upstream, downstream sediment migration out of the Study Area (either in the main stem or Multnomah Channel), and the fate and transport of sediments within the Study Area, such as surface sediment mixing and resuspension, permanent burial at depth in the sediment column, and biological uptake. These processes are addressed below.

6.2.1.3.1 Sediment Flux into/out of the Study Area

Sediment enters Portland Harbor as suspended load carried in the water column and as bedload transported along or very close to the riverbed. Suspended and bedload sediment fluxes are discussed separately below.

Suspended Sediments

A significant volume of suspended sediment data has been collected in the LWR across a range of hydrologic conditions both by the LWG, as part of the surface water sampling program and as a HST model need, and by other entities (e.g., the City of Portland), as part of long-term monitoring programs. These data are described in Section 3.1.4.3 and show that suspended loads are strongly correlated with flow and accordingly vary widely (from approximately 5 to 50 mg/L) seasonally and annually (see Figures 3.1-16 and 3.1-18a–h). The data also suggest higher suspended loads on the rising limbs of the hydrographic events than on the falling limbs. Finally, a series of

in-situ suspended particle size measurements conducted for the hydrodynamic modeling data collection effort indicate that suspended sediment particles sizes are comparable throughout the Study Area, with a median percentile particle diameter between 15 and 30 µm (silt). In contrast, the median grain-size diameter at an upriver location (RM 18) was 78 µm (fine sand), reflecting the higher energy environment in the upper LWR (Figure 3.1-17).

The modeled (FS HST) suspended load fluxes into and out of the Study Area (entering at RM 11.8 and leaving in the main stem at RM 1.2 and down Multnomah Channel) are included in Table 6.1.3. Across the modeled flow years (5th, 25th, 50th, 75th, and 95th percentiles), average total suspended sediment flux into the Study Area from upstream equals about 1.53 billion kg/yr, and the average total suspended sediment flux out of the Study Area equals about 1.26 billion kg/yr, indicating a net accumulation of about 0.28 billion kg/yr in the Study Area. The model suggests, therefore, that averaged over time about 18 percent of the suspended material entering the harbor accumulates somewhere between RM 11.8 and 1.2. As discussed in Section 3.1.4.2 and presented in Table 3.1-6, the average annual net sediment accumulation rate calculated from empirical bathymetric survey data collected between 2003 and 2009 was 0.20 billion kg/year, which is in very good agreement with the model estimates, especially the 50th percentile model estimates of sediment influx and efflux, which correspond to a net accumulation of 0.19 billion kg/year. Net sediment accumulation represents a combination of new material entering the Study Area from upstream and some percentage of bedded sediment that is resuspended from the riverbed within the Study Area and then redeposited further downstream, but before exiting the Study Area. The relative magnitude of these suspended loads, their sediment quality, and the spatial distribution of deposition and resuspension over time determine the potential for recontamination/natural recovery in the harbor.

Bedload Sediments

Noncohesive bedload sediments move downstream along (rolling, sliding) or just above the riverbed (saltation) whenever near-bottom shear stresses exceed the threshold for sediment movement. Unlike suspended load, no direct measurements of bedload have been made as part of the RI/FS because it is extremely difficult to measure in the field. Additionally, as discussed previously, bedload is assumed to be a minor sediment transport mechanism due to the generally cohesive character of the sediment bed of the LWR. In light of that, and in consideration of the presumption that contaminants are not strongly associated with coarse-grained sediments, bedload processes are not quantified in the RI/FS fate and transport evaluation.

6.2.1.3.2 Sediment Fate and Transport in the Study Area

The spatial pattern and extent of deposition and erosion in the Study Area was measured directly with the time-series of precision bathymetric surveys conducted from 2002 to 2004 (described in Section 3.1.4.2). As noted above, the bathymetric change data show distinct zones of deposition and erosion that reflect the cross-sectional area and resultant hydrodynamics of specific portions of the Study Area (see Maps 3.1-10 and 3.1-8).

These hydrodynamic reaches, described in Section 3.1.5, share attributes such as sediment texture and general sediment stability.

Based on surface and subsurface grain-size (percent fines) distribution (Maps 3.1-2 and 3.1-4) and bathymetric features (Map 3.1-5), areas of fine-grained sediment accretion appear to be dominant from RM 8 to 10, along the channel edge from RM 4 to 5, and from RM 1.5 to 3. The downstream and upstream areas noted here are known to be long-term sediment accumulation areas based on historical dredging needs. Upstream depressions (borrow pits) between RM 9.5 and 11, that in combination span the navigation channel, likely capture some suspended and much of the bedload (noncohesive) sediments that are entering the system. The Study Area reaches between RM 5 and 7 and RM 10 and 11.8, where the river is relatively narrow, are dominated by areas of small-scale net erosion, as is the western off-channel area from RM 0 to 3 (outside bend of the LWR as it turns toward the Columbia).

The analysis of the time-series bathymetric change data presented in Section 3.1.4.2.2 indicates that during typical flow conditions only about 10 percent of the riverbed exhibited net bathymetric changes (erosion or accretion) greater than 30 cm, but that relatively "small-scale" scour or accretion from about 8 cm (the limit of bathymetric resolution) to 30 cm in extent was widespread. These data suggest the top 30 cm of the sediment column is relatively unconsolidated and more susceptible to resuspension/erosion than deeper sediments.

As described in Appendix H4.2.2, radioisotope cores were collected in upstream borrow pits at RM 10.5 and 10.9 in an attempt to quantify sedimentation rates in known depositional locations. Detailed evaluation of the radioisotope data from these cores is provided in Appendix H, Attachment H-5; however, generation of a timeline with depth was precluded as these depressions appear to be collecting a complex mix of suspended and bedload sediments with heterogeneous origins. The history of the borrow pits (i.e., the estimated original depths of the borrow pits when excavated in 1988 compared to the observed mudline elevations at the time of sampling in 2007) points to long-term average sedimentation rates in these low-lying portions of the channel of approximately 45 cm/yr (1.5 ft/yr) at RM 10.9 and 30 cm/yr (1 ft/yr) at RM 10.5 (see Appendix H, Attachment H-5). Additional empirical evidence on relatively long-term net sedimentation rates is provided by the harbor-wide measured riverbed elevation changes over the seven-year period from 2002 to 2009 (see Map 3.1-7). The net sediment accumulation rates in these upstream borrow pits at RM 10.9 and 10.5 based on these data are estimated to be approximately 41 and 31 cm/yr at RM 10.9 and 10.5, respectively, over this time-frame; this is consistent with the 19-year estimates noted above. The borrow pits themselves, spanning the channel in this reach and presenting a relatively larger channel cross-sectional area (because of their greater depth) than in reaches immediately upstream, provide conditions that promote sedimentation. However, based on the bathymetric survey data, sedimentation rates in this portion of the LWR outside the borrow pits, i.e., on the large shoal that occupies the western portion of the navigation channel from RM 8 to 10, are comparable in scale (31 cm/yr at

the maximum shoaling point at RM 9.6, see Map 3.1-10). This shoal area has historically required regular maintenance dredging (see Section 3.2.5.2).

The long-term sedimentation rate observations noted above apply to the Study Area channel environment. Based on bathymetric change, SPI data, and limited radioisotope sampling for monitored natural recovery (MNR) assessment (Anchor 2005), nearshore and off-channel areas do not appear to accumulate sediment at these rates. Short-term active sediment deposition and resuspension are indicated by these data sets, likely due in many areas to anthropogenic activity. Seasonal (rainy season) inputs of fine-grained sediments in areas adjacent to the channel are also evident. However, seasonal comparison of surface sediment textures at similar locations in the spring versus the fall suggests that some nearshore deposits can be remobilized over time and dispersed (WEST and Tetra Tech 2009), minimizing net accumulation rates. These observations are supported by the radioisotope data from four nearshore areas in 2004 (Anchor 2005), which show well-mixed surface sediment layers and calculated net sedimentation rates of approximately 1 cm/yr.

As illustrated in Map 3.1-12 and noted previously, the FS HST model prediction of bed elevation change for the high-flow flood scenario depicted in Figure 3.1-19 suggests that the spatial pattern of erosion predicted by the model for the extreme event is generally consistent with measured bathymetric change from 2002 to 2009 under more typical hydrologic conditions. However, in some areas, the magnitude of bed changes during the extreme event is dramatically greater, with erosion or deposition predicted to occur to one or more meters over observed changes (Map 3.1-12).

6.2.1.3.3 Surface Sediment Dynamics

Particles that settle out or move along the bottom are subjected to a wide range of physical, biological, and chemical processes:

- **Sediment mixed-layer turbation** – Biogenic mixing by benthic infauna or bottom-foraging fish can preclude or slow consolidation of surface sediments, as can natural (e.g., wind waves) and anthropogenic (e.g., prop wash) forces. These factors can greatly complicate the spatial and temporal degree of bed erodibility. The SPI survey conducted throughout the LWR in the late fall of 2001 revealed a complex mosaic of surface sediment processes in the top 22 cm of the sediment column (the maximum depth of the SPI images) across the Study Area (SEA 2002b). Areas of fine-grained, low-shear sediments contrasted with coarse-grained, more compacted bottom areas. In the channel environment, these large-scale gradients in gross characteristics coincided with and helped first define the hydrodynamic reaches described in Section 3. In some fine-grained areas, infaunal feeding pockets and worm tubes indicated that biogenic activity approached 20 cm depth. In other areas, minimal biogenic mixing activity was apparent. A well-mixed, biologically active zone appears to be on the order of 5 cm in many images, although this varied widely across the Study Area. Many nearshore areas showed steep onshore-offshore gradients in

physical and biological conditions as a function of water depth, riverbed slope, and/or the degree of shoreline protection (e.g., embayments, structures). In some areas, layers of freshly deposited sediments exceeding 10 to 15 cm in extent were apparent. This survey was conducted during the onset of the rainy season in late November. Overall, the SPI survey in combination with the bathymetric change data point to a dynamic surface sediment bed in much of Portland Harbor that is subjected to, depending on location and season, physical disturbance in the form of deposition or scour (on a multi-centimeter scale) due to natural and anthropogenic forces, biogenic mixing, and geochemical disturbance factors, such a methane bubble ebullition. Under typical (i.e., non-flood event) flow conditions, however, these disturbance factors appear to be limited to a "maximum" extent of the top 30 cm harbor-wide; and frequent, widespread physical and biological surface sediment mixing is likely restricted to much shallower depths over much of the Study Area.

- **Long-term sediment burial beneath the mixed layer** – Particles and associated chemicals that are advectively transported or buried below the mixed layer (less than or equal to 30 cm) are permanently removed from the active transport system throughout most of the Study Area. In portions of the navigation channel upstream of RM 10.5 and between RM 5 to 7, erosion of bedded sediments to about 2 m is predicted to occur during 100-yr flood events, but this deep erosion is very limited in areal extent (see Map 3.1-9b). Consequently, absent anthropogenic disturbance (e.g., dredging), bedded sediment below 30 cm in most of the Study Area is stable under the long-term hydrologic conditions anticipated for Portland Harbor.
- **Sediment ingestion/uptake by biota** – Filter and deposit feeder organisms may actively or passively ingest particles in suspension or on the sediment bed. High densities of filter feeders can biologically enhance transfer of suspended particles to the sediment bed. Also, chemicals associated with ingested particles can enter the food web.

6.2.1.4 Pore Water Physical Transport Processes

Chemicals in pore water are subject to diffusive and advective physical transport processes. These mechanisms are discussed in the following subsections.

6.2.1.4.1 Diffusive Transport

Diffusion is the movement of particles or dissolved chemical species from higher potential energy to lower potential energy (as represented by a difference in concentration in the case of diffusion from the pore water to the overlying water column). This is a spontaneous physical process that requires no additional energy inputs or expenditure. This mechanism is distinguished from advective transport of chemicals in pore water (described in the following section) in that it requires no driving force other than a concentration gradient. Diffusive transport acts on any IC in solution and is therefore potentially relevant to all of the combined loading list ICs.

Diffusive transport of ICs is being assessed for the Study Area as part of the Fate and Transport Model under development for the FS.

6.2.1.4.2 Advective Transport

Advective transport of ICs in the sediment/pore water environment refers to the aggregate movement of chemicals by flow of pore water through the sediments. Flow through the sediments to the water column occurs in the form of groundwater discharge. This mechanism is a transport process for chemicals in the surface sediment/pore water environment to migrate to the water column. As such, it is distinguished from the upland groundwater plume loading term, described in Section 6.1.5, which represents advective loading of chemicals from upland groundwater sources to the pore water, sediments, and water column. In some parts of the Study Area, both mechanisms are likely occurring simultaneously for chemicals present in upland plumes and in sediments from other sources. A detailed semi-quantitative estimation of Study Area-wide loading to surface water via groundwater advective transport is presented in Section 6.1.6 along with an analysis of the external loading of surface sediment via advective transport from subsurface sediment.

6.2.2 Surface Water Fate and Transport Processes

ICs in surface water are present in the dissolved phase and sorbed to suspended solids. Fate and transport processes for ICs in surface water, including chemical partitioning between surface water, air, and suspended sediment; physical transport of surface water and suspended solids; and physiochemical and biological processes are described below in Sections 6.2.2.1 through 6.2.2.3.

6.2.2.1 Chemical Distribution between Surface Water and Suspended Sediment

A general discussion of chemical partitioning processes that can be used to describe and/or predict the equilibrium distribution of a chemical between water (dissolved aqueous phase) and solid (sorbed to sediment or associated organic matter) was provided previously in Section 6.2.1.1 in the context of chemical distribution between pore water and bedded sediments. As described there, the tendency of a chemical to be associated with suspended solids or dissolved in the water column is described by the K_d term. For organic compounds, this term describes the combined effect of all possible driving forces affecting distribution between suspended sediment and surface water, including hydrophobic sorption into organic matter on the solid surface, electrostatic attractions of oppositely charged ionic functional groups, and covalent bonding or complexation of ionic organic molecules with reactive surface groups. For nonionic organic chemicals, which include most of the organic chemicals on the IC loading list (PCBs, pesticides including DDx, PCDD/Fs, PAHs, SVOCs, and VOCs), the primary driving force defining the K_d term is hydrophobic sorption, and K_{oc} is used to describe chemical behavior as a function of suspended sediment f_{oc}. As described in Section 5.3 (Figure 5.3-19), suspended solids in the Study Area during high-flow conditions exhibit lower relative f_{oc}, and the highest relative f_{oc} is observed on

suspended solids during stormwater events. For inorganic ICs in surface water (metals), mechanisms controlling partitioning behavior include precipitation/dissolution reactions and sorption/ion-exchange processes, and partitioning behavior is not considered to be influenced strongly by suspended sediment f_{oc}.

Figures 6.2-4 though 6.2-8 present the observed partitioning between surface water (filtered) and suspended sediment for the LWG-collected surface water samples for PAHs, DDx pesticides, PCDD/Fs, PCB homologs, and non-DDx pesticides. Limited site-specific empirical information for arsenic, copper, lead, and mercury is presented for general comparison purposes on Figure 6.2-9. These figures also show the literature K_{oc} and K_d values compiled for use in the advective loading assessment (Section 6.1.6). For most chemical groups (PCBs, pesticides, PAHs), the observed partitioning between suspended sediment and filtered surface water spans a wider range than the literature K_{oc} values. The source of this variability is unknown, but may be attributable to disequilibrium conditions between surface water and suspended sediment, errors introduced by the estimation method for the f_{oc} content of suspended sediment (see Appendix E2.2.1), filtered surface water samples not reflecting truly dissolved concentrations, or a combination of these factors. Based on visual inspection of Figures 6.2-4 through 6.2-9, the central part of the range of observed partitioning values corresponds with the CT in the literature values for PCB homologs, PCDD/Fs, and non-DDx pesticides. In contrast, the observed partitioning in the RI data set generally appears to be biased high relative to literature K_{oc}/K_d for PAHs and, to a lesser degree, DDx pesticides and metals. The high observed affinity of PAHs to organic carbon in suspended sediment could be explained by studies performed by McGroddy et al. (1995, 1996) on partitioning behavior of PAHs in sediments from Boston Harbor, which showed that only a small fraction of PAHs present in bulk sediment appeared available for equilibrium partitioning. Both their laboratory studies and *in situ* studies produced log K_{oc} values significantly higher than literature values, specifically for phenanthrene and pyrene. Conversely, their data for PCB congeners appeared to show complete equilibration in both the laboratory and *in situ* studies. The general conclusion made by McGroddy et al. (1995, 1996) is that PAH compounds associated with soot particles, as is typical in many coastal and estuarine areas, may be less available to exchange with the pore water than suggested by the literature. McGroddy et al. (1996) also state that in their studies, equilibrium partitioning models greatly overestimated both the pore water and the desorption aqueous-phase PAH concentrations, as much as 100 times in some cases. One implication of this finding on fate and transport modeling could be that modeled aqueous concentrations of PAHs will be significantly overestimated (potentially by up to two orders of magnitude) if they are based on literature values for K_{oc} and an assumption of equilibrium partitioning.

6.2.2.2 Physical Transport of ICs in Surface Water

Advection, the flow of river water in response to gravitational forces, is the primary mechanism for transport of surface water and its load of dissolved and particle-bound ICs. Water velocity and discharge are used to quantify river flow. Water velocity depends on the slope, shape, and physical characteristics of the riverbed and has the

dimensional units length/time (e.g., ft/s). Discharge, the quantity of water passing a given river location over a given time interval, is calculated as the average velocity times the cross-sectional area of the river and has the dimensional units volume/time (e.g., cfs or L/yr). The surface water mass flux of an IC is the product of the chemical concentration and the volumetric flow rate of the river, producing dimensional units of mass/time (e.g., kg/yr), as calculated in Section 6.1.1.1 for surface water load estimates.

The dominant direction of water flow in the LWR is downstream, along the hydraulic gradient. However, the flow direction reverses on flood tides during low-flow periods (see Section 3.1.3.2). Upstream flow has been identified as far upstream as RM 12.8 during low-flow conditions (Figures 3.1-13a–h; Caldwell and Doyle 1995).

Lateral and vertical movement of ICs in surface water occurs primarily as a result of turbulent (eddy) dispersion (mechanical mixing), and to a lesser extent as a result of mixing/diffusion resulting from chemical, thermal, and density gradients. The velocity of river water is greatest near the center of the river and decreases toward the sides and bottom of the river. These differences in velocity of different water parcels result in velocity shear, which gives rise to eddies. Eddies are also caused by channel irregularities, including structures in the water. These processes serve to mix the water and dilute ICs as they move away from the source. The suspended load of particle-sorbed ICs can also decrease due to particles settling out onto the riverbed sediment surface.

Some sources of ICs to surface water (such as industrial point discharges or groundwater plume discharge areas) can result in plume formation as the ICs mix with and diffuse into river water flowing downstream. Mixing patterns and plume sizes depend on differences in density between the effluent and the river water; the depth, velocity, and turbulence of the river; and any density stratification of the river itself. Density is a function of the temperature and salinity of the water.

Suspended particles provide an important vehicle for exchange of ICs between the sediment bed and surface water. Suspended particles can be derived from mineral sources, including eroded and weathered rock, or from organic sources, such as decaying plant material or plankton. The density of mineral particles is generally 2 to 3 g/cm^3, whereas the density of organic particles is close to the density of water (1 g/cm^3). The entrainment and settling of suspended particles are functions of river flow rate, particle size, particle shape, and particle density, as described in Section 6.2.1.3. The sediment-carrying capacity of river water increases with increasing stream flow and turbulence, which vary spatially as well as temporally. Stream flow, turbulence, and TSS loads are greater in areas where the river is narrower (e.g., upstream of RM 10), and throughout the river during high-flow events. Within the water column, concentrations of suspended particles generally decrease semi-logarithmically from the riverbed to the water surface. The empirical data set demonstrates these patterns, as described in Section 5.3. As seen in Figure 5.3-13, TSS in surface water across the Study Area increases with increasing flow rate; further, the range of TSS across

sampling events decreases by RM 2, where the river turbulence decreases and settling increases.

Because of the opposing tendencies of the particles to settle out and the water to entrain them, sediment entrained from the river bottom as bedload may be redeposited on the river bottom downstream. This process may disperse ICs in the sediment as they are transported downstream with the bedload; however, as discussed Section 6.1.1.2 and 6.2.1.3, bedload is not considered a major contributor to IC transport in Portland Harbor relative to the suspended solids load.

6.2.2.3 Physiochemical and Biological Attenuation Processes in Surface Water

Once ICs enter surface water, several physical, chemical, and biological processes (in addition to equilibrium partitioning, discussed separately above) can result in transfer of ICs between abiotic media (water, suspended solids, sediment, and air) or degradation/transformation of ICs. These processes include chemical precipitation, volatilization, abiotic degradation (chemical reaction or photolysis), and biodegradation. With the exception of volatilization and photolysis, these processes also generally pertain to pore water and sediment interactions and were discussed previously in Section 6.2.1.2.

Volatilization, the transfer of ICs dissolved in surface water to the atmosphere, is a process that transports ICs out of the water column. Volatilization is an important factor for small organic molecules, such as VOCs. The extent of volatilization of an IC depends on the temperature of the water and air and is a function of the concentration of the IC dissolved in water and its vapor pressure. The turbulence of the water and wind velocity at the air/water interface also affects the rate and extent of volatilization. Among the ICs on the combined loading list (Table 6.0-1), PAHs (particularly the LPAHs), SVOCs (1,2-DCB more so than BEHP, PCP, or HCB), chlordanes, gamma-HCH, aldrin, dieldrin, and all of the VOCs are subject to volatilization from surface water. A small fraction of the less chlorinated PCB congeners present in the dissolved form are also subject to volatilization out of the water column.[40] Additionally, various forms of mercury (e.g., elemental mercury [Hg^0] and methylmercury) and lead (e.g., tetraethyllead and tetramethyllead) are volatile from the water column.

[40] It should be noted that PCBs are subject to volatilization from surface water (especially the less-chlorinated congeners). A study conducted of the Lower Fox River between Lake Winnebago and DePere, Wisconsin (Steuer 1995) indicated that significant amounts of PCBs could be transferred to the atmosphere. The study predicted that over the course of 25 years, 1,300 kg of PCBs would be transported over the DePere Dam and 200 kg would be emitted to the atmosphere by volatilization. While the findings of this Wisconsin study are not directly applicable to the Study Area, since volatilization loss rates are a function of many site-specific conditions including water turbulence, temperature, wind speed, the fraction of PCBs associated with particulate material, etc., it is important to acknowledge this relevant loss mechanism. This mechanism will be considered in the fate and transport modeling work. Volatilization from surface water is discussed further in Section 6.2.2.3.

Henry's law constant is used to quantify the equilibrium partitioning of ICs between water and air:

$$H = [\text{conc. in air, mg/L}]/[\text{conc. in water, mg/L}]$$

In this formulation of H, the use of the same units for the concentration of the ICs in air and water results in a unitless constant. Henry's law constant is temperature-specific. Henry's law constants are being compiled for the fate and transport modeling effort under development for the FS.

Photolysis (degradation or transformation reactions that occur in response to solar irradiation) can occur either directly or indirectly. Direct photolysis involves the disruption of bonds in the molecule by electromagnetic radiation, particularly high-energy ultraviolet radiation. Organic compounds that are susceptible to direct photolysis include aromatic compounds and various chemicals with double bonds. Indirect photolysis involves the direct photolytic formation of a reactive species such as a hydroxyl radical or oxygen singlet, which subsequently reacts with an organic compound. Indirect photolysis can involve cleavage of aromatic rings, hydrolysis, hydroxylation, or dechlorination. The products of photolytic degradation of ICs are varied and may be either relatively non-toxic or toxic. The degree to which photolysis occurs is affected by the depth and turbidity of the water, and by the intensity and angle of incidence of light. Specifically, among the ICs, photolysis can be significant for aldrin, PAHs (especially LPAHs), PCP, TBT, and organolead compounds. Additionally, ICs associated with (sorbed to) labile organic carbon can be released to the water column through degradation of the dissolved/suspended organic matter. PCBs and PCDD/Fs are also subject to photolysis in surface water, though the process is considered to be minor for PCBs and is only relevant to PCDD/Fs near the water surface (EPA 1994).

6.2.3 Biota-Related Fate and Transport Processes

A number of processes govern how organisms living in the LWR are exposed to contaminants and how contaminants are transformed, excreted, or stored in tissue. Organisms living in the LWR take up contaminants through physically (e.g., diffusion), chemically, and biologically mediated processes, including transfer of waterborne chemicals across gill structures or other tissues, consumption of prey, or ingestion of sediment. Organisms can modify the chemical burden in their tissues through growth, reproduction, excretion, metabolic transformation, or sequestration. Some chemicals are transferred among organisms through trophic interactions, resulting in increases in concentrations of some chemicals at higher trophic levels.

Several of the contaminants (e.g., PCBs, pesticides, PCDD/Fs, and PAHs) that have been measured in invertebrates and fish from the Study Area are hydrophobic and likely to be highly associated with organic materials (i.e., lipids in tissues, dissolved or particulate carbon in the surface water, pore water, and sediment). Exceptions include some VOCs detected in localized areas of groundwater discharges, some more soluble

organic compounds (e.g., phenolic compounds, chlorinated benzenes), and some metals, which are more readily found in a dissolved or aqueous phase. However, some metals (e.g., lead and zinc) also tend to associate with organic and inorganic solids because the geochemical properties (e.g., ionic charge) governing their behavior tend to promote sorption.

Once released to the aquatic environment, chemicals enter the food web in a number of ways; the process is not sequential in that all trophic levels can interact with abiotic media. The behavior of chemicals within an aquatic food web is briefly described below.

Primary producers such as phytoplankton and plants take up chemicals mainly through diffusion from the water to which they are exposed, although the lipid content of phytoplankton also serves as a substrate for the partitioning and binding of organic compounds that may be in a dissolved state or bound to colloidal organic carbon. Phytoplankton metabolic byproducts contribute to the colloidal material in the water column, which can also bind with chemicals dissolved in the water column. These colloidal materials can be directly utilized by bacteria, other phytoplankton, and zooplankton; this serves as an additional chemical uptake and transfer mechanism for recycling chemicals within the water column food chain. Zooplankton prey upon phytoplankton and other zooplankton, further recycling chemicals in the water column. More complex aquatic organisms (e.g., invertebrates and fishes) can take up dissolved or colloidally bound chemicals in surface water and pore water across gill membranes, skin, and other permeable tissues, such as the mantle in clams (shells, exoskeletons, and scales are less permeable). Hydrophobic chemicals bind to sediment particles because of their organic carbon content. Sediment surfaces may be coated with bacteria and bacterial slimes, natural organic polymers, and other amorphous organic molecules that serve as binding sites. Finer-grained sediment (e.g., silt and clay) has a greater surface area to volume ratio and may therefore have a higher percentage of organic carbon (and thus hydrophobic chemical concentrations).

Once sediment or prey is ingested by invertebrates and fishes, chemical absorption across gut membranes is affected by the size of the molecule (larger molecules being more difficult to transfer across membranes), concentration gradients between gut content and surrounding tissues, acidity of the gut, and other physical/chemical conditions in the gut. Organisms can react with absorbed chemicals through various metabolic processes that result in a change in the chemical structure and properties.

Absorbed metals that are not excreted may end up stored in calcium carbonate matrices (invertebrates) or bone (vertebrates); these storage sites tend to reduce the reactivity of the metal. Organic compounds that are not metabolized tend to be stored in organs or fatty tissues, including gametes. These stores can be released within the aquatic and terrestrial food webs when these organisms are ingested by others, upon their death and decomposition, or by transfer to their offspring.

The relative contributions of bedded sediment versus surface water contamination to tissue contamination levels in the Study Area is a fundamental question that is to be addressed in the FS fate and transport modeling effort. Bioaccumulation modeling is part of that effort, and two kinds of bioaccumulation models are being used to predict chemical uptake by invertebrates and fishes (Windward 2009b). A mechanistic model (Arnot and Gobas 2004) that describes the bioaccumulation of hydrophobic organic chemicals is being used to evaluate bioaccumulation of selected COCs (i.e., total PCBs, TCDD TEQ, PCB TEQ, and pesticides including DDx) via water, sediments, and prey. Bioaccumulation of other substances, such as PAHs, is evaluated using a statistical approach based on biota-sediment accumulation factors (BSAFs) or biota-sediment accumulation regressions (BSARs). Under current conditions, the bioaccumulation model preliminarily determined that sediments are an important source of benthic invertebrate and fish tissue concentrations for the bioaccumulative compounds.

The bioaccumulation modeling and the overall FS fate and transport modeling effort is ongoing, and quantitative statements about the relative contribution of sediment versus other sources to tissue residues cannot yet be made.

7.0 DETERMINATION OF BACKGROUND CONCENTRATIONS FOR INDICATOR CHEMICALS

The approach used to determine the suite of background sediment and surface water concentrations reported here was developed with significant input from EPA on issues such as background/reference area definition; statistical methods, including outlier evaluation; and background uses for the RI/FS. The complete development of the approach is documented in a series of RI technical memoranda and associated EPA comment letters (Kennedy/Jenks et al. 2006; EPA 2006c; EPA 2008f,g; LWG 2008a,b). Specific direction provided by EPA on specific technical subtopics in the development of background levels for Portland Harbor, as well as general EPA CERCLA background guidance, are noted in the applicable subsections below.

The Portland Harbor Programmatic Work Plan (Integral et al. 2004) identified upstream sources, including upriver sediment and surface water, as potentially contributing to chemical concentrations in the Portland Harbor Superfund Site. These upstream sources influence regional background conditions that in turn influence chemical loads to, and concentrations within, the RI Study Area.

Background conditions are particularly salient in the case of Portland Harbor. This is because of the urbanized and industrialized setting of the region, and the fact that the lower portion of the river is influenced by many human activities occurring upstream across the broader watershed. Extensive details on the local and regional setting of the Study Area are provided in earlier sections of this report. This section extends upon that information by presenting a quantitative evaluation of background conditions upstream of the Study Area. This evaluation serves as the foundation from which relative comparisons can be drawn regarding chemical concentration within the Study Area versus those typical of regional conditions. This information, in turn, will be used to support the Portland Harbor FS, in which remedial alternatives will be developed and evaluated based on the findings of the risk assessment and in light of background chemical concentrations and chemical loads.

This section describes the identification of relevant background sediment and surface water data sets for the RI/FS, discusses the evaluation of those data sets for use in the RI/FS (including data quality considerations and identification of outliers), presents a statistical analysis of these data sets, and provides the complete, final RI background data sets in an electronic format.

Various statistical techniques—ranging from point values (e.g., upper-bound estimates of CT and upper background threshold values [BTVs]), to hypothesis testing to compare whether background and Site data are drawn from the same population—are available to compare background and site concentrations in the RI/FS process. The analysis presented here focuses on upper-bound estimates of CT (e.g., the 95th percentile upper confidence limit [UCL] on the mean) and upper BTVs (e.g., the 95th percentile upper prediction limit [UPL]). At the direction of EPA, the LWG developed

these estimates using the EPA statistical software package ProUCL Version 4.0 and its supporting technical guidance document (Singh and Singh 2007).

As described in the EPA Office of Solid Waste and Emergency Response (OSWER) guidance document, *Role of Background in the CERCLA Cleanup Program* (EPA 2002d), contamination at a CERCLA site may be due to releases from the CERCLA site itself, as well as contamination from other sources, including natural and/or anthropogenic sources that are not related to the site under investigation. According to the OSWER Guidance, background is a factor that should be considered in risk assessment and risk management at CERCLA sites. Consistent with this, the broad goal of a background evaluation in the context of an RI/FS is to estimate the levels of chemicals that would exist in environmental media at the site in the absence of CERCLA-related releases of hazardous chemicals from the site or releases from other point sources of contamination within the site.

The CSM for Portland Harbor, presented in Section 10 of this RI Report, identifies upriver sediment and surface water as sources of hazardous substances to the Study Area. Chemicals that are evaluated in the BHHRA and BERA have been detected in upstream environmental media collected during the RI and in previous investigations. In order to support the risk assessment, the FS process, and remedy selection for the Portland Harbor Site, background concentrations in upriver sediment and surface water need to be determined for those chemicals that may be found to pose unacceptable risks to human health and the environment within the Study Area.

The discussion presented in this chapter is organized as follows:

- Section 7.1 presents definitions, based on EPA guidance, that are relevant to the determination of background in the RI, along with a discussion of anticipated uses of background concentrations during the RI/FS process.
- Section 7.2 describes the process that was employed to generate appropriate data sets for characterizing background concentrations in bedded surface sediments and surface water (and specifically addresses the identification of chemicals for which background estimates are needed), reference area selection, data quality requirements, and data preprocessing.[1]
- Section 7.3 presents the background analysis for bedded surface sediments including outlier identification and development of estimates of CT and BTVs using ProUCL.
- Section 7.4 presents a parallel analysis for total and dissolved surface water background concentrations.
- Section 7.5 describes and summarizes supporting lines of evidence that may be useful for interpreting and applying background estimates in the context of the

[1] The "reference envelope" concept developed for the assessment of risk to benthic invertebrates is provided in the BERA and is not addressed in this section of the RI Report.

RI/FS, including upriver sediment trap data (RM 11 and 16), upper Study Area borrow pit sediment core profiles (RM 10.5 and 10.9), and suspended solids in the water column (RM 11 and 16).

7.1 DEFINITIONS AND USES OF BACKGROUND IN THE RI/FS PROCESS

The following EPA guidance documents were reviewed to assist in providing a consistent set of definitions, as well as recommended uses, of background data in the Portland Harbor RI/FS:

- *Role of Background in the CERCLA Cleanup Program* (EPA 2002d)
- *Guidance for Comparing Background and Chemical Concentrations in Soil for CERCLA Sites: Appendix B Policy Considerations for the Application of Background Data in Risk Assessment and Remedy Selection* (EPA 2002c)
- *Determination of Background Concentrations of Inorganics in Soils and Sediments at Hazardous Waste Sites* (EPA 1995)
- *ProUCL Version 4.0 Technical Guide* (Singh and Singh 2007).

To achieve a consistent understanding of the background approach, the following definitions provided in EPA (2002d) are adopted for the Portland Harbor RI/FS:

- **Background**—"Substances present in the environment that are not influenced by releases from a site and are usually described as naturally occurring or anthropogenic.
    1. ***Naturally occurring*** – substances present in the environment in forms that have not been influenced by human activity; and,
    2. ***Anthropogenic*** **–** natural and human-made substances present in the environment as a result of human activities (not specifically related to the CERCLA site in question)."
- **Reference Area—**"The area where background samples are collected for comparison with samples collected on site. The reference area should have the same physical, chemical, geological, and biological characteristics as the site being investigated, but has not been affected by activities on the site…. Background reference areas are normally selected from off-site areas, but are not limited to natural areas undisturbed by human activities."

Depending on the specific use of background information, several statistical tools are available for background evaluations in the RI/FS context. BTVs are often estimated using an upper percentile, a UPL, or an upper tolerance limit (UTL). BTVs can be applied in point-by-point comparisons of single concentrations measured within a site with the upper bound of the background concentration range. A BTV can also be used to define a "not-to-exceed" value that can be used in establishing PRGs (Singh and

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Singh 2007). In this Portland Harbor background evaluation, BTVs are provided using both the upper 95th percentile of the data set and the 95th percentile UPL (95 UPL)[2]; both these statistics are calculated based on the distribution of the collective data points. Another relevant statistic in background studies is the 95th percentile UCL (95 UCL) on the sample mean, which provides an upper-bound estimate for the range within which the true (unknown) population mean is likely to occur. The 95 UCL can be used, for example, to compare an average exposure point concentration (EPC) for an area of interest within a site—estimated using a 95 UCL on the mean exposure area concentration—with the background 95 UCL. Finally, where adequate data are available, parametric or non-parametric statistical hypothesis testing is generally the preferred approach for comparing concentrations from a site, or subareas of a site, with background concentrations.

For the Portland Harbor Site, several potential uses of background information have been identified:

- **Risk Characterization—**Background concentrations will be used for comparison purposes in the risk characterization section of the baseline risk assessment. Per EPA (2002d) direction, contaminants of potential concern[3] (COPCs) were determined where detected concentrations of COIs in the Study Area exceeded screening levels, regardless of the magnitude of background concentrations. EPA policy recommends an approach for baseline RAs that involves addressing site-specific background issues at the end of the RA process. Specifically, EPA (2002d) states that "the COPCs with high background concentrations should be discussed in the risk characterization, and if the data are available, the contribution of background to site concentrations should be distinguished." The 95 UCL of background concentrations is the primary background statistic discussed in the risk characterization sections of the BHHRA and BERA.
- **PRG Development—**Background values provide information that is relevant for risk management and establishing PRGs that will be evaluated in the FS. For example, if a risk-based threshold for a given chemical in sediment was determined to be 10 mg/kg, but the background sediment chemical concentration within the Site estimated from upstream chemistry was 100 mg/kg, the PRG would likely be set to background, because the risk level could not be achieved, assuming no attenuation of the background concentration. Various statistical techniques are available to compare background and Site concentrations; all may be relevant in the context of PRG development.

[2] Although the ProUCL 4.0 Technical Guide (Singh and Singh 2007) does not formally recommend the use of one BTV statistic over any other, the developers of ProUCL 4.0 express a preference for the use of the UPL or upper percentile value to perform point-by-point site versus background comparisons.

[3] Prior deliverables and some of the tables and figures attached to this document may use the term "chemical of potential concern," which has the same meaning as "contaminant of potential concern" and refers to "contaminants" as defined in 42 USC 9601(33).

- **Remedy Selection—**Comparison of background and site concentrations may be relevant in the context of remedy selection to evaluate whether post-cleanup chemical concentrations would be similar to background or to evaluate the relative risk reduction among cleanup options.
- **Long-term Monitoring Post Remedy—**Background values are one possible metric for evaluating remedy performance based on long-term monitoring results after the remedy is implemented, including but not limited to areas where monitored natural attenuation is the selected remedy.
- **Cap Material Selection—**Background levels such as the 95 UCL or 95 UPL could be among the criteria for selecting capping material.

Due to the diversity of potential uses of background information in the RI/FS, and the similar diversity in how background information may be applied to serve these uses, the remainder of this section of the RI seeks to provide a set of background tools that can be used, where appropriate, elsewhere in the RI/FS process. This includes development and provision of potentially applicable background data sets, preliminary identification of outlying values, statistical summaries of the background data sets (with and without outliers removed), and calculation of potentially applicable statistics including the 95 UCL on the mean and BTVs (95 UPL and 95th percentile). This information, while not intended to describe the universe of all potential approaches to and applications of background that may be used in the RI/FS process, provides a common foundation and context for describing regional background conditions upstream of the Study Area.

7.2 BACKGROUND DATA SET IDENTIFICATION

Identification of an appropriate background data set is a critical element of a CERCLA background evaluation and involves the overlapping considerations of which chemicals are relevant chemicals for background determination to support RI/FS objectives, the selection of a suitable reference area(s), specification of background data quality requirements, and data preprocessing to develop working background data sets for bedded surface sediments and surface water. Each of these elements is described in subsections 7.2.1 through 7.2.4, below. Identification and treatment of outlying data points that may reflect the influence of point sources of contamination and, therefore, may not be representative of true background conditions, is addressed in Sections 7.3 and 7.4 for sediment and surface water, respectively.

7.2.1 Contaminants Considered in the Background Analysis

All contaminants that are included in the background analysis, i.e., background ICs and their basis for inclusion, are presented in Table 7.2-1 for surface sediment and surface water. The background ICs are based primarily on contaminants that potentially pose an unacceptable risk to human health and the environment as identified in the BHHRA and BERA, and those chemicals in surface water and TZW sampling results that exceed drinking water and surface water quality criteria, without taking into account any spatial

or temporal averaging. The water screening methods and results are provided in Appendix D3.3 and Anchor QEA (2011).

7.2.2 Reference Area Selection

For the Portland Harbor RI/FS, the upriver reach of the LWR, extending from RM 15.3 to 28.4, was selected, in consultation with EPA, DEQ, and the tribes, as the reference area for determining background concentrations of bedded sediments (Maps 7.2-1a–b); sample data from this area were used to derive background values. This area, which extends from the upstream end of Ross Island (just upstream of the downtown Portland area) to approximately 2.5 miles above Willamette Falls, was chosen because it is considered broadly representative of the urban and suburban upland conditions along the banks of the LWR as it flows through Portland and its suburbs, but upstream and uninfluenced by releases from the Portland Harbor Site. Establishing an appropriate background data set in this context differs from settings in which an appropriate background data set is intended to represent "pristine" conditions. In urbanized and other developed settings, reference areas may be influenced by local point sources (e.g., shoreline industrial facilities and overwater structures), as well as by diverse non-point sources of chemicals (e.g., atmospheric deposition and storm runoff from a range of land use types). Procedures employed in this analysis to address these potential complexities in the reference area selected for Portland Harbor are detailed in Section 7.3.1.

For surface water, the LWG and EPA agreed that samples collected from surface water transects at RM 11 and RM 16 (Map 7.2-2) would be the basis for the background data set. Recognizing that RM 11 lies within the upper reach of the Study Area, special procedures were established to ensure that the combined RM 11 and RM 16 data sets represented the same population of upstream data, and that outlying values from RM 11 potentially indicative of a separate population were removed from the background data set. These procedures are discussed in Section 7.4.1 below.

7.2.2.1 Data Quality Requirements

Chemical concentrations in bedded sediments in the reference area have been the subject of LWG and non-LWG characterization efforts. Because an accurate background data set is of importance to project stakeholders, only those data meeting the stringent data quality requirements established for the baseline risk assessments (i.e., Category 1, QA Level 2) were considered for inclusion in the background data set.

Data that meet these criteria for bedded surface sediments in the reference area are available from the following LWG and non-LWG investigations:

- LWG Round 2A Sediment Sampling, 2004
- LWG Round 3B Sediment Sampling, 2007
- 2005 Portland District O&M Sediment Characterization

- Corps Dredged Materials O&M Sediment Characterization, 2004
- McCormick & Baxter RI Phase 3, 1999
- EPA Blue Heron & West Linn Paper Mill Site Investigations, 2007.

Individual bedded sample locations from these investigations and within the reference area are shown on Maps 7.2-1a–b.

For surface water, the background data set consists of total and dissolved surface water data collected by the LWG from transects at RM 11 and 16 (Map 7.2-2). Surface water samples from these two transects were collected during the following three surface water sampling events from the Round 2A sampling effort and four surface water sampling events from the Round 3A effort:

- November 2004 (Round 2A, Low Flow)
- March 2005 (Round 2A, Low Flow)
- July 2005 (Round 2A, Low Flow)
- January 2006 (Round 3A, High Flow)
- September 2006 (Round 3A, Low Flow)
- November 2006 (Round 3A, Stormwater-Influenced Low Flow)
- January 2007 (Round 3A, High Flow).[4]

The Round 2 and Round 3 surface water sampling program was designed to characterize chemical concentrations under low-flow (<50,000 cfs) and high-flow (>50,000 cfs) regimes. The timing of sample events against the river hydrograph was presented previously in Section 5.3.1. The November 2006 stormwater-influenced sampling event was considered a low-flow event for this background analysis. Surface water indicator chemical concentrations from RM 11 and 16 were evaluated to determine chemical concentrations representative of low-flow and high-flow river conditions specific to upstream of the Study Area. Additional details of the surface water sampling events, including the sampling methods specific to each transect location and event, are discussed in Section 5.3 of this report.

7.2.3 Measurement Basis for Surface Sediment Background Estimates

Background values for bedded surface sediment were estimated on a dry weight basis and, for hydrophobic organic chemicals, also on an OC-normalized basis. OC normalization is important because hydrophobic organic chemicals are primarily associated with (i.e., adsorbed to) the organic carbon fraction in sediment. The

[4] The January 2007 high-flow event involved sampling at only three stations (W023M, W024, and W025M) due to an unexpected change in flow conditions. Sampling was suspended and recommenced on February 21, 2007 once high-flow conditions (>50,000 cfs) were once again observed.

bioavailability of organic chemicals is inversely related to sediment organic carbon content (i.e., if a high organic carbon sediment and low organic carbon sediment have the same dry-weight sediment concentration of an organic chemical, the bioavailability of that chemical will be lower in the high organic carbon sediment than the low organic carbon sediment). The summary statistics presented in Section 5 show that both organic carbon and percent fines are higher, in the aggregate, in Study Area sediments (Table 5.1-1) than in the upriver reach (Appendix H, Table H4.2-1). For this reason, background estimates using OC-normalized sediment data for organic chemicals may provide a more meaningful basis for comparing site concentrations to background than background estimates using dry-weight concentrations. OC normalization was performed in accordance with the procedures developed for the BERA and described in Table 2.1-3 of this RI Report.

Further, because sediment remediation goals for Portland Harbor will ultimately be expressed on a dry-weight basis, the dry-weight background values for nonpolar, hydrophobic organic chemicals were also adjusted to reflect the differences between the mean organic carbon content of surface sediments in the background (RM 15.3–28.5) reach and the Study Area. These estimates, termed OC-equivalent dry-weight values, were calculated as follows:

$$C_{dw,eq} = C_{dw,\,bgrnd} \times \frac{TOC_{SA}}{TOC_{bgrnd}}$$

Where,

$C_{dw,eq}$ = OC-equivalent dry-weight sediment concentration
$C_{dw,\,bgrnd}$ = Dry-weight background sediment concentration
TOC_{SA} = Study Area surface sediment mean TOC (1.71%)
TOC_{bgrnd} = Background surface sediment mean TOC (1.11%).

7.2.4 Data Preprocessing

Preprocessing of the background data sets was needed to address field replicates, remove high-biasing non-detect results, and incorporate non-detect values in the calculation of multiple-constituent analytical totals (e.g., total PAHs and total PCBs). For organic chemicals in sediment, additional preprocessing of sediment data was required to create OC-normalized data sets. Additional preprocessing steps were required for surface water collected at RM 11 and 16, as the surface water samples from these locations were collected using a range of sampling approaches. Each of the data preprocessing steps is explained in the subsections below.

7.2.4.1 Field Replicates

Field replicates reported in the background sediment data set were averaged to provide a single reported value for each chemical constituent. This was done to avoid introducing spatial bias into the data set by “double-counting” replicates from the same

station. In contrast to sediment, surface water field replicates, which were collected one or more days apart during the same sampling event, were treated as distinct results and were not averaged, since concentration differences in surface water samples collected at different times and different hydrological conditions in a flowing river are expected to provide distinct snapshots of temporal variability in surface water concentrations and are not expected to introduce spatial bias.

7.2.4.2 High-Biasing Non-Detects

Consistent with EPA (1989) and EPA comments on the Round 2 Report (EPA 2008d), non-detect results with a reporting limit higher than the highest detected result for a given analyte in the surface sediment and surface water background data sets were flagged as high-biasing non-detects and were excluded. The number of high-biasing non-detects for each analyte or analytical sum is provided in Tables 7.2-2, 7.2-3, and 7.2-4a–c.

7.2.4.3 Summing Rules for Multiple-Constituent Totals

Chemical concentrations for multiple-constituent analytical totals were calculated using the rules established for the baseline risk assessments. Specifically, detected constituents were included at their reported concentrations. Non-detects were included at one-half of the reporting limit for those analytes that were detected at least once in the background data set. Chemicals that were never detected in a given background data set, i.e., sediment or surface water, were excluded from the multiple-constituent analytical totals. (Sediment and water were evaluated separately with respect to frequency of detection.) Finally, if all analytes contributing to a sum were not detected in a given sample, then the highest reporting limit for any of the individual constituents within the given sample was reported for the total and qualified with a non-detect flag (i.e., U-qualified).

7.2.4.4 Surface Water Subaveraging

This section describes the additional preprocessing of surface water concentration data at RM 11 and 16 in support of the background analyses, including procedures for arriving at single chemical concentrations at multi-sample transect locations, i.e., data "subaveraging." Subaveraging refers to the process used to generate a single average chemical concentration at a transect location where multiple samples were collected during a given sampling event. Additional details on the subaveraging procedures applied to surface water samples are provided in Appendix E2.2.2.

All RM 11 and 16 transect surface water samples were collected as VI sample composites from multiple lateral substations across the width of the river channel. Transect sampling is designed to estimate integrated water concentration through a cross section of the river or fraction of a cross section at a point in time. The transect sample concentration data in the background surface water data set comprise three different sample collection techniques:

- **EDI sampling**—samples were vertically and horizontally integrated over the entire cross section of the river. EDI samples were collected at RM 11 during Round 2A.
- **Vertically Integrated: East-Middle-West (VI-EMW) sampling**—the cross-river transect was sampled at three discrete points: east bank, middle, and west bank. Each east, middle, and west sample is vertically composited over the depth of the river. VI-EMW samples were collected at RM 11 during Round 3A.
- **NB/NS**—samples were collected from two vertical points in the water column, and integrated horizontally across the width of the river transect. The NB sample was collected at a depth of 1 ft off the river bottom. The NS sample was collected 3 ft below the surface. NB/NS samples were collected at RM 16 during Round 3A.

Subaveraging of the VI-EMW and NB/NS total and dissolved surface water data was performed to generate a single chemical concentration for each individual transect (RM 11 and RM 16) and sampling event. For those locations where field replicate samples were collected, the individual replicates were also subaveraged.

7.2.5 Preliminary Background Data Sets and Summary Statistics

Upon completion of all the data preprocessing steps described above, electronic flat files containing the dry-weight sediment, OC-normalized sediment, and surface water background data sets were developed. The flat files also include flags for potential and primary outliers in the data sets identified as described in Section 7.3.1 and 7.4.1, below. The flat files are provided on the CD accompanying this chapter of the RI Report.

Summary statistics for the entire dry-weight sediment and OC-normalized sediment background data sets, prior to the outlier disposition described in Section 7.3 below, are provided in Tables 7.2-2 and 7.2-3, respectively. Summary statistics for the surface water background data set, prior to the outlier disposition described in Section 7.4 below, are provided in Table 7.2-4a (total concentration), 7.2-4b (dissolved concentration), and 7.2-4c (particulate concentration).

7.3 BEDDED SURFACE SEDIMENT BACKGROUND OUTLIER DISPOSITION AND STATISTICAL ANALYSIS

This section addresses the identification and disposition of outliers in the dry-weight and OC-normalized sediment background data sets, presents summary statistics for the resulting data sets, and describes the statistical procedures and resulting estimates of sediment background CT (95 UCL) and BTVs (upper 95th percentile and 95 UPL). For the reasons described in the subsections below, outlier identification and disposition in the context of establishing background for the Study Area relies on multiple lines of

evidence and explicitly takes into account the diversity of natural and anthropogenic chemical inputs to the upstream watershed that define regional background conditions.

7.3.1 Sediment Outlier Identification

A key element of developing an appropriate background data set is to ensure that the data set is as free as possible of data points that are not representative of the background conditions of interest for a given project. In many background evaluations, a basic assumption is invoked that an appropriate background data set should consist of a single statistical population that represents natural background conditions (i.e., samples obtained from a reference area that has not been influenced by releases from the site or other known point sources of contamination). In practice, however, and particularly in instances when sites are located in regionally developed areas, natural background conditions may no longer exist, and cannot be known with certainty.

In addition, the assumption that an appropriate background data set should represent a single population may not be valid for background data sets that are obtained from urbanized or other developed settings. Such reference areas may be influenced by local point sources (e.g., shoreline industrial facilities and overwater structures) as well by diverse non-point sources of chemicals (e.g., atmospheric deposition and storm runoff from a range of land use types), resulting in the possible presence of high-biasing outliers that are not representative of background. As a result, identification and removal of outliers from the background sediment data set for Portland Harbor is more complex than in many other settings. The ProUCL Technical Guide (Singh and Singh 2007) explicitly recognizes that this type of complexity may exist in many CERCLA contexts and, therefore, provides guidance on the use of professional judgment in the identification and disposition of high-biasing outliers:

> [T]he decision regarding the proper disposition of outliers (e.g., to include or not to include outliers in statistical analyses; or to collect additional verification samples) should be made by members of the project team and experts familiar with site and background conditions.

And:

> In order to assess the influence of outliers on the various statistics (e.g., upper limits) of interest, it is suggested to compute all relevant statistics using data sets with outliers and without outliers, and compare the results. This extra step often helps to see the direct potential influence of outlier(s) on the various statistics of interest (e.g., mean, UPLs, UTLs). This in turn will help the project team to make informative decisions about the disposition of outliers. That is, the project team and experts familiar with the site should decide which of the computed statistics (with outliers or without outliers) represent better and more accurate estimate(s) of the population parameters (e.g., mean, EPC, BTV) under consideration. Since the treatment and handling of outliers is a controversial and subjective topic, it is suggested that the outliers be treated on a site-specific basis using all existing knowledge about the site and the site background (e.g., EA, area of concern [AOC], reference area) under investigation.

To support decisions about the disposition of outliers in the Portland Harbor RI/FS process, and in keeping with guidance by Singh and Singh (2007), outlier identification was performed in two steps: 1) identification of *potential outliers* using classical statistical and graphical analysis tools available in ProUCL, and 2) further investigation of all potential outliers using multiple lines of evidence to identify *primary outliers* that are determined to be unrepresentative of background conditions and should be removed from the background data set. (Note: the outlier identification process described here addresses only potential high-biasing outliers and does not consider the possible existence of statistical outliers at the lower end of the background concentration range.) Additionally, to provide members of the project team with information on the impact of outliers on background estimates, background statistics (i.e., 95 UCL, 95 UPL, and upper 95th percentile) are provided in this chapter for the full background data sets (i.e., with all potential outliers included) and with primary outliers removed.

7.3.1.1 Identification of Potential Outliers

Classical statistical outlier tests are best used to aid in identifying potential outliers that require additional investigation, and should be used accompanied with graphical displays including quantile-quantile (Q-Q) plots and box-whisker plots. Final outlier decisions should be based on review of all relevant information (see *Identification of Primary Outliers,* below) to determine the actual disposition of potential outlying values.

ProUCL includes the Dixon and Rosner tests for outlier identification but notes that those tests are strictly appropriate for normally-distributed data sets only. Given the right-skewness of many environmental data sets, the assumption of normality is frequently violated, and application of the Dixon and Rosner tests may result in numerous false-positive outlier identifications. As such, these tests are appropriate only for preliminary identification of *potential* outliers and not positive confirmation of actual outliers. The Dixon or Rosner outlier test was run on all (non-transformed) dry-weight and OC-normalized data sets, with non-detects set at one-half the reporting limit; ProUCL automatically selects either Dixon's or Rosner's test based on sample size (Rosner's for $n \geq 25$, Dixon's for $n < 25$). All potential outliers identified using the Dixon or Rosner test are listed in Table 7.3-1 (dry-weight basis) and 7.3-2 (OC-normalized basis).

Graphical review of the data was conducted using box-whisker plots, normal Q-Q plots[5] with non-detects set at the full reporting limit, and river mile concentration plots. These graphical tools are shown in Figures 7.3-1 through 7.3-128 for all background sediment chemical concentrations on both a dry-weight and an OC-normalized basis (excluding

[5] On a normal Q-Q plot, normally distributed data plot as a linear pattern. Right-skewed (e.g., lognormally distributed) data plot as an upward-curving pattern. Sharp breaks in slope and/or observations at the upper end of the quantile range that are well separated vertically from the majority of values on a Q-Q plot may indicate that outliers are present and/or that more than one statistical population is present in the data set.

metals). On these figures, potential outliers identified using the Dixon or Rosner outlier test are shown with red symbols.

7.3.1.2 Identification of Primary Outliers

The relative magnitude of each potential outlier identified as described above was evaluated further, on a weight of evidence basis, to determine whether the data points should be considered primary outliers and be removed from the data set. A weight-of-evidence approach is appropriate in recognition of the fact that the treatment and handling of outliers is a site-specific decision, based on all existing knowledge about the site and the background data set under investigation, as discussed previously in Section 7.3.1.

For potential outliers at locations proximal to known or potential point sources (e.g., paper mills, overwater structures) and where chemical evidence suggested the probability of a release from that source, all related compounds were removed from the data set regardless of their magnitude. For example, if chemical evidence indicates the presence of one or more potential outliers for individual PCB congeners or PAHs, then all PCB or PAH data for that station were removed from the data set. Source proximity resulted in the primary outlier identifications tabulated below:

Station	Dry Weight Basis	OC-Norm Basis	Proximal Source
WR10SD	- All PCDD/F congeners - TCDD TEQ	- All PCDD/F congeners - TCDD TEQ	Paper mills are probable point sources of dioxins/furans upstream of Willamette Falls. All dioxin/furan congeners and TCDD TEQ removed from background data set.
WR09SD	- All PCDD/F congeners - TCDD TEQ	- All PCDD/F congeners - TCDD TEQ	Paper mills are probable point sources of dioxins/furans upstream of Willamette Falls. All dioxin/furan congeners and TCDD TEQ removed from background data set.

Station	Dry Weight Basis	OC-Norm Basis	Proximal Source
WR08SD	- All individual PCDD/F congeners - All individual PCB congeners - TCDD TEQ - PCB TEQ - Total TEQ - Total PCB congeners - Total PCBs (combined)	- All individual PCDD/F congeners - All individual PCB congeners - TCDD TEQ - PCB TEQ - Total TEQ - Total PCB congeners - Total PCBs (combined)	W. Linn Paper Mill is a probable point source of dioxins and PCBs. PCBs, dioxin/furans, and TEQs removed from background data set.
UG04B	- All individual cPAHs - Total PAH - Total cPAH - Total LPAH	- All individual cPAHs - Total PAH - Total cPAH - Total LPAH	Suspected cPAH source associated with residential boat dock. All individual cPAHs, total cPAH, and total PAH removed from background data set.

For potential outliers that could not be tied to a known or suspected source, several lines of evidence were considered in a best professional judgment evaluation of primary outliers, including the following:

- The presence (or absence) of sharp breaks in slope and/or well-separated observations at the upper end of the quantile range on a Q-Q plot
- Co-occurrence of potential outliers for multiple chemicals at single stations
- The magnitude of the potential outlier compared to the full data set, expressed as the outlier:mean ratio; potential outliers with an outlier:mean ratio approaching an order of magnitude were examined closely in conjunction with other lines of evidence to assess whether the value represents a primary outlier
- Variability in chemical concentrations at closely clustered locations or between field replicates; spatial clusters of potential outliers provide strong evidence of a localized chemical source, while spatial heterogeneity in concentrations over a small spatial scale suggests that the potential outlier could simply reflect the heterogeneity in background concentrations expected in suburban/urban river systems.

This evaluation was conducted by visual examination and spatial analysis using the river mile plots, box-whisker plots, and Q-Q plots shown on Figures 7.3-1 through 7.3-128 and the mapped distribution of potential outliers by station shown in Maps 7.3-1 and 7.3-2, as well as consideration of the outlier:mean concentration ratios provided in Tables 7.3-1 (dry weight) and 7.3-2 (OC-normalized). This lines-of-evidence evaluation resulted in the identification of additional primary outliers that, while not linked to known or suspected sources, do not appear to be representative of the background data set. These additional primary outliers are tabulated below along with the rationale for their identification:

Station	Dry Weight Basis	OC-Norm Basis	Rationale/Lines of Evidence
WR11SD	--	- PCB TEQ	High outlier:mean ratio for PCB TEQ (OC-normalized); Distinct separation on the Q-Q plot
BH04SD	--	- Benzo(a)pyrene	Very high outlier:mean ratio (OC-norm only)
BH03SD	--	- 1,2,3,6,7,8 HCDD	High outlier:mean ratio (OC-norm only); Distinct separation on the Q-Q plot
U5Q-1	- Alpha-hexachlorocyclohexane	- Alpha-hexachlorocyclohexane	Very high outlier:mean ratio

Station	Dry Weight Basis	OC-Norm Basis	Rationale/Lines of Evidence
WR04SD	- 1,2,3,4,7,8-HCDF - Dibenzo(a,h) anthracene	- 1,2,3,6,7,8 HCDD, 1,2,3,4,7,8-HCDF, 2,3,4,7,8-PeCDF, 2,3,7,8-TCDF - PCB 118, 156, and 157 - TCDD TEQ - Total PCB congeners - Dibenzo(a,h) anthracene - Total PCBs (combined) - Total TEQ.	High outlier:mean ratios for dibenzo(a,h)anthracene and selected PCDD/Fs and PCB congeners (OC-norm only), TCDD TEQ (OC-norm only), total PCB congeners (OC-norm only), and total PCBs (combined); Multiple chemicals at single station
U1C-3	- Bis(2-ethyhexyl) phthalate	- Bis(2-ethylhexyl) phthalate	Very high outlier:mean ratio
D09942	- Total LPAHs	- Total LPAHs	High outlier:mean ratio.
WL01SD	- 4-Methylphenol	--	High outlier:mean ratio (dry weight only).

Tables 7.3-1 and 7.3-2 indicate the full set of primary outliers that were identified and removed from the background data sets. Primary outliers are also circled on the river mile concentration plots provided in Figures 7.3-1 through 7.3-128. Finally, Maps 7.3-1 and 7.3-2 show the mapped distribution of the potential outliers and the primary outliers on a dry-weight and OC-normalized basis, respectively.

In discussions held during the fall of 2008 regarding identification of primary outliers, the LWG and EPA reached different conclusions in the case of two chemical groups—total PCB Aroclors and total DDx. Specifically, the LWG concluded that the four potential outliers for total PCB Aroclors in the vicinity of RM 16 and 17 (Figure 7.3-17 and Map 7.3-1) do not rise to the level of primary outliers, because 1) the outlier:mean ratios are relatively low (ranging from 3.76 to 6.09); 2) samples collocated with and nearby the potential outlier locations are significantly lower, indicating a high degree of spatial heterogeneity in this reach; and 3) no local source of PCB releases to this reach has been identified. In contrast, the EPA concluded that the potential outliers may indicate the influence of a local, albeit unknown, PCB release that may be addressed (i.e., cleaned up) in the future. For total DDx, the LWG concluded that the two potential outliers located near Cedar Island upstream of RM 23 (Figure 7.3-53 and Map 7.3-1) are not potential outliers for the same set of reasons identified above for PCB Aroclors, whereas EPA concluded that these two potential outliers may reflect the influence of an unknown localized DDx release that may be addressed in the future. To resolve these differences, EPA and LWG agreed (Wyatt 2008, pers. comm.) that the background analysis in the RI will present background estimates both with (LWG case) and without (EPA case) these potential outliers retained in the data set. Another element of the resolution is that EPA and DEQ will work to identify what specific point sources may have influenced PCB concentrations in the RM 16 to 17 reach and total DDx concentrations in the vicinity of Cedar Island.

Summary statistics for the dry weight and OC-normalized sediment background data sets, reflecting the removal of primary outliers identified above, are provided in Tables

7.3-3 and 7.3-4, respectively. As described above, each of these tables provide two sets of summary statistics (LWG case and EPA case) for total PCB Aroclors and total DDx, reflecting different treatments of primary outliers for these two multi-constituent chemical sums.

7.3.2 Upper-Bound Average and Background Threshold Value Estimates for Background Sediment

Estimates of upper-bound background CT (95 UCL) and BTV (95 UPL) were generated in ProUCL Version 4.0, as outlined below:

a. Upper-Bound Central Tendency Estimates

   i. Import data set at ND=DL.

   ii. Use ProUCL to calculate the 95th percentile upper confidence limit on the mean (95 UCL) or other appropriate central tendency statistic (e.g., 97.5 UCL) as recommended by ProUCL. Because all data sets contained multiple detection limits and/or were nonparametric, the Kaplan-Meier statistic recommended by ProUCL for the appropriate underlying distribution was selected.

b. Background Threshold Values (Upper Prediction Limits)

   i. Import data set at ND=DL.

   ii. Use ProUCL to calculate the 95th percentile upper prediction limit (UPL95). Because all data sets contained multiple detection limits and/or were nonparametric, the 95% Kaplan-Meier UPL was selected in all cases, as recommended by ProUCL.

Because calculation of the upper 95th percentile value is not available in ProUCL, the Statistica software package (StatSoft 2005) was used to calculate the upper 95th percentile BTV estimate.

Tables 7.3-5a–b present the calculated values of the upper threshold and upper-bound CT statistics for background sediments on a dry weight basis for two cases—with outliers included (all data) and with primary outliers removed. Parallel sets of results for the OC-equivalent dry weight and OC-normalized sediment concentrations are presented in Tables 7.3-6a–b and 7.3-7a–b, respectively. As described above, Tables 7.3-5b, 7.3-6b, and 7.3-7b provide two sets of summary statistics (LWG case and EPA case) reflecting different treatments of primary outliers for total PCB Aroclors and total DDx.

7.4 SURFACE WATER BACKGROUND OUTLIER DISPOSITION AND STATISTICAL ANALYSIS

This section addresses the identification and disposition of outliers in the surface water background data set, presents summary statistics for these refined data sets, and describes the statistical procedures and resulting estimates of surface water background CT (95 UCL) and BTVs (upper 95th percentile and 95 UPL). An analysis of the distribution of selected chemicals between the dissolved and particulate phases and the dependence of these concentrations on flow conditions is presented in Section 5.3 of the RI Report; specifically including the background data set at RM 11 and 16.

7.4.1 Surface Water Outlier Identification

The upstream surface water background data set includes transect sample results collected at RM 11 and 16, subaveraged as described previously in Section 7.2.4 (field replicate, VI-EMW, and NB/NS discrete samples were subaveraged prior to generating the background data sets, as described above). To ensure that the combined RM 11 and 16 data sets represented the same population of upstream data, a graphical comparison of the surface water concentrations (total basis) from both transects was conducted. The RM 11 and 16 analyte concentrations were analyzed on a chemical-by-chemical basis to determine the following:

- Surface water background chemical concentrations to be combined for RM 11 and 16
- Surface water background chemical concentrations to be combined for RM 11 and 16 following exclusion of outlying samples.

To aid in identifying high-concentration samples that represent potential outliers in the surface water data set, Figures 7.4-1 through 7.4-27 present bar chart graphs of RM 11 and 16 chemical concentrations (total basis) for discrete sample concentration data (prior to subaveraging of field replicate, VI-EMW, and NB/NS samples); subaveraged field replicate, VI-EMW, and NB/NS concentrations; and scatter plots showing the final background data sets, grouped by high-flow and low-flow events. The first chart for each chemical, showing the data prior to any subaveraging, was visually analyzed to identify high-concentration samples that were not likely to be representative of conditions upstream of the Study Area. In particular, discrete VI samples collected at the RM 11 east station and RM 11 EDI samples were scrutinized for outlying total concentration values potentially influenced by stormwater discharge that was observed by the sampling crew during sample collection on the east bank of the river near RM 11. Best professional judgment was applied to identify discrete VI samples collected at the RM 11 east station and RM 11 EDI samples that exhibited notably higher total concentrations than other RM 11 or RM 16 samples for a given analyte; these outlying samples excluded from all upstream background calculations are presented in Table 7.4-1. The outlying values are also circled on Figures 7.4-1 through 7.4-27, as applicable. The second chart for each chemical shows the subaveraged

concentrations with and without outliers removed; subaveraged concentrations reflecting the removal of outliers are shown in a checkered pattern. The third chart for each chemical presents these concentrations as scatter plots that are grouped by high-flow and low-flow conditions.

Summary statistics for the surface water background data sets, reflecting the removal of RM 11 outliers identified as described above and summarized in Table 7.4-1, are provided in Table 7.4-2a (total concentration basis),7.4-2b (dissolved concentration basis), and 7.4-2c (particulate concentration basis).

7.4.2 Upper-Bound Average and Background Threshold Value Estimates for Background Surface Water

Estimates of upper-bound background CT (95 UCL) and BTVs (95 UPL and upper 95th percentile) were generated in ProUCL Version 4.0 and Statistica, as described above in Section 7.3.2. Upper threshold and CT statistics for background surface water with outliers included (all data) are presented in Table 7.4-3a (total concentration basis), Table 7.4-3b (dissolved concentration basis), and Table 7.4-3c (particulate concentration basis). Tables 7.4-4a–c show a parallel set of results for total, dissolved, and particulate surface water with primary outliers removed.

7.5 SUPPORTING LINES OF EVIDENCE

This section describes and summarizes several other data sets from the Portland Harbor RI/FS that provide some context for the bedded sediment background estimates from the upriver reach provided above. This evaluation focuses on the four chemical groups (PCBs, PCDD/Fs, DDx, and PAHs) that are most important in the Study Area. For direct comparison with the background values, the summed parameters (e.g., total PAHs) were calculated using the risk assessment summing methods. These supporting lines of evidence and the rationale of their inclusion here are listed below:

Suspended Sediments in Surface Water: This data set includes the measured concentrations of the target chemicals on the particulate fraction sampled in the LWG surface water program. Data generated during all flows sampled at RM 11 and 16 were compiled. These data represent material moving downstream in suspension both above and below the downtown corridor. These vertically integrated water column samples were collected during seven distinct sampling events over three years (November 2004 to March 2007) that captured low, high, and storm-influenced flow regimes (see Section 5.3 for details on the surface water sampling program and results).

In-River Sediment Traps: The data generated from the four in-river sediment traps deployed at approximately RM 11 and RM 16 were compiled. At each of these locations, the traps were deployed on each side of the river from November 2006 to November 2007, and sediments were collected and analyzed quarterly to provide a data set reflecting seasonal variation in chemical concentrations in suspended (and

resuspended) sediments moving through the water column just above the river bed at these upstream locations.

Borrow Pit and Shoaling Sediment Cores: As described in Appendix H, Section H4.2.2, three 10- to 11-ft cores were collected in two upper Study Area borrow pits and on a long-term shoaling area in an attempt to generate, through radioisotope sampling, information on net sedimentation rates. In conjunction with the radiochemistry, conventional and contaminant chemistry samples were obtained from 30-cm-interval vertical composites from the mudline to the bottom of the cores. These cores were collected in February 2007 and, based on borrow pit infilling rates, are estimated to provide approximately a 10-yr profile of sediment quality in these areas of the Study Area. These sample locations were not proximal to any know major source of contaminants. As such, they provide information on the chemical composition of sediments deposited in the upper reaches of the Study Area and therefore are likely to reflect, to some degree, material entering the Study Area from the downtown and upriver reaches of the LWR.

7.5.1 Data Comparisons

The data for total PCBs (congeners and Aroclors, separately), dioxins/furans (total and TEQ, mammals 2005), total DDx, and total PAHs, were preprocessed for the supporting lines of evidence in the same manner described above for the upriver bedded sediment (Section 7.2.4) and compiled with that data. Grain size (percent fines) and TOC were also compiled. Tables 7.5-1 and 7.5-2 provide the summary statistics for these data on a dry weight and OC-normalized basis. Figures 7.5-1 through 7.5-14 are box-whisker plots of these data for each chemical category.

7.5.1.1 Grain Size and Total Organic Carbon

Figures 7.5-1 and 7.5-2, respectively, present box-whisker plots for grain size (expressed as percent fines) and TOC for the upriver sediments and the supporting data sets. Percent fines in the borrow pit and sediment trap samples are higher than in the upriver bedded sediments, consistent with expectations for the lower energy regimes expected for sediment deposition in both the borrow pits and the sediment traps. Grain size data are not available for suspended sediment. TOC patterns are similarly consistent with expectations, with higher median TOC values in the borrow pit, sediment trap, and suspended solids data than in the upriver bedded sediment.

7.5.1.2 PCBs

Figures 7.5-3 and 7.5-4 show the distribution of PCB congeners on a dry-weight and OC-normalized basis for the upriver bedded sediment juxtaposed with the data sets listed above. No congener data were generated for the borrow pit samples. On a dry-weight basis (Figure 7.5-3), the sediment trap and surface water suspended sediment data are comparable and slightly elevated compared with the upriver sediment. OC normalization of these data sets pulls the distributions together (Figure 7.5-4),

suggesting there is little difference in the sediment quality relative to PCBs between these data sets once the physical matrix (and source-influenced data) are accounted for.

The PCB Aroclor plots (Figures 7.5-5 and 7.5-6) show a similar pattern. For PCB Aroclors, there is borrow pit data, but only one non-detect result for suspended sediment. The dry weight data sets show that the sediment trap concentrations (including the source-influenced data) are higher than the borrow pit and upriver sediment concentrations. OC normalization again reduces the separation between data sets overall. PCB Aroclor concentrations in the borrow pit data and upriver sediments are very similar to each other, but the sediment trap concentrations remain higher due to the source-influenced sample.

7.5.1.3 TCDD

Figures 7.5-7 through 7.5-10 compare the distributions of total PCDD/Fs and TCDD TEQ values on a dry-weight and OC-normalized basis. The total TCDD and derived TEQ values show the same patterns between data sets. In general, the upriver sediment, borrow pit, and sediment trap concentration distributions are comparable and overlapping, especially on an OC-normalized basis. The suspended sediment concentrations of total PCDD/Fs and TCDD TEQ are consistently higher than the other data sets.

7.5.1.4 Total DDx

Figures 7.5-11 and 7.5-12 show the dry-weight and OC-normalized total DDx distributions for the upriver sediments and the three supporting lines of evidence. There are relatively small magnitude differences between the data sets; a trend of increasing concentrations from the upriver sediment to borrow pits to sediment traps to suspended sediment is apparent in the dry-weight data. OC-normalization reduces, but does not eliminate this apparent trend. The OC-normalized suspended sediment distribution is much wider than dry-weight distribution and encompasses the ranges seen in the other data sets. Finally, there are two highly elevated total DDx values in the sediment trap data set collected from the source-influenced location ST007 in Quarter 3 and 4 of the sediment trap sampling. As discussed in Section 5.2, 2,4'-DDD was the only detected DDx isomer in the ST007 Quarter 3 and 4 samples, and these detections may be artifacts of PCB interference (false positives). High concentrations of Aroclor 1260 were also detected at this station in these samples (1,800 µg/kg and 2,600 µg/kg). Thus, there is a significant uncertainty associated with these two total DDx values in the sediment trap data set.

7.5.1.5 Total PAHs

Figures 7.5-13 and 7.5-14 show the dry-weight and OC-normalized total PAH values for the four data sets. The dry-weight data suggest that upriver sediment has the lowest total PAH levels, followed by the borrow pit and sediment trap data, which are comparable, and finally the suspended sediment levels. This trend is not evident in the

OC-normalized data sets, which generally overlap with one another. However, the median OC-normalized total PAH concentration remains elevated relative to the other data sets. All of the very low values are seen in the upriver bedded sediment data set in both the dry-weight and OC-normalized data.

7.5.2 Summary

In summary, the comparison of these other lines of evidence with the upriver or background bedded sediment data reveals an overall consistency in the range of concentrations for the major contaminants of concern in Portland Harbor. Recognizing the presence of a known PCB source at RM 11.3E, these data do not indicate a major shift in contaminant levels in the LWR between the upriver area (i.e., upstream of Ross Island and the downtown corridor) and the upper portion of the Study Area. The data suggest that slight increases in PAHs and dioxin levels may occur through this area, particularly in the surface water suspended sediment fraction.

Overall, these other lines of evidence provide corroborative support for the use of the upriver bedded sediment sampling results as a representative background data set for the Portland Harbor RI/FS.

8.0 BASELINE HUMAN HEALTH RISK ASSESSMENT SUMMARY

The BHHRA is an analysis of potential adverse health effects (current or future) caused by hazardous substance releases from a site in the absence of any actions to control or mitigate these releases. The BHHRA is intended to provide information to risk managers on potential risks to human health that are associated with contaminants originating from and present at the Site. Under CERCLA, the BHHRA is one of several pieces of information that is used by risk managers to inform their decisions about potential risks associated with a given site, as well as what can be practically done to reduce risks that may be deemed unacceptable. Other sources of information weighed by risk managers are the ecological risk assessment, fate and transport modeling for sediment, stormwater and other sources of chemicals from outside the Site, as well as various engineering considerations on the feasibility of certain remedial alternatives that can be implemented to reduce risk. The BHHRA is also used to support development of PRGs for protection of human health.

The LWG has worked with EPA to develop the methods and assumptions used in the BHHRA. Consistent with EPA guidance (1989), the BHHRA has been developed to provide a conservative (i.e., health-protective) assessment of risks associated with contaminants present at the Site. The results of this BHHRA should be weighed in a measured and informed fashion in light of the health-protective assumptions.

The approach of the BHHRA is consistent with EPA (1986, 1989, 1991a, 2001b), Region 10 EPA (2000b), and DEQ (2000) HHRA guidance. The methods and inputs for the BHHRA reflect numerous discussions, directives, and agreements with interested parties, including EPA, DEQ, ODHS, and Native American Tribes. The BHHRA was conducted in accordance with the EPA-approved Programmatic Work Plan (Integral et al. 2004) and Human Health Interim Deliverables (Kennedy/Jenks 2004a,b,c; 2006). Additional exposure scenarios that were not included in the Programmatic Work Plan were also evaluated in the BHHRA based on documented direction from EPA, as discussed in Section 8.2.

The remainder of this section presents a summary of the methods used and results of the BHHRA, including the data evaluation, exposure assessment, toxicity assessment, risk characterization, uncertainty analysis, and conclusions. The complete BHHRA is presented in Appendix F to this RI report.

8.1 DATA EVALUATION

The sources of data available for use at the time of the BHHRA are described in Section 2 of this RI Report. The use and evaluation of those data for purposes of the BHHRA are described in Section 2 of Appendix F and in Attachment F2. Data from LWG and non-LWG sampling events were included in the SCRA database, a subset of which was used for the BHHRA. Only data that meet QA2Cat1 data quality objectives were used in the BHHRA. Also, the BHHRA only included data collected between RM 1.0 and

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RM 12.2, including Multnomah Channel. Samples collected between RM 1.9 and RM 11.8 were considered to be within the Study Area, which was the focus of the BHHRA. Samples collected outside of the Study Area were evaluated separately in the BHHRA per an agreement with EPA. The following summarizes the data used in the BHHRA by medium:

- **Beach Sediment:** Composite beach sediment samples that were collected from designated human use areas within the Study Area were included in the BHHRA data set.
- **In-water Sediment:** In-water sediment (i.e., not beach sediment) samples that were collected from the top 30.5 cm in depth between the bank and the navigation channel were included in the BHHRA data set.
- **Surface Water:** All Round 2 and Round 3 surface water data collected from the Study Area, as well as Multnomah Channel, were included in the BHHRA data set.
- **Groundwater Seep:** Data from Outfall 22B, which discharges in a potential human use area, were included in the BHHRA data set. Samples collected from this outfall as part of a stormwater sampling event were excluded from the BHHRA groundwater seep data set.
- **Fish Tissue:** Composite samples, both whole body and fillet with skin (fillet without skin samples were analyzed for mercury only), of target resident fish species (smallmouth bass, brown bullhead, black crappie, and common carp) were included in the BHHRA data set. Composite samples of adult Chinook salmon (whole body, fillet with skin, and fillet without skin), adult lamprey (whole body only), and sturgeon (fillet without skin only) were also included in the BHHRA data set.
- **Shellfish Tissue:** Field-collected composite samples of crayfish and clam tissue (depurated and undepurated) were included in the BHHRA data set.

COIs are all contaminants that were detected in environmental media during investigations within the Study Area. EPA guidance (1989) recommends considering criteria to limit the number of contaminants that are included in a quantitative risk assessment while also ensuring that all contaminants that may contribute significantly to the overall risk are addressed. The contaminants included in the BHHRA for quantitative evaluation are the COPCs. For the purposes of the BHHRA, a comparison to risk-based screening levels was used to select COPCs for further evaluation. COPCs were selected for a medium based on a subset of data determined to represent exposure to a specific human population. If the maximum detected concentration of a COI for a given exposure scenario exceeded the risk-based screening level or if a risk-based screening level was not available, the contaminant was selected as a COPC.

8.2 EXPOSURE ASSESSMENT

The objectives of the exposure assessment were to identify potentially exposed receptor populations, identify and characterize exposure pathways, and estimate the extent of exposure for pathways that are potentially complete and significant. The exposure

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assessment for the BHHRA is found in Section 3 of Appendix F. Additionally, Figure 8.2-1 shows the CSM for the BHHRA, which summarizes all of the exposure scenarios that were evaluated in the BHHRA.

Only potentially complete and significant exposure pathways were quantitatively evaluated for risk in the BHHRA (see Figure 8.2-1). The following are the populations and associated exposure scenarios that were quantitatively evaluated in the BHHRA:

- **Dockside Worker** – Direct exposure to (i.e., ingestion of and dermal contact with) beach sediment, infant ingestion of human breast milk.
- **In-water Worker** – Direct exposure to in-water sediment, infant ingestion of human breast milk.
- **Adult and Child Recreational Beach User** – Direct exposure to beach sediment and surface water (for swimming scenarios).
- **Transient** – Direct exposure to beach sediment, surface water (for bathing and drinking water scenarios), and groundwater seeps.
- **Diver** – Direct exposure to in-water sediment and surface water, infant ingestion of human breast milk.
- **Tribal Fisher** – Direct exposure to beach sediment or in-water sediment, fish consumption, and infant ingestion of human breast milk.
- **Fisher** – Direct exposure to beach sediment or in-water sediment, fish consumption, shellfish consumption, and infant ingestion of human breast milk.
- **Domestic Water User** – direct exposure to untreated surface water hypothetically used as a drinking water source in the future.

Exposures were evaluated on a Study Area-wide basis, as well as on more localized spatial scales as appropriate for each exposure scenario. Exposures to beach sediment were assessed per beach, and exposures to groundwater seeps were assessed per seep. Exposures to in-water sediment, surface water, and fish and shellfish tissue were assessed on both localized and Study Area-wide scales. Except where specifically noted, the exposure assumptions used in the BHHRA were applied uniformly to all of the Study Area, and may or may not be applicable at specific locations within the Study Area depending on factors not specifically addressed in the BHHRA (e.g., accessibility, habitat). The actual exposure at a given location may be less than that assumed for the population and Study Area as a whole due to location-specific conditions.

Several of the exposure scenarios were directed by the EPA, including exposure to divers, clam consumption, fish ingestion for single-species diets, and domestic water use. There is uncertainty to what extent each of these exposures occurs at the Site. Two diver exposure scenarios that differentiate between the use of either a wet suit or dry suit were included in the BHHRA as directed by the EPA. Also as directed by the EPA, consumption of clams was assessed as an exposure pathway. The abundance of clams is limited at the Site as demonstrated by the low numbers of clams found during sampling activities, which were predominantly Asian clams (*Corbicula* sp.). The

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possession and harvest of Asian clams is illegal in the State of Oregon. Even though surface water in the LWR within Portland Harbor is not currently used as a domestic water source, under OAR 340-041-0340 Table 340A, domestic water supply is a designated beneficial use of the Willamette River, with adequate pretreatment.

The exposure assessment incorporated the reasonable maximum exposure (RME) approach described by EPA (1989). The RME is intended to be a conservative exposure level that is still within the range of possible exposures. Consistent with EPA (1989), the exposure assessment also used CT values, which are intended to represent average exposures, for certain exposures that may occur within the Study Area. For some exposure scenarios, such as fish consumption, exposure assumptions were directed by EPA. Study Area-specific information is not available for fish consumption rates for specific species, so a range of ingestion rates and diets composed of various species were evaluated in the BHHRA for both adult and child consumers of fish. The ingestion rates used for an adult fisher of 17.5 g/day, 73 g/day, and 142 g/day are the same as approximately 2, 10, and 19 eight-ounce meals per month, respectively.

Exposure point concentrations (EPCs) were calculated for the 95% upper confidence limit on the arithmetic mean (95% UCL) and the arithmetic mean for each exposure area. In some exposure areas, the maximum concentration was used instead of the 95% UCL. Therefore, the EPCs are referred to as the 95% UCL/max and mean throughout the BHHRA.

8.3 TOXICITY ASSESSMENT

Toxicity values provide a quantitative estimate of the potential for adverse effects resulting from exposure to a chemical. Toxicity values for both cancer and noncancer endpoints were evaluated. Toxicity values used in the BHHRA are presented in Section 4 of Appendix F. The following hierarchy of sources of toxicity values is currently recommended for use at Superfund sites (EPA 2003a), and was used for the BHHRA:

- **Tier 1** – EPA's Integrated Risk Information System
- **Tier 2** – EPA's Provisional Peer Reviewed Toxicity Values
- **Tier 3** – Additional EPA and non-EPA sources of toxicity information.

Some toxicity values are based on exposure to chemical mixtures and not to individual chemicals. As a result, risks were evaluated for the combined exposure to the chemicals and not on an individual chemical basis for the following chemicals:

- Total Chlordanes
- Total DDD, Total DDE, and Total DDT
- Total Endosulfan
- Total PCBs
- Total Dioxins and Furans
- Carcinogenic PAHs (assessed on both an individual and cumulative basis).

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TEFs were used to evaluate carcinogenic and non-carcinogenic effects of dioxin and furan congeners and dioxin-like PCB congeners. PCBs were also evaluated as total PCBs for both carcinogenic and noncarcinogenic effects. Carcinogenic PAHs were evaluated for toxicity based on their PEF, which estimates toxicity relative to BAP (EPA 1993).

8.4 RISK CHARACTERIZATION

Risk characterization integrates the information from the exposure assessment and toxicity assessment, using a combination of qualitative and quantitative information. With this information, risk characterization estimates the potential health risk, based on the dose of a chemical that a person may receive under specific exposure conditions and the toxicity of that chemical. Consistent with DEQ (2000) and EPA (1989) guidance, noncarcinogenic and carcinogenic effects were evaluated separately. COPCs were identified as contaminants potentially posing unacceptable risks if they resulted in a cancer risk greater than 1×10^{-6} or a hazard quotient (HQ) greater than 1 under the exposure scenarios and EPCs evaluated in this BHHRA, regardless of the uncertainties. The methods and results are described in detail in Section 5 of Appendix F and summarized below.

8.4.1 Risk Characterization Methods

The potential for adverse effects resulting from exposure to contaminants with noncarcinogenic effects was addressed by comparing the estimated absorbed dose (i.e., chronic daily intake) for a specific COPC to its corresponding reference dose (RfD) to yield a HQ. The HQs were summed across all chemicals to calculate the cumulative hazard indices (HIs). Per EPA (1989) guidance, HQs should be summed for chemicals with common toxicological endpoints. If the cumulative HIs exceeded 1, estimated endpoint-specific HIs were compared to a target HI of 1 for each exposure area, below which remedial action at a Superfund site is generally not warranted (EPA 1991b).

Potential cancer risks were assessed by multiplying the estimated absorbed dose (i.e., lifetime average daily intake) of a carcinogen by its cancer slope factor. This calculated risk is expressed as the probability of an individual developing cancer over a lifetime as a result of exposure to the potential carcinogen, and is a conservative, health-protective estimate of the incremental probability of excess individual lifetime cancer risk. Estimated total cancer risks (summed across all chemicals) were compared to 1×10^{-4}, 1×10^{-5}, and 1×10^{-6} cancer risk targets based upon the following language in EPA's National Contingency Plan (NCP): "For known or suspected carcinogens, acceptable exposure levels are generally concentration levels that represent an excess upper bound lifetime cancer risk to an individual of between 1×10^{-4} and 1×10^{-6}." The point of departure for cancer risks is 1×10^{-6}.

Cancer risks were calculated separately for adult and child receptors for the recreational beach user, domestic water user, and fisher scenarios. To assess risks to individuals exposed as both a child and an adult, cancer risks were also calculated for a combined

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adult and child receptor for the recreational beach user, domestic water user, and fisher scenarios.

For bioaccumulative chemicals, exposure to the mother can potentially lead to the presence of those chemicals in human milk, which can pose a risk to breastfeeding infants. Per agreement with EPA and DEQ, risks to infants through the consumption of human milk were included for all receptors where PCBs, dioxins, and/or DDx were identified as COPCs. To assess risks to infants, infant risk adjustment factors (IRAFs) were applied to the mother's risk for each of these contaminants. The approach and values for IRAFs used to assess risks to infants were in accordance with DEQ guidance (2010b).

8.4.2 Risk Characterization Results

The ranges of estimated potential risks resulting from the different exposure scenarios are summarized in Table 8.4-1. A summary of the risk characterization results is presented by exposure scenario in the following sections.

8.4.2.1 Fish Consumption

Risks were calculated for the adult and child fishers who consume fish caught within the Study Area, as well as breastfeeding infants of mothers consuming fish, based on the following:

- Three different ingestion rates representing a range of consumption scenarios
- Both single species and multiple species diets (black crappie, common carp, brown bullhead, and smallmouth bass)
- Consumption of both whole body and fillet tissue.

Risks were also evaluated for adult and child tribal fishers who consume fish as a multi-species diet consisting of resident fish species (black crappie, common carp, brown bullhead, and smallmouth bass) as well as sturgeon, lamprey, and salmon; and on consumption of both whole body and fillet with and without skin tissue. All risk estimates were made using both mean and 95 percent UCL/max estimates of contaminant concentrations in tissue. Consequently, minimum risk estimates represent the lowest consumption rate used for the scenario and mean tissue concentrations, and maximum estimates represent the highest consumption rate and 95 percent UCL/max tissue concentrations.

The cancer risks for all of the fish consumption scenarios range from 3×10^{-6} (child consumption) to 7×10^{-2} (combined adult/child consumption). Cumulative HIs for all of the fish consumption scenarios ranged from 0.5 (adult consumption) to 5,000 (child consumption). The maximum cumulative HI for a breastfeeding infant of a fish consumer is 60,000. For many of the fish consumption scenarios, the cancer risks and noncancer hazards exceed the EPA target cancer risk range (10^{-6} to 10^{-4}) and target HI

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(1). The following summarizes the assumptions associated with the highest risk estimates:

- **Fish ingestion rate.** The highest ingestion rates used in the BHHRA for adult tribal fishers and adult fishers are 175 g/day (CRITFC 1994) and 142 g/day (EPA 2002b), respectively. These are equivalent to 23 and 19 meals per month, respectively, based on an 8-ounce serving size, every month of the year exclusively of fish caught within the Study Area.
- **Exposure duration.** Fish consumption is assumed to occur at that same rate for 30 years for adult fishers and 70 years for tribal fishers.
- **Whole-body tissue.** Only whole-body tissue (i.e., the entire fish) is consumed.
- **Single species.** Only one species (i.e., common carp) is consumed (for fishers).
- **Source of fish.** 100 percent of the fish consumed is caught/harvested from the same localized exposure area.
- **Possible effects of cooking methods.** Possible effects of cooking methods, which can reduce concentrations of lipophilic chemicals in fish tissue, were not considered. PCB concentrations have been shown to be reduced from 10 to 87 percent with various cooking methods (Wilson et al. 1998).

Site-specific fish consumption information is not available for the fisher scenarios. As a result, nationwide fish consumption data were used as the source for the fish ingestion rates used in the BHHRA. The specific ingestion rates used in the risk assessment were directed by EPA. If actual exposures within the Study Area were less than the assumptions provided above, estimated risks from consumption of fish would be lower.

PCBs and dioxins/furans contribute approximately 93 and 5 percent, respectively, of the estimated cumulative cancer risk for whole body, multi-species fish consumption for the Study Area. The remaining contaminants potentially posing unacceptable risks account for less than 2 percent of the cumulative cancer risk on a Study Area-wide basis. The contribution of contaminants to the cumulative cancer risks varies on a localized basis. PCBs also resulted in the highest HQs for the Study Area (up to 60,000 for the breastfeeding infant of a fisher consuming a whole-body carp tissue diet).

On a regional basis, risks from exposure to bioaccumulative compounds in tissue exceed the EPA target risk range of 10^{-6} to 10^{-4}. For example, the PCB concentrations detected in resident fish from the Willamette and Columbia rivers (EVS 2000; EPA 2002a) are approximately 20 to 100 times higher than the EPA target fish tissue concentration, which is based on a target risk level of 1×10^{-6}. Regional efforts are underway to reduce fish tissue concentrations. Sources contributing to regional tissue concentrations are unknown.

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8.4.2.2 Shellfish Consumption

The consumption of shellfish was evaluated for adult fishers, and breastfeeding infants of adult fishers, based on two consumption rates representing a range of potential consumption scenarios. Current and potential future shellfish consumption rates for the Site are not known. The shellfish species evaluated for consumption risks were crayfish and clams. Cumulative cancer risks from consumption of shellfish ranged from 9×10^{-7} to 7×10^{-4}. Cumulative HIs for noncarcinogenic risk, ranged from 0.06 to 40 for consumption of shellfish. The maximum cumulative HI for a breastfeeding infant of a shellfish consumer is 800.

8.4.2.3 Direct Exposure to Beach Sediment

Beaches were identified as potential human use areas associated with industrial upland sites (dockside workers), recreation (recreational users or fishers), and/or trespassing or transient use (transients). The extent to which each beach is used and the nature of the contact with sediments/beach are uncertain. However, conservative assumptions were included in the risk analysis of this exposure pathway to provide an estimate of potential risks.

The cumulative cancer risks for all of the CT beach sediment exposure scenarios were below 10^{-4}. The CT scenarios for exposure to beach sediment resulting in risks above 1×10^{-6} were the dockside worker (6×10^{-6}), tribal fisher (2×10^{-6}), and combined adult and child, as well as child-only, recreational beach user (2×10^{-6}) scenarios. The RME scenarios for exposure to beach sediment resulting in cumulative cancer risks above 1×10^{-6} include: dockside worker, adult and child recreational beach user, tribal fisher, low-frequency fisher, and high-frequency fisher. None of the RME scenarios for exposure to beach sediment resulted in risks greater than 1×10^{-4} or cumulative HIs exceeding 1. Risks above 1×10^{-6} resulting from exposures to beach sediment are due at least in part to arsenic, which is likely present at naturally occurring background concentrations. For instance, the highest cumulative risk from exposure to beach sediment at beaches used by transients, for recreation, and/or by fishers is for the tribal fisher RME scenario (2×10^{-5}), and approximately 50 percent of this risk is associated with arsenic concentrations that are at or below the background arsenic concentration of 7 mg/kg (DEQ 2007). Carcinogenic PAHs were the only other contaminant identified as potentially posing unacceptable risks for beach sediment.

8.4.2.4 Direct Exposure to In-Water Sediment

Risks from in-water sediment exposure were estimated separately for each of the half-mile river segment exposure areas on each side of the river, and for Study Area-wide exposure. In-water sediments within the navigation channel were not included in the risk evaluation. Risks from in-water sediment exposure were evaluated for exposures by in-water workers, tribal fishers, fishers, divers, and breastfeeding infants of adults exposed to in-water sediment.

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The cumulative risks for all of the CT scenarios were below 1 x 10^{-4}, and only the tribal fisher and breastfeeding infant CT scenarios had cancer risks above 1 x 10^{-6}. For the RME scenarios, cumulative cancer risks were greater than 1 x 10^{-6} but were below 1 x 10^{-4}, with the exception of cancer risks above 1 x 10^{-4} for direct exposure to in-water sediment by a tribal fisher and breastfeeding infant of a tribal fisher at exposure areas RM 6W (risk is 2 x 10^{-4} due primarily to cPAHs) and RM 7W (risk is 3 x 10^{-4} due primarily to dioxins). RM 7W also results in a cumulative HI greater than 1 for the tribal fisher (HI is 3), breastfeeding infant (HI is 5), and high frequency fisher (HI is 2) RME scenarios.

8.4.2.5 Direct Exposure to Surface Water

Risks were evaluated for direct surface water exposures by transients, divers and adult and child recreational beach users. The only scenarios resulting in cumulative cancer risks greater than 1 x 10^{-6} were the diver in wet suit (1 x 10^{-5}) and the diver in dry suit (2 x 10^{-6}) RME scenarios at RM 6W due primarily to cPAHs. None of the direct surface water exposure scenarios resulted in HIs greater than 1.

Risks were also evaluated for hypothetical exposure to untreated surface water used as a domestic water source by future residents. The maximum cumulative cancer risk for hypothetical exposure to untreated surface water was 9 x 10^{-4} for the combined adult/child RME scenario, due primarily to cPAHs, and benzo(a)pyrene specifically. The child RME scenario for hypothetical exposure to untreated surface water as a domestic water source was the only scenario with a HI that exceeded 1. The exceedance occurred at RM 8.5, primarily from exposure to 2-(4-chloro-2-methylphenoxy)propanoic acid (MCPP) (HQ for MCPP was 2).

8.4.2.6 Direct Exposure to Seeps

Risks from direct contact with groundwater seeps were evaluated for exposure by a transient for one exposure point. The transient exposure scenario did not result in cumulative cancer risks greater than 1 x 10^{-6} or HIs greater than 1.

8.4.2.7 Identification of Contaminants Potentially Posing Unacceptable Risks

Contaminants were identified as potentially posing unacceptable risks if they resulted in a cancer risk greater than 1 x 10^{-6} or a HQ greater than 1 under any of the exposure scenarios for any of the EPCs evaluated in the BHHRA, regardless of the uncertainties. There were 33 contaminants identified as potentially posing unacceptable risks for the exposure scenarios listed above. Only a subset of these contaminants were associated with cancer risks exceeding 1 x 10^{-4} or HQs exceeding 1, and an even smaller number of contaminants contributed to most of the relative percentage of total risk. In some cases, the contaminants were identified as potentially posing unacceptable risks based only on the highest ingestion rate, a single exposure point, and/or the maximum detected concentration. Four of the contaminants (alpha-, beta-, and gamma-hexachlorocyclohexane and heptachlor) were identified as potentially posing

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unacceptable risks on the basis of N-qualified data only. The use of an "N" qualifier indicates that the identity of the analyte is not definitive. These four chemicals are not recommended for further evaluation of potential risks to human health. The remaining 29 contaminants identified as potentially posing unacceptable risks to human health are evaluated further in the Human Health Risk Management Recommendations document. These contaminants are presented in Table 8.4-2.

8.5 UNCERTAINTY ANALYSIS

Uncertainty is associated with every step of a risk assessment, from the sampling and analysis of chemicals in environmental media to the assessment of exposure and toxicity and the risk characterization. Uncertainty can have two components: 1) variability in data or information, and 2) lack of knowledge. The uncertainty analysis conducted as part of the BHHRA focused on issues of variability and knowledge uncertainty associated with each of the inputs and models used to derive the risk estimates. In general, the approach and methodologies used in a risk assessment are designed to provide a margin of conservatism in human health risk estimates.

A detailed analysis of the uncertainties associated with the BHHRA is found in Section 6 of Appendix F. Uncertainties were evaluated relative to their potential impact on the conclusions of the BHHRA and resulting significance to risk management decisions. The following discussion presents the sources of uncertainty with the highest significance to risk management decisions.

Exposure Parameters for Fish and Shellfish Consumption Scenarios. The exposure factors used in estimating potential human health risks were purposefully selected to be conservatively protective, and the range of fish consumption rates varied by an order of magnitude to reflect the uncertainty in consumption rates. Furthermore, assumptions about the species and tissue consumed, the source of fish, and lack of preparation methods add to the magnitude of uncertainty. These particular exposure assumptions are likely to overestimate actual exposure conditions at the Study Area and, therefore, overestimate human health risks and hazards.

Using the Maximum Concentration to Represent Exposure. In cases when there were fewer than five samples with a detected concentration for a given analyte for a given exposure area, the sample size was not sufficient to calculate a 95% UCL on the mean, so the maximum concentration detected was used as the EPC. Using maximum detected concentrations of infrequently detected analytes to represent individual exposure areas, and especially Study Area-wide exposure, results in a conservative estimate of risk for the Study Area. The use of the maximum detected concentration likely overestimates the actual human health risks. The number of samples used to calculate EPCs for each scenario are listed in Tables 3-2 through 3-4 and 3-6 through 3-25 of Appendix F.

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Risks from Background. Arsenic and mercury were found to result in risks greater than 1×10^{-6} or an HQ of 1 for at least one of the exposure scenarios evaluated in the BHHRA. Metals are naturally occurring chemicals and may be present in tissue, water or sediment due to background concentrations. For beach sediment, the arsenic EPCs ranged from 0.7 to 9.9 mg/kg and are consistent with the default background soil concentration for arsenic of 7 mg/kg used by DEQ (2007). In addition to naturally occurring metals, anthropogenic background may contribute to the overall risks. Further discussion of background concentrations is presented in Section 7.

Background tissue concentrations were not established for the Study Area, which increases the uncertainty in estimating risks from ingestion of tissue that are attributable to sources within the Study Area.

The risks were presented in the BHHRA without accounting for contributions from background, so it is important to recognize that background concentrations may result in unacceptable risks based on the exposure assumptions used in the BHHRA.

Regional Tissue Concentrations. PCBs and dioxins/furans have been detected in fish tissue collected in the Willamette and Columbia rivers, outside of the Study Area. In the Columbia River Basin Fish Contaminant Survey, the basin-wide average concentrations of total PCBs in resident fish ranged from 0.032 to 0.173 parts per million (ppm) for whole body samples and from 0.033 to 0.190 ppm for fillet with skin samples (EPA 2002a). In the middle Willamette River (RM 26.5 to 72), the average concentrations of total PCBs in resident fish ranged from 0.086 to 0.146 ppm for whole body samples and from 0.026 to 0.071 ppm for fillet with skin samples (EVS 2000). These concentrations are lower than the concentrations detected in the Study Area where average concentrations ranged from 0.16 to 2.8 ppm in whole body samples and from 0.17 to 2.5 ppm in fillet with skin samples (for PCBs as total congeners). The fish species included in the studies were different than those collected within the Study Area, so the concentrations may not be directly comparable. Sources contributing to the PCBs and dioxins/furans detected in fish collected outside of the Study Area are unknown and may not be relevant to the Study Area.

8.6 SUMMARY AND CONCLUSIONS

The following presents the major findings of the BHHRA[1]:

- Risks resulting from the consumption of fish or shellfish are generally orders of magnitude higher than risk resulting from direct contact with sediment, surface water, or groundwater seeps. Risks from fish and shellfish consumption are within or above the cumulative cancer risk range of 1×10^{-6} to 1×10^{-4} and exceed an HI of 1. With the exception of two half-mile river segments for the

[1] However, the identification of the contaminants presenting the most significant risk in various areas of the site consistent with EPA risk assessment guidance is not intended to suggest that other contaminants in those areas and at the site generally do not also present potentially unacceptable risk.

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tribal fisher scenario and the hypothetical use of untreated surface water as a drinking water source by a future resident, direct contact with in-water sediment, surface water, and seeps results in risks within or below the EPA target cancer risk range of 1×10^{-6} to 1×10^{-4}. The evaluation of shellfish consumption was done at the direction of EPA.

- Fish consumption is the exposure scenario that is considered the primary contributor to risk for the Study Area. PCBs are the primary contributor to risk for fish consumption, and dioxins/furans are a secondary contributor to risk for fish consumption. Risks from PCBs based on consumption of fish within the Study Area exceed the EPA target risk range of 1×10^{-6} to 1×10^{-4}, with a maximum estimated risk from PCBs of 7×10^{-2} for whole body consumption and 4×10^{-2} for fillet consumption. The maximum cancer risks from consumption of dioxins/furans in fish tissue are 7×10^{-3} for whole body tissue and 1×10^{-3} for fillet tissue.
- The uncertainties associated with the tissue consumption scenarios should be considered when using the results of the BHHRA in risk management decisions. The fish tissue consumption risks in the BHHRA incorporate assumptions that may underestimate or more likely overestimate the actual risks.
- The contribution of background sources of contaminants is an important consideration in risk management decisions. For example, approximately 50 percent of the highest risk to tribal fishers from exposure to beach sediment is associated with arsenic concentrations that are at or below the background arsenic concentration of 7 mg/kg (DEQ 2007).

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9.0 BASELINE ECOLOGICAL RISK ASSESSMENT SUMMARY

A draft final BERA has been prepared following the ecological risk assessment (ERA) approach presented in the Portland Harbor RI/FS Programmatic Work Plan (Integral et al. 2004) and direction in the EPA-prepared February 2008 Problem Formulation document (EPA 2008e) (included in the BERA [Appendix G] as Attachment 2). The approach was prepared by the LWG based on the requirements of the scope of work and AOC (EPA 2001a) entered into with EPA for conducting the RI/FS. The approach is also consistent with EPA guidance for conducting ERAs (EPA 1997a, 1998).

The Portland Harbor RI/FS has two broad ecological risk assessment objectives:

1. Identify the potentially unacceptable risks posed by contaminants to aquatic and aquatic-dependent ecological receptors in the Study Area under baseline conditions.[1]
2. In the event that potentially unacceptable ecological risks are found, provide information that risk managers can use to support decisions about remediation goals (RGs), remedial action levels (RALs), areas of concern, and methods to analyze remedial action alternatives in the FS for the protection of ecological receptors.

Incorporating the results of the BERA and the BHHRA, these RGs and RALs will provide estimates of the long-term goals to be achieved by any cleanup actions in Portland Harbor. During the FS process, RGs will be refined based on background sediment quality, technical feasibility, and other risk management decisions. EPA will identify the final sediment remediation goals for the site in the ROD following the completion of the FS. Given that the Portland Harbor Superfund Site is located in an urban and industrialized area, the regional land uses and physical and chemical baseline conditions will play a role in risk management decisions. For most ecological receptors, the BERA assumed that the entire Study Area represents potential habitat; further evaluation of specific habitat areas should be another key component considered when making future risk management decisions.

The evaluation of potentially unacceptable risks to ecological receptors at the Portland Harbor Superfund Site has been an ongoing and iterative process involving both the LWG and EPA, with oversight and direction from EPA. This process has been documented by numerous reports and technical memoranda since 2001. Major ERA documents include those listed above and the following:

- *Portland Harbor RI/FS Programmatic Work Plan for the Portland Harbor* (Integral et al. 2004)

[1] Baseline conditions are the conditions represented by the BERA data set, which consists of data collected between June 2002 and November 2007. The BERA data set is included in Attachment 4 of the BERA (Appendix G).

- *Portland Harbor Superfund Site Ecological Risk Assessment: Comprehensive Synopsis of Approaches and Methods* (Windward 2004)
- *Portland Harbor Superfund Site Ecological Risk Assessment Interpretive Report: Estimating Risks to Benthic Organisms Using Predictive Models Based on Sediment Toxicity Tests (Draft)* (Windward et al. 2006)
- *Portland Harbor Superfund Site Proposed Ecological Risk Assessment Decision Framework (Draft)* (Windward 2006)
- *Portland Harbor Benthic Toxicity Reanalysis Technical Memorandum* (Windward 2009a)
- *Portland Harbor RI/FS Site-Specific SQGs Based on Individual Bioassay Endpoints* (Windward 2010).

Estimates of risks were made based on preliminary data sets in the following documents:

- *Portland Harbor RI/FS, Ecological Preliminary Risk Evaluation* (Ecological PRE; Windward 2005)
- *Portland Harbor RI/FS Comprehensive Round 2 Site Characterization Summary and Data Gaps Analysis Report* (Integral et al. 2007).

Data from the Study Area were collected by LWG during three sampling rounds (Rounds 1, 2, and 3). The initial sampling (Round 1) was conducted concurrent with the preparation of the Programmatic Work Plan from summer 2002 until spring 2004. The Ecological PRE evaluated preliminary risks to ecological receptors based on Round 1 data. Round 2 data were collected from summer 2004 until December 2005 to support the Comprehensive Round 2 Report and fill data gaps from Round 1 sampling.

During the preparation and following the submittal of the Comprehensive Round 2 Report, EPA and LWG identified additional data gaps that were filled through a third round of sampling. Round 3 sampling was conducted from January 2006 until February 2008. Data from all LWG sampling rounds as well as other relevant and acceptable sources combined with a series of exposure assumptions and effects thresholds formed the basis of the risk estimates in the BERA. The approach applied in the BERA and the risk results and conclusions supersede prior approaches and estimates of risk.

The BERA identifies ecological COPCs that pose potentially unacceptable risks. These COPCs were drawn from a longer list of COIs. Risk estimates are stated as HQs, which are calculated as the concentration at the point of exposure divided by the concentration that defines a threshold for adverse effects, typically a toxicity reference value (TRV). Any COPC with an HQ $\geq$ 1 in at least one line of evidence (LOE) at any location in the Study Area is identified as posing potentially unacceptable risk. The spatial distribution of risks is also presented. The results of the BERA will be used in the FS to identify contaminants, receptors, and areas of concern for the purpose of assessing the protectiveness of potential remedies.

In total, 89 contaminants (as individual contaminants, sums, or totals) with HQs ≥ 1 pose potentially unacceptable risk.[2] The primary risk of significant adverse effects on ecological receptors in the Study Area is from four groups of contaminant mixtures: PCBs, dioxins and furans, PAHs, and total DDx. The identification of the primary contributors to risk is consistent with EPA risk assessment guidance (EPA 1997a) and is not intended to suggest that other contaminants do not also present potentially unacceptable risk.

The BERA identified potentially unacceptable benthic risks in about 7% of the Study Area. The COPCs in sediment that pose potentially unacceptable risk to the benthic community or populations are PAHs, PCBs, and DDx compounds. PCB HQs ≥ 1 occurred throughout the Study Area for river otter, mink, spotted sandpiper, bald eagle and osprey, indicating possible population-level effects. The potential for adverse effects in fish due to exposure to total PCBs is low based on the limited extent and frequency of TRV exceedances and the likelihood that uncertainties contribute to overestimates of risk. The combined toxicity of dioxins/furans and dioxin-like PCBs, expressed as TEQ, poses a risk of reduced reproductive success in mink, river otter, sandpiper, bald eagle, and osprey. PCBs are responsible for the majority of total TEQ exposure, but the dioxin/furan TEQ also exceeds its TRV in some locations of the Study Area. DDx compounds pose low to negligible risk to individual bald eagles and Pacific lamprey ammocoetes and only within limited portions of the Study Area. Total DDx HQs ≥ 1 occurred for sculpin for certain LOEs and for spotted sandpiper. The weight of evidence (WOE) indicates that DDx likely poses low risk to populations of these receptors because of the low magnitude and limited frequency of exceedances and likelihood that uncertainties contribute to overestimates of risk. Thirty-eight TZW COPCs, including 6 metals, 16 PAHs, 2 SVOCs, 2 pesticides, 10 VOCs, cyanide, and perchlorate, pose potentially unacceptable risk in localized areas. Given their feeding habits and oxygen requirements, benthic organisms have relatively low exposure to TZW compared with surface water in the hyporheic zone. This was not accounted for in the BERA's TZW exposure estimates, which therefore are likely to overestimate TZW exposure levels. For many COIs, the available exposure and effects data preclude a quantitative risk evaluation. These COIs have nonetheless been identified as posing potentially unacceptable risk.

These findings can be used in the FS as the framework for analyzing and comparing alternatives to remediate areas of elevated risk to ecological receptors.

By design, risk assessments are conservative in the face of uncertainty. In this context, conservative means efforts made to minimize the chances of underestimating exposure or risk. Consistent with the methods of EPA's Problem Formulation (EPA 2008e), receptor-contaminant pairs posing potentially unacceptable risk were identified using conservative methods and assumptions. Examples of conservatism include assumptions that contaminants are bioavailable and assumptions that yielded such low effect thresholds

[2] The likelihood and ecological significance of the potentially unacceptable risk varies across contaminants and LOEs from very low to high. Therefore, the potentially unacceptable risks range from negligible to significant.

(TRVs) that, in the case of essential metals, they had to be adjusted upward because they were below nutritional requirements for some fish species.

Not all uncertainties create a conservative bias. Some can lead to underestimation of risk, including the unavailability of exposure or effects data, thresholds that do not account for untested sensitive species, synergistic interactions among the multiple contaminants present at a site, and metabolic processes that increase the toxicity of accumulated contaminants.

The remainder of Section 9.0 provides an overview of the BERA.

9.1 ECOLOGICAL SETTING OF THE BERA

In accordance with CERCLA and ERA guidance (EPA 1997a, 1998), the BERA identifies chemical stressors that occur at concentrations that have the potential to cause adverse effects on the populations or communities of valued ecological receptors. A physical description of the Study Area is useful to understand the ecological communities and populations and the other stressors that may influence their abundance and distribution.

Within the Study Area, the LWR is typical of an industrialized urban river corridor, with highly developed shoreline and channel modified for marine uses, manufacturing, and navigation. Much of this development has affected the physical habitat characteristics of the natural system, especially shallow off-channel wetlands, nearshore shallow areas, and riparian cover. Wharves and piers extend out toward the federally maintained navigation channel, and bulkheads and riprap revetments armor the riverbank. Active dredging has produced a uniform channel with little habitat diversity. However, some segments of the Study Area are more ecologically complex, with small embayments, shallow water areas, gently sloped beaches, small localized accumulations of wood, and less shoreline development, providing habitat for a suite of local fauna.

The numerous organisms that use the LWR can be divided into the following general groups: invertebrates, fishes, birds, mammals, amphibians, reptiles, and aquatic plants. These groups include native species and species that are exotic to the river but have high ecological impact (e.g., centrarchids). Each group makes an important contribution to the ecological function of the river based on its trophic level, abundance, and interaction with the physical environment. Details on the ecological habitat types and species present at the Study Area are presented in Section 3.1.6 and Appendix G of the RI.

9.2 BERA PROBLEM FORMULATION

In accordance with CERCLA and ERA guidance (EPA 1997a, 1998), the problem formulation step of the ERA process was completed for the BERA and included the following:

- Refinement of preliminary ecological COPCs
- Review and refinement of information on fate and transport, complete exposure pathways, and ecosystems potentially at risk

- Selection of assessment and measurement endpoints
- Development of a CSM and risk questions
- Development of an analysis plan.

Elements of the problem formulation were provided as part of Appendix B of the Programmatic Work Plan (Integral et al. 2004), in the draft Ecological PRE (Windward 2005), and in Appendix G of the Comprehensive Round 2 Report (Integral et al. 2007). EPA developed and directed LWG to use its Problem Formulation document (EPA 2008e), which provided the methods for completing the BERA, addressed implementation of the five steps identified above, and accounted for data and information collected to date. EPA's Problem Formulation is provided in Attachment 2 of the BERA (Appendix G).

9.2.1 Identification of COPCs

Ecological COPCs were identified by screening the BERA dataset in a two-step screening process (i.e., the screening-level ERA [SLERA] and refined screen) per EPA's Problem Formulation (EPA 2008e). The BERA contaminant dataset includes data for biological tissue, surface sediment, surface water, and TZW.

Whole-body fish tissue, invertebrate tissue, and bird egg tissue were included in the BERA dataset. Field-collected tissues were available for evaluating tissue-residue concentrations or dietary-exposure concentrations of ecological receptor species. Whole-body tissue data from the following receptors were used in the BERA: clams, epibenthic invertebrates, crayfish, mussels, black crappie, brown bullhead, carp, juvenile Chinook salmon, largescale sucker, northern pikeminnow, peamouth, sculpin, smallmouth bass, juvenile white sturgeon, and Pacific lamprey ammocoetes. Concentrations of contaminants in the stomach contents of juvenile Chinook salmon and juvenile white sturgeon were used to evaluate dietary exposure of these receptors. Concentrations of contaminants in osprey eggs were used to evaluate risks to both eagles and osprey.

Laboratory-exposed tissues (based on 28-day bioaccumulation testing) were included in the BERA dataset for freshwater oligochaetes and clams. Bioaccumulation testing was conducted using freshwater oligochaetes to estimate tissue concentrations in other common sediment-exposed benthic invertebrates that could not be collected in the field. Bioaccumulation testing was conducted using clams to facilitate evaluation of the two different exposure regimes (field and laboratory) and the subsequent tissue concentrations. Concentrations of detected neutral organic contaminants in laboratory-exposed tissues were adjusted to estimate steady-state concentrations using the process based on McFarland (1995) and described in the inland testing manual (EPA and USACE 1998).

Surface sediment (0 to a maximum of 30 cm), surface water, and, at the direction of EPA, shallow TZW collected from the Study Area during multiple sampling rounds were also evaluated as part of the BERA data set.

The following COPCs were identified for each media type based on the SLERA and refined screening step using conservative screening-level TRVs, maximum concentrations, and exposure assumptions:

- **Surface Sediment –** Sixty-seven COPCs were identified. Surface sediment COPCs were evaluated as part of the benthic invertebrate risk assessment.
- **Tissue –** Seventeen COPCs were identified. Tissue COPCs were evaluated in the benthic invertebrate and fish risk assessments as part of the tissue-residue LOE.
- **Diet –** Eight dietary COPCs were identified for fish, and 24 dietary COPCs were identified for wildlife (birds and mammals). Dietary COPCs, which were identified from both tissue and sediment data, were evaluated as part of the fish and wildlife risk assessments**.**
- **Surface Water –** Fourteen COPCs were identified. Surface water COPCs were evaluated as part of the benthic invertebrate, fish, amphibian, and aquatic plant risk assessments.
- **TZW –** Fifty-eight COPCs were identified. TZW COPCs were evaluated as part of the benthic invertebrate, fish, amphibian, and aquatic plant risk assessments.

9.2.2 Refined CSM

The CSM describes relationships between contaminants and the resources potentially affected by their release. Consistent with EPA Superfund guidance (EPA 1997a), the ecological receptors selected for assessment in the Portland Harbor BERA were identified from among the organisms using the site by considering the following criteria:

- Societal and cultural significance (i.e., species valued by society or that have special regulatory status – threatened or endangered)
- Ecological significance (i.e., species that serve a unique ecological function)
- Level of exposure to likely COPCs at the site (i.e., site use)
- Relative ability to bioaccumulate likely COPCs at the site
- Sensitivity to likely COPCs at the site
- Availability of sufficient data to assess risks to specific organisms.

Based on these criteria, the following ecological receptors were selected for assessment:

- **Benthic invertebrate community**[3] – benthic macroinvertebrate community as a whole, bivalves (clams), and decapods (e.g., crayfish)

[3] Clams and crayfish are members of the benthic macroinvertebrate community but were also evaluated separately to provide a population-level assessment.

- **Omnivorous fish populations –** largescale sucker, carp, and pre-breeding white sturgeon
- **Invertivorous fish populations –** sculpin, peamouth, and juvenile Chinook salmon[4]
- **Piscivorous fish populations –** smallmouth bass and northern pikeminnow
- **Detritivorous fish individuals –** Pacific lamprey ammocoetes
- **Sediment-probing invertivorous bird populations** – spotted sandpiper
- **Omnivorous bird populations**[5] – hooded merganser
- **Piscivorous bird populations**[6] – osprey and bald eagle
- **Aquatic-dependent carnivorous mammal populations** – mink and river otter
- **Amphibian and reptile populations** – amphibians (e.g., frog and salamander species)
- **Aquatic plant community** – aquatic plant community (e.g., phytoplankton, periphyton, macrophyte species).

The assessment endpoints for all receptors are based on the protection and maintenance of their populations and the communities, with the exception that organism health was designated by EPA as the assessment endpoint for juvenile Chinook salmon, Pacific lamprey ammocoetes, and bald eagle.

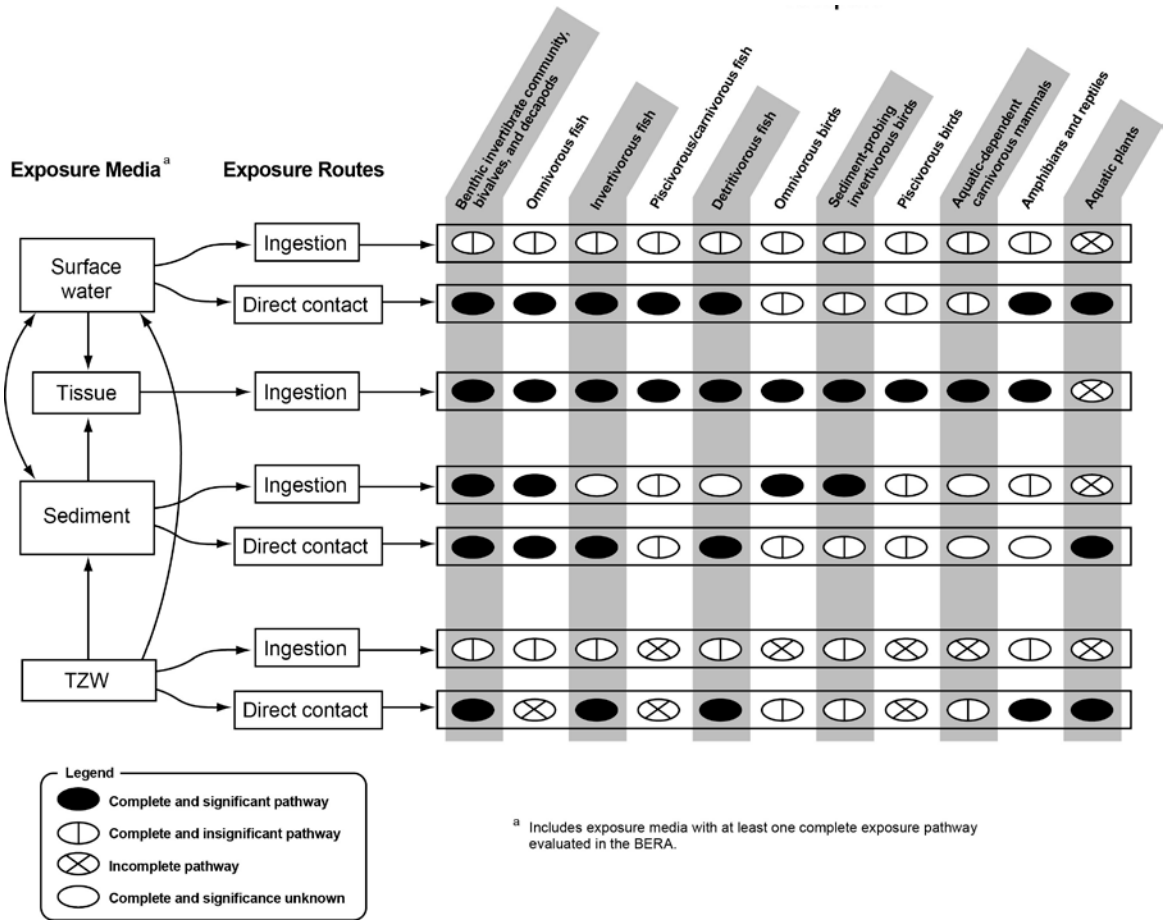
The ecological CSM illustrates the pathways that contaminants may follow from exposure media to the ecological receptors through potential exposure pathways. The exposure pathways were classified as one of four categories for each receptor: complete and significant, complete and significance unknown, complete and insignificant, and incomplete. Complete and significant pathways were quantitatively assessed in the BERA. Pathways that are complete but of unknown significance were qualitatively assessed to a level of certainty dependent on available toxicological studies and exposure data. Insignificant pathways were not evaluated in the BERA.

The refined ecological CSM (EPA 2008e) was derived from previous ecological CSMs for the Study Area (Windward 2005; Integral et al. 2007; Integral et al. 2004). A simplified version of EPA's refined ecological CSM is presented in Figure 9-1, according to the assessment endpoints and measurement endpoints identified by EPA.

[4] Juvenile Chinook salmon were evaluated at the organism level; the other selected invertivorous fish receptor species were evaluated at the population level.

[5] Belted kingfisher were evaluated in the uncertainty assessment.

[6] Bald eagle were evaluated at the organism level.



Note: The complete and detailed ecological CSM per EPA is provided in Attachment 2 of the BERA (Appendix G).
Figure 9-1. Simplified Ecological CSM

9.2.3 Refined Assessment and Measurement Endpoints

Assessment endpoints and hypotheses (expressed as risk questions) are summarized below. These endpoints and risk questions are based on those provided in EPA's Problem Formulation (EPA 2008e). Some modification of the language has been made to clarify the framework used for conducting the BERA.

Assessment endpoint: Survival, growth, and reproduction of benthic invertebrates (e.g., amphipods, isopods, bivalves, gastropods, oligochaetes, insects, decapods)

- Are the survival and biomass of benthic invertebrates, as indicated by *Hyalella azteca* and *Chironomus dilutus* exposed to bulk sediment from Portland Harbor, below biological effect thresholds that represent unacceptable effects?
- Do contaminant concentrations in bulk surface sediment from Portland Harbor exceed sediment quality values (SQVs) derived from site-specific models that

reliably predict effects based on the survival and biomass of benthic macroinvertebrates exposed to Portland Harbor sediment?

- Do contaminant concentrations in bulk sediment from Portland Harbor exceed sediment quality guidelines (SQGs) that reliably predict reductions in the survival, reproduction, or biomass of benthic macroinvertebrates, bivalves, or decapods?
- Are the survival and growth of bivalves, as indicated by the bivalve *Corbicula fluminea*, exposed to whole sediments from Portland Harbor, below biological effect thresholds that represent unacceptable effects?
- Are contaminant concentrations in Willamette River surface water or shallow TZW from Portland Harbor greater than the toxicity thresholds that are protective of the survival, growth, or reproduction of benthic macroinvertebrates (including bivalves and decapods)?
- Are contaminant concentrations in whole-body tissues of laboratory-exposed or field-collected benthic macroinvertebrates, bivalves, or decapods higher than tissue-residue thresholds for survival, reproduction, or growth of benthic invertebrates?

Assessment endpoint: Survival, growth, and reproduction of fish

- Are contaminant concentrations measured in field-collected or predicted for whole-body tissues of invertivorous, omnivorous, piscivorous, or detritivorous fish in Portland Harbor higher than tissue-residue benchmarks for survival, reproduction, or growth?
- Do tissue concentrations in prey or other potentially ingested media (i.e., sediment or water) from Portland Harbor exceed the acceptable concentrations for the survival, reproduction, or growth of invertivorous, omnivorous, or piscivorous fish consuming those media?
- Are contaminant concentrations in surface water from Portland Harbor greater than the toxicity thresholds for survival, growth, or reproduction of invertivorous, omnivorous, piscivorous, and detritivorous[7] fish?
- Are detritivorous fish more or less sensitive to waterborne contaminants than the species used to develop existing water quality criteria and TRVs?
- Are contaminant concentrations in shallow TZW greater than the toxicity thresholds for survival, growth, or reproduction of invertivorous or detritivorous fish?

Assessment endpoint: Survival, growth, and reproduction of birds and mammals

- Do tissue concentrations in fish and/or benthic invertebrate prey and other potentially ingested media (i.e., water or sediment) from Portland Harbor exceed the acceptable concentrations for the survival, reproduction, or growth of piscivorous, omnivorous, or invertivorous birds or aquatic mammals consuming those media?

[7] Detritivorous fish (i.e., lamprey ammocoetes) were evaluated only for growth and survival because they do not reproduce in the Study Area.

- Do contaminant concentrations in eggs of piscivorous birds exceed egg-based toxicity thresholds for hatchability and survival of chicks?

Assessment endpoint: Survival, growth, and reproduction of amphibians

- Do contaminant concentrations in surface waters of the Willamette River or shallow TZW exceed TRVs for the survival, reproduction, or growth of amphibians?

Assessment endpoint: Survival, growth, and reproduction of aquatic plants

- Do contaminant concentrations in surface waters of the Willamette River or shallow TZW exceed TRVs for the survival, reproduction, or growth of aquatic plants?

9.2.4 Analysis Plan

A summary of the major components described in the BERA analysis plan is provided below.

Exposure assessment – All exposure pathways classified as complete and significant in the ecological CSM (Figure 9-1) were evaluated quantitatively. Exposure concentrations in sediment, water, and tissue were based on concentrations in environmental samples at an ecological scale relevant to the receptor being evaluated or as required by EPA.

As stipulated in EPA's Problem Formulation (EPA 2008e), all COPCs first were evaluated on a sample-by-sample basis. Because a sample-by-sample scale is not ecologically relevant for most of the receptors evaluated in the BERA, COPCs were then evaluated at an exposure scale that is ecologically relevant for each specific receptor. For dietary risks to fish and wildlife, exposure estimates were also determined for a diet consisting of multiple prey species, using prey portions reported in the literature. Exposure concentrations are based both on contaminant concentrations quantified in the analytical laboratory (empirical concentrations) and, for some LOEs, on predicted values (i.e., for the tissue-residue LOE, the dietary LOE for shorebirds, and the bird egg LOE). Exposure of benthic invertebrates was assessed based on contaminant concentrations in individual samples of sediment, water, and TZW. For several selected COPCs, contaminant concentrations in tissues of fish and benthic invertebrates were estimated from sediment concentrations using a bioaccumulation model.

Effects assessment – The effects assessment involved two general approaches. For most ecological receptors, the effects of COPCs were assessed by comparing contaminant concentrations in each environmental medium to contaminant- and medium-specific toxicity benchmarks. These benchmarks are contaminant concentrations that, if not exceeded, should protect ecological receptors from adverse effects on survival, growth, or reproduction. The second approach used sediment toxicity bioassays as a direct measure of the effects of sediment chemistry on the survival and growth of benthic invertebrates in the laboratory. Consistent with EPA's Problem Formulation (EPA 2008e), for all receptors and receptor groups evaluated at the community or population level, the lowest-observed-adverse-effect level (LOAEL) was used. The no-observed-adverse-effect level (NOAEL)

was used for receptors evaluated at the organism level (i.e., bald eagle, juvenile Chinook salmon, and Pacific lamprey ammocoete).

EPA (2008b) specified the TRVs that were used in the BERA. Tissue-residue TRVs were calculated using EPA-prescribed methodology (EPA 2008c). However, some TRVs selected by EPA are associated with significant uncertainty; these TRVs were further evaluated as part of the risk characterization process. Examples include water TRVs based on dietary thresholds for wildlife that were used to assess direct exposure of aquatic organisms to surface water (i.e., total PCBs and total DDx), dietary TRVs that were at or near nutritional requirements (i.e., copper), and tissue TRVs based on species sensitivity distributions that included field-based studies that reported other co-occurring contaminants (i.e., total PCBs).

As directed by EPA in its Problem Formulation (EPA 2008e), two types of models were used to develop site-specific SQVs. Several published sets of non-site specific SQGs were also evaluated to predict unacceptable risks to the benthic community. All sets of SQGs were tested to establish their reliability as predictors of benthic toxicity in Portland Harbor.

Risk characterization and uncertainty analysis – Risk was characterized primarily as HQs calculated as the receptor- and medium-specific exposure concentrations divided by the respective effect threshold. The risk characterization integrated information on contaminant exposure and effects to identify COPCs posing potentially unacceptable risk.

Potentially unacceptable risks were identified through an iterative process of analyzing exposure and effects data for the various contaminants and ecological receptors, with increasing realism at each step in the process. For most receptors, multiple LOEs were evaluated. For each LOE, risk characterization began with a SLERA and progressed iteratively through the final step in the risk characterization. Throughout the process, receptor-contaminant pairs that showed the potential for adverse effects were further analyzed, and those that did not were screened out. The final step in the process reflects the most realistic risk estimates. Each receptor-LOE-COPC combination with an HQ $\geq$ 1 based on the final step in the risk characterization constitutes evidence of potentially unacceptable risk. In total, 89 ecological contaminants (as individual contaminants, sums, or totals) were identified in the BERA as posing potentially unacceptable risk.

Table 9-1 presents a summary of the contaminants posing potentially unacceptable risk for each ecological receptor group. Additional information was used to characterize the numerical risk estimates relative to assessment endpoints at the population or community level. Uncertainties in the exposure and effects data, spatial extent of HQs $\geq$1, magnitude of HQs, level of effect represented by the TRV, and WOE (e.g., consistency across multiple LOEs) were considered in formulating risk conclusions.

Table 9-1. Contaminants Posing Potentially Unacceptable Risk to Ecological Receptors

Line of Evidence	COPCs with HQs ≥ 1
BENTHIC INVERTEBRATES	
Macroinvertebrates (e.g., amphipods, isopods, bivalves, gastropods, oligochaetes, insects, decapods)	
Survival and biomass measured in sediment bioassays	Not applicable; responses based on contaminant mixtures
Survival and biomass predicted from site-specific SQVs	6 metals, TBT, 19 PAHs or group sums, dibutylphthalate, 3 SVOCs, 2 phenolic compounds, PCBs, 15 pesticides or group sums
Sediment concentrations compared with SQGs	8 metals, 14 PAHs or group sums, 2 PCBs, 9 pesticides or group sums
Surface water concentrations compared with water TRVs	Zinc, monobutyltin, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, total DDx, ethylbenzene, trichlorethene
TZW concentrations compared with water TRVs	14 metals, 16 PAHs, 3 SVOCs, 4,4′-DDT, total DDx, 16 VOCs, gasoline–range hydrocarbons, cyanide, perchlorate
Tissue concentrations in epibenthic organisms compared with tissue TRVs (field-collected, laboratory-exposed, and predicted)	Arsenic, copper, zinc, TBT, PCBs, total DDx
Bivalves (clams, mussels)	
Tissue concentrations compared with tissue TRVs (field-collected, laboratory-exposed, and predicted)	Copper, zinc, TBT, BEHP, PCBs, total DDx
Survival in bioaccumulation tests	Not applicable; responses based on contaminant mixtures
Survival and biomass in sediment bioassays	Not applicable; responses based on contaminant mixtures
Surface water concentrations compared with water TRVs s	Zinc, monobutyltin, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, total DDx, ethylbenzene, trichlorethene
TZW concentrations compared with water TRVs	14 metals, 16 PAHs, 3 SVOCs, 4,4′-DDT, total DDx, 16 VOCs, gasoline–range hydrocarbons, cyanide, perchlorate
Sediment concentrations compared with SQGs	8 metals, 14 PAHs or group sums, 2 PCBs, 9 pesticides or group sums
Decapods (crayfish)	
Tissue concentrations compared with tissue TRVs (field-collected, predicted)	Copper, total PCBs, total DDx
Sediment concentrations compared with SQGs	8 metals, 14 PAHs or group sums, 2 PCBs, 9 pesticides or group sums

Table 9-1. Contaminants Posing Potentially Unacceptable Risk to Ecological Receptors

Line of Evidence	COPCs with HQs ≥ 1
Surface water concentrations compared with water TRVs s	Zinc, monobutyltin, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, total DDx, ethylbenzene, trichlorethene
TZW concentrations compared with water TRVs	14 metals, 16 PAHs, 3 SVOCs, 4,4′-DDT, total DDx, 16 VOCs, gasoline–range hydrocarbons, cyanide, perchlorate
FISH	
Omnivores (white sturgeon, largescale sucker)	
Measured tissue concentrations compared with tissue TRVs	Total PCBs
Dietary dose compared with dietary TRVs	Copper
Surface water concentrations compared with water TRVs	None
Correlation of lesion prevalence with areas of contamination, comparison with lesion-based TRVs (if relevant to receptor species)	Inconclusive for PAHs
Invertivores (juvenile Chinook salmon, peamouth, sculpin)	
Tissue concentrations (measured and predicted) compared with tissue TRVs	Copper, lead, total PCBs, total DDx
Dietary dose compared with dietary TRVs	Cadmium, copper, TBT
Surface water concentrations compared with water TRVs	Zinc, monobutyltin, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, total DDx, trichlorethene
Shallow TZW concentrations compared with water TRVs	14 metals, 16 PAHs, 3 SVOCs, 4,4′-DDT, total DDx, 16 VOCs, gasoline-range hydrocarbons, cyanide, perchlorate
Piscivores (northern pikeminnow, smallmouth bass)	
Tissue concentrations (measured and predicted) compared with tissue TRVs	Antimony, lead, total PCBs
Surface water concentrations compared with water TRVs	Zinc, monobutyltin, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP
Dietary dose compared with dietary TRVs	Copper
Detritivores (Pacific lamprey ammocoete)	
Measured tissue concentrations compared with tissue TRV	Copper
Surface water concentrations compared with water TRVs	None

Table 9-1. Contaminants Posing Potentially Unacceptable Risk to Ecological Receptors

Line of Evidence	COPCs with HQs ≥ 1
Shallow TZW concentrations compared with water TRVs	14 metals, 16 PAHs, 3 SVOCs, 4,4′-DDT, total DDx, 16 VOCs, gasoline-range hydrocarbons, cyanide, perchlorate
BIRDS	
Invertivores (spotted sandpiper)	
Dietary dose compared with dietary TRV	Copper, benzo(a)pyrene, dibutyl phthalate, total PCBs, PCB TEQ, dioxin/furan TEQ, total TEQ, sum DDE, total DDx, aldrin
Omnivores (hooded merganser)	
Dietary dose compared with dietary TRV	Total PCBs
Piscivores (osprey, bald eagle)	
Dietary dose compared with dietary TRV	Lead, mercury, total PCBs
Measured concentrations in osprey eggs compared with egg- or embryo-based TRVs (DDT and metabolites, PCBs, and dioxin-like compounds only)	Total PCBs, PCB TEQ, dioxin/furan TEQ, total TEQ, sum DDE
AQUATIC-DEPENDENT MAMMALS (mink, river otter)	
Dietary dose compared with dietary TRVs	Aluminum, lead, total PCBs, PCB TEQ, dioxin/furan TEQ, total TEQ
AMPHIBIANS	
Surface water concentrations compared with water TRVs	Zinc, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, total DDx
Shallow TZW concentrations compared with water TRVs	11 metals, 8 PAHs, the SVOC 1,2-dichlorobenzene, 4,4′-DDT, total DDx, 8 VOCs, gasoline-range hydrocarbons, cyanide, perchlorate
AQUATIC PLANTS	
Surface water concentrations compared with water TRVs	Zinc, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, total DDx
Concentrations in shallow TZW compared with water TRVs	11 metals, 8 PAHs, the SVOC 1,2-dichlorobenzene, 4,4′-DDT, total DDx, 8 VOCs, gasoline-range hydrocarbons, cyanide, perchlorate

Notes: Potentially unacceptable risk is indicated by HQ ≥ 1. A more comprehensive version of the information summarized here is available in Table 11-1 of the BERA (Appendix G). The assessment endpoints are based on protection and maintenance of populations, except for species of special status, (i.e., juvenile Chinook salmon, Pacific lamprey ammocoete, bald eagle), whose assessment endpoint is based on the protection of individual organisms.

BEHP – bis(2-ethylhexyl) phthalate
BERA – baseline ecological risk assessment
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SQG – sediment quality guideline
SQV – sediment quality value
SVOC – semivolatile organic compound
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value
TZW – transition zone water
VOC – volatile organic compound

9.3 BENTHIC INVERTEBRATE RISK ASSESSMENT

Fifty-five contaminants (individual contaminants, sums, or totals) were identified as posing potentially unacceptable risk to benthic invertebrates because they exceeded site-specific SQVs, tissue TRVs, or surface water TRVs. COPCs for TZW were identified by comparing undiluted TZW concentrations with water TRVs. These COPCs include eight metals (arsenic, cadmium, chromium, copper, lead, mercury, silver, and zinc), TBT, monobutyltin, 20 individual PAHs and PAH sums, BEHP, dibutyl phthalate, phenol, 4-methylphenol, three SVOCs (dibenzofuran, benzyl alcohol, and carbazole), total PCBs, two VOCs (trichloroethene and ethylbenzene), and 15 pesticides (various DDx forms, dieldrin, endrin, endrin ketone, endosulfan, chlordane, and beta- and delta-hexachlorocyclohexane).

TZW COPCs with HQs ≥ 1 include 14 metals (barium, beryllium, cadmium, cobalt, copper, iron, lead, magnesium, manganese, nickel, potassium, sodium, vanadium, zinc), 16 individual PAHs (2-methylnaphthalene, acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, pyrene), three SVOCs (1,2-dichlorobenzene, 1,4-dichlorobenzene, dibenzofuran), pesticides (4,4′-DDT, total DDx), 16 VOCs (1,1-dichloroethene, 1,2,4-trimethylbenzene, 1,3,5-trimethylbenzene, benzene, carbon disulfide, chlorobenzene, chloroethane, chloroform, cis-1,2-dichloroethene, ethylbenzene, isopropylbenzene, m,p-xylene, o-xylene, toluene, total xylenes, trichloroethene), gasoline-range hydrocarbons, cyanide, and perchlorate.

SPI data were examined to determine whether locations associated with sediment toxicity tended to have a less mature benthic community structure than would be expected for the physical characteristics of the location. The evaluation focused on the locations where community response was worse than expected (potentially due to non-physical factors). Specifically, the evaluation focused on locations where immature (Stage 1) and transitional (Stage 2) communities were present in fine-grained depositional environments. An attempt was made to identify other factors influencing the community, including bottom slopes and sediment chemistry. The SPI data analysis was used as corroborative evidence in the BERA. The analysis suggested that the physical environment in the Study Area can explain the condition of the benthic community throughout this area of the river. In over 90 percent of the images evaluated, the successional stage matched the expected community based on the

physical regime, when slope was included as a habitat characteristic. Of the 31 cases where the community stage was not as might be predicted by the physical environment, 19 appear to be related to sediment toxicity. These qualitative results suggest that the benthic community is typical of a large river system that is predominantly influenced by physical processes. Impacts from sediment contamination appear to be primarily associated with depositional areas that have received historical releases of contamination.

The potential for unacceptable risk to the benthic invertebrate community was determined spatially using a WOE framework that overlaid the results of the primary LOEs. First, bioassay results were mapped for four test endpoints (*Chironomus dilutus* survival, *Chironomus dilutus* biomass, and the same for *Hyalella azteca*). Next, sediment COPC concentrations were compared with site-specific SQVs based on both the floating percentile model and the logistic regression model to predict several different levels of benthic toxicity in locations without bioassay data. Next, these sediment chemistry data were used (with bioaccumulation models[8]) to predict where organisms might accumulate COPCs to concentrations above tissue TRVs. Locations where estimated sediment concentrations exceed SQVs or where empirical or predicted tissue concentrations exceed TRVs were identified as potential benthic risk areas. Empirical tissue-residue data were mapped, allowing a visual assessment of concordance across LOEs. Water (both surface water and TZW) TRV exceedances were considered along with sediment SQV and tissue TRV exceedances. Water TRV exceedances were found to co-occur with areas that had the greatest SQV and TZW TRV exceedances. Based on the spatial analysis, PAHs, PCBs, and DDx were found to pose potentially unacceptable risks to the benthic invertebrate community. The spatial evaluation identified approximately 7 percent of the Study Area that poses potentially unacceptable risk to the benthic community.

9.4 FISH RISK ASSESSMENT

Four primary quantitative LOEs were used to characterize risks to fish: the tissue-residue LOE, the dietary-dose LOE, the surface water LOE, and the TZW LOE. Benthic fish exposure to PAHs in sediment was also evaluated as a qualitative LOE per EPA's Problem Formulation (EPA 2008e) and included an assessment of the apparent health of pre-breeding sturgeon; this LOE was inconclusive.

Fifty-nine contaminants were identified as posing potentially unacceptable risk to at least one fish species based on the tissue, dietary, surface water, and/or TZW LOEs (Table 9-1). Of these, 44 COPCs had HQs ≥ 1 only for the TZW LOE. The exposure and effects data are insufficient to evaluate risk to fish from 17 tissue-residue COIs, 11 dietary COIs, 5 surface water COIs, and 9 TZW COIs.

Risk conclusions for each fish receptor were reached by evaluating the risk estimates and the reliability of each LOE. Total PCBs were found to pose low risk to populations of piscivorous fish (smallmouth bass and northern pikeminnow) and the small-home-range invertivorous fish sculpin. The potential for adverse effects on fish assessment endpoints

[8] The analysis used the same bioaccumulation models created for developing the PRGs to be used in the FS.

from total PCBs established by the tissue-residue LOE was assessed to be low: the other LOE used to evaluate PCB risks—surface water—resulted in HQs < 1, tissue-residue HQs ≥ 1 occurred over a moderate spatial extent (or in relatively few samples for large-home-range fish), and uncertainty in the tissue-residue TRV is more likely to over- than underpredict risk.

Several TZW COPCs, including benzo(a)anthracene, benzo(a)pyrene, naphthalene, total DDx, ethylbenzene, and trichloroethene, have high concentrations in localized areas that could adversely affect sculpin or Pacific lamprey ammocoetes at those locations. Because the spatial extent of such exceedances is limited, adverse effects on populations are not expected.

Thirty-eight[9] TZW COPCs, including six metals (barium, iron, manganese, sodium, vanadium, and zinc),[10] 16 PAHs (2-methylnaphthalene, acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, pyrene), two SVOCs (1,2-dichlorobenzene, 1,4-dichlorobenzene), the pesticides 4,4-DDT and total DDx, 10 VOCs (benzene, carbon disulfide, chlorobenzene, chloroform, cis-1,2dichloroethene, ethylbenzene, o-xylene, toluene, total xylenes, and trichloroethene), cyanide, and perchlorate pose potentially unacceptable risk to Pacific lamprey ammocoetes in localized areas associated with groundwater discharges to the river. However, the water TRVs were derived to be protective of highly sensitive species and probably overpredict potentially unacceptable risk to lamprey ammocoetes, whose sensitivity to contaminant toxicity across several modes of action was average or lower than that of most aquatic species (Andersen et al. 2010). Also, given their feeding habits and the low oxygen levels at the depths represented by the TZW samples, lamprey ammocoetes have relatively lower exposure to TZW than to surface water. The exposure assessment conservatively assumed that lamprey ammocoetes are exposed to undiluted TZW.

Risks to fish from other COPCs with HQs ≥ 1 in the final step of the risk characterization were found not likely to result in ecologically significant adverse effects at the population level. In some cases, the selected TRV probably underestimates the threshold for ecologically significant adverse effects; and in other cases, the great majority of samples used to estimate exposure resulted in HQs < 1. Furthermore, TZW exposure assumptions probably overestimate risk.

[9] Because petroleum compounds are not CERCLA contaminants, gasoline-range aliphatic hydrocarbons (C10 – C12) have been excluded from the count, even though they may be contributing to potentially unacceptable risk. These contaminants were evaluated as an uncertainty, have HQ > 10, and may pose risk to individual lamprey.

[10] There is substantial uncertainty as to whether the source of barium, iron, and manganese in TZW is anthropogenic.

9.5 WILDLIFE RISK ASSESSMENT

Risks to wildlife receptors were evaluated using two LOEs. Dietary dose was used for all six wildlife receptors. Ingestion of prey tissue and incidental ingestion of sediment are reflected in dietary-dose estimates. Tissue residues in bird eggs were used as a second LOE for bald eagle and osprey.

Twelve contaminants (copper, lead, mercury, benzo(a)pyrene, dibutyl phthalate, total PCBs, PCB TEQ, total dioxin/furan TEQ, total TEQ, aldrin, 4,4′-DDE, and total DDx) were identified as posing potentially unacceptable risk for at least one bird receptor. Six contaminants (aluminum, lead, total PCBs, PCB TEQ, total dioxin/furan TEQ, and total TEQ) were identified as posing potentially unacceptable risk to mink or river otter.

Of these COPCs, total PCBs pose the primary risk. Calculated risk estimates indicate that populations of both mink and river otter in the Study Area might be experiencing reduced reproductive success because of exposure to PCBs. Reproductive success in spotted sandpiper, bald eagle, and osprey might also be reduced because of PCB exposure. A greater degree of uncertainty is associated with PCB risk estimates for birds than for mammals because of greater uncertainties about exposure and the effects data. Uncertainty is higher for otter than for mink because otter-specific effects data are lacking.

Total TEQ exposure also poses the potential for reduced reproductive success in mink, river otter, spotted sandpiper, bald eagle, and osprey. The PCB TEQ is responsible for the majority of total TEQ exposure, but the total dioxin/furan TEQ also exceeds its TRV in some locations of the Study Area. As is the case for total PCBs, and for the same reasons, a greater degree of uncertainty is associated with risk estimates for birds and otter than for mink.

Osprey egg data (used as a surrogate for bald eagle) suggest that DDx compounds could pose low risk of reduced reproductive success to individual bald eagle organisms within limited portions of the Study Area. Concentrations of total DDx in egg tissue from two of five exposure areas exceeded the NOAEL for bald eagle, but not where the highest sediment DDx concentrations occurred. In all five exposure areas, egg concentrations were below both the bald eagle-specific LOAEL TRV and the dietary effect threshold. There is significant uncertainty about the source of DDx in the osprey eggs collected from the Study Area, given that adults nest and lay eggs shortly after returning to the LWR from their overwintering grounds in Mexico and Central America.

Wildlife COPCs with HQs ≥ 1 in the final step of the risk characterization were found unlikely to result in ecologically significant adverse effects in the receptor populations: the HQs are of low magnitude and over a limited spatial extent, with uncertainties in exposure and effects likely to result in overestimated risk.

9.6 AMPHIBIAN AND REPTILE RISK ASSESSMENT

The ecological CSM (Figure 9-1) classified ingestion of prey, direct contact with surface water, and direct contact with TZW as complete and significant exposure pathways for

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amphibians. Although the CSM identifies dietary exposure as a complete and significant exposure pathway for amphibians, this pathway was not evaluated because it was not specified in EPA's Problem Formulation (EPA 2008e) and because dietary-based TRVs for quantitative assessment are not available. Because amphibians had been selected as a receptor that is protective of reptiles, risk conclusions for amphibians also apply to reptiles.

Two LOEs were used to characterize risks to amphibians and aquatic plants: surface water and TZW. The same exposure and effects data were used to assess risk to amphibians and aquatic plants, although several uncertainties apply to only one of the receptor groups. Thirty-three contaminants were identified as posing potentially unacceptable risk to amphibians. Five of these COPCs have HQs ≥1 in both surface water and TZW (zinc, benzo(a)anthracene, benzo(a)pyrene, naphthalene, and total DDx); one is a COPC based on surface water only (BEHP), and 27 are COPCs based on TZW only (Table 9-1). None of these COPCs is expected to have ecologically significant adverse effects on amphibian or aquatic plant populations in the Study Area. In general, surface water concentrations of COPCs were below amphibian-specific thresholds, exceeded surface water thresholds by a low magnitude and at low frequency, or both. These conditions indicate low to negligible risk. Furthermore, surface water contaminants with HQs ≥ 1 occur in samples collected during the less-sensitive, non-reproductive periods (when amphibians may not be present in the Study Area), again indicating negligible risk to amphibian populations. For the TZW LOE, the great majority of samples result in HQs < 1, indicating a limited spatial extent of exceedance. Although highly uncertain, risk to amphibians from TZW is likely negligible because of a low level of exposure to TZW. There is some uncertainty concerning the relevance of selected TRVs: aquatic plant, algae, or amphibian-specific thresholds for several COPCs are either limited or not available.

9.7 AQUATIC PLANT RISK ASSESSMENT

As presented in the ecological CSM (Figure 9-1), three complete and significant exposure pathways were identified for the aquatic plant community. These pathways are direct contact with surface water, direct contact with sediment, and direct contact with TZW. The direct sediment contact pathway was considered complete but was not evaluated per EPA's Problem Formulation (EPA 2008e) because no appropriate studies reporting adverse effects on aquatic plants from sediment-associated contaminants were identified in the literature.

The same exposure and effects data were used to assess risk to amphibians and aquatic plants, and the contaminants posing potentially unacceptable risk are the same (Table 9-1). In general, surface water concentrations were below algae-specific thresholds or exceed thresholds by a low magnitude and at low frequency, indicating low to negligible risks to the aquatic plant community. For the TZW LOE, the great majority of samples result in HQs < 1, indicating a limited spatial extent of exceedance. Although highly uncertain, risk to aquatic plants from TZW is likely negligible because of a low level of exposure to TZW. There is high uncertainty concerning the relevance of the selected TRVs for zinc, benzo(a)anthracene, BAP, and BEHP to aquatic plants because algae-specific thresholds are highly variable or because limited or no aquatic plant- or algae-specific thresholds are available for comparison.

9.8 RISK MANAGEMENT RECOMMENDATIONS

Section 12 of the BERA (Appendix G) provides the LWG's ecological risk management recommendations to develop and evaluate remedial alternatives that are protective of ecological resources. In summary, the risk management recommendations are as follows:

- Total PCBs and total TEQ are the recommended contaminants of concern (COCs) for assessing risk to all ecological receptors except benthic organisms (including fish). Mink is the recommended receptor of concern.
- 4,4′-DDT, total DDx,[11] chlorobenzene, benzo(a)anthracene, benzo(a)pyrene, naphthalene, carbon disulfide, cyanide, cis-1,2-dichloroethene, and trichloroethene are the recommended TZW COCs. These recommendations presume that groundwater source control measures will be implemented prior to sediment remedies. DEQ is working with upland property owners to implement groundwater source control measures prior to sediment remedies.
- Most of the contaminants posing potentially unacceptable risk were not recommended as COCs for non-benthic receptors based on risk characterization considerations (magnitude, spatial extent, and ecological significance of HQs $\geq$ 1). This list includes all the metals, butyltin, phthalate, pesticide, and VOC COPCs.
- The benthic risk assessment methodologies are designed to address contaminant mixtures and do not conclusively identify contaminants causing toxicity. For protection of the benthic invertebrate community, methodologies for delineating benthic areas of concern (rather than identifying COCs) and for evaluating remedial action alternatives are recommended. Recommended benthic areas of concern were mapped by applying the comprehensive benthic approach based on EPA's April 21, 2010, guidelines for assessing benthic risk in the FS (EPA 2010). These maps are presented in Section 12 of the BERA (Appendix G).

[11] There is uncertainty associated with 4,4′-DDT and total DDx as COCs because HQs based on filtered samples are less than 100. This suggests that the risk from DDx compounds in TZW may be lower than indicated by the maximum concentrations in unfiltered samples because of the lower bioavailability of the particulate-bound fraction of the contaminant.

10.0 RI CONCEPTUAL SITE MODEL FOR SELECT INDICATOR CHEMICALS

The updated CSM for the Portland Harbor Study Area (RM 1.9 to 11.8) is presented in this section. A CSM is a written or pictorial representation of an environmental system and the biological, physical, and chemical processes that affect the transport of contaminants from sources through environmental media to human and ecological receptors in the system. CSMs are a critical component of the RI/FS process because they establish the link between investigation data and the assessment of risk (ASTM 1995). Importantly, CSMs establish a context for evaluating potential site-associated sources and risk versus non-site-associated sources and risk.

Within this framework, this section presents a CSM update for the Portland Harbor RI/FS Site that draws on and synthesizes supporting information presented previously in this RI Report or in associated RI/FS documents. A preliminary CSM for the Portland Harbor ISA was provided in the Programmatic Work Plan (Integral et al. 2004); an updated CSM, which focused on groundwater sources and pathways, was submitted later in 2004 (Integral and GSI 2005a,b,c). An interim CSM was presented in the Round 2 Report (Integral et al. 2007) that incorporated all data collected through Round 2 of the RI. The CSM update presented in this section integrates all Study Area data and information available through the final round of RI sampling (Round 3). Future steps involving updates and refinements of the CSM for the RI/FS are noted in Section 11.

Section 10 is divided into two subsections. Section 10.1 presents a Study Area-wide overview of the physical setting; contaminant distribution in sediments; contamination sources identified to date; external loading and internal fate and transport mechanisms; human health risk drivers and potentially complete exposure pathways/scenarios; and ecological risk drivers and ecological receptors/exposure pathways. Section 10.1 is not focused on specific contaminants but, instead, discusses potential sources and upland activities that may have used and/or released the contaminants measured in Study Area sediments at levels that potentially pose unacceptable risk. Although this and earlier sections identify many specific sources of contamination, neither this section nor this RI report generally is intended by the LWG as an exhaustive list of current or historical sources of contamination. Identification and evaluation of potential sources is still ongoing. All historical sources may never be known, and current sources likely will continue to be discovered into the future. However, sufficient information about likely significant historical and current sources is available for preparation of the FS to begin.

Section 10.2 presents and briefly describes a detailed, graphical CSM presentation for the 13 specific CSM ICs selected, in consultation with EPA, in the fall of 2008, for focus in this RI. The development of separate CSMs for different individual contaminants is consistent with EPA (2005a) guidance. The 13 ICs were identified from the extensive list of COIs to represent the nature and extent of the range of contaminants that potentially pose risk to human health and the environment in

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sediment, surface water, TZW, and biota. Additional COIs beyond the CSM IC list are present in the Study Area and may pose risk to human health and the environment. The CSM IC list was generated based on the contaminants that emerged from the Round 2 risk screening and preliminary risk evaluation process and consideration of the following non-risk-based factors:

- Frequency of detection—Chemicals with a high frequency of detection (e.g., generally >50%) were selected as ICs.
- Cross media comparisons—Preference was given to chemicals that would allow comparisons across media.
- Representative chemicals—Several ICs were selected to represent a suite of compounds (e.g., BEHP was selected to represent the phthalate group).
- EPA requests—EPA requested the inclusion of several additional contaminants for sediment and biota based on its review of the Round 2 Report (Integral et al. 2007). EPA also requested the inclusion of certain other contaminants with widespread sources in the harbor (e.g., several metals) during the IC lists development discussions in 2008.

The 13 CSM ICs selected, mapped, and discussed in Section 10.2 are:

- Total PCBs (congeners or Aroclors)
- TCDD TEQ (mammalian 2005 TEFs [Van den Berg et al. 2006])
- Total DDx
- Total PAHs
- Bis(2-ethylhexyl)phthalate
- Total chlordanes
- Aldrin
- Dieldrin
- Arsenic
- Copper
- Zinc
- Chromium
- Tributyltin ion.

Section 10.2 includes a series of contaminant-specific maps of the Study Area's abiotic and biotic data sets that illustrate relationships between the observed contaminant distributions and known and likely historical and current sources and pathways. These displays are intended to provide a picture of the distribution, transport, and fate of

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contaminants in the Study Area across a range of physical, chemical, and biological processes, as well as potential sources. PCBs are the most widespread organic contaminant group in the Study Area and present the most significant risk to human health. TCDD TEQ, total DDx, and total PAHs also are widespread throughout the Study Area, and in most cases, the distribution of these contaminants, together with PCBs, create the outer boundaries of potentially unacceptable contaminant-related human health and environmental risk in Portland Harbor. Consequently, these four contaminants receive a detailed narrative in Section 10.2 of the CSM, with briefer narratives provided for the remaining nine ICs that are found throughout the Study Area but are not as widespread as these four contaminant groups. The baseline risks from the remaining nine chemicals are also lower.

The general objective of this CSM is to illustrate our understanding of the sources and fate and transport mechanisms that determine the observed distribution of individual contaminants in affected abiotic and biotic media across the Study Area, based on the information and data collected, compiled, and evaluated in this RI. The emphasis of this CSM is on the area-wide distribution of the 13 selected ICs. Specific objectives of this CSM update include the following:

- Updating the CSM based on the updated RI data set and other updated information, e.g., on sources
- Providing an overview of the relative importance of exposure and contaminant fate and transport pathways to overall risk for the individual ICs
- To the extent possible with current information, identifying general sources of contamination and determining whether the sources are current or historical in nature to support the FS (e.g., recontamination and MNR potential assessment)
- Identifying and characterizing significant fate and transport processes that could affect risk management decisions and/or remedial design (to be further evaluated by FS fate and transport modeling).

The CSM provided will be refined as necessary as part of the FS and will be focused on individual sediment management areas identified in the FS.

10.1 SITE CONCEPTUALIZATION

This section summarizes the Study Area CSM, including an overview of in-river physical environments; contaminant distribution and site sources; loading, fate, and transport mechanisms; and relevant exposure pathways for human and ecological receptors. Section 10.2 presents detailed CSMs for the 13 selected CSM ICs.

A pictorial representation illustrating the major elements of the CSM (sources, pathways, fate and transport mechanisms, and human and ecological receptors) for the Portland Harbor Study Area is shown in Figure 10.1-1, while Figure 10.1-2 presents a graphical conceptualization of the sources, release mechanisms, transport media, and

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exposure media of the CSM. The detailed human health and ecological CSMs for the Portland Harbor Site are summarized in Appendix F, Figure 3-1 (also RI Section 8, Figure 8.2-1) and Appendix G, Attachment 2, Figure 1 (also RI Section 9, Figure 9.1), respectively, and focus on exposure routes and receptor groups.

10.1.1 Physical Setting

The physical setting of the LWR is detailed in Section 3. A conceptual-level summary of that information is presented here as a basis for the CSM physical environment.

10.1.1.1 Physical Alterations of the River

The Portland Harbor RI/FS Study Area (RM 1.9 to 11.8 of the Willamette River) is located at the downstream end of the LWR, which extends from the Willamette Falls at RM 26 to its convergence with Columbia River at RM 0. In its natural, undisturbed state, the Study Area reach was a relatively shallow, meandering portion of the LWR, surrounded by uplands, forested wetlands, and floodplains. As the Portland Harbor was developed as a shipping harbor over the last century, the channel has been redirected, dredged, and straightened (see Map 3.1-6). Nearshore and riparian areas have been filled, channelized, and modified with flood and erosion control measures to allow industrial development of uplands adjacent to the harbor. Section 3.2.1 provides a detailed summary of physical alterations of the river over time.

Much of the original riverbank in Portland Harbor has been filled, stabilized, and/or engineered for commercial, industrial, and marine operations with riprap, bulkheads, and overwater piers and docks. Consequently, shallow-profile beaches, nearshore benches, and submerged or overwater vegetation are limited. Currently, a federal navigation channel authorized to –43 ft CRD but maintained to –40 ft CRD, approximately double the depth of the natural riverbed, runs to RM 11.6, nearly the entire length of the Study Area. In some locations of the Study Area, the navigation channel extends nearly bank-to-bank, resulting in an altered river cross section. This extensive physical alteration and the associated anthropogenic activities (e.g., industry, vessel traffic, roads and bridges, and other urban uses), as well as upstream river-stage control through the construction and management of dams, have resulted in a river reach that little resembles its pre-industrialized character in terms of hydrodynamics, sediment processes, and ecological habitat.

10.1.1.2 Flow Regime, Hydrodynamics, and Sedimentation

The Willamette River is the thirteenth largest river in the contiguous United States in terms of discharge, averaging about 40,000 cfs. Flows are highly variable, however, both seasonally and year-to-year as a function of rain and snowpack levels in the region. Discharge typically varies seasonally by a factor of 10, with late-summer, dry-season levels at or below 10,000 cfs and rainy season December/January averages that approach and periodically exceed 100,000 cfs. Thirteen federal dam/reservoir systems on the upper Willamette River and its tributaries are used to stabilize river flow by storing water in the winter months and releasing it in the summer. Nonetheless,

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discharge events approaching 200,000 cfs occur every few years, and exceptionally large precipitation and snow melt events can still result in major floods. The February 1996 event nearly flooded downtown Portland as the Willamette River discharge exceeded 400,000 cfs (40–50 times greater than typical low-flow levels). This combination of river regulation, high seasonal flow variability, and high levels of anthropogenic activity within the Study Area results in potentially complex and variable sediment transport dynamics over time.

In a regional watershed context, the Study Area is a relatively low-energy, depositional reach of the LWR and the entire Willamette River watershed. The upstream portion of the LWR (from approximately RM 11 in the upper Study Area to the Willamette Falls at RM 26) is markedly narrower, more confined by bedrock outcrops in places, and faster flowing than the Portland Harbor reach. The reach immediately downstream of the Study Area (RM 1.9 to the Columbia River) also narrows as the river turns toward north and converges with the Columbia. In addition, the Multnomah Channel exits the LWR at RM 3, considerably reducing the LWR discharge downstream of this point. This physical setting and the associated hydrodynamic interactions result in deposition and accumulation of some of the suspended, and most of the bedload, sediments that enter the Study Area from upstream over time.

Within the Study Area, there are distinct sub-reaches that share similar hydrodynamic characteristics (see Section 3.1.5). The primary factors controlling river flow dynamics and riverbed character appear to be the river cross-sectional area and navigation channel width. In the upper end of the Study Area, from RM 11.8 downstream to RM 10, the river gradually widens from the downtown reach located immediately upstream (approximately RM 12 to 16), and some areas of substantial deposition are observed, especially in borrow pits and along the western portion of the channel (see Map 3.1-7). However, flows are high enough in this section that much of the area can be subject to scour, especially the eastern half of the channel, and the sub-reach is generally dominated by sandy sediments. RM 5 to 7 shows similar characteristics, and the river in this region is narrow compared to the reaches just upstream and downstream. The channel in the remainder of the Study Area, from RM 7 to 10 and downstream of RM 5 to 1.9, is generally dominated by fine-grained sediments and appears to be stable or depositional over time (see Map 3.1-12), with the possible exception of the outside, western bend of the river from the head of the Multnomah Channel (RM 3) to RM 1.9.

Relatively long-term net sedimentation rates can be estimated for the Study Area based on empirical information. Map 3.1-7 shows the measured riverbed elevation changes over the seven-year period from 2002 to 2009 and illustrates the pattern of general shoaling in the reaches from RM 7 to 10 and RM 2 to 5 and no change or scour in the higher energy reaches upstream of RM 10, and between RM 5 and 7. The maximum net sedimentation accumulation occurs in the navigation channel between RM 9 and 10 and in the upstream borrow pits at RM 10.9 and 10.5. Maximum sedimentation rates over the seven-year period approach and, in some places, exceed 30 cm/yr in these areas. Independent estimates of net sedimentation rates over a 29-year period have been

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approximated from infilling rates estimated for the borrow pits at RM 10.9 and 10.5, which were dredged in 1988 to a depth of –75 ft NAVD88 (see Section H4.2.2, Appendix H). Based on the mudline elevation in these borrow pits, measured in January 2009, long-term rates of sedimentation (which appears to be a combination of bedload and suspended load inputs) are estimated to be approximately 41 and 31 cm/yr at RM 10.9 and 10.5, respectively.

Shoaling on a similar scale along the western half of the navigation channel, from RM 8 to 10, is evident from the 2002 to 2009 bathymetric change data set (e.g., a maximum accumulation rate of 31 cm/yr on the shoal at RM 9.6); this area has historically required regular maintenance dredging. Bathymetric change data from 2002 to 2009 in the downstream channel shoaling area, which begins at RM 2.8 and extends downstream towards the confluence with the Columbia River, showed a maximum sediment rate of about 18 cm/yr at RM 2 over this seven-year time frame. The decrease in net sedimentation rates between upstream and downstream channel shoaling areas is consistent with a single major source of sediments (both suspended sediments and bedload sediments) that enter the Study Area from upstream and then settle out or are trapped in depressions and shoaling areas from upstream to downstream.

Estimates of net sedimentation rates for nearshore (i.e., shoreward of the federal navigation channel and off-channel areas in the Study Area (e.g., Swan Island Lagoon) are based on bathymetric change data, SPI observations (SEA 2002b), and limited radioisotope sampling for MNR assessment (Anchor 2005). These data indicate that sediments do not accumulate in nearshore areas at the rates noted above for the major shoals in the channel environment. While many nearshore areas exhibit fine-grained sediments and appear to be depositional (e.g., based on SPI interpretation), little net elevation change and/or small-scale scour was observed in many nearshore areas, such as RM 9 to 11E, areas within Swan Island Lagoon and Willamette Cove, RM 6.5 to 7.5W, and RM 5 to 6.5E (see Map 3.1-7) from 2002 to 2009. Short-term active surface sediment mixing is also suggested by these data sets; this may be due to anthropogenic disturbance factors in many areas of the working harbor. These observations from the bathymetric and SPI survey data are supported by the radioisotope data from four nearshore areas collected in 2004 in water depths of –4 to –28 ft NAVD88, which showed well-mixed surface sediment layers and calculated net sedimentation rates of approximately 1 cm/yr. The HST modeling effort being conducted for the FS will further evaluate net sedimentation rates throughout the Study Area over various flow conditions on a more refined spatial scale.

One objective of the HST modeling effort for the RI was to predict large-scale erosion in the Study Area under major flood events. The empirical bathymetric change data from 2002 to 2009 suggest that during typical seasonal high flows, net erosion is largely limited to 30 cm or less in extent. The flood scenario output from the revised HST model predicts that significantly deeper gross and net erosion, as much as two meters in localized areas of the navigation channel, could occur in some portions of the Study Area (Map 3.1-9b). In general, these areas are limited in extent to the relatively high-

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energy reaches of the Study Area (RM 10–11.8 and RM 5–7) mentioned above, which are dominated by sands. These areas also exhibited evidence of short-term, smaller-scale net erosion from 2002 to 2009 (Map 3.1-12). Visual comparison of Maps 3.1-9a (predicted net change) and 3.1-9b (predicted maximum erosion) shows the areal extent of backfilling that occurs over the course of the modeled high flow event.

In summary, based on the seven-year bathymetric change data (Map 3.1-7), most of the channel and off-channel areas appear to accumulate sediment or show minimal change over time. While most nearshore areas appear to be depositional environments, some areas seem to be subject to frequent disturbance and sediment resuspension. Net sedimentation rates in nearshore areas appear to be relatively low (e.g., ~1 cm/yr). The revised FS HST model suggests that during extreme events, such as the 1996 flood, relatively deep (to two meters) scour occurs only in the sandier, high-energy, in-channel reaches. The remainder of the Study Area stays relatively stable or continues to accumulate sediments.

10.1.2 Overview of Contaminant Distribution

This section provides a brief overview of the overall distribution of ICs in the Study Area sediments. The CSM data presentations that follow in Section 10.2 focus on the distributions of individual ICs separately. As mentioned above and in Section 3.1, the Study Area is generally a depositional reach of the LWR, and sediments entering from upstream accumulate there, particularly from RM 1.9 to 5 and from RM 7 to 10, as well as in any depressions, such as formerly dredged borrow areas that are deeper than the federal navigation channel depth, currently maintained at –40 ft CRD. Point and non-point sources from within the Study Area and upstream of the Study Area have contributed to the sediment contaminant distributions detailed in Section 5 for the 36 nature and extent ICs and are summarized briefly here for the 13 ICs selected for the CSM.

Contaminant concentrations for sediment and other media are presented in Panels 10.2-1 through 10.2-13.[1] Sediment concentrations are grouped into concentration ranges based on the data distributions (see Section 5.1.3) and are graphically presented in Thiessen polygons. Based on examination of the contaminant distribution trends shown in the panels (Panels 10.2-1 through 10.2-13), as well as in the Section 5 nature and extent surface and subsurface sediment chemistry maps (Maps 5.1-1 through 5.1-28), some general patterns emerge among subsets of different ICs that may be indicative of Study Area fate and transport processes, as well as the relative importance of regional (e.g., upstream) versus Study Area sources. These general patterns are discussed below.

Sediment contaminant concentrations are greatest in nearshore areas: As shown in the Section 10 panels and Section 5 maps, for synthetic organic compounds, sediment

[1] The contaminant distribution shown in these panels does not include data obtained after June 2008. Appendix H provides updated maps using data obtained through July 2010.

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concentrations are generally higher in localized nearshore and off-channel areas as compared to sediments in the federal navigation channel, Multnomah Channel, and downstream areas. In contrast, metals generally show a much narrower concentration range throughout the LWR than the organic compounds.

Contaminant concentrations are generally greater in subsurface sediments: For most organic contaminants, concentrations also tend to be higher in subsurface sediments than in surface sediments, but there are chemical-specific and localized exceptions (i.e., subareas where surface concentrations are higher). Of the CSM ICs, the concentrations of total PCBs, total DDx, total PAHs, total chlordanes, aldrin and dieldrin, and TBT are generally higher in subsurface sediments than in surface sediments, indicating that historical inputs were likely greater than current inputs. In contrast, arsenic, copper, chromium, and zinc do not have large concentration ranges and generally show similar levels in surface and subsurface sediments, although again some local variations in these patterns are present.

Regional inputs exhibit uniform concentrations across the area: Contaminants that may be derived predominantly from regional or upstream inputs show widespread surface sediment distributions without distinct, isolated elevated areas. Examples of this are arsenic and chromium (Panels 10.2-9A–B and 10.2-12A–B), which occur at relatively low concentrations throughout the Study Area, and no strong concentration gradients are apparent.

Areas of high concentrations are present throughout the Study Area and generally are associated with known upland sources: A number of ICs show distinct isolated areas of relatively high sediment concentrations offshore of known or likely sources, but these well-defined areas are separated by large areas with relatively low levels lacking obvious concentration gradients. These ICs include total PCBs, TCDD TEQ, BEHP, total chlordanes, copper, zinc, and TBT.

Areas of high concentrations are more common in the lower (downstream) half of the Study Area: Two contaminants, total DDx and total PAHs, exhibit distinct areas of relatively high concentrations at locations in the center and downstream portions of the Study Area. While there are some small-scale, localized exceptions in the upstream portion of the Study Area, concentrations of both of these contaminants are relatively low outside of these distinct mid-Study Area locations. Some areas of elevated concentrations of PAHs and DDx are associated with known or likely upland sources; however, the source(s) associated with elevated levels of these contaminants in other areas, particularly in the upstream and downstream portions of the Study Area are not readily apparent. This pattern is not true for PCBs, which are found in nearshore (outside of the federal navigation channel) and off-channel areas throughout the Study Area that are both more quiescent and proximal to likely or known historical sources.

Concentrations of some metals are related to sediment grain size: A comparison of metals concentrations to the distributions of percent fines in the Study Area shows that

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where sandy sediments (less than 40 percent fines) are predominant in surface sediments, chromium and copper concentrations are relatively low (i.e., above RM 10, between RM 5 and 7, and in the Multnomah Channel; compare Map 3.1-2 with Panels 10.2-12A and 10.2-10A). A similar, but less pronounced, correspondence exists between sandy sediments and zinc concentrations (Panel 10.2-11A).

Sediment contaminant co-occurrence: Several locations within the Study Area have relatively high surface sediment concentrations of more than one IC. These areas and the co-occurring contaminants are as follows:

- **RM 9.7W:** total PCBs, TCDD TEQ, BEHP, zinc
- **RM 8.7–9.3W:** total PCBs, TCDD TEQ, total PAHs, total chlordanes, copper, zinc
- **RM 8.3W:** total PCBs, total PAHs, BEHP, total chlordanes, dieldrin, copper
- **Swan Island Lagoon:** total PCBs, TCDD TEQ, total PAHs, BEHP, total chlordanes, copper, zinc, chromium, TBT
- **RM 6.8–7.5W:** TCDD TEQ, total DDx
- **RM 6.7–6.8E:** total PCBs, TCDD TEQ, copper
- **RM 5.6–5.7E:** TCDD TEQ, total PAHs, total chlordanes, copper, zinc, chromium
- **RM 4.3–4.5E:** total PCBs, TCDD TEQ, total PAHs, total chlordanes, zinc
- **International Slip:** total PCBs, TCDD TEQ, total PAHs, BEHP, total chlordanes, copper, zinc, chromium, TBT.

This degree of contaminant co-occurrence in specific Study Area locations reflects the history of upland site development, including waste and stormwater conveyance systems and industrial and commercial activities, as described in Section 3 and summarized in Section 10.1.3 below.

10.1.3 Site Sources

The following is a summary of information presented in Section 4 on the general nature of historical and current sources and associated pathways to the Study Area.

10.1.3.1 Historical

Historical sources dating back to the early 1900s contributed to the majority of the observed contaminant distributions in sediments within the Study Area. This is reflected in the extent and degree of subsurface sediment contamination as discussed in the previous section. Nearly all the identified chemical pathways have an historical component.

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In the early 1900s, rivers in the United States were generally used as open sewers, which was also true for the Willamette (Carter 2006). Untreated sewage, contaminated stormwater runoff from various land uses, as well as process water from a variety of industries, including slaughterhouses, lumber mills, paper mills, and food processors, was discharged directly into the river, as were pollutants from less conspicuous (non-point) sources, including agricultural fields, oil spills, rubber and oils, and garbage dumps. With the general exception of manufactured gas operations and bulk fuel storage, which began in the early 1900s, most chemical manufacturing and use began in the 1930s.

Commercial and industrial development in Portland Harbor accelerated prior to World War I and again during World War II. In May 1918, Portland contained 4 steel and 17 wooden shipyards; one of these shipyards, the Grant Smith-Porter shipyard at the present-day Mar Com North site, produced 25 wooden ships for the WWI supply fleet (Hopkins 1994). During the WW II, numerous Liberty and Victory ships, minesweepers, T-2 tankers and other vessels were built at military shipyards located in Portland Harbor. A number of these shipyards were also involved in ship repair and ship conversion. Following the war, many of the ship building facilities closed, but a few were repurposed for scrapping the military's surplus and obsolete vessels and for ship repair work.

The years following the war saw an increase in industrial development, which continued to spread throughout the Study Area. While not an exhaustive discussion of historical operations adjacent to the LWR, summaries of the major industrial operations over the past century are provided in Section 3 and shown on Maps 3.2-3 through 3.2-12. These industrial operations and their associated COIs are summarized below:

- **Ship Building, Dismantling, and Repair.** Map 3.2-3 shows the approximate locations of historical shipyards that are observable on aerial photographs taken between 1936 and 1969. Approximate areas of former shipyards include RM 4E, 5.6E, 7E, 7.4E, Swan Island, RM 9W, 10W, and 11E. Ship building continues at a much smaller scale in Portland Harbor today, with most work focused on ship maintenance and repair. VOCs, SVOCs, PAHs, PCBs, TPH, copper, zinc, chromium, lead, mercury, phthalates, and butyltins are identified as common sediment contaminants associated with shipyards in studies by the National Shipbuilding Research Program (NASSCO 1999) and EPA's Office of Compliance (EPA 1997b), as well as EPA's Industrial Stormwater Fact Sheet Series (EPA 2006l).
- **Wood Products and Wood Treating.** Lumber mills and wood treatment facilities operated at various locations within the Study Area historically (Map 3.2-4). M&B, a large wood-treating facility, existed at RM 6.9–7.2E. COIs associated with wood treatment include creosote/diesel oil mixtures, PCP, and a variety of water- and ammonia-based solutions containing arsenic, chromium, copper, and zinc (EPA 2006d). PCP wood treatment products routinely contain dioxin/furans as contaminants and these are an additional COI of wood

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treatment facilities (EPA 2004b). Many other lumber mills and plywood manufacturers were found throughout the Study Area, including Linnton Plywood, St. Johns Lumber (which operated on the present-day Crawford Street and BES WPCL sites), Kingston Lumber, and former mills in Willamette Cove. While most of the byproducts of the lumber and sawmill operations were organic materials, COIs typically associated with sawmills include metals, TPH, and PAHs. In addition to these COIs, plywood manufacturing could include VOCs and SVOCs, as well as possibly pesticides and fungicides (Eaton et al. 1949; U.S. Forest Service 1964; Moore and Loper 1980; Stellman 1998).

- **Chemical Manufacturing and Distribution.** Study Area chemical plants (RM 6.8–7.5W) that manufacture pesticides and herbicides were in place as early as 1941 (Map 3.2-5). COIs typically associated with these operations include pesticides, herbicides, VOCs, dioxins/furans, and metals (EPA 2006f).
- **Metal Recycling, Production and Fabrication.** Metal salvage and recycling facilities operated at RM 4E, 5.8W, 7.3W (Schnitzer-Doane Lake), 8.5W (Calbag/Acme), 8.9W (Gunderson – Former Schnitzer Steel auto dismantling), and 10W (Calbag) in the Study Area, and several scattered locations upriver (Map 3.2-6). COIs commonly found in waste streams from metal recycling facilities include VOCs, TPH, PCBs, phthalates, cyanide, and a variety of metals (EPA 2006i,j). Metal production and fabrication, currently takes place in the Burgard Industrial Park and several sites in the RM 8 to 10.3W reach. COIs associated with metal production and fabrication include metals, PAHs, and TPH (EPA 2006e,h). For high-temperature applications such as die-casting machines, hydraulic oil with PCBs was often used (EPA 2004a). Metal plating also has occurred at a few locations in the Study Area, including Columbia American Plating at RM 9.5W. COIs associated with metal plating activities include VOCs, PAHs, TPH, cyanide, and several metals (EPA 2006h).
- **Manufactured Gas Production.** Manufactured gas production operations took place between 1913 and 1955 at RM 6.2W (Map 3.2-7). The Pintsch Compressing Company Gas Works at RM 7.3W operated between 1890 and the mid-1930s and manufactured compressed gas from crude oil for railroad train lighting. Prior to 1913, gas production also occurred just upstream of the Study Area at the Portland MGP site at RM 12.2E. COIs associated with manufactured gas operations include VOCs, SVOCs, TPH, PAHs, metals, and cyanide (Integral et al. 2007).
- **Electrical Production and Distribution.** Electrical transformers and capacitors are associated with all of the major industries in the harbor. Some of these transformers and capacitors may contain PCBs. Seven current and one historical substation are found in the Study Area. Transformer repair, servicing, and salvaging operations were found on the east bank from RM 11.3 to 11.5 (Tucker Building, Westinghouse, and PacifiCorp Albina Properties), at RM 3.7W (ACF Industries), RM 9.5E (Portable Equipment Salvage), RM 9.5W (GE Decommissioning), and the GE facility at NW 28th Ave (TSCA site). Map 3.2-8

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shows the locations of former servicing and salvage operations. COIs linked with these types of operations include PAHs, TPH, and PCBs (Pfafflin and Ziegler 2006).

- **Bulk Fuel Distribution and Storage and Asphalt Manufacturing.** Bulk fuel facilities have a long history in Portland Harbor. By 1936, most of the facilities currently in place had been established between RM 4 and 8 on the west side of the river (Map 3.2-9). COIs typically associated with bulk fuel storage operations include VOCs, SVOCs, PAHs, TPH, and metals (EPA 2006g).
- **Steel Mills, Smelters, and Foundries.** The harbor hosted several foundries at RM 11.4W (Gender Machine Works), at RM 9.7W (Schmitt Forge), and at RM 2.7E (Consolidated Metco). Several smelters were located at RM 7.2W (Gould), at RM 9W (National Lead/Magnus Smelter), and at RM 11.6W (RiverTec Property). Steel mills are or were located at RM 2.4E (Evraz, aka Oregon Steel Mill) and at RM 8.3W (former Oregon Steel Mill operation at Front Ave LP) (Map 3.2-10). Besides metals, other COIs associated with these types of operations include TPH, PCBs, and PAHs. PCBs were a component of hydraulic fluid for high temperature applications (machining and die casting) where fire resistance was important. PCBs were also a component of heat transfer fluid used in big applications like heat exchangers and recirculating cooling systems (EPA 2004a, 2006h).
- **Commodities Maritime Shipping and Associated Marine Operations.** In addition to the Port of Portland's large presence in Portland Harbor with three deep-water terminals committed to import/export, currently there are or have been several other commodity shipping facilities in the harbor (Map 3.2-11). These include the grain handling operations at CDL Pacific Grain (RM 11.4E) and Centennial Mills (RM 11.3W), edible oils at the former Premier Edible Oils facility (RM 3.6E), scrap metal export at International Terminals (RM 3.7E), cement import and distribution at Glacier NW (RM 11.3E), anhydrous ammonia and solid and granular urea at JR Simplot in the South Rivergate Industrial Park (RM 3E), and alumina, electrode binder pitch, and grain at the former Goldendale Aluminum property (RM 10E). Supporting maritime activities include over-water tug and barge moorage, maintenance and repair facilities, overwater bunkering and lightering, tug-assisted and independent maneuvering of vessels in and around marine facilities, and stevedoring (loading and discharging) product at vessels. Incidental spills into the river from commodities maritime shipping include organic materials, VOCs, PAHs, and TPH (EPA 2006k).
- **Rail Yards.** As shown on Map 3.2-12, the Study Area has hosted several types of rail yard and freight car repair operations. Active facilities are located at approximately RM 9.8 to 11.1E (UPRR Albina Yard), RM 8.6 to 9.5W (PTRR Guilds Lake Yard), and RM 4.8E (UPRR – St. Johns Tank Farm). Historical rail yard operations were located at and around RM 11.6W (BNSF Hoyt Street

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Railyard, and UPRR Union Station operations). Historical rail car maintenance operations were located at RM 3.6 (ACF Industries). At rail yards used for fueling or rail car maintenance, COIs may include VOCs, SVOCs, TPH, PCBs, and metals (DEQ 2011c, EPA 2011). Railcar switching yards (RM 8.1W BNSF Willbridge Switching Yard) are locations where trains are assembled and disassembled, and this moving of railcars typically does not result in releases or produce waste streams.

COIs related to these facilities and operations reach the in-water Portland Harbor media through several migration pathways, including stormwater, industrial wastewater, overland flow, groundwater, bank erosion, and overwater releases. COIs associated with these pathways are typically related to site-specific operations, but in the case of shared stormwater conveyance systems, COIs may be associated with a number of facilities. The outfalls and drainage basins for the historical facilities largely have not been evaluated. Most early discharges of stormwater and sanitary and industrial wastewater went directly to the Willamette River (Bridgewater 2000; see Section 4.3.1). In addition to direct stormwater discharges from facilities, public and private shared conveyance systems drained areas away from the shoreline and, in many instances, included CSO discharges (as described in Section 4). COIs identified for these outfalls could include the entire range of Portland Harbor COCs, depending on the site-specific discharges to the system.

Some contaminated surface soils exposed in the upland areas and along riverbanks can be carried directly to the river as riverbank erosion and in stormwater sheet runoff (i.e., overland transport). The greatest erosional events occur during high flows and floods. As development continued through the 1900s, the bank was armored in many areas. The occurrence and relative importance of riverbank contamination is not well characterized for all parts of the Study Area, but it is a focus of DEQ's Joint Source Control investigations. Contamination in riverbank soils can result from upland activities or from contaminated material used in construction fill activities. In some locations, contaminated dredged material may have been placed in low-lying areas subject to erosion. Section 3.2.2 details how fill was used in many locations in the Study Area to create nearshore uplands. While the quality of this fill material is generally undocumented, the time periods involved and the history of sediment contamination from a range of industrial and maritime sources (including as discussed in Section 4 and Maps 3.2-20a-f) suggest that contaminated sediment could have been included in the fill material. Bank erosion and overland transport of soil to the river were likely more important historically, prior to the development of extensive stormwater conveyance systems and paving of upland areas adjacent to the Study Area.

Migration of contaminants from upland areas to the river via the groundwater pathway is also a historical source of contamination to the river at a limited number of upland sites within the Study Area based on available information. At a subset of these sites, the historical groundwater pathway has contributed significant loads of upland contaminants to sediment and TZW. While some complete historical groundwater

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transport pathways have been mitigated or eliminated through source control actions, others remain complete, as identified in Section 10.1.3.2 below.

Historically, overwater releases were common occurrences for industries on the banks of the Willamette that relied on maritime shipping to get commodities to and from market. Overwater releases are likely important historical contributors to in-water contamination at sites that have long histories of overwater operations (e.g., ship building and repair, dock facilities, fuel facilities) and product transfers. However, as noted in the section below, records of overwater spills over the past 30 years generally do not include documentation of large releases.

While historical releases in the Study Area are generally not quantifiable, the processes and operations responsible for releases and the locations of historical releases are generally well understood, as discussed on a chemical-specific basis in Section 10.2.

Upstream sources also contributed to the historical contamination of the LWR. These sources included sewerage, stormwater runoff, and direct discharge of industrial wastes from upstream cities, towns, and industrial areas; agricultural runoff; and aerial deposition on the water surface and drainage areas within the Willamette Valley.

10.1.3.2 Current

Many of the large historical operations in the harbor have ceased operating over the past 50 years. These former operations include widespread ship building and scrapping operations; large-scale chemical manufacturing; manufactured gas production and wood treatment; and the manufacturing, repair, and storage of PCB-containing electrical equipment. However, some historical operations continue to exist today, including bulk fuel storage, barge building, ship repair, automobile scrapping, recycling, steel manufacturing, cement manufacturing, transformer reconditioning, operation and repair of electrical transformers (including electrical substations), and many smaller industrial operations. Maps 3.2-3 through 3.2-12 show the locations of both current and historical major industrial operations in Portland Harbor.

Stormwater and wastewater discharges are regulated and permitted for many of the sites adjacent to the Study Area. However, sampling for RI-related chemicals in stormwater and catch basins only began in recent years and, for the most part, has only been done for those facilities that have voluntarily conducted a JSCS stormwater source control evaluation. In addition, under the 2003 Intergovernmental Agreement between DEQ and the City, the City is continuing through storm drain sampling to identify sites discharging RI-related chemicals to the Study Area. Significant examples of the City's work under its Portland Harbor Program are the identification of the GE Decommissioning and the Calbag-Nicolai sites as PCB sources to stormwater. Known or likely complete pathways for stormwater have been identified at many sites (see Section 4). As continued sampling is conducted under the JSCS and City programs, additional sites with known problematic stormwater discharges may be identified.

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With the construction of stormwater treatment systems and wastewater treatment systems over the years, overland transport has been largely abated at most sites. A current overland transport pathway has been identified as likely complete at very few sites, although more such sites may continue to be discovered.

With respect to the groundwater pathway, available groundwater information for more than 120 upland sites bordering and/or in close proximity to the river was reviewed during the RI and under JSCS programs. Based on this data review, and on further sampling information collected and evaluated during the RI groundwater pathway assessment, current known complete or likely complete pathways have been identified for 11 sites, 51 sites have insufficient data to make a determination, and 58 sites have been identified as not having a complete pathway. The groundwater pathway assessment conducted during the RI developed detailed groundwater discharge and TZW sampling information at nine high priority sites. Based on these efforts, a current complete groundwater pathway with influence on TZW and sediment chemistry was confirmed at four sites, migration of groundwater was found to have no significant influence on TZW and sediment chemistry at four other sites, and the effect of upland groundwater on TZW and sediment chemistry could not be established at one site (see Appendix C2).

Riverbank erosion from contaminated and unstabilized bank areas may represent an ongoing release mechanism in the Study Area. Currently about 75 percent of the riverbanks within the Study Area are stabilized and armored with various materials, including seawalls, riprap, and engineered and non-engineered soil. Known or likely complete riverbank pathways have been identified at a few sites with unstabilized banks, although more such sites may continue to be discovered. The occurrence and relative importance of riverbank contamination is a focus of DEQ's JSCS investigations.

The activities most commonly associated with current overwater spills in the Study Area are product handling, overwater activities such as refueling, and spills from vessels. Overwater releases are likely important contributors to in-water contamination at sites that have long histories of overwater operations and product transfers. Spill records collected over the past approximately 30 years do not generally record large releases, but there have been some exceptions.

DEQ's JSCS program focuses on the abatement of current and threatened future releases of contaminants to the Study Area. The current status of that program is summarized in Section 4.6.

As with historical sources, current upriver sources also play a role in the contaminant distribution in the LWR. Current upstream loading is discussed in the following section.

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10.1.4 External Loads to the Study Area and Internal Fate and Transport Processes

This section summarizes the information presented in Section 6 of the RI on contaminant mass inputs (i.e., loads from external sources to the Study Area), as well as internal contaminant mass transfer mechanisms within the Study Area. The boundaries of the Study Area include the upstream and downstream river mile designations (RM 11.8 and 1.9), the surface of the river, the riverbank sediment/riparian soil boundary at an elevation of +13 ft NAVD88, and the surface sediment/subsurface sediment boundary at 30 cm bml. Contaminant mass passing through these boundaries into the Study Area are external loads. Section 10.1.4.1 assesses the current and historical external loads to the Study Area; these include upstream loading (via surface water and sediment bedload), stormwater, permitted industrial discharges, upland groundwater (plume transport to the river), atmospheric deposition (to the river surface), direct upland soil and riverbank erosion, groundwater advection through subsurface sediments (partitioning and advection from subsurface to surface sediment), and overwater releases. Many of these external loading terms have been quantified for the CSM ICs and so their relative magnitudes can be compared.

Section 10.1.4.2 describes internal fate and transport or transfer processes that act on contaminants in abiotic and biotic media within the Study Area. Internal transfer mechanisms are those that involve the transport of contaminant mass from one media to another within the Study Area as defined above, but which do not add new contaminant mass to the Study Area. Internal fate and transport mechanisms include sediment resuspension/transport/deposition, solid/aqueous-phase partitioning, abiotic/biotic transformation and degradation, biological uptake and depuration, and groundwater advection through surface sediments (partitioning from surface sediment to surface water). Groundwater advection is the only internal process that is quantified here. Some internal processes may be significant in the transfer of contaminants between abiotic media and biota (e.g., sediment resuspension) and may be quantified as part of the FS fate and transport modeling effort.

10.1.4.1 External Loading Mechanisms

The generally higher contaminant concentrations in subsurface sediments throughout much of the Study Area, particularly in nearshore areas that appear to be long-term depositional/stable sedimentary environments, indicate that historical sources were significant contributors to the contaminant concentrations measured in Portland Harbor sediments during the RI. However, understanding ongoing sources and their associated contaminant loads to the Study Area is important in assessing recontamination potential, identifying the need for source control activities, and evaluating remedial technologies in the FS. Identification and control of current sources is being implemented by DEQ through the JSCS program. The current status of the JSCS program is summarized in Section 4.6.

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Estimated *external* contaminant loads from current sources to the Study Area are summarized in the following subsections.[2] Current contaminant loading to the Study Area includes the following (the type of evaluation conducted in this RI is noted in parentheses):

- Upstream inflows of water and sediment (quantitative to semi-quantitative)
- Stormwater and CSO discharges from upland areas (quantitative)
- Direct permitted discharges (quantitative to semi-quantitative)
- Atmospheric deposition (semi-quantitative)
- Upland groundwater (plume transport to river; quantitative)
- Groundwater advection through sediments (partitioning from deep sediment, advecting to shallow sediments; semi-quantitative)
- Upland soil and riverbank erosion (qualitative)
- Overwater releases from ongoing activities (qualitative).

The detailed evaluation used to estimate loads is provided in Section 6, and the contaminant-specific findings are provided in the chemical-specific CSM presentations in Section 10.2. It should be noted that the empirical loads reported in the RI are based on measurements, e.g., upstream surface water concentrations, collected over the range of flows experienced during the RI sampling period. While every effort was made to capture a full range of seasonal flow levels (i.e., less than 10,000 to 168,000 cfs) as described in Section 5.3, no empirical data from extreme or flood conditions were collected. This adds some uncertainty to the empirical loading estimates. Some climate change models for the Pacific Northwest suggest that high flow events may occur more frequently in the future. No attempt to quantify this uncertainty was conducted in the RI. However, a visual comparison of the net empirical bathymetric change measured from 2002 to 2009 (a period without a major flood) with the HST modeled net bathymetric change results for a major flood scenario presented in Section 3.1.4 indicate that, while the depth of scour in erosion areas is notable greater in the flood simulation, the subareas experiencing erosion or deposition are very similar during both flood and typical flow conditions in the LWR (e.g., compare the middle and lower panels on Map 3.1-12). Finally, the calibrated FS HST model will evaluate fate and transport patterns beyond the range of the observed hydrologic conditions experienced during the RI and will be used in the FS, through sensitivity analyses, to address loading uncertainties as part of the evaluation of remedial alternatives.

The following paragraphs summarize the approach used to estimate the current loads for each term and the general findings. For all loading terms, the target loading units are mass per year to the entire Study Area for a typical water year. Each estimate of current

[2] Internal loads and fate and transport processes, such as the movement of contaminants from surface sediments to surface water, are summarized in Section 10.1.4.2 and detailed in Section 6.2.

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loading has a corresponding historical component that may be significant to the current contamination profile in the Study Area. Historical loading is defined as any loading of ICs to the Study Area that occurred prior to collection of the data sets used to assess the current loading terms. Historical loading is expected to have occurred by all loading mechanisms; however, very limited quantitative data are available to support estimates of these historical terms. For many locations and contaminants, historical loading is likely to have been higher than current conditions due to environmental regulations and controls that have been implemented in the past 30 years or so. As a result, it is likely that a large proportion of the contaminant mass in sediments of the Study Area is due to historical loading. Although historical loading is likely to be a significant contributor to sediment contamination, it is primarily discussed qualitatively.

10.1.4.1.1 Upstream Loading

For most contaminants evaluated, upstream loading is the largest current loading term for the Study Area. While upstream surface water and suspended sediment concentrations are typically lower than those measured in the Study Area, the very large flow volume of the river compared to the flow volumes for the other loading terms results in a relatively large mass load of contaminants compared to other current sources. Figure 10.1-3 shows estimated flow volumes used for the various loading terms. Upstream loads are likely to have been larger historically when fewer environmental laws and regulations and engineering controls were in place for upstream areas.

Upstream loading is defined as the contaminant mass that entered and continues to enter the Study Area at RM 16 and 11 (the upstream surface water program sampling locations) via dissolved and suspended particulate transport mechanisms. Upstream loading includes surface water loading (dissolved and suspended solids fractions) and was assessed quantitatively.[3] Per agreement with EPA (see Section 6.1.1), upstream loading estimates do not include potential bedload inputs (i.e., rolling, sliding, and saltating of sediment grains along the bottom). During scoping of the Fate and Transport Model developed for the FS, quantification of sediment bedloads into the Study Area and associated contaminant transport was deemed unnecessary because the sediment bed of the LWR is mainly cohesive and bedload transport processes are not significant components of upstream loading.

[3] While upstream loading terms are presented here simply as surface water (dissolved and particulate) loads, it is important to recognize that these loads represent the combined input to the Study Area from a variety of loading processes in the upstream watershed. These inputs include upstream point sources, upstream stormwater runoff, upstream atmospheric deposition, upstream in-river sources, etc. Distinguishing these individual contributions to the combined upstream load is beyond the scope of the loading evaluations conducted by LWG.

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The loading analysis results (Section 6) indicate that upstream surface water loading is the largest current loading term, typically by 1 to 3 orders of magnitude, for all of the CSM ICs, except PAHs and TBT.[4]

For the FS, upstream loading will be included in the chemical fate and transport analysis and modeling for MNR and recontamination evaluations. This will include scenarios commensurate with these ongoing source levels.

10.1.4.1.2 Stormwater Loading

Based on the stormwater loading calculations (Section 6.1.2), current stormwater runoff is the second largest quantified annual external loading term to the Study Area for all CSM ICs, with the exceptions of total PAHs (current stormwater runoff is the lowest loading term) and arsenic (stormwater is the fourth highest loading term).[5] It is expected that the overall contribution of stormwater loading to the Study Area may have been more significant historically, prior to implementation of current management practices and stormwater runoff controls. Loading from CSO discharges is also a factor in stormwater loading, however, at a much reduced rate than in the past.

Stormwater is a migration pathway for contaminants in upland areas to reach the river via runoff from the local watershed. Contaminants present in stormwater runoff may be present in the upland watershed as a result of upland soil contamination, atmospheric deposition, and a wide range of anthropogenic activities. Stormwater-related chemicals are transported mostly via conveyance systems and discharged through numerous outfalls along the river shoreline within the Study Area. Overland flow of stormwater to the river also occurs in some relatively limited areas.

As detailed in Section 6.1.2.3, the estimated stormwater annual load is generally highest for the heavy industrial land use category and individual (non-representative) industrial sites as compared to the other land uses, which include light industrial, parks/open space, residential, and major transportation (see Tables 6.1-5a and b). For two of the three bounding ICs measured in the LWG stormwater sampling program, PCBs and DDx, the individual non-representative industrial outfalls contribute over half of the total stormwater load to the Study Area; however, there is likely some uncertainty associated with the DDx annual load (see Section 6.1.2). Among the representative land use types, the heavy industrial use category is responsible for the dominant loads. For PAHs, the heavy industrial land use category is also the dominant load, but the light industrial and the major transportation land use category also are important (i.e., about 20 percent of the total estimated composite water load, combined). Individual non-representative industrial outfalls contribute about half of the total PAH stormwater load

[4] For PAHs, advective loading, groundwater plume loading, and atmospheric deposition to the river surface are the largest estimated external loading terms. This reflects the range of chemical behavior within the PAH chemical group, as well as the effect of significant localized source terms. For TBT, advective loading is the largest estimated loading term, likely reflecting the largely historical inputs and high subsurface sediment concentrations of this chemical in some areas.

[5] PCDD/Fs and TBT were not sampled in stormwater.

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to the Study Area. PCDD/Fs were not measured in stormwater. Among the other CSM ICs, the heavy industrial category also dominates loading from representative land use types, followed by light industrial, with minor contributions from all other categories for the metals.

For total PCBs, the annual estimated load via stormwater is about one half of the upstream surface water load. For the remaining CSM ICs, estimated loads from stormwater to the Study Area are typically 1 to 2 orders of magnitude lower than upstream surface water loads.

For the FS, stormwater loading will be included in the contaminant fate and transport analysis and modeling for MNR and recontamination evaluations. This will include scenarios commensurate with these ongoing source levels, as well as scenarios assuming certain decreases in source levels due to the presumed effects of ongoing upland source control efforts.

10.1.4.1.3 Direct Permitted Non-Stormwater Discharges

NPDES-permitted, non-stormwater discharges to the Study Area include private and municipal outfalls. Current annual load estimates for this source term were generated for the 14 NPDES wastewater permitted discharges in the Study Area listed as either Individual or GEN 15A Permits. Semi-quantitative annual loads were estimated from discharge and flow data in Discharge Monitoring Reports for the 10 dischargers with available information for CSM ICs. Of the CSM ICs, analytical data for only total PAHs, arsenic, chromium, copper, and zinc were generated as part of the permits. The current direct-permitted discharge loading analysis is provided in Section 6.1.3, and results are discussed on a chemical-specific basis in Section 10.2. The City's CBWTP NPDES permit also permits discharges from CSO and SSOs into the Study Area, as discussed in Section 4.4.1.4. However, those discharges are not monitored for CSM ICs.

Overall, it is expected that loading via permitted non-stormwater discharges, as defined and assessed here, is not currently a major source of CSM ICs to the Study Area since it represents regulated and monitored loads. This is supported by the chemicals for which there are data (PAHs and metals) where the annual estimated loads via permitted discharges are 2 to several orders of magnitude smaller than upstream surface water loads.

Historical loading to the Study Area from industrial discharges, however, was likely more significant prior to adoption and implementation of laws and regulations for discharge permits and controls.

For the FS, direct discharge loading will be included in contaminant fate and transport analysis and modeling for MNR and recontamination evaluations because it appears likely this is overall a minor source, and will not greatly impact results in any direction.

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10.1.4.1.4 Atmospheric Deposition Loading

Atmospheric deposition of chemicals refers to both wet deposition (via precipitation) and dry deposition (not via precipitation) of chemicals present in the atmosphere directly to the Study Area river surface. Chemicals are present in the atmosphere from local and global sources as a result of emissions from stationary sources (e.g., industrial smokestacks), mobile sources (e.g., vehicle emissions), and non-point sources (e.g., fugitive dust). For a riverine system such as the LWR, with small surface water areas relative to the contributing watershed, atmospheric deposition to the watershed plays a greater role. Atmospheric deposition in the Study Area watershed is captured by the stormwater loading assessment.

Current atmospheric deposition to the Study Area (the sum of wet and dry deposition) was estimated semi-quantitatively using a literature-based deposition velocity and, depending on the chemical, local or non-local air and precipitation chemical concentrations. For most CSM ICs, local atmospheric concentration data and/or precipitation concentration data were limited or could not be found; therefore, these loading term estimates are considered more uncertain than semi-quantitative loading estimates for other loading terms. Total atmospheric deposition could not be estimated for BEHP or TBT due to lack of applicable local or literature values. The atmospheric deposition loading analysis is provided in Section 6.1.4, and results are discussed on a chemical-specific basis in Section 10.2.

For total PCBs, TCDD TEQ, total PAHs, and total DDx, annual estimated loads via atmospheric deposition are 1 to 2 orders of magnitude smaller than upstream surface water loads. For the remaining CSM ICs, estimated loads from atmospheric deposition to the Study Area are typically 2 to 4 orders of magnitude lower than upstream surface water loads.

It is likely that overall atmospheric loading to the Study Area for PCBs and DDx pesticides was more significant historically, prior to widespread adoption of controls by many countries on chemical production and usage. For these contaminants, the atmospheric concentrations of these now-banned and/or controlled substances have decreased relative to historical levels; however, they are still present in the atmosphere.

Because of the uncertainties associated with such estimates, atmospheric deposition will not be modeled in the FS chemical and fate transport analyses.

10.1.4.1.5 Upland Groundwater Plume Loading

Upland groundwater plume loading is defined as the current contaminant mass entering the Study Area transition zone/surface water via upland groundwater plumes flowing toward the river. Empirical seepage rate and TZW concentration data from the nine GWPA study sites were applied to generate quantitative annual loads.[6] Because

[6] It is acknowledged that these Study Area loading estimates are based on empirical information from only nine study sites; however, the site selection process for the study was designed to identify all sites with a reasonable likelihood of exhibiting a complete transport pathway for upland groundwater plumes to the Study Area. It is

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contaminant concentrations in TZW may include contributions from sources other than just upland plumes (e.g., desorption from contaminated sediment), this approach for calculating upland plume loads may result in overestimates. This possibility is recognized and accepted in this analysis to allow for approximation of the loading potentially associated with upland plumes and comparison with other estimated loads. Of the 13 CSM ICs, upland groundwater plume loading was assessed for only total DDx, total PAHs, arsenic, copper, and zinc. The other CSM ICs were not identified as having a known or likely complete pathway from upland groundwater to the river; therefore, they were not sampled in transition zone water. The upland groundwater plume loading analysis is provided in Section 6.1.5, and results are discussed on a chemical-specific basis in Section 10.2.

Of the five CSM ICs sampled in groundwater, upland groundwater plume loading from the nine TZW sites appears to significantly contribute to the overall Study Area external loading for only total PAHs. On a localized basis, however, this loading term is relevant compared to other external loading terms for total DDx, total PAHs, arsenic, chromium, copper, and zinc (discussed further in Section 10.2). For total PAHs, annual estimated loads via upland plumes are approximately one-half of the upstream surface water load. For the remaining CSM ICs for which loads were estimated, the loads from upland plumes to the Study Area are typically 2 to 3 orders of magnitude lower than upstream surface water loads.

It is possible that groundwater plume loading to the transition zone was more significant historically. However, the time delay in transport of chemicals from upland groundwater to the river can be significant, making it difficult to predict or generalize about the duration of migration or the timing of the peak concentrations in the transition zone.

For the FS, groundwater plume loading will be considered in the contaminant fate and transport analysis and modeling for MNR and recontamination evaluations.

10.1.4.1.6 Groundwater Advection through Sediment Loading

Subsurface advective loading refers to the migration of chemicals associated with subsurface sediments to shallow surface sediments via desorption and groundwater advection. This term was assessed semi-quantitatively from the surface sediment concentration data set by applying an assumption of equilibrium partitioning and a uniform groundwater flux. The range of values presented for sediment advective loading demonstrates the uncertainty associated with these estimates based on the large range of equilibrium partitioning constants available in the literature. The groundwater advection loading analysis is provided in Section 6.1.6, and results are discussed on a chemical-specific basis in Section 10.2.

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expected that these sites capture the large majority of the load to the Study Area based on the upland source control program information established by DEQ. There may be additional sites that lack upland groundwater data but have complete groundwater pathways; however, there is no way to identify or assess these sites at this time. Detailed discussion of the site identification process is presented Appendix C2.

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Except for total PAHs and TBT, the results indicate that groundwater advection through surface sediment is not a significant loading term compared to other loading terms for the CSM ICs (e.g., one to several orders of magnitude lower than the upstream surface water loads). Advective loading of total PAHs is comparable in magnitude to upstream surface water and upland groundwater plume loading terms, and is dominated by the LPAH fraction. For TBT, loading was estimated for only the upstream surface water and sediment advective loading; the advective loading estimate was slightly higher but comparable to the upstream surface water loading estimate.

The relative historical advective loading from subsurface sediments to surface sediments is uncertain; however, the patterns in sediment concentration with depth provide some insight. For PCBs, DDx, and PAHs, subsurface sediment concentrations exceed surface sediment concentrations in most of the Study Area, suggesting that these compounds were released in greater amounts in the past and subsequently covered with lower concentration sediments. Higher shallow subsurface sediment concentrations in the past would result in greater subsurface advective loading to past surface sediment horizons.

For the FS, groundwater advection through sediment loading will be considered in the chemical fate and transport analysis and modeling for MNR and recontamination evaluations.

10.1.4.1.7 Upland Soil and Riverbank Erosion Loading

Erosion of bank soils is a potential source of chemicals to the Study Area. Although it is estimated that approximately 15 to 25 percent of the banks within the Study Area are potentially vulnerable to erosion (i.e., they are not stabilized or armored), it is not possible to estimate erosion rates or a range of rates that might apply to these areas given the wide range of conditions present. Further, bank soil chemistry data are extremely limited and not suitable for extrapolation to the Study Area because an erodible bank soil may not be a source of CSM ICs. Consequently, current bank soil loading estimates could not be developed and could not be assessed relative to other loading terms.

There is limited information regarding historical releases of bank soils; therefore, the overall impact of historical bank erosion is unknown. Bank conditions and available data sets are presented and discussed in Section 6.1.7.

Because bank erosion is a location-specific condition dependent on both the erodibility of and contaminant concentrations in the bank area, the potential role of bank erosion below mean high water level may need to be evaluated as a part of the remedial design process for each SMA. DEQ is actively addressing cleanup of many upland sites adjacent to the river, and erodible banks above mean high water level are being evaluated under the DEQ's Joint Source Control program.

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For the purposes of the FS, it will be assumed that potential bank erosion sources above mean high water level will be controlled before remedial action proceeds.

10.1.4.1.8 Overwater Releases

The activities most commonly associated with current spills in the Study Area are product handling, overwater activities such as bunkering (overwater fueling), lightering and vessel repair and maintenance, and vessel spills and leaks. All sites with active overwater facilities have the potential for releases to the river. Spill records collected over the past approximately 30 years do not generally include records of large releases, but there have been some exceptions.

Contaminant releases from current and/or historical overwater activities (e.g., sandblasting, paint removal and painting, material transfer, bunkering, maintenance, repair, and operations at riverside docks, wharfs, or piers), discharges from vessels (e.g., gray, bilge, or ballast water), fuel releases, and spills are not considered quantifiable and were not specifically assessed in the loading analysis. However, historical releases of this nature are important in the overall CSM because these releases are believed to be a potentially significant source of the existing contamination in some locations in the Study Area, especially in the deeper sediments. While improved BMPs are likely to have reduced the occurrence of overwater releases significantly, it is acknowledged that current and future releases could occur.

For the FS, overwater release loading will be assumed to be zero for the contaminant fate and transport analysis and modeling for MNR and recontamination evaluations because these are sporadic localized events that cannot be predicted in any consistent way.

10.1.4.1.9 Surface Water and TZW Quality in the Study Area

The spatial and temporally variable loading processes discussed above, as well as internal processes, such as sediment resuspension described below in Section 10.1.4.2, determine the contaminant levels in Study Area surface water. A time-integrated "snapshot" of surface water quality is provided in Appendix D3.3. In this appendix, all RI Study Area surface water data is screened against a suite of human health-based screening criteria values used in the BHHRA, e.g., Safe Drinking Water Act Maximum Contaminant Levels (EPA 2009b).

Map D3.3-1 summarizes the results of that screen for all analytes. Analyte groups (e.g., metals, PCBs) are color coded in Map D3.3-1 for presentation purposes. Many contaminants at multiple locations exceed one or more of these criteria throughout the Study Area. This is not unexpected, given the low threshold values of the selected screening criteria. For the subset of CSM ICs some notable general patterns were observed.

The same group of CSM ICs (i.e., arsenic, PCBs, dieldrin, DDx, and TCDD TEQ) show exceedances at both RM 11 and at RM 2, the upstream and downstream ends of the

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Study Area. In terms of contaminant profiles, this suggests that water quality entering the Study Area is qualitatively similar to water leaving the Site. A second general, and contrasting, pattern is that the number of individual PAHs exceeding screening criteria is much greater from RM 3 to 8 (and in Swan Island Lagoon) than upstream or downstream of this reach. This appears to reflect the concentration of known and potential PAHs sources in this portion of the river.[7]

Maps D3.3-2a–h in Appendix D3.3 show the list of chemicals measured in RI TZW samples that exceed the human health-based screening criteria used in the BHHRA. Each TZW location had a focused list of target analytes based on the COIs in the upland groundwater at each site. In general, the chemicals which exceed the screening values are consistent with the known and potential upland sources (e.g., DDx exceeds screening values at Arkema and Rhone Poulenc; PAHs exceed screening values at Gasco, Arco, Exxon/Mobil, Kinder Morgan, Siltronic, and Willbridge Terminal). Arsenic is present at concentrations exceeding screening levels in TZW off of every upland area location investigated. Further, arsenic exceeds screening levels at most of the individual TZW sampling locations. This, combined with the ubiquity of arsenic exceedances in surface water (Map D3.3-1), suggests that arsenic levels in TZW and surface water are likely naturally occurring and not due to upland sources.

10.1.4.2 Internal Fate and Transport Processes

Once a chemical enters the Study Area it may be transported or transformed by physical, chemical, and biological processes that occur within the Study Area. Identifying these fate and transport processes in the CSM is important because it provides a framework for assessing critical issues such as the following:

- Significance of current external loads in regards to recontamination
- Erosion/resuspension and movement of contaminated sediments within the site
- Transport of contaminants from buried deep sediments to biologically active sediments and pore water
- Partitioning of contaminants between sediments and surface water and the impact of contaminants in one media on the quality of another media
- Transformation by biotic and abiotic processes to contaminants that may be more or less toxic
- Bioavailability and uptake of contaminants into biota.

A detailed description of these processes is beyond the level of detail of this CSM and is presented in Section 6. A brief description of important site elements and how they may affect contaminant distribution and site risks is presented below.

[7] The Siltronic surface water sampling effort, sample labels beginning with GP on the inset in Map D3.3-1, were not analyzed for a full suite for SVOCs (e.g., naphthalene was the only PAH measured). This explains the limited list of contaminations which exceed criteria in that RM 6.5 area.

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10.1.4.2.1 Solid/Aqueous Partitioning

The transport, degradability, and bioavailability of a contaminant often relates to its tendency to associate with particulate material within the system. Many of the CSM ICs are hydrophobic organic compounds, which tend to partition preferentially to the dissolved and particulate organic matter associated with the solid and aqueous phases of surface water, sediments, and pore water. Because the particulate organic matter within the solid phase of sediments represents the largest available pool of organic carbon in the Study Area, contaminated sediments represent the largest repository (by mass) of CSM ICs of all environmental matrices addressed by this RI.

10.1.4.2.2 Surface Water/Sediment Physical Transport

The flow of river water is the primary mechanism for transport of dissolved and particle-bound chemicals. Lateral and vertical movement of chemicals in surface water occurs primarily as a result of turbulent (eddy) dispersion (mechanical mixing). Higher flow velocities typically cause greater mixing and increased transport of suspended and bedload sediments. As described in Section 3, with the exception of the channel environment upstream of RM 10 and between RM 5 and 7, the Study Area appears to be a depositional or relatively stable sedimentary environment.

Relevant processes that influence sediment transport include deposition, erosion/resuspension, mixed-layer turbation, long-term burial, and ingestion/uptake by biota. The relative significance of these transport and fate mechanisms varies by CSM IC, depending on source locations and the IC's other physical/chemical properties. A potentially important mass transfer mechanism is surface sediment resuspension and movement of contaminants from bedded sediment to the water column with a resultant increase in mobility and bioavailability. The FS Fate and Transport Model includes the following processes—deposition/burial, erosion/resuspension due to natural forces, surface sediment mixing by river currents and/or bioturbation, and uptake to biota (via the food web model linkage). Erosion/resuspension due to prop wash and wave action will also be evaluated in the FS. This model and evaluation will be used to assess the relative importance of these internal transfer terms across the Study Area under a range of natural hydrologic conditions.

10.1.4.2.3 Abiotic and Biotic Transformation/Degradation

A variety of abiotic and biotic (microbially mediated) degradation processes are relevant for transformation and degradation of CSM ICs in the Study Area. The relevant processes vary by chemical and location in the Study Area. Relevant abiotic degradation/transformation/loss mechanisms include abiotic oxidation/reduction, hydrolysis, dehalogenation, volatilization (primarily from dissolved phase in surface water), and photolysis (primarily in upper levels of surface water). Microbially mediated degradation (biodegradation) involves the metabolic oxidation or reduction of organic compounds and is carried out predominantly by bacteria in aqueous environments, but yeasts and fungi may also contribute to biodegradation. Biodegradation can proceed to full mineralization of the compound, with end products

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of carbon dioxide and water, or an intermediate compound may be formed that is not easily biodegraded further.

Transformation and degradation rates were assessed qualitatively for the RI. For the FS, a literature review has been completed to find appropriate biodegradation rate constants for use in the FS fate and transport model; these were compiled in cooperation with and were agreed to by EPA. The FS fate and transport modeling will incorporate these biodegradation rate estimates.

10.1.4.2.4 Uptake into Biota

A number of processes govern how organisms living in the Study Area are exposed to contaminants and how contaminants are transformed, excreted, or stored in tissue. Organisms living in the Study Area may take up (bioaccumulate) contaminants through physical (e.g., diffusion), chemical, and biological processes, including transfer of water-borne contaminants across gill structures or other tissues, consumption of prey, or ingestion of sediment. Organisms can modify the contaminant burden in their tissues through growth, reproduction, excretion, metabolic transformation, or sequestration. Some contaminants are transferred among organisms through trophic interactions. For some contaminants, tissue concentrations may increase (biomagnify) at progressively higher trophic levels in the food chain.

The relative contributions of bedded sediment versus surface water contamination to tissue contamination levels in the Study Area is addressed by the FS fate and transport modeling effort. Bioaccumulation modeling is part of that effort, and two kinds of bioaccumulation models are being used to predict contaminant uptake by invertebrates and fishes (Windward 2009b). A mechanistic model (Arnot and Gobas 2004) that describes the bioaccumulation of hydrophobic organic chemicals is being used to evaluate bioaccumulation of selected COIs (i.e., total PCBs, TCDD TEQ, PCB TEQ, and pesticides including DDx) via water, sediments, and prey. Bioaccumulation of other substances, such as PAHs, is evaluated using a statistical approach based on BSAFs or BSARs. Under current conditions, the bioaccumulation model preliminarily determined that sediments are an important source of benthic invertebrate and fish tissue concentrations for the bioaccumulative compounds.

The bioaccumulation modeling and the overall FS fate and transport modeling effort is ongoing, and quantitative statements about the relative contribution of sediment versus other sources to tissue residues cannot yet be made.

10.1.5 Relevant Site Receptors and Pathways

The following subsections briefly summarize the major findings of the BHHRA (Section 8 and Appendix F) and BERA (Section 9 and Appendix G) to the extent necessary for purposes of the CSM, including summaries of human use and ecology of the LWR. Summaries of risk and identified receptors for each IC are provided in Section 10.2.

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10.1.5.1 Site Use and Human Health Risk Assessment Findings

Information on human uses of the river, its shorelines, and resources, as detailed in Section 3.2.5, was used to determine potential receptor populations for the BHHRA. Figure 8.2-1, the BHHRA CSM, lists the human receptor populations.

People interact with the river in a number of ways. Portland Harbor is a major industrial water corridor and working harbor, and the majority of the Study Area waterfront is currently zoned for industrial land use (City of Portland 2006b). Worker activities that may include contact with sediments and surface water at industrial and commercial facilities in the Study Area are somewhat limited due to the sparse beach areas and high docks associated with most of the industrial and port facilities. In the Study Area, some commercial diving occurs, although the practice of using dry suits would clearly limit exposure via sediment ingestion or dermal contact.

The Study Area also contains some natural areas and provides recreational opportunities, both on the water and along the riverbanks, including boat ramps, beaches, and waterfront parks. Recreational/non-commercial fishing is conducted throughout the LWR basin and in the Study Area, both by boaters and from shore. Non-tribal and Native American anglers may catch and eat fish from the LWR. For Native American anglers, the Willamette River provides a ceremonial and subsistence fishery for Pacific lamprey and spring Chinook salmon. The extent to which commercial fishing occurs within the Study Area is not known, but it is thought to be negligible.

In theory, shellfish consumption could occur throughout the Study Area wherever shellfish are found. However, it is not known to what extent shellfish consumption occurs by recreational fishers or transients. The Linnton Community Center project (Wagner 2004, pers. comm.) stated that some transients reported eating clams and crayfish; however, many of the individuals indicated that they were in the area temporarily, move from location to location frequently, or have variable diets based on what is easily available. The ODHS occasionally receives calls from citizens who are interested in harvesting crayfish from local waters and are interested in fish advisory information. According to a member of the Oregon Bass and Panfish club, crayfish traps are placed in the Portland Harbor Superfund Site boundaries and collected for bait and possibly consumption (ATSDR 2006). Even if collection does occur within the Study Area, it is not known whether those crayfish are consumed by humans or used as bait. The shellfish consumption pathway was evaluated at the direction of EPA.

Based on this site use information, the following potentially complete exposure pathways were quantitatively evaluated in the BHHRA:

- Direct exposure to (i.e., ingestion of and dermal contact with) beach sediment
- Direct exposure to in-water sediment
- Direct exposure to surface water

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- Direct exposure to groundwater seeps
- Ingestion of fish
- Ingestion of shellfish.
- Infant consumption of human milk.

A summary of the estimated cancer risks and non-cancer hazards (expressed as an HI) for each of these potential exposure pathways is provided below. The risks to infants through consumption of human milk are presented with the mother's potential exposure pathway. While some of these exposure pathways discussed below result in an estimated cancer risk greater than 1 x 10^{-6}, the exposure scenarios with estimated cancer risks exceeding 1 x 10^{-4} or an estimated HI greater than 1 are fish and shellfish ingestion throughout the Study Area, exposure to untreated surface water as a hypothetical drinking water source at two exposure areas, and direct exposure to in-water sediment for two half-river-mile segments.

10.1.5.1.1 Direct Exposure to Beach Sediment

None of the beach sediment exposure scenarios resulted in cancer risks above the EPA target range (i.e., greater than 10^{-4}) or HIs greater than 1. For CT assumptions, dockside worker, child recreational beach user, combined adult/child recreational beach user, and tribal fisher scenarios resulted in cancer risks above 1 x 10^{-6}. The RME scenarios for exposure to beach sediment resulting in cumulative cancer risks above 1 x 10^{-6} include: dockside worker, adult and child recreational beach user, tribal fisher, low frequency fisher, and high frequency fisher.

10.1.5.1.2 Direct Exposure to In-water Sediment

None of the in-water sediment exposure scenarios based on CT assumptions resulted in cancer risks above the EPA target range (i.e., greater than 10^{-4}) or HIs greater than 1; the tribal fisher and breastfeeding infant CT scenarios had cancer risks above 1 x 10^{-6}. For the RME scenarios, cumulative cancer risks were greater than 1 x 10^{-6} but were below 1 x 10^{-4}, with the exception of cancer risks above 1 x 10^{-4} for in-water sediment exposures to a breastfeeding infant (at RM 7W) and to a tribal fisher (at RM 6W and 7W). RM 7W also presents a cumulative HI greater than 1 for the tribal fisher, high frequency fisher, and breastfeeding infant RME scenarios.

10.1.5.1.3 Direct Exposure to Surface Water

None of the surface water exposure scenarios (i.e., transients, divers, and adult and child recreational users) resulted in cancer risks above the EPA target range (i.e., greater than 10^{-4}) or HIs greater than 1. The only scenarios resulting in cumulative cancer risks greater than 1 x 10^{-6} were the diver in wet suit and the diver in dry suit RME scenarios at RM 6W.

For hypothetical exposure to untreated surface water used as a domestic water source by future residents, both the CT and RME scenarios resulted in cancer risks above 1 x 10^{-4} at RM 6.1. None of the other locations resulted in cancer risks above the EPA target

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range (i.e., greater than 10^{-4}). The child RME scenario for hypothetical exposure to untreated surface water as a domestic water source was the only scenario with a HI that exceeded 1, which occurred at RM 8.5.

10.1.5.1.4 Direct Exposure to Groundwater Seep

Risks from direct contact with groundwater seeps were evaluated for exposure by a transient for one exposure point. The transient exposure scenario did not result in cancer risks greater than 1×10^{-6} or HIs greater than 1.

10.1.5.1.5 Ingestion of Fish

The exposure scenarios evaluated for ingestion of fish in the BHHRA resulted in ranges of estimated cancer risks and HIs. Many of the exposure scenarios resulted in cancer risks greater than 10^{-4} and HIs greater than 1. The highest cancer risks and HIs were associated with the assumptions of the highest fish ingestion rate used in the BHHRA (142 g/day) and a diet consisting of only whole-body carp.

10.1.5.1.6 Ingestion of Shellfish

The exposure scenarios evaluated for ingestion of shellfish in the BHHRA resulted in ranges of estimated cancer risks and HIs. Some of the exposure scenarios resulted in cancer risks greater than 10^{-4} and HIs greater than 1. The highest cancer risks and HIs were associated with the assumptions of the highest shellfish ingestion rate used in the BHHRA (18 g/day) and a diet consisting of only non-depurated clams. Although the harvest and possession of Asian clams, which is the clam species that was found in the LWR during sampling events, is illegal in the State of Oregon (OAR 635–056–0050), EPA has stated that it has information that harvesting occurs. The extent to which ingestion of shellfish may occur within the Study Area, both currently and in the future, is unknown, as discussed in Section 10.1.5.1.

10.1.5.1.7 BHHRA Summary

The primary exposure pathway accounting for the majority of risk for human health in Portland Harbor is ingestion of fish. PCBs are the primary contributor to risk for fish consumption, and dioxins are a secondary risk contributor for fish consumption of both whole body and fillet tissue diets. PCBs and dioxins/furans both resulted in cancer risks greater than 1×10^{-4} and HQs greater than 1 for fish ingestion for both localized and Study Area-wide exposures. PCBs and dioxins/furans contribute approximately 98 percent of the cumulative cancer risk for whole-body fish ingestion and approximately 99 percent of the cumulative cancer risk for fillet fish ingestion on a Study Area-wide basis. The contribution of contaminants to the cumulative cancer risks varies on a localized basis. Other ICs that resulted in cancer risks greater than 1×10^{-4} or HQs greater than 1 on a limited spatial scale and/or for a specific exposure scenario included cPAHs, DDx, BEHP, arsenic, and zinc.

At the direction of EPA, the BHHRA was developed to provide a health-protective assessment of risks associated with contaminants present at the Site. To derive risk estimates in the BHHRA, exposure parameters based on health-protective assumptions

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were multiplied together, resulting in compounded uncertainty that may underestimate or overestimate the actual risks that may exist within the Study Area (see Section 6 of Appendix F for an analysis of the uncertainties associated with the BHHRA). The results of the BHHRA should be weighed in a measured and informed fashion in light of the health-protective assumptions. Moreover, the contribution of background sources is an important consideration in risk management decisions. On a regional scale, fish consumption results in risk estimates exceeding cumulative risks of 1×10^{-4} or HI of 1 based on fish tissue data collected from the Willamette and Columbia rivers outside of the Study Area. However, concentrations are higher within the Study Area than in the regional tissue. Regional efforts are underway to reduce fish tissue concentrations. Sources contributing to regional tissue concentrations are unknown.

10.1.5.2 Site Ecology and Baseline Ecological Risk Assessment Findings

Details on the Study Area's ecology, provided in the BERA (Appendix G) and Section 3.1.6, are summarized here, followed by a summary of the findings of the BERA. The BERA is summarized in more detail in Section 9. The majority of the Study Area is industrialized, with modified shoreline and nearshore areas (e.g., wharfs, piers extending out toward the channel, bulkheads, and riprap-armored banks). The federal navigation channel, authorized to –43 ft and currently maintained to –40 ft CRD, has little habitat diversity. However, some segments of the Study Area are more complex, with small embayments, shallow water areas, gently sloped beaches, localized small wood accumulations, and less shoreline development, providing some habitat for a suite of local fauna. By area, the major habitat in the Study Area is associated with the river bottom or water column. Riparian, shallow-water, and vegetated habitats are limited to the nearshore area or shoreline, and are much less extensive. Although limited in area, these habitats provide important ecological services, such as refuge for juvenile fish species, importation of organic material and nutrients from terrestrial and shallow-water vegetation, breeding for amphibians (in low velocity areas), foraging for migratory waterfowl and shorebirds, and where unarmored, contribution of sediment that affects the character and stability of benthic habitats.

The organisms that use the LWR include invertebrates, fishes, birds, mammals, amphibians, reptiles, and aquatic plants. Each group contributes to the ecological function of the river based on trophic level, abundance, biomass, and interaction with the physical-chemical environment and other species. Riverine invertebrates are predominantly benthic, living on or in such substrates as fine-grained sediments, gravel and cobble, plant roots, or large woody debris. Much of the infaunal invertebrate community within the LWR feeds on and processes organic material imported from upriver areas.

The LWR is an important migration corridor for anadromous fish, such as salmon and lamprey, and provides habitat for numerous resident fish species (more than 40 species have been collected in many historical and recent studies) that represent four feeding guilds: herbivores, invertivores (either from the water column or bottom habitats),

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piscivores, and detritivores. A number of species are omnivores[8] and utilize multiple food types. The type of riverbank present in the Study Area influences the fish species occurrence and use of a given area.

Limited suitable habitat for amphibians and reptiles is present in the LWR. Amphibians prefer undisturbed, shallow-water areas with adjoining ephemeral wetlands and emergent vegetation. Reptiles prefer shallow, quiescent aquatic areas and wet vegetated terrestrial habitats. Current conditions in the LWR prevent the widespread development of dense, submerged, floating-leaved, and emergent aquatic plant communities along the riverbanks because of high turbidity, few quiescent areas, and the presence of riprap and other bank modifications.

For semi-aquatic mammals that depend on shoreline plant cover, habitat in the Study Area is limited because of past human modification of riparian habitats. The upland environment near the LWR is primarily urban, with fragmented areas of riparian forest, wetlands, and associated upland forests. Historical development and filling of channels and wetlands has left only small strips or isolated pockets of riparian wildlife habitat.

Mink and river otter, both semi-aquatic species, were evaluated in the BERA. The Study Area offers at least marginally suitable habitat, and both species have been collected nearby (Elliott et al. 1999; Henny et al. 1996). For birds, the fragmentation of habitat may not be as critical as for mammals. Numerous aquatic and shorebird species, such as cormorants and spotted sandpipers, use the habitats, where available, in the Study Area.

Based on the ecology of the LWR, multiple complete exposure pathways were evaluated for risk to various ecological receptor groups in the BERA (Section 9). The exposure pathways evaluated in the BERA were identified based on behaviors and characteristics of selected ecological receptors, or were required by EPA for evaluation in the BERA. The following complete and significant exposure pathways were quantitatively evaluated in the BERA using multiple lines of evidence:

- **Benthic invertebrates** – Direct contact with sediment and surface water, ingestion of biota and sediment, and direct contact with shallow TZW
- **Fish** – Direct contact with surface water, direct contact with sediment (for benthic fish receptors), ingestion of biota, incidental ingestion of sediment, and direct contact with shallow TZW (for benthic fish receptors)
- **Birds and mammals** – Ingestion of biota and incidental ingestion of sediment
- **Amphibians and aquatic plants** – Direct contact with surface water and shallow TZW.

[8] Omnivores have often been included with the herbivore guild since they may feed on plants or benthic algae and diatoms, but may also be considered a separate guild.

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In total, 89 contaminants (as individual chemicals, intermediate sums, or totals) were identified as posing potentially unacceptable ecological risks. Potentially unacceptable ecological risks were identified from metals, TBT, PAHs and other SVOCs (including chlorinated benzenes, benzyl alcohol, dibenzofurans, carbazole, phenols, and phthalates), PCBs, dioxins and furans, DDx and other pesticides, and VOCs. The risk of ecologically significant adverse effects on populations or communities of ecological receptors in the Study Area is greatest from four groups of chemical mixtures: PAHs, PCBs, dioxins and furans, and total DDx.

The BERA found potentially unacceptable benthic risks in about 7 percent of the Study Area based on all lines of evidence. The COPCs in sediment that pose potentially unacceptable risk to the benthic community or populations are PAHs, PCBs, and DDx compounds. The phenolic compound 4-methylphenol may also be contributing to unacceptable benthic community risk. Widely distributed throughout the Study Area, this contaminant is found in both contaminated and otherwise uncontaminated areas. With a half-life on the order of days, its presence suggests ongoing sources. Whether and to what extent the source is legacy contamination is not known.

PCB HQs ≥ 1 occurred throughout the Study Area for river otter, mink, spotted sandpiper, bald eagle, and osprey, indicating possible population-level effects. The potential for adverse effects in fish due to exposure to total PCBs is low based on limited extent and frequency of TRV exceedances and the likelihood that uncertainties contribute to overestimates of risks. The combined toxicity of dioxins/furans and dioxin-like PCBs expressed as TEQ poses risk of reduced reproductive success in mink, river otter, sandpiper, bald eagle, and osprey. PCBs are responsible for the majority of total TEQ exposure, but the dioxin/furan TEQ also exceeds its TRV in some locations of the Study Area.

Total DDx HQs ≥ 1 occurred for sculpin for certain LOEs and for spotted sandpiper. The weight of evidence indicates the DDx likely poses negligible risk to populations of these receptors because of the low magnitude and limited frequency of exceedances and likelihood that uncertainties contribute to overestimates of risks. For many COIs, the available exposure and effects data preclude a quantitative risk evaluation. These COIs have nonetheless been identified as posing potentially unacceptable risk.

10.2 CHEMICAL-SPECIFIC CONCEPTUAL SITE MODELS

This section presents graphical and detailed narrative CSMs for the 13 ICs (which were derived from the full list of COIs as described in Section 10.0)—total PCBs, TCDD TEQ, total DDx, total PAHs, BEHP, total chlordanes, aldrin/dieldrin, arsenic, copper, zinc, chromium, and TBT. For each of these ICs, the discussion starts with a brief presentation of the IC's general characteristics followed by contaminant distribution, potential sources and pathways, and loading, fate, and transport. The discussion concludes with a summary for each of these ICs.

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In the context of these chemical-specific CSMs, "relatively elevated" (also referred to simply as "elevated" in the sections that follow) concentration ranges are defined for each chemical based on the observed concentration range in sediments across the Study Area, in accordance with the general mapping color-ramp approach used in Section 5 (i.e., samples color-ramped orange and red are referred to as "elevated" in this section).

The basis of each chemical-specific CSM is the graphical presentation. For each CSM chemical, a three-section panel series is provided that presents cross-media contaminant distributions and available source information. Each panel is an oversized map (24 in. by 36 in.) that presents a subset of the contaminant distribution data. All show the entire Study Area, upland site property boundaries, outfall locations, and river mile markers. The presentation on the panels shows concentrations among proximal samples from other abiotic and biotic media, and known or likely complete source pathways. An electronic version of the three-section CSM panels is included in Appendix I.

The first panel (A) presents summary information on the observed concentration of the chemical in surface sediment (Thiessen polygons are used to spatially represent concentrations between data points), sediment traps (mean concentration and maximum quarterly concentration), riparian soil/sediment, surface water (mean and maximum concentrations at points and transects), and TZW from RM 1.9 to 11.8, excluding dredge and cap sample locations.[9] A histogram of the surface sediment data is included in the top left-hand corner of each panel to show the distribution of the data within the color ramps selected for display. For TZW, unfiltered push probe, filtered push probe, and peeper results are displayed. Because surface water data were collected using two different sampling methods, it is important to note which data are mapped for each chemical. Surface water XAD data are presented for total PCBs, TCDD TEQ, total DDx, total PAHs, total chlordanes, aldrin, and dieldrin. Surface water peristaltic pump data are presented for arsenic, chromium, copper, zinc, and TBT. The BEHP data presented are a combination of the XAD and peristaltic pump data.

The second panel (B) presents a summary of the subsurface sediment concentrations and large-scale (>30 cm) erosional/depositional areas predicted for a major flood based on the FS HST model for the LWR (Chen 2011, pers. comm.) from RM 1.9 to 11.8. The Thiessen polygons on Panel B represent the concentration in the sediment interval just below the surface sediment (typically the B interval), and the dot on each polygon is color coded to indicate the maximum concentration of the subsurface samples at that location (i.e., this allows for indication of concentrations at depths below the first subsurface interval). A histogram of the subsurface sediment data is included in the top

[9] The maps of contaminant distribution in Study Area surface (Panel A) and subsurface (Panel B) sediments necessarily involve estimates of concentrations in areas between discrete data points that were not actually sampled. The maps use Thiessen polygons to depict these areal estimates. Each Thiessen polygon contains one sample point. Each polygon edge is a perpendicular bisector between neighboring discrete sample points. The size of each polygon is indicative of the density of sample points, e.g., the surface sediment polygons are generally smaller than subsurface polygons due to greater density of surface samples. Lower or higher actual concentrations may be found in some areas as additional data are gathered in the remedial design process or during remedial action implementation.

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left-hand corner of each panel to show the distribution of the data within the color ramps selected for display. Panel B also includes icons depicting the locations of 10 major types of historical industries that are or were active in the Study Area, as discussed in Section 3.2.1.

In addition to the erosional/depositional information shown on Panel B, Map 10.2-1 shows areas in the Study Area at risk for surface sediment disturbance from incidental anthropogenic activities based on water depth and in-water operations. This includes all areas above the –5 ft NAVD88 contour that are potentially subject to boat wakes, areas in the immediate vicinity of docks and berths, and any additional areas where sediment scour that did not appear to be due to natural forces was evident in the 2002–2009 bathymetric time-series data set.[10] This map does not include an analysis of those activities that are specifically intended to move sediments (i.e., shoreline/structure construction, maintenance dredging, or remedial capping/dredging).

The third panel (C) presents whole-body concentrations in field-collected smallmouth bass, clams, crayfish, and sculpin. The smallmouth bass data are presented with a symbol for the location of each fish; the color of the symbol corresponds to the concentration of the composite sample. Concentrations for the individual fish for all forms of smallmouth bass tissue (i.e., whole-body, fillet, fillet without skin, stomach contents, etc.) are shown on Map 5.5-12a–e. A detailed view of the composite groupings can be found on Map 2.2-10a–d.

For each upland site that has undergone sufficient investigation to identify known or likely complete pathways, a box is shown on the panels. Within each box, the applicable pathways are listed for the site, noting whether they are complete or likely complete. Information used to populate the boxes is based on Table 4.2-2 (source table). The panels show only known or likely complete pathways for a given chemical and do not include pathways when there were insufficient data to make a determination or when a complete pathway was determined to be not present (see Tables 10.2-1 and 10.2-3 through 10.2-14). If a particular site is located within a shared conveyance basin, the outfall number and location are also included in the panel. Sites with only partial drainage to a shared outfall are denoted with an asterisk. Additionally, all outfalls, whether associated with specific upland sites or not, are shown, including private and public stormwater, process water outfalls, and City CSOs.

As noted in Section 4.2, the source information presented here is highly dependent upon whether a site is involved in DEQ's cleanup program and the degree of investigation and data generation. As shown on Table 4.2-1, several sites adjacent to or near the Study Area are not in the cleanup program, and it is likely that many sites, particularly those that are the location of historical facilities that operated outside the

[10] Map 10.2-1 is a qualitative presentation of areas where there is a reason to believe that anthropogenic disturbance risk may be relatively higher than other areas. A quantitative evaluation of prop wash and wave wake generated disturbance of bottom sediments in the Study Area is being conducted as part of the FS contaminant fate and transport evaluation.

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boundaries of current sites, are not fully addressed in DEQ files. As a result, this section does not represent a complete inventory of sites and operations that contribute or have contributed to contamination in Portland Harbor. These limitations on source information primarily affect historical sources. The understanding of current sources may also be affected by these limitations; however, available information appears to be adequate for the purposes of the FS.

Along with the panels, three figures are provided for each CSM chemical to portray loading, fate, and transport processes under current conditions in the Study Area[11] in a typical year. The first figure consists of a pair of box and whisker plots; the first (1a) compares the range of the estimated external and internal annual loads to the Study Area for each of the loading terms quantified for a given CSM chemical, while the second (1b), compares the concentrations of the chemical in surface sediment, sediment traps, and suspended solids in surface water for the entire Study Area. The second figure is a box-and-arrow diagram[12] depicting relevant loading, fate, and transport processes for each CSM chemical at the Study Area scale. The third figure provides a graphical comparison, by river mile, of the quantified external and internal loading terms, including central, upper, and lower estimates, and affords additional resolution of spatial patterns in loading to the Study Area. As documented in Section 6.1 and Appendix E of the RI, external loading (gains and losses) of each CSM chemical to the Study Area was estimated quantitatively or semi-quantitatively for upstream surface water,[13] stormwater, atmospheric deposition to the water surface, and groundwater advection through subsurface sediments. Quantitative to semi-quantitative estimates were also generated for upland groundwater plumes and permitted point-source discharges, for a subset of the CSM chemicals for which these terms may be significant. Unquantified loading terms, including bedload, volatilization, and riverbank erosion, are represented qualitatively on the box-and-arrow diagrams. The only contaminant fate and transport mechanism internal to the Study Area for which quantitative estimates were developed in the RI (see Section 6.2.1) is pore water advection from surface sediment to the overlying surface water column. Other internal fate and transport mechanisms, including sediment erosion, sediment deposition, sediment burial, and degradation are represented qualitatively on the box-and-arrow diagrams.

Finally, it is important to note that these CSM presentations represent an attempt to graphically illustrate spatial patterns between abiotic and biota media and potential sources based on the understanding gained during the RI of in-water contaminant

[11] For the purpose of this analysis, the Study Area includes surface water and bedded sediments in the surface mixed layer (0–40 cm bml).

[12] This diagram does not attempt a mass balance because sufficient data are not available and because of the varying levels of quantification (qualitative to quantitative) of each term.

[13] As discussed in Section 6.1, estimated upstream surface water loads were developed using data from both RM 11 and RM 16. Because of the complex hydrodynamics on the LWR between its confluence with the Columbia River (RM 0) and the entrance to Multnomah Channel at RM 3 (e.g., frequent flow reversals, see Section 3.1.3.3), surface water chemical loads leaving the Study Area at RM 1.9 could not be estimated using the simplified approach described in Section 6.1. The furthest downstream surface water loads for the LWR were estimated at RM 4. Surface water loads exiting the Study Area via Multnomah Channel were also estimated.

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distributions, known and potential upland sources, historical and current loading estimates, and fate and transport processes.

10.2.1 Total PCBs

Panels 10.2-1A–C present the Study Area graphical CSM for PCBs and support the discussions in the following subsections. These panels, per the format described in Section 10.2, present observed PCB distribution in sampled media (biota, surface water, sediment, sediment traps, riparian soils), as well as the current understanding of known and potential historical and current pathways of PCBs to Portland Harbor. The data from recent sediment studies at RM 11E and in the downtown reach are included in the updated RI data set and presented in Appendix H. These data are not included on Panels 10.2-1A–C, but are considered in the narrative discussion below as part of the revised CSM for PCBs.

PCBs are a class of nonpolar, synthetic, halogenated hydrocarbons that were manufactured in the United States between 1929 and 1977 and widely used for a variety of purposes.[14] Current allowed uses include transformers, heat transfer systems, natural gas pipelines, existing carbonless copy paper, and electrical switches. Historical PCB uses included dielectric fluids in transformers and capacitors, electrical cables, cutting oils, hydraulic oils, lubricants, heat transfer fluids, plasticizers, flame retardants, additives to pesticides, paints, carbonless copy paper, caulk, adhesives, sealants, in heat transfer systems, electromagnets, and for dust suppression. PCB sources in waste materials include scrap metal recycling, auto salvage, used oil, recycled paper, asphalt roofing materials, building demolition, and in the repair and salvaging of ships, locomotives, heavy equipment, and manufacturing equipment. Although PCBs are ubiquitous in the environment, commercial PCB production in the United States ended in 1977. The Toxic Substances Control Act, which became law in 1976, bans the manufacture of PCBs, controls the phase-out of their use, and regulates safe disposal.

PCBs do not readily degrade in the environment and are lipophilic, persistent, and tend to bioaccumulate (Erickson 1997). PCBs are hydrophobic organic toxicants that accumulate in organisms both by uptake from the environment over time (bioaccumulation) and along the food chain. PCBs biomagnify with each trophic level in the food web (biomagnification). For aquatic organisms, the rate and physiological mechanism of PCB metabolism depend on the species and the specific type of PCB. PCBs cause cancer in animals and are probably carcinogenic in humans (group 2A classification, International Agency for Research on Cancer; ATSDR 2000).

As PCBs are no longer produced and sold, significant primary releases to the environment are uncommon; however, relatively small PCB releases still occur. Consequently, most of the mass of PCBs found in the Study Area sediments is derived from historical sources. In addition, secondary sources may introduce PCBs to the

[14] See DEQ (2003a) for additional information on PCB sources.

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LWR through a variety of environmental pathways such as atmospheric deposition (Hope 2008), as described in Section 10.2.1.2.

PCBs are present at a wide range of concentrations in the Study Area, in all media (Panels 10.2-1A–C). The PCB-related risks to humans and ecological receptors were evaluated in the BHHRA (Section 8 and Appendix F) and BERA (Section 9 and Appendix G), and the overall risks are briefly summarized below. Overall, PCBs are the most significant risk-posing contaminant in Portland Harbor.

The highest human health risks from exposure to PCBs in Portland Harbor were associated with the fish consumption pathway, for which PCBs contribute approximately 93 percent of the cumulative cancer risk for whole-body consumption on a Study Area-wide basis. Fish consumption risks from PCBs exceed both a target cancer risk of 1×10^{-4} and an HQ of 1 for both localized and Study Area-wide exposures. PCBs also result in cancer risks greater than 1×10^{-4} and/or HQs greater than 1 for shellfish consumption and for breastfeeding infants under in-water sediment direct exposure scenarios. Cancer risks from the fish consumption pathway are greater than 1×10^{-4} are projected for all areas of the Study Area, including areas where the sediment concentrations are less than upriver background levels. Direct exposure to PCBs in in-water sediment resulted in risks greater than 1×10^{-6} but less than 1×10^{-4} for at least one localized exposure area for non-tribal fishers, tribal fishers, and divers in wet suits.

Total PCBs pose ecologically significant risk to the protection and maintenance of the mink, otter, sandpiper, and osprey populations, to the protection of individual bald eagle organisms, and to the protection of the benthic community. PCBs pose low risk to fish. Risks to the benthic community are found in localized areas and, in all but three areas, co-occur with other contaminants potentially posing risks to the benthic community. The highest risk estimates are for mammalian predators, which are represented by mink and river otter. Based on PCB concentrations and toxicity parameters used in the BERA, potentially unacceptable risks to mink are projected throughout the Study Area. Because of the high sensitivity of mink to site contaminants, evaluation of risk to mink is used as a conservative surrogate for other mammalian predators such as river otters, for which data on sensitivity are unavailable.

10.2.1.1 PCB Contaminant Distribution

For the purposes of this CSM discussion, PCB concentrations greater than 300 µg/kg in sediment, 100 µg/kg in biota samples, and greater than 400 pg/L in surface water samples are considered elevated (the orange or red symbols and polygons on Panels 10.2-1A–C). The distribution of PCBs is summarized by media below.

10.2.1.1.1 Sediment

On a harbor-wide basis, the elevated PCB sediment concentrations occur, with minor exceptions, in nearshore areas outside the navigation channel and proximal to local upland sources (Map 5.1-2a–m and Panels 10.2-1A–B). Relatively high concentrations

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of PCBs are also often found in riparian sediments, sediment trap samples, surface waters, and biota samples in the areas with elevated sediment concentrations. The natural neighbors surface weighted average concentration (SWAC) for PCBs in the Study Area is 85 µg/kg.[15,16] For comparison, as presented in Section 7 and summarized in Table 7.3-5b, the 95 UPL of background total PCB concentrations in the upriver reach of the LWR (RM 15.3 to 28.4) is 17 µg/kg dry weight. An adjusted "OC-equivalent" background 95 UPL of 26 µg/kg dry weight was also estimated for total PCBs to account for the difference in the mean organic carbon content of surface sediments in the upriver reach (1.11 percent TOC), which is lower than the Study Area organic carbon content (1.71 percent TOC; see Table 7.3-6b).

As shown in Figure 5.1-1 and on Panel 10.2-1A, there are several areas with total PCB concentrations above 300 µg/kg in the eastern and western nearshore zones, in Swan Island Lagoon, and in a few scattered areas in the federal navigation channel. Similar spatial and concentration trends are observed for subsurface sediments (see Panel 10.2-1B). However, the >300 µg/kg surface and subsurface samples are sometimes at different locations (Figures 5.1-1 and 5.1-2 and Panels 10.2-1A–B), likely pointing to spatially and temporally variable inputs and sources or to different influences from sediment transport mechanisms. The largest areas of elevated PCBs in sediment include the areas offshore of the former Albina Shipyard (RM 11.3E), Gunderson (RM 8.8W area), Swan Island Lagoon, International Slip (RM 3.7–3.8E), and EOSM (RM 2.1–2.5E). Additional subareas with elevated surface sediment concentrations (300–999 µg/kg) include Balch Creek Cove (RM 9.7W; 322–600 µg/kg), and RM 4.0 to 4.1E (699 µg/kg).

Total PCB concentrations are generally higher in subsurface sediments within the Study Area as a whole (Panels 10.2-1A–B, Maps 5.1-2a–m, and Figure 5.1-33), pointing to predominantly historical total PCB sources and higher past loads. Notable exceptions to this general trend include the area between RM 11.2E and 11.6E. Sampling conducted by the City of Portland following the final round of RI sampling by the LWG (see Appendix H, Map H3.1-2a-b) indicates elevated surface and subsurface sediment concentrations between the shoreline and the federal navigation channel, with several exceptions that extend into the federal navigation channel. The area off the Glacier dock (RM 11.3E) was dredged in 2004 and pre-dredging, dredge prism testing identified elevated PCB concentrations. The post-dredging results showed that high PCB concentrations were left at the exposed surface. Dredging also occurred at the CDL Pacific Grain docks (RM 11.4E) in 2002 and 2009, which could also have affected

[15] The SWAC concentrations for the CSM chemicals were calculated using the natural neighbor interpolation method with 10-ft cells using the RI convention for the treatment of non-detects, i.e., in summed parameters, non-detects were assumed to be 0, if all parameters in a sum were non-detects, the value was set to the highest reported detection limit.

[16] The Study Area SWAC reported in this paragraph is based on the original RI data set (to June 2008) and was not recalculated to include the more recent Study Area data reported in Appendix H. Given the limited amount of more recent data relative to the size of the original RI data set, the Study Area SWAC is unlikely to have changed markedly. The upriver reach data set did not change between the original RI and the updated RI data sets.

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the relationship between surface and subsurface concentrations. The pattern of elevated surface and subsurface sediment levels in this area indicates a local source of PCBs and subsequent anthropogenic sediment disturbance (i.e., prop wash/dredging) which has altered the distribution of PCBs with depth in the sediment column and potentially re-exposed and re-mobilized subsurface contamination.

Collocated surface and subsurface samples from the inner portion of Willamette Cove also show higher surface PCB concentrations than at depth (Map 5.1-2g) suggesting higher recent inputs. Finally, in Swan Island Lagoon mean surface and subsurface total PCBs concentrations are approximately the same, based on Figure 5.1-33. Examination of vertical trends on a sample-by-sample basis (Map 5.1-2i) indicates a complex mix of samples exhibiting mostly higher surface levels or similar surface and subsurface levels, but some higher subsurface concentrations also occur. This lack of a clear surface versus subsurface pattern may reflect a combination of time-varying inputs, low net sedimentation rates, and localized high surface sediment mixing rates that result in variable spatial trends in sediment quality with depth.

Spatial variations in PCB composition (based on congener data) are evident throughout the Study Area, and areas of elevated PCB sediment concentrations often exhibit congener homolog patterns that are distinct from surrounding areas of lower PCB concentrations (Maps 5.1-32 and 5.1-33; Figures 5.1-35a–c and 5.1-36a–c). In general, these areas of lower concentrations are predominated by similar proportions of the tetraCB, pentaCB, hexaCB, and heptaCB homolog groups, with some localized exceptions. Areas of higher concentration (greater than 1,000 µg/kg) tend to be predominated by two or three of these homolog groups and may have a higher proportion of other homolog groups. In the eastern nearshore zone, the overall chlorination level of PCBs in the surface and subsurface sediments tends to be higher at the upstream end of the Study Area and lower downstream. Homolog patterns in areas of high PCB concentration tend to be more variable in the western nearshore zone.

For the areas of elevated concentration noted above, the PCB homolog patterns in surface and subsurface sediment, as well as in the sediment traps and in the particulate portion of the surface water samples, are often similar within each area. Subsurface sediment patterns are less consistent with surface sediment homolog patterns for the areas at RM 6.9 to 7.5W and RM 2.1 to 2.5E compared with the other elevated areas. Also, at RM 6.9 to 7.5W, the surface water and biota homolog patterns had more variations than the homolog patterns across media at the other elevated areas. Across elevated areas, homolog distributions vary spatially between areas, further suggesting multiple sources of PCBs for these areas of elevated concentration.

Relatively low levels of PCBs are widespread in portions of the harbor away from the localized areas of elevated concentrations (Panel 10.2-1A and Map 5.1-1). The histogram inset on Panel 10.2-1A shows that 57 percent of the detected surface sediment results have total PCB concentrations less than or equal to 34 µg/kg; these levels are comparable to the levels measured in the upriver, background reach upstream

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of Ross Island (Section 7) where, as noted above, the 95 UPL is 17 µg/kg, or 26 µg/kg adjusted OC-equivalent (see Table 7.3-6b). The homolog patterns in these widespread, low level PCB areas are generally similar in both depositional and erosional areas, have less distinct variations than areas of higher concentration, and may reflect inputs from upstream and transport within the Study Area.

10.2.1.1.2 Surface Water and Suspended Sediments

Relatively high concentrations in surface water are generally found in areas with elevated sediment concentrations. The majority of the highest total PCB concentrations (six of seven >2,000 pg/L) were associated with single point samples collected at RM6.7 (Station W013) within the Willamette Cove during low-flow conditions (Figures 5.3-20 and 5.3-21). The fourth highest single point sample was collected at RM 5.5. These data suggest that local PCB sources may exist in these regions of the Study Area. Total PCB concentrations at RM 11 were consistently higher than at RM 16, suggesting a source or sources between these locations. The range of total PCB concentrations within the complete data set across the Study Area was fairly consistent between RM 11 and 2, excluding the highest single-point concentrations, and elevated concentrations near the east side of the river at RM 6.7.

In the sediment trap data, PCB concentrations were generally higher in samples collected between RM 6.7 and 11.3 compared to other locations. PCB concentrations upstream of RM 11.3 did not exceed 24 µg/kg for total Aroclors or 13 µg/kg for total congeners.

Aside from large differences in PCB levels in sediment traps between ST007 (RM 11.3E) and its paired location ST008 (RM 11.5W), there was little relative difference in PCB concentrations between other cross-river sample pairs, suggesting suspended sediments are laterally homogenized in most of the Study Area. Total PCB congener concentrations in the Study Area samples were all higher than the average PCB concentrations from upstream locations—one- to five-fold greater than upstream concentrations, in most cases. These trends were generally also reflected in the Aroclor data.

10.2.1.1.3 TZW

Because of their hydrophobic character, PCBs are not expected to migrate significantly in groundwater. PCBs have been detected in upland groundwater at a few sites, but have not been identified as COIs in groundwater with a complete pathway to the Study Area at any upland sites that were included in the LWG TZW sampling program.

10.2.1.1.4 Biota

Relatively high concentrations of PCBs are often found in biota samples in areas with elevated sediment concentrations. Generally, in areas with elevated surface sediment concentrations, clam, sculpin, and smallmouth bass samples are elevated; however, the crayfish samples are not.

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PCBs were detected in all fish samples from the Study Area. The maximum total PCB concentration in whole-body fish occurred in a carp composite sample from the RM 4 to 8 fishing zone. PCBs were detected in all invertebrate samples from the Study Area that were analyzed for PCB congeners. PCBs were not detected in 10 crayfish tissue composites that were only analyzed for Aroclors, which have a higher detection limit than PCB congeners. The maximum total PCB value was measured in worms exposed in the laboratory to sediment from RM 8.8W.

10.2.1.2 Potential PCB Sources and Pathways

This section provides a summary of the various potential historical and current PCB sources to the Study Area, based on the types of industries that have existed in Portland Harbor that are known to have handled and used PCBs, and summaries of the various pathways (e.g., stormwater/waste water/overland transport, riverbank erosion, atmospheric deposition) by which PCBs can enter the Study Area.

Potential upland and overwater sources and identified known and likely complete migration pathways are identified on Table 10.2-1 and Panels 10.2-1A–C. These sources and pathways, identified on the basis of the process described in Section 4.2, focus on ECSI sites and are based on a review of information in the associated DEQ ECSI files and other readily available site information, including, in the case of LWG-member sites, information provided by the site owner.

10.2.1.2.1 Chemical Usage

In Portland Harbor, PCBs were widely associated with materials used in many historical manufacturing processes and, during WWII, to build ships and tankers (NASSCO 1999, EPA 1997b) (see Panel 10.2-1B and Map 3.2-3). Historically PCBs were used in electrical transformers, electric equipment manufacture, and roofing material manufacturing. PCB use was common in hydraulic machinery, especially for uses where heat- and fire-resistant fluids were needed (EPA 2004a, 2006h), and in automobiles, and they were potentially released in machinery repair, ship salvage, and auto salvage operations. Use of PCB-containing waste oils and paints was also widespread historically. Following WWII, a few of the shipbuilding facilities were repurposed for scrapping the military's surplus and obsolete vessels.

The most significant migration pathways for PCBs in the Study Area are historical and include industrial wastewater, stormwater, overland transport, overwater releases, and riverbank erosion (Table 10.2-1). Historical atmospheric deposition and upstream inputs may have also contributed PCBs to the Study Area. The historical releases from these pathways are not quantifiable and, in many cases, are difficult to distinguish. The effects of these historical releases are often expressed in older subsurface sediment as shown in Panel 10.2-1B.

Stormwater PCB loads have decreased substantially from historical levels since implementation of stormwater laws and regulations and the statutory ban on PCB manufacture in the 1970s. Overland transport was likely to have been more important

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historically, prior to the development of extensive stormwater conveyance systems. Bank erosion is also likely to have been more important historically, particularly during upland activities, or when contaminated material was used in construction fill activities. With one exception, historical PCB overwater releases have not been identified through the file review process, but are likely to have occurred in association with overwater operations, such as ship building and dismantling, ship repair and maintenance, and with the use of hydraulic fluids in dock operations. The locations of elevated PCB concentrations in sediments coincide in some cases with historical ship construction, dismantling, and repair operations (e.g., in the International Slip and the riverside areas offshore of Schnitzer/Calbag, the former Oregon Shipbuilding site), at the former Mar Com shipyard, in Willamette Cove, in Swan Island Lagoon, at Gunderson, the former WISCO shipyard, and offshore of the former Albina Engine and Machine Works shipyard at RM 11.3E), and it is likely that overwater releases occurred concurrently as a result of historical activities in these locations. PCBs are also detected in sediments near outfalls draining facilities historically engaged in electrical equipment manufacturing, such as the PacifiCorp facilities at RM 11.3E and two former GE transformer decommissioning facilities draining to OF-17. Stormwater discharges and riverbank erosion associated with fill soil from offsite and/or steel manufacturing activities at RM 2.3 have also resulted in elevated PCB concentrations in sediments.

Overall, current PCB inputs to the Study Area are much lower than historical inputs because PCBs are no longer manufactured and because waste management practices have been greatly improved. However, measured elevated levels of PCBs in surface sediments and other media, including biota in specific areas (e.g. International Slip, Swan Island Lagoon), indicate ongoing localized, inputs and/or internal mass transfer of historical PCB inventory from subsurface to surface sediments and then to other media by processes such as sediment resuspension (due to both natural and anthropogenic disturbance factors) and biological uptake. While surface sediments generally show lower PCB levels than subsurface sediments, the temporal persistence of elevated PCB levels in surface sediments in many nearshore and off-channel areas (e.g., Swan Island Lagoon) suggests that net sedimentation rates may be low in many nearshore areas (this is supported by the bathymetric change data and the limited radioisotope data from the Study Area; Anchor 2005), sediment column mixing rates are high, and inputs of PCBs still occur. Potentially important current pathways include stormwater and, locally, riverbank erosion. The effects of current releases and these physical site features are potentially expressed in surface sediment levels (Panel 10.2-1A).

As shown in Table 10.2-1 and Panels 10.2-1A–C, historical known or likely complete pathways for PCBs have been identified at 40 sites. Of these, stormwater is included at 38 sites, overland transport and riverbank erosion at 6 sites, and overwater at 1 site. Historical known or likely complete pathways have not been identified for groundwater migration.

Current known or likely complete pathways for PCBs have been identified at 18 sites. Of these, stormwater is included at 16sites, overland transport at 2 sites, riverbank

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erosion at 3 sites, and groundwater at 1 site. Current known or likely complete pathways have not been identified for the overwater pathway.

10.2.1.2.2 Stormwater/Wastewater/Overland Transport

Based on the external loading analysis presented in Section 6.1, stormwater transport is estimated to be the largest current pathway for PCBs to enter the Study Area from lateral sources (i.e., areas adjacent to the Study Area rather than upstream; see Figure 10.2-3). Atmospheric deposition of PCBs is estimated to contribute a total PCB load approximately one-third that of stormwater at the Study Area scale. Stormwater is identified as a likely pathway for PCBs to enter the Study Area at several areas with elevated PCBs in surface sediment and is expected to be identified as a complete pathway at additional locations with future stormwater sampling. Data gathered in Round 3A and 3B stormwater sampling in the harbor detected PCBs in stormwater from each land use area sampled—heavy industrial, light industrial, major transportation, residential, and parks/open space, however, outfalls from the heavy industrial land use category and select individual non-representative outfalls contributed the majority of the estimated PCB stormwater load (see Section 6.1.2.3 and 10.1.4.1.2).

Numerous upland sites have been identified as being known or likely historical and/or current sources of PCBs. These sites discharge directly to the river or discharge through shared conveyance systems (see Table 4.4-1).

Historical and current known or likely complete pathways for PCBs in stormwater have been identified at several properties associated with former shipyards and sites where transformers were serviced and/or dismantled.

Wastewater discharges are currently regulated primarily through NPDES permits. PCBs are not identified as a monitoring parameter for individual or general NPDES permits (see Tables 4.4-6 and 4.4-7). Wastewater discharges in CSO areas, where wastewater could overflow during CSO events, are regulated through municipal pretreatment permits. PCBs are not identified as a COI at any sites with a pretreatment permit (see Table 4.4-5). Historical wastewater discharges associated with ship building and decommissioning, electrical component manufacturing, and leaks and spills from equipment that used PCBs (e.g., hydraulic oils) are likely, but have not been specifically identified or quantified.

Sampling of the overland transport pathway is generally limited, and overland transport has been identified as a current known or likely complete pathway at only two sites, Gunderson and Triangle Park.

10.2.1.2.3 Overwater Discharge

No current known or likely complete PCB overwater pathways have been identified. Current overwater releases may be locally important at sites with continuous waste handling or operational activities, but are considered a minor current pathway overall.

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10.2.1.2.4 Groundwater Discharge

Because of PCBs' hydrophobic nature, groundwater is unlikely to be a significant historical or current pathway for PCB migration into the Study Area. However, PCBs have been detected in groundwater at Triangle Park, which has a current known complete pathway to the river. Triangle Park has been acquired by the University of Portland. A Bona Fide Prospective Purchaser Agreement (BFPPA) was signed with EPA by the University in December 2008. Under the agreement, EPA is overseeing the work being undertaken by the University to ensure actions are consistent with long-term cleanup remedies for the Superfund site.[17]

10.2.1.2.5 Riverbank Erosion

Like overland transport, sampling of the riverbank erosion pathway for PCBs is limited. Current known or likely complete PCB riverbank erosion pathways have been identified for EOSM, Gunderson, and Triangle Park. Historical known or likely complete pathways exist for these three sites and for Crawford Street and Willamette Cove, as well.

10.2.1.2.6 Atmospheric Deposition

Overall, atmospheric deposition in the Study Area for PCBs was more significant historically, prior to widespread adoption of controls by many countries on PCB production and usage. However, PCBs are still present in the atmosphere. Current research suggests that the major source of PCB releases to the atmosphere is an environmental cycling process that involves volatilization from water and/or soil into the atmosphere with subsequent removal via wet/dry deposition and then revolatilization. PCBs can also be released from landfills containing PCB waste materials, incineration of municipal waste or sewage sludge, and illegal disposal of waste transformer liquids,[18] although there are no known current or historical landfills or incineration in the Study Area associated with PCBs.

According to EPA's Toxic Chemical Release Inventory database, of the PCB releases from 1987 to 1993, the bulk occurred in 1990 from non-ferrous wire drawing and insulating industries, and the releases primarily occurred in California.[19]

10.2.1.2.7 Upriver Watershed Sources

Because of their ubiquitous nature, PCBs are found throughout the upriver watershed basins that feed into the Willamette River. Table 4.5-1 lists ECSI sites upstream of the Study Area (between RM 11.8 and the Tualatin River) and associated COIs. Information for these sites is from the DEQ ECSI database. Environmental cleanup of sediments or upland areas have been ordered for some of these properties, indicating that they could be now, or in the past have been, significant sources of PCBs to the Willamette River. Potential pathways are identified if enough information was available

[17] See http://yosemite.epa.gov/r10/cleanup.nsf/sites/trianglepark.
[18] See http://www.epa.gov/safewater/pdfs/factsheets/soc/tech/pcbs.pdf
[19] See http://www.epa.gov/tri/index.htm.

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in the database. PCBs were identified as COIs for 19 of these sites based on their hazardous substances/waste types. Several of the sites were or are involved in electrical equipment maintenance and repair. Of these 19 sites, PCBs were detected in sampled media at the following 11 sites:

- PGE Willamette River Sediment Investigation (RM 13.1E)
- Rexel/Taylor Electrical Supply (RM 13.2E)
- Portland General Electric Station L (RM 13.4E)
- Zidell Marine Corporation (RM 14W)
- Ross Island (RM 14.6E)
- OHSU - Moody Ave. Units A, B, C (RM 13.7W)
- Heath Oregon Sign Company (RM 14.8E)
- Martin Electric (RM 20.2W)
- Gross Property Disposal Site (RM 25W)
- Jeff Lohr Residence (RM 25.5E)
- West Linn Paper Company (RM 26W).

10.2.1.3 Loading, Fate, and Transport of PCBs

This section discusses current PCB loading, fate, and transport in the Study Area based on the detailed evaluation of these processes presented in Section 6. Consideration of these current and ongoing processes and their approximate relative magnitudes is an important element of the CSM. In particular, this empirical information (and its refinement) will be used in conjunction with the fate and transport modeling effort to assess recontamination and natural recovery potential in the FS.

Figures 10.2-1a-b, 10.2-2, and 10.2-3 summarize the PCB loading, fate, and transport assessment for the Study Area in a typical year. Estimates are for current conditions and likely differed historically. Much of the PCB mass in the Study Area, especially deeper sediments, is attributable to historical loading that occurred under different loading conditions and rates.

As shown in Figures 10.2-1a and 10.2-2, upstream surface water is the largest estimated current loading term for PCBs to the Study Area. Upstream surface water PCB loads are associated with both the dissolved and suspended fractions, with the majority of the PCB mass load entering the Study Area during low-flow conditions as compared to high-flow periods (see Figure 6.1-2), on a annually averaged basis.[20] This reflects a combination of factors. First, low-flow conditions occur for approximately two-thirds of the year compared to one-third of the year for high-flow conditions. Second, high-

[20] See Appendix E for a detailed discussion of the approach for surface water loading.

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flow surface water samples consistently exhibited lower concentrations of PCBs than low-flow samples across the site, suggesting that inflow concentrations at high flow rates overwhelm local effects at the scale of the surface water sample collection program. Figure H4.1-9 indicates that total PCB concentrations of sediments accumulating in upstream borrow pits, which are likely a mixture of upstream bedload and suspended load, are comparable to the upriver bedded sediment background levels (ranging from <5 µg/kg [non-detects] to about 20 µg/kg). There is an apparent increase in PCB loads in surface water between the upstream (composite of RM 11 and RM 16) and the downstream boundary of the Study Area (combined loads at RM 2 and the Multnomah Channel entrance) in both the particulate and dissolved fractions. The apparent increase is likely attributable in part to other quantified external loads (i.e., stormwater, atmospheric deposition, and advection through subsurface sediments).

Also, some of the apparent increase is likely due to fate and transport processes, in particular sediment resuspension, which have not been quantified for the RI, but which will be assessed for the FS. Figure 10.2-1b compares the distribution of total PCBs in surface sediments, sediment trap samples, and the particulate fraction of surface water samples on a dry-weight and OC-normalized basis. Summary statistics for the Study Area-wide concentrations of these media are presented in Table 10.2-2a and evaluated statistically to assess whether the data sets vary significantly from each other (Table 10.2-2b, non-parametric Mann-Whitney U test). Surface sediment concentrations across the Study Area are significantly greater than sediment trap sample concentrations, which are greater than surface water suspended solids concentrations. On an OC-normalized basis, the mean surface sediment concentration on a Study Area-wide basis (18.6 mg/kg) is about 1.4 times higher than sediment trap mean concentration (12.9 mg/kg), which, in turn, is about 19 times higher than surface water particulate mean concentration (0.66 mg/kg).

After the upstream surface water load, the second largest estimated current PCB loading term is stormwater, which is approximately a factor of 2 lower than the upstream load, followed by atmospheric deposition directly to the Study Area river surface (atmospheric deposition to the local watershed is included in the stormwater term), which is nearly an order of magnitude lower than the upstream surface water load (Figure 10.2-2). The final quantified external load is groundwater advection through subsurface sediments; this term has the smallest central estimate of the quantified terms, but is subject to a relatively high degree of uncertainty due to the variability in published organic-carbon partitioning values for PCBs. PCB loading terms that were assessed qualitatively are also shown on Figure 10.2-2. As discussed in Section 6.1.1.2, bedload into and out of the Study Area is qualitatively expected to be low relative to dissolved and particulate surface water loading. PCB volatilization from the water column is relevant for only a small fraction of the less chlorinated PCB congeners and is expected to be low.

Fate and transport processes internal to the Study Area for total PCBs are also shown on Figure 10.2-2. Among these, the only process for which quantitative estimates were

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developed in the RI is PCB transport to the water column due to pore water advection through surface sediments, which is estimated to be similar in magnitude to the subsurface advective loading term and is subject to the same high degree of uncertainty associated with PCB partitioning behavior. Other internal fate and transport processes are depicted on Figure 10.2-2 on a qualitative basis only. Sediment erosion, deposition, and burial are a function of locally and temporally variable hydrodynamic conditions and the surface sediment mixing rate (driven by both natural and anthropogenic forces). PCB partitioning between suspended sediment and surface water depends on the relative chemical concentrations associated with suspended particulate organic carbon and the dissolved surface water fraction, as well as reaction kinetics. The fate of PCBs within the Study Area may also be influenced to a minor degree by very slow microbially facilitated degradation and photolysis. These internal fate and transport processes are expected to be evaluated in the fate and transport modeling being conducted as part of the FS.

Estimates of PCB loads from upland groundwater plumes are not expected to be significant and were not generated.[21] PCB loads from permitted point source discharges (i.e., permitted stormwater and wastewater discharges) were not estimated because PCBs are not regulated and monitored under any active discharge permits within the Study Area (and permitted stormwater discharges are already taken into account in the stormwater loading term). Estimates of PCB loading from upland soil and riverbank erosion also were not assessed in the RI due to a paucity of data for riparian soil PCB concentrations and erosion rates.

The data shown in Figure 10.2-3 suggest that the total PCB load in surface water increases moving downstream through the Study Area to RM 4.[22] This suggestion is qualified by uncertainties inherent to the numerical load estimate calculations. The stormwater loading data indicate that the largest PCB stormwater inputs enter the Study Area between RM 3 and 4. As described in Section 6.1.2, the estimated load (1 kg/yr) to this reach is largely due to loading from a non-representative (unique) site (Outfall WR-384) and exceeds the next highest stormwater load, between RM 8 and 9, by a factor of 10. Estimated atmospheric deposition loading of PCBs to the water surface varies only as a function of water surface area by river mile. The estimated current PCB loads from the subsurface and surface sediments mirror the patterns of subsurface PCB sediment contamination (Panel 10.2-1B), with the highest loads between RM 8 and 9.

10.2.1.4 Relationship of Sources to the Distribution of PCBs

PCBs are no longer produced; however, they are common in the environment due to widespread historical sources, such as dielectric fluids in transformers, hydraulic oils,

[21] PCBs are not expected to migrate significantly in groundwater. PCBs have been detected in upland groundwater at a few sites, but have not been identified as COIs in groundwater with a complete pathway to the Study Area at any upland sites. Therefore, PCBs were not sampled in the LWG TZW sampling program.

[22] At approximately RM 3, the Columbia River and Multnomah Channel hydraulically influence the flow regime complicating interpretation of load conditions in this area (see Section 3.1.3.3).

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lubricants, and additives, and current sources such as waste materials from scrap metal recycling and the repair of ships and heavy equipment. PCBs are detected in Portland Harbor surface and subsurface sediments, other abiotic media (such as surface water), and fish and shellfish tissues. PCB concentrations in sediments across much of the Study Area are comparable to upriver background levels (e.g., 95 UPL of 17 µg/kg and 26 µg/kg corrected for the TOC difference between the upriver reach and the Study Area). In contrast, a number of distinct subareas of the Study Area exhibit surface and/or subsurface sediment total PCB concentrations that greatly exceed these levels. With minor exceptions, these subareas are located in nearshore (outside of the navigation channel) and off-channel (off the mainstem of the river, such as Swan Island Lagoon) areas that are both more quiescent and proximal to likely or known historical sources. PCB composition (homolog patterns) shows variations between many of the subareas, further suggesting localized sources. The largest areas of elevated PCBs in sediment include the areas offshore of the former Albina Shipyard at RM 11.3, Gunderson, Swan Island Lagoon, International Slip, RM 4E (International Terminal), and EOSM.

In most of the subareas with elevated PCB sediment concentrations, and throughout the Study Area generally, subsurface PCB concentrations exceed surface levels, indicating greater inputs or releases historically that have been reduced or eliminated over time. Small-home-range fish and shellfish tissue samples collocated with these subareas generally exhibit elevated levels, indicating ongoing exposure and biological uptake. The localized long-term accumulation of sediments (and PCBs) in the sediment column, as well as the physical site characteristics (e.g., time-series bathymetric changes) and HST modeling, indicate that most of these subareas are depositional/stable over time. Empirical data (time-series bathymetry, sediment-profile data, and limited radioisotope cores) further suggest that many of these nearshore areas have low long-term net sedimentation rates and the surface sediment layer is well-mixed; this likely contributes to the persistence of relatively elevated surface sediment levels over time.

One noteworthy exception to these general patterns is the elevated PCB subarea at RM 11.3E. Though there are exceptions, several sampling points in this area exhibit higher surface sediment than subsurface levels along the eastern nearshore area and adjacent channel edge. The nearshore sediment PCB distribution, as well as the collocated surface water and sediment trap data, suggest a local, recent input and/or redistribution of PCBs historically released into this area and present in the sediments. Data from recent sediment studies in the RM 11E area, provided in Appendix H, indicate that ship traffic and recent maintenance dredging at the Glacier NW dock (RM 11.3E) and the CDL Pacific Grain dock (RM 11.4E) have resulted in resuspension, exposure, and redistribution of PCB contaminated sediments in this nearshore area. Also, the federal navigational channel in this relatively dynamic segment of the Study Area reach is subject to high flows and both short- and long-term sediment scour. While the highly elevated PCB subareas in the Study Area appear to result from releases from specific localized sources (both historical and/or current), PCBs also enter the Study Area from more general sources, such as private and public stormwater and

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sewer outfalls, and sources upstream of the Study Area in the Willamette River. Also, not all sources for all areas of elevated concentrations have been identified. Harbor-wide loading estimates indicate that the highest current external inputs to the Study Area are from upstream surface water and, to a lesser degree, stormwater runoff and atmospheric deposition within the Study Area. The same estimates indicate that the mass flux of PCBs in surface water (dissolved and particulate) exiting the Study Area at RM 2 and at the Multnomah Channel entrance exceed the estimated fluxes entering the Study Area from all quantified external loading terms. This suggests an internal mass transfer of PCBs from bedded sediment to the water column, likely through sediment resuspension.

PCBs are the primary contributor to risk in Portland Harbor. The highest human health risks from exposure to PCB concentrations were associated with the fish consumption pathway, for which PCBs contribute approximately 93 percent of the cumulative cancer risk for whole-body consumption and approximately 97 percent of the cumulative risk for fillet consumption, on a Study Area-wide basis

Total PCBs pose significant risk to mink, river otter, and spotted sandpiper, and low risk to osprey, bald eagle, sculpin, and smallmouth bass. PCBs were identified as one of the primary contaminants in sediment likely to pose potentially unacceptable risk to the benthic community or populations.

The relationships between tissue body burdens and abiotic concentrations across the Study Area are a primary focus of the fate and transport modeling to be conducted as part of the FS. Elevated levels of PCBs in tissue are generally correlated with elevated levels of sediment contamination, surface water contamination, and sediment trap contamination. Furthermore, elevated surface water contamination seems to be associated with elevated sediment contamination. The reasons for this are complex. In some cases, surface water contamination may be associated with stormwater, and in other cases it may be associated with sediment. Indications are that PCB levels in surface sediments in most areas are decreasing over time. The rate of this improvement is a function of localized net sedimentation rates, erosion potential, surface sediment mixing rates, and source control actions.

10.2.2 TCDD TEQ

Panels 10.2-2A–C present the Study Area graphical CSM for TCDD TEQ per the format described in Section 10.2 and support the discussions in the following subsections. Recent sediment studies at RM 11E and upriver are included in Appendix H.

As a group, PCDDs represent 75 different positional isomers, while PCDFs comprise over 135 compounds (Eisler 1986). These two chemical classes are generally referred to as dioxins and furans, respectively. The toxicity of PCDD/Fs is frequently expressed using a toxicity equivalency procedure in which the toxicities of individual congeners (expressed relative to the toxicity of 2,3,7,8-TCDD) are summed to yield the 2,3,7,8-

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TCDD dioxin TEQ. This procedure involves multiplying TEFs by the analytical results for individual dioxins/furans and calculating the 2,3,7,8-TCDD TEQ for each sample. TEFs developed by the World Health Organization (Van den Berg et al. 1998, 2006) for seven dioxin congeners and 10 furan congeners were used to calculate dioxin TEQs for this RI/FS. The TEQ expresses the assumed combined contribution of the individual congeners to total dioxin/furan toxicity, expressed in terms of the toxicity of 2,3,7,8-TCDD. This section discusses PCDD/Fs in terms of the TCDD TEQ and includes total PCDD/F concentration information, where relevant, to further support the CSM.

PCDD/Fs are byproducts of combustion, incineration, certain industrial chemistry processes, and natural sources. They are not intentionally produced and have no known use. They can be anthropogenically and naturally produced through combustion, bleached paper production, polyvinyl chloride (PVC) production, ink/dye production, metal smelting, and production of chlorine, chlorophenols, chlorinated herbicides, and commercial Aroclor (PCB) mixtures (ATSDR 1998). Examples of combustion and incineration that may lead to the formation of PCDD/Fs include waste (hazardous, medical) incinerators, cement kilns, boilers and industrial furnaces, vehicle emissions, fossil fuel power plants (e.g., coal), and backyard burning (e.g., refuse piles, burn barrels). PCDD/Fs are naturally produced from forest fires, volcanic eruptions, and sedimentary deposits. Currently the largest source of TCDD TEQ is from backyard burning as documented in EPA's 2006 inventory of sources (EPA 2006b). The primary source of PCDD/Fs in environmental media is generally atmospheric deposition. When released into the air, some PCDD/Fs may be transported long distances, even around the globe. In the atmosphere, it has been estimated that 20 to 60 percent of 2,3,7,8-TCDD is in the vapor phase. Sunlight and atmospheric chemicals can break down a very small portion of the PCDD/Fs, but most PCDD/Fs will be deposited on land or water (ATSDR 1998).

Behavior of PCDD/Fs in the environment is characterized by extremely low vapor pressures, high octanol-water and organic carbon partitioning coefficients (K_{ow} and K_{oc}, respectively), and extremely low water solubilities. These factors indicate a strong affinity for sediments, particularly sediments with high organic matter content; as a result, transport of PCDD/Fs in aquatic systems is closely tied to fine-grained sediment transport processes. Some PCDD/Fs, present near the water surface and not bound to solids, may be subject to photodegradation (EPA 1994), but this is only relevant for a small fraction of the PCDD/Fs in the system. In general, PCDD/Fs are highly stable in all environmental media, with persistence measured in decades. Chemical degradation of PCDD/Fs through reductive chlorination can also occur, but it is a very slow process. PCDD/Fs have been shown to biomagnify in aquatic food webs and associated avian and mammalian species (ATSDR 1998).

Dioxins/furans are contributors to risks in the BHHRA and are considered secondary to PCBs, the primary contributor to risk on a Study Area wide-basis. In the BHHRA, total dioxin TEQ resulted in risk estimates that exceed 1×10^{-4} and/or HQ of 1 for both localized and Study Area-wide exposures for fish and shellfish consumption. For direct

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contact with in-water sediment, cancer risks exceed 1 x 10^{-4} for tribal fishers on a localized basis in the vicinity of RM 7W, and exceed 1 x 10^{-6} for in-water workers and divers (localized basis) and for high- and low-frequency fishers (localized and Study Area-wide basis). HQs also exceed 1 on a localized basis in the vicinity of RM 7W for tribal fishers, high frequency fishers, and breastfeeding infants.

The combined toxicity of dioxins/furans and dioxin-like PCBs expressed as TEQ poses significant risk to mink and river otter, and low risk to spotted sandpiper, osprey, and bald eagle.

10.2.2.1 TCDD TEQ Contaminant Distribution

Panels 10.2-2A–C present the Study Area graphical CSM for TCDD TEQ. For TCDD TEQ, concentrations greater than 5 pg/g in sediment, 10 pg/g in biota samples, and greater than 0.25 pg/L in surface water samples are considered elevated.

Care should be taken in interpreting the TCDD TEQ Panels 10.2-2A–B because relatively few sediment samples from the federal navigation channel were analyzed for PCDD/Fs. The size of a Thiessen polygon is determined by one-half the distance to the nearest sampling point in any direction, and when no data exist for the federal navigation channel, the polygon is drawn such that the concentration extends across the channel (e.g., see red and yellow surface sediment polygons above RM 11). The full surface and subsurface data sets shown in Maps 5.1-5 and 5.1-6a–m are useful in interpreting these panels.

10.2.2.1.1 Sediment

The natural neighbors SWAC for TCDD TEQ in the Study Area is 7.6 pg/g.[23] For comparison, the 95 UPL of background total TCDD TEQ concentrations in the upriver reach of the LWR (RM 15.3 to 28.4) is 2.16 pg/g dry weight (Table 7.3-5b). An adjusted "OC-equivalent" background 95 UPL of 3.32 pg/g dry weight (Table 7.3-6b) was also estimated for TCDD TEQ to account for the difference in the mean organic carbon content of surface sediments in the upriver reach (1.11 percent TOC), which is lower than in the Study Area (1.71 percent TOC).

Elevated TCDD TEQ concentrations occur in sediment generally in the eastern and western nearshore zones, i.e., outside the boundaries of the navigation channel (Maps 5.1-5 and 5.1-6a–m and Panels 10.2-2A–C).

Post-RI sampling indicates an area of elevated TCDD TEQ concentrations in the eastern nearshore areas between RM 11.2E and 11.6E (see Appendix H, Map H3.1-6a-b). Both surface and subsurface sediments are elevated between the shoreline and the navigation channel; however, there is no consistent vertical trend.

[23] The SWAC concentrations for the CSM chemicals were calculated using the natural neighbor interpolation method with 10-ft cells and non-detected results included at the detection limit.

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At RM 6.6 to 6.7W, elevated concentrations in surface sediment extend into the edge of the navigation channel. Relatively high TCDD TEQ concentrations are also found in subsurface sediment, sediment trap samples, surface waters, and in biota samples in this area.

As shown in Figure 5.1-9 and on Panel 10.2-2A, there are areas with clusters of TCDD TEQ concentrations greater than 5 pg/g in the eastern and western nearshore zones, in Terminal 4, Slip 1, Swan Island Lagoon, and in the navigation channel. Similar spatial and concentration trends are observed for subsurface sediments (see Panel 10.2-2B and core plots 5.1-6a–m), although the sample density in the subsurface results in fewer individual subsurface polygons. Sediment traps are elevated at RM 11.3E and at the head of Swan Island Lagoon.

As noted above, the lower density of PCDD/F data make the surface to subsurface concentration gradient difficult to interpret from Panels 10.2-2A–B. Maps 5.1-4a–m show that most of the elevated areas lack strong or consistent vertical concentration gradients. Some exceptions to this include the area under and just upstream of the Railroad Bridge at RM 6.9, where surface layers show higher concentrations than at depth (Map 5.1-4g) and the northwest corner of Willbridge Terminal bulk petroleum operations and tank farms, where higher levels are evident at depth (Map 5.1-4h). Figure H4.1-10 shows that PCDD/F concentrations (expressed as TCDD TEQ) in sediments accumulating in upper Study Area depositional areas (and likely representing the quality of material entering the site from upstream over approximately the 10+ years) show low and relatively constant concentrations with depth.

As noted in Section 5, OCDD and HpCDDs are the predominant homolog groups in sediment (Figures 5.1-40a–c and 5.1-41a–c), sediment trap (Figure 5.2-14), and surface water samples (Figures 5.3-56 and 5.3-57). OCDD also accounts for a significant percent of the total PCDD/F concentration in invertebrate biota samples (Figure 5.5-27). PCDD/F homolog distributions in fish and invertebrate samples are variable by species with a higher contribution from PCDFs; however, trends are apparent within species (Figure 5.5-26). The dominance of OCDD and HpCDDs in most media and samples limits the utility of the PCDD/F composition data for detecting relationships within and between the subareas of elevated TCDD TEQ concentration.

10.2.2.1.2 Surface Water and Suspended Sediments

Surface water samples are elevated in Willamette Cove, across the river from Willamette Cove at RM 6.8W, and in the RM 4 transect sample. The highest concentration was measured in the single-point sample collected at RM 6.7 during the July 2005 low-flow sampling event. The second highest PCDD/F concentration was measured in the transect sample from RM 11 on the east side of the river, collected during the stormwater-influenced event (November 2006). From sampling event to sampling event, total PCDD/F concentrations within the Study Area were variable and did not display consistent trends from upstream to downstream.

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An isolated elevated PCDD/F concentration was observed in the 4th quarter (August to November) in Swan Island Lagoon, but samples from previous quarters were not analyzed for PCDD/Fs at this location. Elevated concentrations were also measured in the 3rd quarter (May to August) and 4th quarter at RM 11.3E, the 3rd quarter RM 1.8W, and the 4th quarter at RM 7.5W. There were no other strong spatial or temporal gradients evident in total PCDD/F concentrations measured in suspended sediments collected in sediment traps within the Study Area (Figure 5.2-14). In general, Study Area sediment trap samples had higher total PCDD/F concentrations than the upstream locations near RM 15.6.

10.2.2.1.3 TZW

Because of the hydrophobic nature of PCDD/Fs, these chemicals were not included in the TZW groundwater sampling program.

10.2.2.1.4 Biota

PCDD/Fs were detected in all fish and invertebrate tissue samples from the Study Area. Elevated biota TCDD TEQ concentrations are limited to the area between RM 6.5 and 7.5 (Panel 10.2-2C). The maximum whole-body concentration was measured in a sculpin composite sample from RM 7.3W. The homolog distribution in fish tissues was quite variable, with differences evident between species as well as between locations and tissue types within species. For invertebrates, the maximum concentration was measured in worms exposed in the laboratory to sediment from RM 6.9W.

10.2.2.2 Potential PCDD/Fs Sources and Pathways

This section summarizes the various potential historical and current PCDD/Fs sources to the Study Area based on the types of industries that have existed in Portland Harbor that are known to have generated PCDD/Fs and the various pathways by which PCDD/Fs can reach the Study Area.

As shown in Table 10.2-3 and Panels 10.2-2A–C, historical known complete or likely complete pathways for PCDD/Fs have been identified at four sites. Of these, the stormwater pathway is included at four sites, groundwater at one site, overwater at one site, overland transport at two sites, and riverbank erosion at two sites.

Current known complete or likely complete pathways for PCDD/F have been identified at two sites. Of these, the stormwater pathway is included at both sites, groundwater and overwater at neither of the sites, overland transport at one site, and riverbank erosion at one site.

10.2.2.2.1 Chemical Usage

In Portland Harbor, PCDD/Fs are currently and historically associated with combustion, incineration, cement kilns, boilers and industrial furnaces, vehicle emissions, fossil fuel power plants (e.g., coal), and backyard burning (e.g., refuse piles, burn piles). Other potential sources include structure fires, especially wood-product facilities, and trash burning on upland sites adjacent to the river. Several significant fires have occurred in

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the harbor, including M&B in 1966, a grain elevator on the Goldendale Aluminum property in 1966, and a cooperage on the central portion of Willamette Cove in 1967 (Oregonian 1966a,b; 1967). Historical operations associated with PCDD/Fs include smelters, wood treatment facilities, pesticide/herbicide formulation industries, power plants, pulp and paper facilities, and certain chlorine production processes (see Panel 10.2-2B). All of these historical local operations have ceased and the dominant current external loading of PCDD/Fs to the Study Area is via upstream surface water. PCDD/Fs also are naturally produced by a variety of mechanisms including forest fires, volcanic eruptions, and sedimentary deposits of some clays (EPA 2006b).

Historical pathways to the Study Area identified for PCDD/Fs include stormwater, groundwater, overland transport, overwater releases, and riverbank erosion (Table 10.2-3); these are discussed below. The historical releases from these pathways are not quantifiable and, in many cases, are difficult to distinguish.

Stormwater TCDD TEQ loads have likely decreased substantially from historical levels since implementation of stormwater regulations and the cessation of facility operations. Due to the hydrophobic nature of PCDD/Fs, groundwater is not expected to have been a significant historical pathway. Furthermore, any contribution of TCDD TEQ via groundwater to the Study Area was probably greater prior to the initiation of groundwater remediation efforts at the former herbicide, wood treating, and chlorine manufacturing facilities along the harbor. Bank erosion is also likely to have been more important historically, particularly during upland activities, or when contaminated material was used in construction fill activities.

The most extensive area of elevated TCDD TEQ concentrations in sediments occurs just upstream of RM 7 offshore of the former Arkema (RM 7.3W), Rhone Poulenc (RM 7.2W), and Gould properties (RM 7.2W). The highest concentration is located at a single sampling location with elevated furans (TCDF). Trace levels of PCDFs (but not PCDDs) may have been a byproduct of an older chlor-alkali technology used in the Arkema facility chlor-alkali process. Both PCDDs and PCDFs were byproducts in production of the herbicides 2,4-D and 2,4,5-T at the former Rhone Poulenc herbicide manufacturing facility (RM 7.2W), and secondary lead smelting operations, such as those historically conducted at the Gould Site. Additional areas with high sediment PCDD/F concentrations in this portion of the river include the former M&B facility (RM 7E), where wood products were treated with creosote, PCP, and inorganic preservatives. These wood treating processes are known sources of PCDD/Fs, particularly wood preservatives containing PCP. Most of this area was remediated in 2004 as part of the M&B (federal) Superfund site cleanup, and the elevated PCDD/F concentrations in sediments have been isolated under a cap. Also, the nearby Triangle Park property (RM 7.4E) has been used for lumber milling, marine services, and electrical power generation, possibly fueled with wood wastes, coal, bunker fuel, and diesel fuel.

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Other areas with elevated PCDD/Fs include Terminal 4 (RM 4.5W), RM 8.8W, Willamette Cove/M&B, and Balch Creek Cove. While no sources of PCDD/Fs have been identified in some of these areas, M&B has been identified as a historical source of PCDD/Fs.

Current PCDD/F inputs to the Study Area are potentially much lower relative to historical inputs because all of the known historical industries that produced PCDD/Fs as a byproduct are no longer in operation and because waste management practices have been greatly improved.

However, elevated levels of TCDD TEQ in surface sediments and other media, including biota, in the RM 6.8 to 7.5W and the RM 6.8 to 7.5E reaches indicate localized, current and/or ongoing internal mass transfer of historical TCDD TEQ inventory from subsurface/surface sediments to other media by processes such as sediment resuspension and biological uptake. There are elevated levels of TCDD TEQ, particularly in surface sediments, in other areas of the Study Area (e.g., along the western shoreline from RM 9 to 10, in Swan Island Lagoon, at the head of International Slip, etc.; Panel 10.2-2A). Given the ubiquity of dioxins and furans in the environment and large number of potential natural and anthropogenic activities that can inadvertently produce them, many as yet unidentified sources and pathways certainly exist or have existed in the Study Area, as well as the LWR region.

As shown in Table 10.2-3 and Panels 10.2-2A–C, historical known or likely complete pathways for PCDD/Fs have been identified at four sites. Of these, groundwater is included at one site, stormwater at four sites, overland transport and riverbank erosion at two sites, and overwater at one site.

Current known or likely complete pathways for PCDD/Fs have been identified at one site and pathways include stormwater, riverbank erosion and overland transport.

10.2.2.2.2 Stormwater/Wastewater/Overland Transport

In general, stormwater transport containing atmospheric depositional PCDD/Fs from regional combustion sources is expected to be the most significant current pathway for PCDD/Fs to enter the Study Area from adjacent upland sites. As stated above, a primary source of PCDD/Fs in the environment is also atmospheric deposition from global sources. PCDD/Fs were not sampled for during the Round 3A and 3B stormwater sampling program but, based on their ubiquitous nature, PCDD/Fs would very likely be detected in stormwater from all areas sampled. A current known complete pathway has been identified for Triangle Park and City of Portland Outfall OF-22B. The Triangle Park property is undergoing cleanup according to a BFPPA with EPA. Gould Electronics, Rhone Poulenc, and M&B, are identified as having known complete historical pathways. Other potential sources include structure fires, especially at wood-product facilities, and trash burning on upland sites adjacent to the river. Several significant fires have occurred in the harbor, including M&B in 1966, a grain

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elevator on the Goldendale Aluminum property in 1966, and a cooperage on the central portion of Willamette Cove in 1967 (Oregonian 1966a,b; 1967).

Sampling of overland transport is generally limited, and therefore no conclusions can be reached regarding this potential pathway. Within the Study Area, only Triangle Park has a current complete overland transport pathway for PCDD/Fs.

Wastewater discharges are currently regulated primarily through NPDES permits. PCDD/Fs are included as a monitoring parameter for Rhone Poulenc's (StarLink Logistics [SLLI]) individual NPDES permit for discharge of treated stormwater and wastewater after biological treatment, settling, and carbon filtration. Wastewater discharges in CSO areas, where wastewater could overflow during CSO events, are regulated through municipal pretreatment permits. PCDD/Fs are not identified as a COI at any sites with a pretreatment permit (see Table 4.4-5).

10.2.2.2.3 Overwater Discharge

No current known or likely PCDD/F overwater pathways have been identified. However, M&B is identified as having a known historical pathway.

10.2.2.2.4 Groundwater Discharge

Groundwater is unlikely to be a significant historical or current pathway for PCDD/F migration into the Study Area due to the hydrophobic nature of PCDD/Fs, and no sites have been identified as having current known or likely complete groundwater pathways for PCDD/Fs. M&B had a historical complete groundwater pathway for PCDD/Fs.

10.2.2.2.5 Riverbank Erosion

Sampling of bank areas is also generally limited and, therefore, no conclusions can be reached regarding this potential pathway. A current known pathway for PCDD/Fs exists at the Triangle Park site. M&B had a historical known complete pathway and Triangle Park had a historical likely complete pathway for PCDD/Fs.

10.2.2.2.6 Atmospheric Deposition

As noted earlier, the primary source of PCDD/Fs in environmental media is generally atmospheric deposition from regional combustion sources. When released into the air, some PCDD/Fs may be transported long distances, even around the globe.

10.2.2.2.7 Upriver Watershed Sources

As noted below in Section 10.2.2.3, TCDD TEQ loads entering the Study Area in upstream surface water (dissolved and particulate fraction) are estimated to be the dominant current external TCDD TEQ loading term. As noted previously, however, Figure H4.1-10 indicates the TCDD TEQ concentration of sediments accumulating in upstream borrow areas is low (<1 pg/g).

Table 4.5-1 lists upstream ECSI sites (between RM 11.8 and the Tualatin River) and associated COIs. Information for these sites is from DEQ ECSI database. PCDD/Fs

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were identified at two paper mills (Blue Heron and West Linn) listed on Table 4.5-1, based on their hazardous substances/waste types. These mills, located at the base of Willamette Falls, were the focus of an EPA Site Investigation in 2007. Data from that site investigation were included in the Portland Harbor RI/FS background data set (Section 7) and are discussed in the background data outlier identification evaluation (Section 7.3.1.2). PCDD/F sediment data were removed from the upstream background set due to these local, potential sources of PCDD/Fs to sediments in this portion of the river.

10.2.2.3 Loading, Fate, and Transport of PCDD/Fs

This section discusses loading, fate, and transport of PCDD/Fs in the Study Area based on the detailed evaluation of these processes presented in Section 6. Consideration of these current and ongoing processes and their approximate relative magnitudes is an important element of the CSM. In particular, this empirical information (and its refinement) will be used in conjunction with the fate and transport modeling effort to assess recontamination and natural recovery potential in the FS. This discussion addresses PCDD/F mass flux in terms of total PCDD/Fs, the sum of 17 target dioxins and furans. TCDD TEQ loads are also presented to allow for comparison with contaminant distribution and source discussions, which are generally presented in terms of TCDD TEQ. Total PCDD/Fs and TCDD TEQ data were available for surface water and sediments and were used to estimate surface water and advective (surface and subsurface sediment) loading terms. For atmospheric deposition to the river surface, only TCDD TEQ data were available. PCDD/F and TCDD TEQ loads from permitted point source discharges (i.e., permitted stormwater and wastewater discharges) were not estimated because PCDD/Fs are infrequently regulated and monitored under active discharge permits within the Study Area; Rhone Poulenc has an active NPDES permit that requires monitoring for PCDD/Fs (see Table 4.4-6). Upland groundwater plume and stormwater PCDD/F load estimates were also not generated.[24] Estimates of PCDD/F loading from upland soil and riverbank erosion also were not assessed in the RI due to a lack of data for riparian soil PCDD/F concentrations and erosion rates.

Figures 10.2-4a, 10.2-5a, and 10.2-6a summarize the total PCDD/F loading, fate, and transport assessment for the Study Area in a typical year. TCDD TEQ loads are presented in Figures 10.2-4a (includes both PCDD/F and TCDD TEQ), 10.2-5b, and 10.2-6b. Loading estimates are for current conditions and likely differed historically. The PCDD/Fs mass in the Study Area deeper sediments (Panel 10.2-2B) is likely attributable to historical loading that occurred under different past loading conditions and rates.

Based on the data available, only two PCDD/F loading terms were quantified, upstream surface water and advection through subsurface sediments. As shown in Figures

[24] PCDD/Fs were listed as target analytes for only one TZW site (Rhone Poulenc) and were detected in one of two sampling locations where PCDD/Fs were analyzed; transport of PCDD/Fs via upland groundwater plumes is expected to be insignificant compared to other loading terms. PCDD/Fs were not included in the stormwater sampling analyte list.

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10.2-4a and 10.2-5a, upstream loading contributes much larger (5 orders of magnitude) estimated loads of total PCDD/Fs to the Study Area than advection through subsurface sediments. Consistent with the highly hydrophobic nature of PCDD/Fs, the upstream surface water loads are associated primarily with the suspended particulate fraction, with similar total PCDD/F mass loads entering the Study Area during high-flow and low-flow conditions (see Figure 6.1-4). The total PCDD/F loads in upstream surface water (composite of RM 11 and 16) and the downstream boundary of the Study Area (combined loads at RM 2 and the Multnomah Channel entrance) are generally comparable. TCDD TEQ quantitative load estimates show similar patterns (Figures 10.2-4a and 10.2-5b), with upstream surface water loads 3 orders of magnitude greater than subsurface sediment advective loads. Estimated loading from atmospheric deposition to the river surface, based on TCDD TEQ, falls between upstream surface water and subsurface sediment advective loading. This loading estimate, however, is considered highly uncertain due to the dearth of local data. PCDD/F loading terms that were assessed qualitatively are also shown on Figure 10.2-5a. As discussed in Section 6.1.1.2, bedload into and out of the Study Area is qualitatively expected to be low relative to upstream surface water loading. Dioxin and furan compounds are subject to slow abiotic and photolytic degradation processes. These processes, though currently unquantified, are expected to be minor relative to other terms. A cross-media comparison and statistical assessment of surface sediment, sediment trap samples, and the particulate fraction of surface water samples on a Study Area-wide basis (Tables 10.2-2a-b and Figure 10.2-4b) show that the mean PCDD/F concentrations differ significantly across all media, with the surface sediment showing the highest average concentration, followed by surface water particulate concentrations and then sediment trap concentrations.

Fate and transport processes internal to the Study Area for total PCDD/Fs and TCDD TEQ are also shown on Figures 10.2-5a and 10.2-5b, respectively. Among these internal terms, the only process for which quantitative estimates were developed in the RI is PCDD/F transport to the water column due to pore water advection through surface sediments, which is estimated to be comparable in magnitude to the subsurface advective loading term for total PCDD/Fs and TCDD TEQ. As with advective loading from subsurface sediments, advection through surface sediments is subject to a high degree of uncertainty associated with the partitioning behavior of individual dioxins/furans. Other internal fate and transport processes are depicted on Figures 10.2-6a and 10.2-6b on a qualitative basis only. Sediment erosion, deposition, and burial are a function of locally and temporally variable hydrodynamic conditions and the surface sediment mixing rate (driven by both natural and anthropogenic forces). PCDD/F partitioning between suspended sediment and surface water depends on the relative chemical concentrations associated with suspended particulate organic carbon and the dissolved surface water fraction, as well as reaction kinetics. The fate of PCDD/Fs within the Study Area may also be influenced to a minor degree by slow abiotic and photolytic degradation.

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The loading estimates shown on Figure 10.2-6a suggest that the external PCDD/F loads to surface water within the Study Area are several orders of magnitude smaller than upstream surface water loads. The estimated current internal PCDD/F loads from advection through subsurface and surface sediments generally follow the patterns of PCDD/F sediment concentrations (Figure 6.1-58), with the highest loads between RM 7 and 8. Similarly, the estimated subsurface and surface sediment advective loads based on TCDD TEQ were also highest from RM 7 to 8 (Figure 10.2-6b). Estimated atmospheric deposition loading of PCDD/Fs to the water surface varies only as a function of water surface area by river mile.

10.2.2.4 Relationship of Sources to Distribution of TCDD TEQ

PCDD/Fs are currently and historically associated with combustion, incineration, cement kilns, boilers and industrial furnaces, vehicle emissions, fossil fuel power plants (e.g., coal), and backyard burning (e.g., refuse piles, burn barrels); historical operations (EPA 2006b) also include smelters, wood treatment facilities, herbicide formulation industries, power plants, pulp and paper, and PCDFs from chlorine production processes (see Panel 10.2-2B). Detected TCDD TEQ levels in most Study Area sediments are comparable to upriver background levels (OC-equivalent 95 UPL of 3.3 pg/g dry weight was determined for the upriver reach). However, there are a number of distinct nearshore subareas scattered throughout the Study Area that exhibit elevated TCDD TEQ levels. One of these subareas is between RM 6.5 and 7.5 along both the east and west banks of the river. These areas coincide with several currently identified known or likely historical industrial dioxin and/or furan sources. In addition, some surface water and biota (bass, sculpin, and crayfish) samples from this reach also show somewhat elevated concentrations.

Elevated TCDD TEQ sediment levels elsewhere in the Study Area (e.g., RM 11E, nearshore adjacent to Gunderson, Swan Island Lagoon, International Slip, EOSM) are not associated with documented sources and pathways. This pattern indicates that not all point sources of PCDD/Fs to the Study Area have been identified. However these locations are coincident with areas showing elevated concentrations of other ICs, particularly PCBs and/or PAHs (see Section 10.2.1 and 10.2.4). The relatively low density of TCDD TEQ data (compared to other ICs) makes surface to subsurface concentration gradients difficult to discern on a Study Area-wide basis. In general, however, there do not appear to be strong trends with depth. Surface layers do exhibit higher TCDD TEQ concentrations than deep sediment along the west side of the navigation channel at Railroad Bridge (RM 6.9), reflecting the known sources there. Scattered cores along the western bank from Willbridge Terminal to RM 9 show higher subsurface TCDD TEQ concentrations; this is generally a long-term sediment accumulation area and suggests higher historical inputs in this vicinity or from upstream. Sediments accumulating in upper Study Area depositional areas, which likely represent the past 10+ years of deposition, show low TCDD TEQ levels (see Figure H4.1-10, Appendix H).

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The quantification of current external loading estimates to the Study Area was limited to upstream surface water, subsurface sediment advection, and atmospheric deposition. The highest estimated current input to the Study Area is from upstream surface water. An apparent increase in the estimated surface water flux of PCDD/Fs from upstream to downstream within the Study Area may reflect contributions from a combination of other external loading terms (e.g., stormwater, advective transport, and atmospheric deposition) or internal fate and transport processes (e.g., sediment resuspension).

In the BHHRA, total dioxin TEQ resulted in risk estimates that exceeded a cancer risk of 1×10^{-4} and/or a non-cancer HQ of 1 for both localized and Study Area-wide exposures for fish and shellfish consumption. For direct contact with in-water sediment, cancer risks exceed 1×10^{-4} for tribal fishers on a localized basis in the vicinity of RM 7W, and exceed 1×10^{-6} for in-water workers and divers (localized basis) and for high- and low-frequency fishers (localized and Study Area-wide basis). In the BERA, total TEQ was recommended as a COC because exposure poses a risk of ecologically significant adverse effects to mink populations. Total TEQ also poses risk of adverse effects to river otter, spotted sandpiper, and osprey populations and to bald eagles. These latter risks are lower than the risk to the mink population.

The relationships between tissue body burdens and abiotic concentrations across the Study Area are a primary focus of the fate and transport modeling to be conducted at the Site for the FS. TCDD TEQ levels in surface sediments in elevated subareas located proximal to likely sources do not appear to be decreasing over time (i.e., surface and subsurface sediment concentrations are similar) despite the observation that sediment entering the Study Area appears to be comparable to upriver background levels.

10.2.3 Total DDx

DDx pesticides refer to the group of nonionic, organochlorine pesticides containing DDT and its principal metabolites (DDD and DDE). This list includes both the 2,4'- and 4,4'-isomers of DDT, DDD, and DDE, for a total of six chemicals in the group. Each chemical is analyzed individually and the concentrations are summed to obtain the total DDx value.

DDT was used as an insecticide from about 1943 (Porter 1962) until 1972, when it was banned for use in the United States because of its toxicity to wildlife. DDE and DDD are the primary metabolites of DDT, but technical-grade DDT may also contain DDE and DDD as impurities (ATSDR 2002d). DDT was released historically to air and soil through widespread spraying of crops and forests, and for mosquito control. Releases also occurred at more local scales at pesticide manufacturing and storage facilities. Ongoing releases occur in countries where its use is not banned, and some of these releases can be transported globally through the atmosphere. Because DDT is no longer produced and sold in the United States, significant new releases to the environment are uncommon. Consequently, most of the mass of DDx found in the Study Area is derived from historical sources. However, DDx compounds continue to be introduced to the LWR through a variety of environmental pathways, as described in Section 10.2.3.2.

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DDT degrades slowly via abiotic and microbially mediated processes to the more persistent DDx compounds, DDE (under aerobic conditions) and DDD (in anoxic systems; EPA 2000b). In the environment, DDx compounds are persistent and are readily bioaccumulated in aquatic organisms (EPA 2000b). While there is no clear evidence that DDT, DDD, or DDE causes cancer in humans, there is sufficient evidence of carcinogenicity of these substances in rodents, which has led to their classification as probable human carcinogens (ATSDR 2002d).

DDx compounds are present at a wide range of concentrations throughout the Study Area, in all media (Panels 10.2-3A–C). The DDx-related risks to humans and ecological receptors were evaluated in the BHHRA (Section 8 and Appendix F) and BERA (Section 9 and Appendix G), and the overall risks are summarized below.

DDD, DDE, and DDT were identified as contaminants potentially posing unacceptable risks for the BHHRA due to exceedances of 1 x 10-6 cancer risk threshold and a HQ of 1 from ingestion of fish. However, they are not considered primary contributors to risk on a Study Area-wide basis because they did not exceed the upper bound of the risk management range of 1×10^{-6} to 1×10^{-4} or HQs of 1 for any of the exposure scenarios evaluated on a Study Area-wide basis. DDD, DDE, and DDT resulted in cancer risks of 1×10^{-4} and HQs greater than 1 for fish consumption scenarios on a localized basis.

In the BERA, sum DDE was found to pose low to negligible risk to bald eagle. 4,4'-DDT, and total DDx were found to pose localized risk to individual Pacific lamprey ammocoetes due to potential exposure to contaminated shallow transition zone water. Total DDx was identified as one of the primary COPCs in sediment likely to pose potentially unacceptable risk to the benthic community or populations. The BERA concludes that DDx compounds in sediment pose potentially unacceptable risk to the benthic community only on the western side of the river between approximately RM 6.8 and 7.4.

TZW exceedances are localized, indicating that DDx in TZW is not likely to pose risk to Study Area benthic invertebrate communities or fish populations. Concentrations of total DDx at RM 7.3W may be sufficiently high to adversely affect local benthic fish and invertebrates. Risks to amphibians and plants from TZW are not expected because the species in the Study Area are unlikely to use the habitats where contaminated groundwater discharges.

10.2.3.1 DDx Contaminant Distribution

The following sections describe the distribution of DDx compounds in all media. The discussions are focused on data compiled/collected in the Study Area, but comparisons to upstream levels are noted for those media for which upstream data exist. Graphical summaries of the distribution of DDx compounds in all media, as well as physical site information and DDx source and loading information, are presented in Panels 10.2-3A–C and Figures 10.2-7a-b through Figure 10.2-9.

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10.2.3.1.1 Sediment

The SWAC for total DDx in the Study Area is 31 µg/kg.[25,26] For comparison, the 95 UPL of background total DDx concentrations in the upriver reach of the LWR (RM 15.3 to 28.4) is 3.6 µg/kg dry weight (Table 7.3-5b). An adjusted "OC-equivalent" background 95 UPL of 5.5 µg/kg dry weight (Table 7.3-6b) was also estimated for total DDx to account for the difference in the mean organic carbon content of surface sediments in the upriver reach (1.11 percent TOC), which is lower than in the Study Area (1.71 percent TOC).

Elevated DDx concentrations in surface sediments are defined here as exceeding 1,000 µg/kg (the orange or red sediment polygons on the panels). Overall, the elevated total DDx concentrations in sediments in the Study Area are limited to localized areas in nearshore zones. The largest area of elevated surface sediments is in the western nearshore zone between RM 7.1 and 7.5, proximal to the Arkema property, where DDT was historically manufactured, and near the former Rhone-Poulenc property, where DDT formulation and packaging operations were conducted.. Elevated DDx concentrations are also observed in subsurface sediments downstream to RM 6.8.

Other areas of elevated total DDx sediment concentrations are smaller in extent and are located at RM 8.8W, RM 6.5W, at the mouth of Swan Island Lagoon, and in subsurface sediments only, at RM 4.8W and the head of the International Slip. DDx has not been identified as an upland COI at sites in these areas.

At elevated locations, total DDx concentrations are consistently higher in subsurface sediments within the Study Area (Panels 10.2-3A–B; Figure 5.1-42). The greatest difference between surface and subsurface sediment concentrations, when averaged on a river-mile basis, is seen in the western nearshore zone between RM 7 and 8, where the highest sediment total DDx concentrations are found at depths exceeding 10 ft (Figures 5.1-11 and 5.1-12; Panels 10.2-3A–B). This points to predominantly historical sources and higher past loads. Areas of high concentrations of DDx are more common downstream of RM 8 (where concentrations are typically in the 5 to 100 µg/kg range) compared with upstream of RM 8, where much of the river, except for Swan Island Lagoon, exhibits levels below 5 µg/kg.

10.2.3.1.2 Surface Water and Suspended Sediments

Total DDx concentrations are relatively elevated in surface water transect samples at the upstream end of the Study Area and upstream of the Study Area (Panel 10.2-3A) in high-flow conditions (Figure 5.3-59), indicating inputs from upriver areas. Single-point

[25] The SWAC concentrations for the CSM chemicals were calculated using the natural neighbor interpolation method with 10-ft cells with non-detected results included at the detection limit.

[26] The Study Area SWAC reported in this paragraph is based on the original RI data set (to June 2008) and was not recalculated to include the more recent Study Area data reported in Appendix H. Given the limited amount of more recent data relative to the size of the original RI data set, the Study Area SWAC is unlikely to have changed markedly. The upriver reach data set did not change between the original RI and the updated RI data sets.

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surface water samples downstream of RM 7.5 and the transect at RM 6.3 exhibit elevated concentrations of total DDx in low-flow and high-flow conditions (Figure 5.3-59; Panel 10.2-3A). Surface water samples at the downstream end of the Study Area (RM 2) do not exhibit elevated concentrations; however, the surface water transect at the mouth of Multnomah Channel exhibited an elevated total DDx concentration in low-flow conditions (Panel 10.2-3A; Figure 5.3-59).

Total DDx concentrations in the sediment trap samples were generally low throughout most of the Study Area and upriver at RM 16 in all four quarters sampled (Figure 5.2-17). Relatively elevated DDx concentrations were measured in sediment traps from RM 6 and 7.5W and in Swan Island Lagoon in the summer and fall periods. The elevated DDx levels reported at RM 11E in the fall period are likely an artifact resulting from analytical interferences associated with the elevated PCB levels in that sample (see Section 5.2.7).

10.2.3.1.3 TZW

Elevated total DDx concentrations in TZW samples and are found near RM 7.2W; however, no other TZW samples from other areas were analyzed for DDx so that comparisons with other areas cannot be made.

10.2.3.1.4 Biota

While some biota samples exhibit elevated concentrations of total DDx in other portions of the Study Area, the peak concentrations are found where sediments levels are elevated and most extensive near RM 7 (Figure 5.5-12a-j; Panel 10.2-3C).

10.2.3.2 Potential DDx Sources and Pathways

This section provides a summary of the potential historical and current DDx sources to the Study Area based on the types of industries that have existed in Portland Harbor and the more general use and application of DDx compounds in the watershed (Chemical Usage) and the various pathways (e.g., stormwater/waste water/overland transport, atmospheric deposition) by which DDx can enter the Study Area.

10.2.3.2.1 Chemical Usage

In Portland Harbor, DDx was historically associated with pesticide manufacturing and storage facilities. DDT also was released historically to air and soil through widespread spraying of crops and forests, and for mosquito control. Some riverbank facilities and other upland properties likely applied DDT for mosquito control before the pesticide was banned. Bulk storage of DDx has also been identified at one upland site (Shell Willbridge Terminal bulk petroleum operations and tank farms). Map 3.2-5 shows known historical chemical manufacturing and storage sites. These sites include Arkema, Rhone Poulenc, and the Shell Terminal at Willbridge (storage only).

The most significant current influx of DDx to the Study Area is upstream surface water. Historical DDx sources to the Study Area included upstream surface water, stormwater,

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and riverbank erosion (Table 10.2-4). The historical releases from these pathways are not quantifiable.

As shown on Table 10.2-4 and Panels 10.2-3A–C, historical known complete or likely complete pathways for DDx have been identified at four sites. Of these, the groundwater pathway is included at two sites, stormwater at four sites, and riverbank erosion at two sites. Historical known complete or likely complete pathways for DDx were not identified for overwater releases or overland runoff.

Current known complete or likely complete pathways for DDx have been identified at three sites. Of these, the groundwater pathway is included at one site, stormwater at three sites, and riverbank erosion at one site. Current known complete or likely complete pathways were not identified for overwater releases or overland runoff of DDx.

10.2.3.2.2 Stormwater/Wastewater/Overland Transport

Current and/or historical known or likely complete pathways for DDx in stormwater have been identified at Arkema, Rhone Poulenc, Metro Central Transfer Station, the Willbridge Terminal, and City of Portland Outfalls OF-22 and OF-22B. Most of the facilities either drain stormwater or infiltrate contaminated groundwater to shared conveyances (Saltzman Creek, OF-22B, or OF-22). A historical likely complete pathway for DDx in stormwater has also been identified for Willbridge Terminal.

The locations of the highest DDx concentrations in nearshore sediments are between RM 6.8W and 7.3W. Identified sources include the former Arkema and Rhone Poulenc sites. DDT was manufactured at Arkema from 1947 to 1954. At the Arkema facility, manufacturing process residue was initially discharged to floor drains, which may have been connected to a historical storm sewer that drained to the Willamette River. Manufacturing process residue was also discharged to an unlined pond, a MCB recovery system, and a trench to increase capacity of the pond. The riverbank area, mainly between Docks 1 and 2, received miscellaneous fill for several years. The fill included materials from spent chlorine cells (ERM 2005). In addition, dredge spoils that may have contained DDx were deposited on the riverbank. Under an administrative order with EPA, Arkema is currently conducting a non-time critical removal action to address high concentrations of DDx in sediment adjacent to the Arkema site.

DDx has been detected in groundwater and stormwater associated with the Rhone Poulenc site. DDT was formulated or manufactured at the Rhone Poulenc facility starting in the early 1950s through the 1960s, and insecticide formulation and distribution operations overall ceased in 1969. Between 1943 and 1955, stormwater and untreated wastewater from the Rhone Poulenc property were discharged to Doane Lake. From 1955 to 1966, wastewater was treated for phenols prior to discharge to Doane Lake. Between 1966 and 1977, treated wastewater was redirected to directly discharge through a pipe from the plant site to the river (AMEC 2008). Contaminated

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groundwater infiltrated into City stormwater OF-22B. In 2009, SLLI removed accumulated sediment from OF-22B stormwater lines and grouted the lines to at least partially prevent contaminated groundwater from entering the lines. In the second half of 2009, SLLI cleaned out the lines and installed impermeable liners in the stormwater lines to further prevent groundwater infiltration (DEQ 2010c).

The Metro Central Transfer Station is adjacent to the eastern border of Rhone Poulenc, and is located within the OF-22B outfall basin. Historically, this site hosted a warehouse for a succession of steel companies since at least 1924. Metro has operated a solid waste transfer station on this property since the late 1980s. Contaminated groundwater from Rhone Poulenc has infiltrated Metro's stormwater pipes, and, combined with Metro stormwater COIs, created a preferential pathway. Cleanup efforts conducted by Rhone Poulenc are expected to eliminate the preferential groundwater pathway to stormwater.

DDx has also been detected in stormwater and riverbank soils at the Willbridge Terminal. DDx was stored at the Willbridge Terminal facility and it has a likely complete pathway for stormwater. As noted below under riverbank soils, elevated surface sediment, subsurface sediment, TZW, and biota are not found for DDT in the Willbridge Terminal cove.

Other areas of elevated DDx concentrations are found downstream of RM 7, and may be related, in part, to downstream transport of sediment in the relatively higher energy that extends from about RM 7 to RM 5 (see Section 3.1.5). Additional areas, both upstream and downstream, have DDx concentrations that are not associated with known sources (Panels 10.2-3A–B).

Overall, current DDx inputs to the Study Area are much lower than historical inputs because DDT is no longer manufactured or used in widespread spraying applications and because waste management practices have greatly improved. However, elevated levels of DDx in surface sediments and other media, including biota, in the RM 6.8 to 7.5W reach and at RM 8.8W indicate localized, external inputs or ongoing internal mass transfer of historical DDx from subsurface/surface sediments to other media by processes such as sediment resuspension and biological uptake.

Stormwater transport is expected to be the most significant current pathway for DDx to enter the Study Area from adjacent upland sites. Data gathered in Round 3A and 3B stormwater sampling in the harbor detected DDx in stormwater from each land use area sampled—heavy industrial, light industrial, residential, parks/open space, and transportation; however, outfalls associated with the heavy industrial land use category and selected individual non-representative outfalls contributed the majority of the estimated DDx stormwater load (see Section 6.1.2.3).

No sites with a known or potentially complete current or historical overland transport pathway have been identified.

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10.2.3.2.3 Overwater Discharge

There are no known current or historical overwater releases of DDx in the Portland Harbor associated with ECSI sites evaluated by LWG. In-water access to the Arkema facility was historically provided from three docks—from upstream to downstream—the Salt Dock, Dock 1, and Dock 2. Inadvertent spills that may have occurred during transfer activities (sodium chloride, sodium hydroxide, sodium chlorate solution and chlorine) are not likely to have caused sediment contamination, as these materials are highly water-soluble or volatilize immediately upon release to the atmosphere. Nevertheless, leakage of chemicals may have been sources of in-river contamination.

10.2.3.2.4 Groundwater Discharge

Known or likely complete current and historical DDx groundwater pathways have been identified for the Arkema site and a historical likely pathway for the Rhone Poulenc site. In-water COIs that are associated with upland groundwater plumes at the Arkema site include DDx, chloroform, and MCB. The DDx and VOC (primarily MCB) plumes overlap in the Acid Plant area and are present in nearshore wells. The Round 2 Groundwater Pathway Assessment identified a nearshore groundwater discharge zone and an intermediate variable groundwater discharge zone at the Arkema site (Figure 10-7 from Integral 2006a). Farther offshore, a zone of low-to-no groundwater discharge was identified. These zones span both the Acid Plant area and Chlorate Plant area. The analysis of Round 2 TZW and sediment chemistry within these zones indicates a complete pathway for transport of upland groundwater COIs to the transition zone.

As part of ongoing groundwater source control at Arkema, a focused feasibility study was submitted to DEQ for groundwater/NAPL in summer 2008. DEQ selected a slurry wall/groundwater extraction system as the SCM in 2009, and the SCM is in design. DEQ (2010b) anticipates SCM construction to start shortly.

The Rhone Poulenc facility (RM 7.2W) formulated organochlorine insecticides during the 1950s and 1960s including DDT, aldrin, malathion, toxaphene, heptachlor, chlordane, lindane, dieldrin, and rotenone. According to historical records, the facility made 16 to18 tons of DDT per day in the dust plant from the 1950s until 1965. Volume-wise, DDT was the biggest product. A likely complete historical pathway for groundwater and a known complete historical pathway for stormwater have been identified for DDx at the site.

According to the 2010 DEQ Milestone Report, SLLI submitted a comprehensive SCE report in early 2008. SLLI performed pilot testing of several SCMs to treat and/or control its most significant groundwater plume. SLLI has completed an extensive, long-term groundwater pumping test to support the design for the North Front Avenue SCM, which targets contaminated groundwater moving in the highly conductive fractured basalt zone. The pumping test includes a number of extraction wells that could largely comprise the SCM. Construction of any supplemental portions of the SCM is anticipated for 2011. As mentioned above, SLLI has also implemented a SCM to prevent further groundwater infiltration into the OF-22B stormwater lines.

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10.2.3.2.5 Riverbank Erosion

Historical known or likely complete DDx riverbank erosion pathways have been identified for the Arkema and Willbridge Terminal sites. The riverbank area at Arkema, mainly between Docks 1 and 2 received miscellaneous fill for several years. The fill material included clean soil, asphalt, concrete, metal piping, and miscellaneous materials from spent chlorine cells (ERM 2005). In addition, dredge spoils that may have contained COIs, were deposited on the riverbank. Riverbank erosion at the site has also been identified as a known complete pathway between Docks 1 and 2, both historically and currently, for DDx.

DEQ's (2010c) Milestone Report states that Arkema has evaluated their riverbank and the threat that this portion of the site poses to the river. Riverbank source control between Docks 1 and 2 is anticipated to be incorporated into the EPA-led, in-water Early Action at Arkema.

Riverbank erosion has been identified as a historical likely complete pathway for the Willbridge Terminal site. Elevated surface sediment, subsurface sediment, transition zone water, and biota are not found in the Willbridge Terminal cove.

10.2.3.2.6 Atmospheric Deposition

DDx is a persistent organic pollutant and can be transported, over long distances through the atmosphere. It vaporizes and condenses, depositing on oceans and freshwater bodies, where it begins the cycle again. DDx is a legacy pollutant in the US, Canada and Mexico, as it is no longer used or produced. There are no known atmospheric sources of DDx within the Study Area.

10.2.3.2.7 Upriver Watershed Sources

Table 4.5-1 lists ECSI sites upstream of the Study Area (between the confluence of the Tualatin River at RM 29 and RM 11.8) and associated COIs. Information for these sites was obtained from the DEQ ESCI database. Potential pathways were identified if sufficient information was available.

DDx was identified as a COI for two sites based on hazardous substance/waste type characterizations. Of these two sites, DDx was detected in sampled media at the OHSU Moody Ave. Units A, B, C (including #1401 SW Moody Right-of-Way) property. Potential pathways to the river include groundwater and stormwater/surface runoff from the site. Upstream DDx sources with pathways potentially impacting surface water may be important since upstream surface water represents the largest current loading term for DDx to the Study Area.

10.2.3.3 Loading, Fate, and Transport of DDx

This section discusses total DDx loading, fate, and transport in the Study Area based on the detailed evaluation of these processes presented in Section 6. Consideration of these current and ongoing processes and their approximate relative magnitudes is an important element of the CSM. In particular, this empirical information (and its

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refinement) will be used in conjunction with the fate and transport modeling effort to assess recontamination and natural recovery potential in the FS.

Figures 10.2-7a-b through 10.2-9 summarize the total DDx loading, fate, and transport assessment for the Study Area in a typical year. Estimates are for current conditions and likely differed historically. Much of the total DDx mass in the Study Area, especially in deeper sediments, is attributable to historical loading that occurred under different loading conditions and rates.

As shown in Figures 10.2-8 and 10.2-9, upstream surface water is, by a substantial margin, the largest estimated current loading term for total DDx to the Study Area. Upstream surface water DDx loads are associated with both the dissolved and suspended particulate fraction, with the majority of the DDx mass load entering the Study Area during high-flow conditions, as compared to low-flow periods (see Figure 6.1-6). The total DDx loads in upstream surface water (RM 11) and the downstream boundary of the Study Area (combined loads at RM 2 and the Multnomah Channel entrance) are comparable. The next largest current total DDx loading terms, based on central estimates, are stormwater[27] and atmospheric deposition to the Study Area river surface (atmospheric deposition to the watershed is included in the stormwater term). The lowest quantified external loads, based on the central estimates, are groundwater advection through subsurface sediments and upland groundwater plume discharge, which are essentially equivalent. However, the subsurface sediment advection loading term is subject to a very high degree of uncertainty due to the wide variability in published organic-carbon partitioning values for DDx pesticides. DDx loading terms that were assessed qualitatively are also shown on Figure 10.2-8. As discussed in Section 6.1.1.2, bedload into and out of the Study Area is qualitatively expected to be low relative to dissolved and particulate surface water loading. DDx compounds are subject to slow abiotic and microbial degradation processes, with degradation of DDD and DDE isomers being extremely slow. This loss term, though currently unquantified, is expected to be minor relative to other terms.

As noted above, DDx loads entering the Study Area in upstream surface water (dissolved and particulate fraction) are estimated to be the dominant current external DDx loading term. For surface water, higher concentrations of total DDx were observed for the high-flow sampling events, compared to the low-flow and stormwater-influenced event, suggesting erosion of upstream agricultural areas as an upstream source of DDx to the Study Area. Some additional mass of DDx also enters and remains in the Study Area with upstream bedload sediments, but this additional load has not been quantified. Figure H4.1-11 (Appendix H) indicates that total DDx concentrations of sediments accumulating in upstream borrow pits, which are likely a

[27] The DDx stormwater loading term developed in the RI is dominated by the estimated load from a single "non-representative" site. As discussed in Section 6, the estimated loads from this non-representative site are subject to a high degree of uncertainty related to extrapolation of stormwater concentrations measured in the non-representative outfalls (<10 percent of the drainage sub-basin) to areas that are larger than the catchments that drain to the non-representative outfalls.

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mixture of upstream bedload and suspended load, are comparable to upriver background levels (ranging from <1 to 5 µg/g).

The cross-media comparison and statistical assessment of surface sediments, sediment traps, and the particulate fraction of surface water samples (Tables 10.2-2a-b and Figure 10.2-7b) show that the overall surface sediment concentrations are significantly greater than the concentrations in suspended solids in surface water and sediment trap samples on a Study Area-wide basis; surface water particulate and sediment trap concentrations are not statistically different across the entire Study Area. The OC-normalized mean surface sediment DDx concentration is about 22 and 13 times higher than sediment trap and surface water solids means, respectively (Table 10.2-2a).

Fate and transport processes internal to the Study Area for total DDx are also shown on Figure 10.2-8. Among these internal terms, the only process for which quantitative estimates were developed in the RI is DDx transport to the water column due to pore water advection through surface sediments, which is estimated to be slightly lower in magnitude than the subsurface advective loading term for total DDx. As with advective loading from subsurface sediments, advection through surface sediments is subject to a high degree of uncertainty associated with DDx partitioning behavior. Other internal fate and transport processes are depicted on Figure 10.2-8 on a qualitative basis only. Sediment erosion, deposition, and burial are a function of locally and temporally variable hydrodynamic conditions and the surface sediment mixing rates (driven by both natural and anthropogenic forces). DDx partitioning between suspended sediment and surface water depends on the relative chemical concentrations associated with suspended particulate organic carbon and the dissolved surface water fraction, as well as reaction kinetics. These internal fate and transport processes will be evaluated quantitatively in the fate and transport modeling being conducted as part of the FS.

Estimates of DDx loads from permitted non-stormwater point source discharges (i.e., wastewater discharges) are not expected to be significant and were not generated.[28] Estimates of DDx loading from upland soil and riverbank erosion also were not assessed in the RI due to a paucity of data for riparian soil DDx concentrations and erosion rates.

The data shown on Figure 10.2-9 suggest that the total DDx load in surface water increases moving downstream through the Study Area.[29] The apparent increase is likely attributable to some combination of the other loading, fate, and transport processes in the Study Area. This suggestion is qualified by uncertainties inherent to the numerical load estimate calculations. The stormwater loading data indicate that the largest DDx stormwater inputs enter the Study Area between RM 6.8 and 7.4. Upland groundwater plume loads are also highest between RM 7 and 8. Estimated atmospheric deposition

[28] NPDES permitted wastewater discharge is not expected to contain significant amounts of total DDx relative to other loading terms.

[29] At approximately RM 3, the Columbia River and Multnomah Channel hydraulically influence the flow regime, complicating interpretation of load conditions in this area (see Section 3.1.3).

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loading of DDx to the water surface varies only as a function of water surface area by river mile. The estimated current DDx annual loads from advection through subsurface and surface sediments follow the patterns of DDx sediment concentrations (Figure 6.1-59), with the highest loads between RM 7 and 8.

10.2.3.4 Relationship of Sources to Distribution of DDx

DDx compounds are widely detected at relatively low levels in Portland Harbor surface and subsurface sediments; other abiotic media, such as surface water, sediment traps, and TZW; and fish and shellfish tissues. Total DDx concentrations in sediments in much of the channel and in most of the Study Area upstream of RM 8 are comparable to upriver background levels (95 UPL of background data is 3.6 µg/kg).

Historical known DDx sources (Arkema, Rhone Poulenc, and Willbridge Terminal) are located along the western shore of the river from about RM 6.8 to 7.5, and nearshore surface, and especially subsurface, sediments in this area show elevated total DDx levels. Concentrations of DDx compounds in surface sediments throughout the Study Area are generally lower than subsurface sediment concentrations, indicating greater inputs or releases historically that have been reduced or eliminated over time. These overall patterns are consistent with the location and historical operations of the major known sources of DDx in Portland Harbor.

Relatively elevated DDx levels are also seen in the surface water, sediment traps, TZW, and biota samples from the RM 6.8 to 7.5 area. Sediment concentration gradients along this western nearshore area and channel edge downstream from this source area suggest the downstream transport of DDx in sediments and possible inputs from other point sources. This pattern is consistent with the hydrodynamic and sediment transport characteristics in this portion of the Study Area, particularly from RM 5 to 7. Less extensive areas with elevated DDx occur at Gunderson, at the head of International Slip, RM 4.8W (subsurface only), and at the Cascade General Shipyard at depth in the sediment column.

Elevated DDx concentrations in sediments within the Study Area are primarily the result of localized releases from pesticide manufacturing and storage facilities and stormwater runoff from local areas where DDx was applied for pest control.

Other areas of elevated total DDx sediment concentrations are smaller in extent and are located at RM 8.8W, RM 6.5W, at the mouth of Swan Island Lagoon, and in subsurface sediments only at RM 4.8W and the head of the International Slip. There are no known or potential sources of DDx in these areas and this may represent upland source data gaps. Elevated subsurface concentrations, e.g., in International Slip, likely reflect historical sources.

Estimates of quantifiable external loading terms indicate that the highest current external inputs to the Study Area are from upstream surface water and, to a much lesser degree, from Study Area stormwater and atmospheric deposition. Estimates of DDx

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fluxes in surface water at RM 2, the downstream end of the Study Area, and in Multnomah Channel suggest that slightly more DDx mass may be leaving the Study Area downstream in surface water than entering the Study Area from all quantified sources. This possible increase, which is consistent with the observed increase in total DDx surface water concentrations between RM 11 and 4 in six of seven sampling events (see Section 5.3.8), may reflect an internal mass transfer of DDx from bedded sediment to the water column, likely through sediment resuspension. The relationships between tissue body burdens and abiotic concentrations across the Study Area are a primary focus of the fate and transport modeling to be conducted at the Site for the FS.

The baseline RAs reflect the observed distribution of DDx compounds in the Study Area with the highest risks associated with the RM 7W area. In the BHHRA, DDD, DDE, and DDT were identified as contaminants potentially posing unacceptable risks because of estimated cancer risk exceedances of 1×10^{-6} and HQs greater than 1 from the ingestion of fish. The BERA concludes that DDx compounds in sediment pose potentially unacceptable risk to the benthic community only on the western side of the river between approximately RM 6.8 and 7.4.

10.2.4 Total PAHs

Panels 10.2-4A–C present the Study Area graphical CSM for PAHs, per the format described in Section 10.2.1. Recent sediment studies at RM 11E and the downtown reach are included in Appendix H.

PAHs are a large chemical group composed of more than 100 chemicals that are constituents of crude and refined oil, shale oil, coal tar, and creosote. PAHs are also formed during the incomplete combustion of organic materials including coal, oil, gas, wood (wood stoves, fireplaces), garbage (municipal waste incineration), or other organic substances, such as tobacco. Natural sources of PAHs include volcanoes and forest fires. The largest historical sources of the high concentrations of PAHs in Study Area sediments were localized releases, such as historical industrial direct discharges, from facilities along the Study Area that formerly or currently use, manufacture, and/or store products and waste materials containing PAHs (e.g., petroleum oils and coal-based products, petroleum product use and storage, used oil, and asphalt roofing materials).

PAHs may accumulate in benthic organisms, fish, and other organisms that ingest sediments while feeding. However, biomagnification through the food chain is expected to be minimal for PAHs due to the high rate of PAH metabolism in fish, mammals, and birds. The EPA has determined that some PAHs are probable human carcinogens, and studies on animals have shown that PAHs can cause harmful effects following both short- and long-term exposure (ATSDR 1997a).

Although they are not considered primary contributors to risks on a Study Area wide basis, PAHs were identified as contaminants potentially posing unacceptable risks in the BHHRA based on fish and shellfish consumption and direct contact exposures. Cancer risk estimates in the BHHRA for total cPAHs exceeded 1×10^{-4} for clam consumption

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at two locations (RM 5W and 6W), for direct contact with in-water sediment at one location (RM 6W) for tribal fishers, and at one location (RM 6.1) for hypothetical use of untreated surface water as a domestic water source. Total cPAHs were identified as contaminants potentially posing unacceptable risks, based on cancer risk estimates exceeding 1×10^{-6}, for fish and shellfish consumption, direct contact with beach sediment, direct contact with in-water sediment, hypothetical use of surface water as a drinking water source, and exposure to surface water by divers (RM 6W).

In the BERA, PAHs were identified as likely to pose potentially unacceptable risk to the benthic community or populations. PAHs are strongly associated with the benthic risk area from RM 5.1 to RM 6.9 on the west side of the river, where concentrations are elevated in sediment. Three individual PAHs (benzo(a)anthracene, benzo(a)pyrene, and naphthalene) were identified as contaminants posing potentially unacceptable risk in TZW at sampling locations within this same portion of the river.

TZW exceedances are localized, indicating that PAHs in TZW are not likely to pose risk to Study Area benthic invertebrate communities or fish populations. Concentrations of PAHs in the vicinity of the Gasco and Siltronic locations may be sufficiently high to adversely affect local benthic fish and invertebrates. Risks to amphibians and plants from TZW are not expected because the species in the Study Area are unlikely to use the habitats where contaminated groundwater discharges.

10.2.4.1 PAH Contaminant Distribution

Panels 10.2-4A–C present the Study Area graphical CSM for PAHs and support the discussions in the following subsections. These panels present observed PAH distribution in sampled media (biota, surface water, sediment, sediment traps, riparian soils), as well as the current understanding of known and potential historical and current sources of PAHs to Portland Harbor.

PAHs are present at a wide range of concentrations throughout the Study Area, in all media (Panels 10.2-4A–C). For PAHs, concentrations greater than 10,000 µg/kg in sediment, 50,000 pg/L in surface water, 100 µg/L in TZW, and 100 µg/kg in biota are considered elevated (the orange or red sediment symbols and polygons on Panels 10.2.4A–C). The PAH-related risks to humans and ecological receptors were evaluated in the BHHRA (Section 8 and Appendix F) and BERA (Section 9 and Appendix G), and the overall risks are briefly summarized above in Section 10.2.4.

10.2.4.1.1 Sediment

On a harbor-wide basis, elevated PAH concentrations in sediments generally occur downstream of RM 7 in nearshore areas proximal to local upland sources (Maps 5.1-9 and 5.1-10a–m; Panels 10.2-4A–B). Elevated PAH concentrations are also observed in surface and subsurface sediments in the navigation channel between RM 4 and 6.5.

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The SWAC[30,31] for total PAHs in the Study Area is 20,000 µg/kg. For comparison, as presented in Section 7 and summarized in Table 7.3-5b, the 95 UPL of background total PAHs in the upriver reach of the LWR (RM 15.3 to 28.4) is 187 µg/kg dry weight. An adjusted "OC-equivalent" background 95 UPL of 288 µg/kg dry weight was also estimated for total PAHs to account for the difference in the mean organic carbon content of surface sediments in the upriver reach (1.11 percent TOC), which is lower than the Study Area organic carbon content (1.71 percent TOC; see Table 7.3-6b). As shown in Figure 5.1-13 and on Panel 10.2-4A, there are seven areas with clusters of total PAH concentrations above 10,000 µg/kg in the eastern nearshore zone, one large area in the western nearshore zone and navigation channel, and three smaller areas in the western nearshore zone. The largest area of elevated PAHs in sediments occurs from RM 5.1 to 6.9W and extends into the navigation channel. This area is offshore of Siltronic, Gasco, Marine Finance, and Foss Brix. Other areas of elevated total PAH concentrations in surface sediments include Mar Com South (RM 5.5–5.6E), Terminal 4 Slip 3 and Wheeler Bay (RM 4.3–4.6E), Slip 1 (RM 4.3E), and the International Slip (RM 3.7–3.8E).

Total PAH concentrations are generally higher in subsurface sediments within the Study Area as a whole (Panel 10.2-4A–B; Maps 5.1-10a-m; Figure 5.1-45), pointing to higher historical inputs to the Study Area. The most notable exception to this pattern is the navigation channel at RM 5 to 6.5 where the surface sediments show elevated levels (Map 5.1-9) and are often elevated relative to deeper samples collected in that reach (Maps 5.1-10f-g), suggesting possible downstream transport of PAHs in surface sediments once they reach the channel from adjacent or upstream nearshore zones. This is consistent with the relative dynamic channel environment found from RM 5 to 7 (see Section 3.1.5). Other exceptions to the general pattern of higher subsurface total PAHs include Swan Island Lagoon and Multnomah Channel, where the upper portion of the sediment column (not just the surface layer) often shows higher PAH levels than the deeper layers.

The PAH composition of sediments in the Study Area are generally dominated by four-ring and five-ring PAHs (Figures 5.1-46a–j and 5.1-47a–f), with localized exceptions. Surface sediments from the western nearshore zone generally exhibited higher proportions of two-ring and three-ring PAHs than sediments from the eastern nearshore zone and the navigation channel. Subsurface sediments throughout the Study Area have greater contributions from two- and three-ring PAHs than the surface sediments, but generally have a PAH composition similar to the surface sediments. Some areas of high total PAH concentration have PAH composition that differs from the prevailing trend,

[30] The SWAC concentrations for the CSM chemicals were calculated using the natural neighbor interpolation method with 10-ft cells and non-detected results included at the detection limit.

[31] The Study Area SWAC reported in this paragraph is based on the original RI data set (to June 2008) and was not recalculated to include the more recent Study Area data reported in Appendix H. Given the limited amount of more recent data relative to the size of the original RI data set, the Study Area SWAC is unlikely to have changed markedly. The upriver reach data set did not change between the original RI and the updated RI data sets.

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most noticeably between RM 6.3 and 6.8; however, in general, little variation in PAH composition was seen between samples of differing total PAH concentrations.

10.2.4.1.2 Surface Water and Suspended Sediments

The sediment trap and high-flow surface water particulate samples had PAH compositions that were generally similar to that of the sediments, perhaps suggesting increased bedded sediment resuspension during higher flow periods, whereas the PAH composition of the low-flow and stormwater-influenced surface water particulate samples generally did not, suggesting that current, lateral and upstream sources differ somewhat compositionally from the PAHs in the bedded sediments. Surface water total PAH concentrations are elevated predominately during low-flow conditions (Figure 5.3-77). These observations further illustrate the localized nature of the PAH sources in the Study Area. The PAH surface water signal appears to be diluted during higher flow conditions. PAH concentrations in sediment trap samples are notably elevated at RM 6W in all seasons sampled (Figure 5.2-19). Less distinct spikes were measured from May to August in Swan Island Lagoon, and from August to November in Multnomah Channel.

10.2.4.1.3 TZW

Relatively high concentrations of PAHs are also found in TZW in areas that correspond with elevated areas of sediment concentrations. Of the sites sampled, total PAH concentrations were found to be highest offshore of the Gasco and Siltronic sites. Total HPAH, total cPAH, and BAP results showed similar distribution. For total PAHs, total HPAHs, total cPAHs, and BAP, the unfiltered deeper (90 to 150 cm bml) Trident samples consistently showed higher concentrations than corresponding unfiltered shallow (0 to 38 cm bml) samples. Review of the fractional composition of the 17 individual chemicals that comprise total PAHs shows a clear pattern of high naphthalene concentrations associated with high total PAH concentrations offshore of the Gasco and Siltronic sites.

10.2.4.1.4 Biota

Similar to TZW, relatively high concentrations of PAHs are also found in some biota samples in areas that correspond with elevated areas of sediment concentrations. PAHs are generally metabolized in vertebrates and do not bioaccumulate. However, they do bioaccumulate in invertebrates. PAHs were detected in all clam tissue samples collected from the Study Area or exposed to sediment from the Study Area. Clam tissue from the Study Area generally contained relatively higher proportions of phenanthrene, fluoranthene, pyrene, and chrysene, with lesser proportions of benzo(a)anthracene.

10.2.4.2 Potential PAH Sources and Pathways

This section provides a summary of the various historical and current PAH sources to the Study Area, based on the types of industries that have existed in Portland Harbor that are known to have generated PAHs, and the various pathways by which PAHs can enter the Study Area.

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Potential upland and overwater sources and identified known and likely complete migration pathways are identified on Table 10.2-5 and Panels 10.2-4A–C. These sources and pathways, identified per the process described in Section 4.2, focus on ECSI sites and are based on a review of information in the associated DEQ ECSI files and other readily available site information, including, in the case of LWG-member sites, information provided by the site owner.

10.2.4.2.1 Chemical Usage

In Portland Harbor, PAHs are associated with bulk fuel storage (Map 3.2-9), historical MGP (Map 3.2-7), and materials used in many manufacturing and waste handling processes. Some of these processes include ship and automobile scrapping, asphalt roof manufacturing, use of lubricants, hydraulic, and fuel oils in a large number of industrial and commercial operations, and leakage from vehicles and machinery.

As shown in Table 10.2-5 and Panels 10.2-4A–C, historical known complete or likely complete pathways for PAHs have been identified at 56 sites. Of these, the stormwater pathway has been identified at 48 sites, groundwater at 11 sites, overwater at 21 sites, overland transport at 10 sites, and riverbank erosion at 14 sites.

Current known complete or likely complete pathways for PAHs have been identified at 30 sites. Of these, the stormwater pathway has been identified at 16 sites, groundwater at 9 sites, overwater at 14 sites, overland transport at 3, and riverbank erosion at 4 sites.

10.2.4.2.2 Stormwater/Wastewater/Overland Transport

Stormwater is identified as a known pathway for PAHs to the Study Area at several areas with elevated PAHs in surface sediment. Data gathered in Round 3A and 3B stormwater sampling in the harbor detected PAHs in stormwater from each type of land use area sampled—heavy industrial, light industrial, major transportation, residential, and parks/open space; however, outfalls associated with the heavy and light industrial and transportation land use categories, as well as individual non-representative outfalls, contributed the majority of the estimated PAH stormwater load (see Sections 6.1.2.3 and 10.1.4.1.2).

The locations of high PAH concentrations in sediments, often in subsurface sediments only (Panel 10.2-4B), coincide with a former MGP site (i.e., Gasco), drydock operations (offshore of Cascade General in Swan Island Lagoon, in the International Slip, and offshore of Mar Com), bulk fuel terminals (ARCO, ExxonMobil, Kinder Morgan), and in Willamette Cove (potentially related to historical releases from M&B). It is likely that stormwater/wastewater/overland transport releases occurred concurrently as a result of historical activities in these locations.

PAHs were also detected at elevated concentrations in sediments near outfalls draining facilities historically engaged in manufacturing operations, vehicle maintenance operations, and petroleum product handling/storage. Examples of outfalls with nearby elevated surface or subsurface sediment concentrations (and identified sources) include:

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WR-123 (Burgard Industrial Park), OF-22C (Siltronic, Gasco), and OF-19 (Greenway Recycling, PGE-Forest Park, Willbridge Terminal, Front Avenue LP, Chevron Asphalt).

Stormwater PAH loads have likely decreased substantially from historical levels since implementation of stormwater regulations and onsite stormwater treatment systems at some sites. Overland transport was likely to have been more important historically, prior to the development of extensive stormwater conveyance systems.

Wastewater discharges are currently regulated primarily through NPDES permits. PAHs are included as monitoring parameters for Kopper's individual NPDES permits (see Table 4.4-6). Wastewater discharges in CSO areas, where wastewater could overflow during CSO events, are regulated through municipal pretreatment permits. PAHs are not identified as a COI at any sites with a pretreatment permit (see Table 4.4-5).

10.2.4.2.3 Overwater Releases

Overwater releases are likely to have occurred in association with overwater operations such as fuel transfers and spills, drydock and berth operations, overwater maintenance operations, vessel servicing and emissions, ship repair and maintenance activities, direct discharges of PAH contaminants (tar, oil) to the river, releases during product loading/unloading at docks, and tug and barge operations. As shown in Tables 4.2-3 and 4.3-4, most of the reported spills from sites and vessels and other spills in the Study Area include oil, diesel, hydraulic oil, and sheen, all of which commonly contain PAHs.

10.2.4.2.4 Groundwater Discharge

As shown in Table 10.2-5, current known and likely complete pathways for groundwater to impact in-river media are present at 9 facilities, most of which were or are involved in bulk fuel storage. Of the six sites sampled for PAHs during the GWPA, complete PAH groundwater pathways were identified at the Siltronic and Gasco sites. Potentially complete PAH groundwater pathways could not be ruled out at the four other sites—Willbridge Terminal, Kinder Morgan Linnton, ARCO, and ExxonMobil. The concentrations of PAHs in TZW offshore of these four sites, however, appear to be controlled by direct chemical partitioning from PAH-contaminated sediment to pore water. This suggests that the role, if any, of groundwater transport of PAHs to the transition zone at these sites is minor and is not significantly influencing TZW and sediment chemistry.

Groundwater discharge is likely to have been a more important source historically and known or likely complete historical pathways have been identified at eight sites.

10.2.4.2.5 Riverbank Erosion

Bank erosion likely played a bigger role historically as well, particularly during construction in places where contaminated sediments or manufacturing material and debris from upland activities were used as fill. Based on limited riverbank sampling,

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riverbank erosion is a historical known or likely complete pathway for PAHs at 13 sites, 4 of which are not current sources. Current and historical known or likely pathways include Crawford, Gasco, Gunderson, Mar Com South Parcel, Marine Finance, Port of Portland Terminal 4, Slip 1 and Slip 3, Premier Edible Oils, Siltronic, Sulzer Bingham, Triangle Park, Willamette Cove, and the Willbridge Terminal facility.

10.2.4.2.6 Atmospheric Deposition

PAHs enter the atmosphere primarily from the incomplete combustion of fossil fuels such as coal, oil and natural gas, and incineration of other organic substances such as wood and garbage. Airborne PAHs in the dry phase (aerosols or gases) are short-lived and are deposited close to emission sources. When carried by rain and snow, PAHs can survive many hours and can travel thousands of miles from distant industrial sources (ATSDR 1997a). Numerous localized sources of PAHs exist in Portland Harbor that formerly or currently use, manufacture, and/or store products and waste materials containing PAHs.

10.2.4.2.7 Upriver Watershed Sources

Table 4.5-1 lists ECSI sites upstream of the Study Area (between RM 11.8 and the Tualatin River) and associated COIs. Information for these sites is from DEQ ECSI database. Environmental investigation or cleanup of sediments or upland areas have been ordered for some of these properties, suggesting that they could be now, or in the past have been, significant sources of PAHs to the Willamette River. PAHs were identified as COIs for 22 sites based on their hazardous substances/waste types, and PAHs were detected in sampled media at 14 sites.

10.2.4.3 Loading, Fate, and Transport of PAHs

This section discusses PAH loading, fate, and transport in the Study Area in a typical year, based on the detailed evaluation of these processes presented in Section 6. Figures 10.2-10a, 10.2-11a, and 10.2-12 summarize the loading, fate, and transport assessment for total PAHs. Figures 10.2-11b and 10.2-11c present relative loads for LPAHs and HPAHs, respectively. Loading estimates are for current conditions and, as discussed above, likely differed considerably in the past. Much of the total PAH mass in the Study Area, especially in deeper sediments, is attributable to historical loading that occurred under different loading conditions and rates.

As shown in Figures 10.2-10a and 10.2-11a, advection through subsurface sediments, upstream surface water, and upland groundwater plumes are estimated to contribute comparable (within an order of magnitude) total PAH loads to the Study Area. Estimated external PAH loads associated with stormwater, atmospheric deposition to the river, and direct discharges from permitted non-stormwater point sources were 1 to 2 orders of magnitude lower than the other external terms. Figures 10.2-11b and 10.2-11c show that LPAHs contribute the bulk of the total PAH load for all loading terms evaluated quantitatively. For upstream surface water, the majority of the total PAH mass load enters the Study Area during high-flow conditions, as compared to low-flow periods (see Figure 6.1-8). Upstream surface water total PAH and LPAH loads are

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associated primarily with the dissolved fraction, whereas HPAH loads are slightly higher in the particulate than in the dissolved fraction. There is an apparent increase in total PAH loads in surface water between RM 11 and the downstream boundary of the Study Area (combined loads at RM 2 and the Multnomah Channel entrance) in both the particulate and dissolved fractions (Figure 10.2-12); this trend was observed under all flow conditions. PAH loading terms that were assessed qualitatively are also shown on Figures 10.2-11a–c. As discussed in Section 6.1.1.2, bedload into and out of the Study Area is qualitatively expected to be low relative to dissolved and particulate surface water loading. PAH volatilization from the water column may be significant for LPAHs, and PAHs are subject to degradation (photodegradation, abiotic, and microbial) processes in the surface water column. In addition, the susceptibility to weathering and degradation is dependent on the type of hydrocarbon source (e.g., a PAH compound bound within a soot particle is much more resistant than in a coating on a particle).

The Study Area-wide, cross-media comparisons of surface sediments, sediment traps, and suspended solids in surface water (Table 10.2-2a-b and Figure 10.2-10b) show that the overall surface sediment concentration is significantly greater than the surface water particulate and sediment trap sample concentrations. Surface water particulate and sediment trap concentrations are not significantly different from each other. The mean PAH concentrations for surface sediments, sediment trap samples, and surface water in these data sets are 1190, 99, and 27 mg/kg, respectively (Table 10.2-2a).

Fate and transport processes internal to the Study Area for total PAHs, LPAHs, and HPAHs are also shown on Figures 10.2-11a, 10.2-11b, and 10.2-11c, respectively. Among these internal terms, the only process for which quantitative estimates were developed in the RI is PAH transport to the water column due to pore water advection through surface sediments, which is estimated to be lower in magnitude than the subsurface advective loading term for total PAHs and LPAHs, and similar in magnitude for HPAHs. As with subsurface sediment advection, advection through surface sediments is subject to a high degree of uncertainty associated with the partitioning behavior of individual PAHs. Other internal fate and transport processes are depicted on Figures 10.2-11a–c on a qualitative basis only. Sediment erosion, deposition, and burial are a function of locally and temporally variable hydrodynamic conditions and the surface sediment mixing rate (driven by both natural and anthropogenic forces). PAH partitioning between suspended sediment and surface water depends on the relative chemical concentrations associated with suspended particulate organic carbon and the dissolved surface water fraction, as well as reaction kinetics. The fate of PAHs within the Study Area is also influenced by both abiotic and microbially facilitated degradation and photolysis. These internal fate and transport processes will be evaluated quantitatively in the fate and transport modeling being conducted as part of the FS.

Estimates of PAH loading from upland soil and riverbank erosion were not assessed in the RI due to a paucity of data for riparian soil PAH concentrations and erosion rates.

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The data shown on Figure 10.2-12 suggest that the total PAH load in surface water increases moving downstream through the Study Area,[32] particularly downstream of RM 7. The apparent increase, which is qualified by uncertainties inherent to the numerical load estimate calculations, may be due to other external loads of PAHs to the Study Area. Most of the load from the two largest external lateral loading terms, subsurface sediment advection and upland groundwater plumes, enters the Study Area between RM 6 and 7. In this reach, the estimated subsurface advective load (478 kg/yr) exceeds the next highest load (between RM 5 and 6) by more than an order of magnitude; the upland groundwater plume load from RM 6 to 7 (165 kg/yr) represents almost the entirety of the PAH load to the Study Area from currently known upland groundwater plumes. Estimated atmospheric deposition loading of PAHs to the water surface is low and varies only as a function of water surface area by river mile, although this load may have been substantially higher historically prior to controls on atmospheric emissions. Accumulations of historical soot emissions are typically persistent in sediments. Estimated stormwater and permitted direct non-stormwater discharge loads are also relatively low across the Study Area. The estimated current internal PAH annual loads from advection through surface sediments generally follow the patterns of surface PAH sediment concentrations (Panel 10.2-4A), with the highest loads between RM 5 and 7.

10.2.4.4 Relationship of Sources to Distribution of PAHs

The concentration of total PAHs in Portland Harbor sediments varies widely from levels that are likely within the range of urban background to very high concentrations in localized areas associated with historical sources. Most of the elevated areas are located downstream of RM 7. The portion of the Study Area upstream of RM 7 (not including Swan Island Lagoon) is characterized by widespread low PAH levels (i.e., 0–500 µg/kg). In contrast, downstream of RM 7 (away from the high concentration areas associated with known sources), and also in Swan Island Lagoon, ambient PAHs levels are typically 10 times higher (1,000–5,000 µg/kg) than the upper Study Area levels. As noted in Section 10.1.2, the elevated PAH areas not currently undergoing early action generally exhibit elevated levels of other chemicals, especially PCBs and dioxins.

Areas of elevated PAH concentrations are well-characterized and associated with known or likely historical and current sources. Two elevated PAH areas, offshore of the Gasco former MGP site at RM 6.5W and the Port's Terminal 4, Slip 3, are the focus of early cleanup actions independent of the harbor-wide RI/FS. Releases associated with Gasco and other known and potential sources along the west side of the river between RM 6 and 7 have resulted in elevated nearshore PAH levels in both surface and subsurface sediments. Other potential sources occur on both sides of the river downstream from this area, and downstream concentration gradients both nearshore and in the navigation channel in this relatively high-energy portion of the river suggest downstream transport of PAHs extending approximately to the Multnomah Channel

[32] At approximately RM 3, the Columbia River and Multnomah Channel hydraulically influence the flow regime complicating interpretation of load conditions in this area (see Section 3.1.3).

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entrance. In the navigation channel from RM 5 to 6, high surface sediment concentrations relative to subsurface levels also suggest transport of material through this reach rather than long-term accumulation. Other, less extensive elevated PAH areas in the Study Area, are associated with known or likely sources; these areas include offshore of Cascade General, in Willamette Cove (subsurface sediments), offshore of Mar Com, at the head of the International Slip, and at several nearshore locations along the west bank from RM 3 to 5.

In general, subsurface PAH levels exceed surface levels, indicating greater inputs or releases historically that have been reduced or eliminated over time. Elevated PAH levels in other media—TZW, surface water, sediment traps, and biota (particularly clams)—are largely restricted to the most extensive area of elevated sediment concentrations, along the western shore around RM 6.

Overall, current PAH inputs to the Study Area are much lower than historical inputs because material handling and waste management practices have greatly improved. The effects of current releases are potentially expressed in surface sediment levels (Panel 10.2-4A). Measured elevated levels of PAHs in surface sediments and other media including surface water (e.g., Swan Island Lagoon, RM 7.4W, RM 6.8W) and biota in specific areas (e.g., International Slip; Terminal 4, Slips 1 and 3; offshore of Gasco; and Swan Island Lagoon) indicate that localized inputs and/or internal mass transfer of sediment PAH inventory from surface sediments by processes such as sediment resuspension likely affect other media. Potentially important current pathways from upland sources include groundwater plumes, overwater releases, overland transport, and riverbank erosion.

PAHs were identified as contaminants potentially posing unacceptable risks for the BHHRA based on fish and shellfish consumption and direct contact exposures. Human health and ecological risks from PAHs are geographically limited in extent, reflecting the limited areas of very high sediment levels, and are concentrated in the RM 4 to 6W area. In the BERA, PAHs are strongly associated with the benthic risk area from RM 5.1 to 6.9 on the west side of the river, where concentrations are elevated in sediment and TZW. Negligible risks to receptors other than the benthic community are expected based on the low magnitude and frequency of exceedances in other areas of the river.

The relationships between tissue body burdens and abiotic concentrations across the Study Area are a primary focus of the fate and transport modeling to be conducted in the FS. Current estimates of external PAH loads (mass/yr) to the Study Area indicate that advection through subsurface sediments, upstream surface water, and upland groundwater plumes contribute comparable total PAH loads to the Study Area, whereas loads associated with stormwater, direct atmospheric deposition to the river, and direct discharges from permitted non-stormwater point sources are considerably less important. These data also suggest that surface water PAH mass loads increase from upstream to downstream, likely reflecting inputs from the other external loading terms,

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which peak at RM 6 to 6.9. Internal mass transfer from bedded surface sediments to surface water/biota from sediment resuspension erosion was not quantified but likely also contributes to this pattern.

10.2.5 Bis(2-ethylhexyl)phthalate

Panels 10.2-5A–C present the Study Area graphical CSM for BEHP per the format described in Section 10.2. Recent sediment studies at RM 11E and the downtown reach are included in Appendix H.

BEHP is a phthalate compound. Phthalates are manufactured, colorless liquids with little or no odor. The primary sources of phthalate emissions are the industries that manufacture it or use it in productions, such as the chemical industry, the plastics industry, the cosmetic industry, machinery manufacturers, and manufacturers of plywood and millwork. Phthalates are commonly added to plastics and paint to make the finished product more flexible (ATSDR 2002c) and are widely used as plasticizers in PVC resin. In addition, phthalates are common components of detergents and carriers in pesticide formulations (Xie et al. 2005).

Releases to the environment can occur as direct spills from industrial facilities that manufacture or use these chemicals. More commonly, releases occur by leaching of low volumes of phthalates from the wide variety of products that contain them (ATSDR 2002c).

Despite its low vapor pressure, BEHP is ubiquitous in the atmosphere due to its widespread use in plastics. BEHP, present in the atmosphere in both the vapor phase and associated with particulates, is subject to both wet (rain and snow) and dry (wind and settling) deposition on the earth's surface (ATSDR 2002c).

The behavior of BEHP in the Study Area environment is largely defined by its high hydrophobicity (the central estimate of log K_{oc} is 7.4). Due to this hydrophobicity, BEHP has a strong tendency to sorb to solids and organic matter in surface water and in sediment-pore water environments. As such, it is not expected to migrate significantly in groundwater. Because of its low vapor pressure, volatilization is a minor loss mechanism for BEHP, particularly when sorbed to solids. BEHP is subject to fairly rapid degradation in the atmosphere, but much slower abiotic and microbially mediated degradation processes under aerobic conditions in sediment and surface water (HSDB 2006; ATSDR 2001).

HSDB (2006) indicates that BEHP would be expected to bioconcentrate in aquatic organisms; however, Study Area biota results suggest that BEHP is readily metabolized. Human exposure to phthalates is through consumption of food and drinking water (ATSDR 2002c).

The BHHRA found that BEHP resulted in cancer risk estimates greater than 1×10^{-4} and HQs greater than 1 for consumption of whole body smallmouth bass at RM 4.

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BEHP resulted in cancer risk estimates greater than 1 x 10^{-6} for consumption of whole-body fillet smallmouth bass and whole body brown bullhead, based on a limited number of detected concentrations. BEHP was not identified as a contaminant potentially posing unacceptable risks for any other exposure scenario.

The BERA found that BEHP poses potentially unacceptable risk based on the benthic invertebrate and fish tissue-residue and surface water LOEs. It is not recommended as a COC for several reasons:

- Low frequency of surface water TRV exceedance (2 of 190 samples [1.1%])
- Low frequency of fish tissue-residue TRV exceedance (1 of 38 sculpin samples [2.6%], 2 of 32 smallmouth bass samples [6.3%]) and low frequency of the benthic invertebrate tissue-residue TRV exceedance (1 of 35 clam samples or 3%)
- Low magnitude of exceedance for fish tissue TRV (maximum HQ = 2.9) and for benthic invertebrate TRV (maximum HQ = 2.8)
- Absence of toxicological effects associated with the tissue TRV (which is based on an unbounded NOAEL)
- Absence of relationship between concentrations in co-located sediment and tissue samples.

10.2.5.1 BEHP Contaminant Distribution

For BEHP, concentrations greater than 3,000 µg/kg in sediment, 100 µg/kg in biota samples, and greater than 0.8 pg/L in surface water samples are considered elevated (the orange or red symbols and polygons on Panels 10.2-5A–C).

Several sediment and surface water samples were reported as non-detects with high detection limits for BEHP; some of these high detection limits are at a level that meets the above definition of elevated (Panels 10.2-5A–B). BEHP is a common field and laboratory contaminant due to its frequent use in plastics and other materials, and contaminated field and laboratory blanks can lead to high detection limits for BEHP when concentrations reported by the laboratory are restated as undetected during data validation (e.g., due to blank contamination). Other reasons for elevated detection limits are related to mass spectrometer identification of BEHP. High detection limits may obscure the presence of BEHP at a concentration above the method detection limit, but a high detection limit does not imply the presence of BEHP.

10.2.5.1.1 Sediment

The Study Area-wide SWAC based on natural neighbor interpolation for BEHP is approximately 325 µg/kg; this compares to upriver surface sediment 95 UPL of

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118 µg/kg (see Table 7.3-5b).[33,34] An adjusted "OC-equivalent" background 95 UPL of 182 µg/kg dry weight (see Table 7.3-6b) was also estimated for BEHP to account for the difference in the mean organic carbon content of surface sediments in the upriver reach (1.11 percent TOC), which is lower than the Study Area organic carbon content (1.71 percent TOC).

On a harbor-wide basis, the elevated BEHP concentrations in sediments occurred, with minor exceptions, in nearshore areas outside the navigation channel and proximal to local upland sources (Maps 5.1-12a–m and Panels 10.2-5A–B). With few exceptions, concentrations were generally less than 100 µg/kg throughout the navigation channel.

BEHP surface sediment concentrations above 3,000 µg/kg are generally limited to widely-spaced isolated samples in the eastern and western nearshore zones, and clustered distributions in Swan Island Lagoon and in International Slip (RM 3.7–3.8E) and along the riverside of Schnitzer/Calbag site (RM 3.8–4.1E). Surface concentrations were also elevated, often by a single sample, within Balch Creek Cove (RM 9.7W), RM 8.8W, RM 8.3W, Willbridge Bulk Terminal (RM 7.6W), RM 7.6E, and offshore of RM 7.1, and RM 10 in the navigation channel. Elevated subsurface concentrations occurred in a subset of areas with elevated surface concentrations, except at RM 10.5W and at RM 5.7W where a single elevated subsurface sample was not associated with an elevated surface concentration. Areas where elevated concentrations occurred in both surface and subsurface sediments were at RM 8.8W and the mouth of Swan Island Lagoon. The highest surface and subsurface BEHP concentrations occurred at RM 8.8W.

10.2.5.1.2 Surface Water and Suspended Sediments

Only one surface water sample collected during high-flow conditions at RM 8.6W showed elevated concentrations of BEHP. However, elevated surface water concentrations were not observed at a station just downstream. BEHP concentrations in sediment trap samples did not vary widely spatially or temporally throughout and upstream of the Study Area, with the exception of notably elevated levels measured in the only two samples (summer, fall) collected in Swan Island Lagoon (see Figure 5.2-22).

10.2.5.1.3 TZW

Because of BEHP's hydrophobic nature, groundwater is unlikely to be a significant historical or current pathway for BEHP migration into the Study Area and was not included as a COI in the TZW sampling program.

[33] The SWAC concentrations for the CSM chemicals were calculated using the natural neighbor interpolation method with 10-ft cells with non-detected results included at the detection limit.

[34] The Study Area SWAC reported in this paragraph is based on the original RI data set (to June 2008) and was not recalculated to include the more recent Study Area data reported in Appendix H. Given the limited amount of more recent data relative to the size of the original RI data set, the Study Area SWAC is unlikely to have changed markedly. The upriver reach data set did not change between the original RI and the updated RI data sets.

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10.2.5.1.4 Biota

BEHP was most frequently detected in laboratory-exposed clams (82.2 percent), followed by laboratory-exposed worms (60 percent), field mussels (57.1 percent), and fish (16 percent). BEHP was not detected in crayfish, juvenile Chinook, or carp.

The highest concentrations (greater than 1,000 µg/kg) in biota were generally detected on both sides of the river near RM 4 and above RM 9.5 and one location at the downstream end of Swan Island. With the exception of the surface sediment on the east bank near RM 4 and subsurface sediment at the downstream end of Swan Island, elevated BEHP concentrations in biota do not correlate well with elevated concentrations in sediment.

10.2.5.2 Potential BEHP Sources and Pathways

This section provides a summary of the various potential historical and current BEHP sources to the Study Area, based on the types of industries that have existed in Portland Harbor that are known to have handled and used BEHP, and summaries of the various pathways (e.g., stormwater/waste water/overland transport, atmospheric deposition) by which BEHP can enter the Study Area.

10.2.5.2.1 Chemical Usage

Within the Study Area, several historical and current sources contributed BEHP to the river primarily through the stormwater pathway. Elevated BEHP concentrations in sediment are primarily found in Swan Island Lagoon, International Slip, and along the riverside of Schnitzer/Calbag, where ship construction, repair, painting, and maintenance operations have been ongoing since the 1940s, as well as other manufacturing sites with stormwater drainage to the lagoon. BEHP concentrations also correspond to locations in the harbor where there are equipment manufacturing, marine salvage, and scrap metal recycling operations.

Because BEHP is associated with plastic products and uses, and commonly used in paint as a plasticizer, this chemical is likely found in many of the industrial sectors discussed in Section 3, primarily shipbuilding (Map 3.2-3) and wood products (e.g., plywood production; Map 3.2-4).

As shown on Table 10.2-6 and Panels 10.2-5A–C, historical known complete or likely complete pathways for BEHP have been identified at 30 sites. Of these, the stormwater pathway is included at 28 sites, overwater releases at three sites, overland runoff at one site, riverbank erosion at two sites, and groundwater at one site.

Current known complete or likely complete pathways for BEHP have been identified at 16 sites. Of these, the groundwater pathway is included at three sites, the stormwater pathway is included at 12 sites, and overwater releases at two sites. No known or likely complete pathways have been identified for overland transport or riverbank erosion.

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10.2.5.2.2 Stormwater/Wastewater/Overland Transport

Numerous upland sites have been identified as being known or likely historical and/or current sources of BEHP. These sites discharge directly to the river or discharge through shared conveyance systems (see Table 10.2-6). These facilities include equipment manufacturing, marine salvage, galvanizing, bulk fuel facilities, and scrap metal recycling that reflect the myriad uses and presence of phthalates in their operations.

BEHP is elevated in surface sediment in current and former shipyards such as Swan Island and International Slip. BEHP is likely to have been released to Swan Island Lagoon and the Portland Shipyard for many years, and continued inputs may occur from known sources and contributions from the numerous outfalls in this area. The lack of elevated levels at depth in the sediment column in the lagoon suggests low burial rates, surface sediment mixing, and/or higher recent inputs. The subsurface maximum levels near the shipyard docks indicate historically high levels and burial over time.

There are no known sources of BEHP associated with the ECSI sites that discharge stormwater to the International Slip. However, this area has an auto shredding facility and BEHP has been identified with metal scrapping (see Section 3.2.1.4). It was also the former location of a large shipyard owned by the Oregon Shipbuilding Corporation.

Other areas identified with more than a single data point were located at RM 8.3W and RM 8.8W. Several metals facilities (either fabrication or scrapping) and a historical paint spill area have been identified as BEHP sources that discharge to OF-19 at RM 8.3W. Identified sources of BEHP at RM 8.8W include Gunderson (outfalls from the Marine Barge Paint and Blast Area) and multiple facilities discharging to OF-18.

Wastewater discharges are currently regulated primarily through NPDES permits; BEHP is not a monitoring parameter for individual or general NPDES permits (see Tables 4.4-6 and 4.4-7). Wastewater discharges in CSO areas, where wastewater could overflow during CSO events, are regulated through municipal pretreatment permits. BEHP is not identified as a COI at any sites with a pretreatment permit (see Table 4.4-5). Historical wastewater discharges associated with ship building and decommissioning, metals recycling and scrapping, electrical component manufacturing, and leaks and spills from equipment that used BEHP are possible, but have not been specifically identified or quantified. Overland transport sampling is very limited in the Study Area and of the sites that have been sampled, BEHP migration has been identified at only one site, Mar Com South Parcel, as a historical likely complete pathway.

10.2.5.2.3 Overwater Discharge

Current overwater releases may be locally important at sites with continuous waste handling or operational activities, but are considered a minor current pathway overall. Except for Cascade General and Mar Com South, no current known or likely complete BEHP overwater pathways have been identified. These properties have hosted

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extensive shipbuilding and repair operations where phthalates were possibly used in wood finishes, paints, adhesives, plastic plumbing pipes, lubricants, solvents, and other building materials during their tenures.

10.2.5.2.4 Groundwater Discharge

BEHP was not included as a COI in the TZW sampling program because of its affinity to sorb onto solids and organic matter in water. BEHP has been identified as having a current or historical complete pathway at Premier Edible Oils, Triangle Park, and the Willbridge Terminal facility.

10.2.5.2.5 Riverbank Erosion

Riverbank sampling data within the Study Area are very limited, and there are no current known or likely complete riverbank erosion pathways identified from the collected data. Crawford Street and Mar Com South both have a likely complete historical pathway. However, elevated sediment concentrations are not present adjacent to these sites.

10.2.5.2.6 Atmospheric Deposition

BEHP is ubiquitous in the atmosphere due to its widespread use in plastics. The primary sources of phthalate emissions to the air are the industries that manufacture or use them in the production of chemicals and plastics. In the atmosphere, phthalates occur in both gas and particulate phases, with deposition occurring primarily through rainfall. There are no known atmospheric sources of BEHP within the Study Area.

10.2.5.2.7 Upriver Watershed Sources

Table 4.5-1 lists upstream ECSI sites (between RM 11.8 and the Tualatin River) and associated COIs. Information for these sites is from DEQ's ECSI database. Potential pathways are identified if enough information was available in the database. BEHP was identified as a COI for only one of the 69 sites listed on Table 4.5-1 (Winkler Scrap Metal – ECSI #986), based on its hazardous substances/waste types. However, no upland sampling has been performed at this site. Appendix H summarizes the recent data collected in the RM 11E and downtown reaches.

10.2.5.3 Loading, Fate, and Transport of BEHP

Figures 10.2-13a through 10.2-15 summarize the BEHP loading, fate, and transport assessment for the Study Area in a typical year. These calculations indicate that the highest relative current external inputs are from upriver surface water,[35] with relatively minor additional contributions from Study Area stormwater and advection through subsurface sediments. The total BEHP loads in upstream surface water (composite of RM 11 and 16) and the downstream boundary of the Study Area (combined loads at RM 2 and the Multnomah Channel entrance) are generally comparable. Although the current upstream surface water load estimate exceeds the other loading terms, there is

[35] The surface water data set only has total concentrations for BEHP; therefore, Figure 10.2-13a presents only the total estimated surface water loading rates for this chemical.

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no indication that the surface water load is responsible for spatial distribution of the BEHP observed in Study Area sediments, and much of the surface water load appears to pass through the Site. As noted above, the off-channel and nearshore elevated BEHP concentrations appear to be associated with localized upland sources and pathways. Additional assessment of BEHP loading, fate, and transport within the Study Area will be undertaken in the FS.

A cross-media comparison and statistical assessment of surface sediment, sediment trap samples, and particulate suspended solids on a Study Area-wide basis (Tables 10.2-2a-b and Figure 10.2-13b) show that the mean BEHP concentrations differ significantly across all media, with the surface sediment showing the highest average concentrations, followed by surface water particulates and then sediment trap samples.[36]

10.2.5.4 Relationship of Sources to Distribution of BEHP

The larger areas of elevated BEHP concentrations are located in off-channel areas (Swan Island Lagoon, International Slip, and the riverside along Schnitzer/Calbag). Sediment trap samples were elevated in Swan Island Lagoon. Other areas of elevated sediment concentrations are limited to a few, typically small, widely scattered nearshore areas. With the exception of sediments near the Cascade General shipyard, there are generally more elevated samples in the surface than subsurface sediments, suggesting recent or current sources exist and that historical inputs were not generally elevated relative to current levels. Clams, crayfish, and sculpin tissue BEHP levels are generally not elevated in the Study Area; smallmouth bass samples were elevated at selected areas, including RM 9.5 to 11.5 and near RM 4.

Within the Study Area, several historical and current sources contributed BEHP to the river primarily through the stormwater pathway. Elevated BEHP concentrations in sediment are primarily found in Swan Island Lagoon, where ship construction, repair, painting, and maintenance operations have been ongoing since the 1940s, as well as other manufacturing sites with stormwater drainage to the lagoon. BEHP concentrations also correspond to locations in the harbor where there are equipment manufacturing, marine salvage, and scrap metal recycling operations.

The BHHRA found that BEHP resulted in cancer risk estimates greater than 1×10^{-4} and HQs greater than 1 for consumption of whole-body smallmouth bass at RM 4. BEHP resulted in cancer risk estimates greater than 1×10^{-6} for consumption of whole-body and fillet smallmouth bass and whole-body brown bullhead. In the BERA, BEHP was identified as posing potentially unacceptable risk to benthic invertebrates, fish, amphibians, and aquatic plants based on the surface water LOE; and to benthic invertebrates and fish based on the tissue residue LOE. Negligible risks are expected based on the low magnitude and frequency of exceedances.

[36] BEHP had a detection frequency of only 13% in surface water particulate samples. Non-detected values are included in the data summaries and box-whisker plots using a substitution of one-half the detection limit to allow comparisons across media.

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The loading calculations indicate that the highest relative current external inputs are from upriver surface water,[37] with relatively minor additional contributions from Study Area stormwater and advection through subsurface sediments.

10.2.6 Total Chlordanes

Panels 10.2-6A–C present the Study Area graphical CSM for total chlordanes per the format described in Section 10.2 and support the discussions in the following subsections. Recent sediment studies at RM 11E and the downtown reach are summarized in Appendix H.

Chlordane is a manufactured chemical that was used as a pesticide on crops, including corn and citrus, and on home lawns and gardens in the U.S. from 1948 to 1988. It was also used from the 1950s to the 1980s to prevent or eliminate termites. Some of its trade names are Octachlor and Velsicol 1068. Chlordane does not occur naturally in the environment. The EPA banned all uses of chlordane, with the exception of termite control, in 1983; all uses of chlordane were banned in 1988 (ATSDR 1995). The use of chlordane is prohibited or restricted in most countries. In Canada, since 1985 the registration of chlordane is limited to use as a restricted class termiticide.

Like the other pesticides on the CSM chemical list, chlordanes are highly hydrophobic. As such, they have a strong tendency to sorb to solids and organic matter in surface water and sediment. Chlordanes are persistent in sediments, and subject to very slow abiotic degradation processes. The dissolved fraction in surface water is subject to volatilization. Chlordanes can bioaccumulate in the tissues of fish, birds, and mammals (ATSDR 1995).

In the BHHRA, total chlordanes was identified as a contaminants potentially posing unacceptable risks for fish consumption based on cancer risk estimates above 1×10^{-6} but did not result in cancer risk estimates greater than 1×10^{-4}. Total chlordanes was not a contaminant potentially posing unacceptable risks for any other human health exposure scenarios. The BERA did not identify total chlordanes as a contaminant potentially posing unacceptable risks. Although total chlordanes appeared to help explain some of the toxicity to benthic organisms, the floating percentile model was unable to identify a threshold that described the onset of toxicity, so no SQG was derived. Chlordane constituents were included in the logistic regression model, but samples predicted to be toxic overlapped extensively with other chemicals (e.g., other pesticides) or were spatially isolated (single prediction of toxicity surrounded by non-toxic samples). Chlordanes did not rise to the level of a contaminant posing potentially unacceptable risk for any other ecological receptor.

[37] The surface water data set only has total concentrations for BEHP; therefore, Figure 10.2-13a presents only the total estimated surface water loading rates for this chemical.

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10.2.6.1 Total Chlordanes Contaminant Distribution

Total chlordanes are the sum of oxychlordane, trans-chlordane, cis-chlordane, trans-nonachlor, and cis-nonachlor. Each of these chemicals is analyzed individually and the concentrations are summed to obtain the total chlordanes value. For total chlordanes, concentrations greater than 10 µg/kg in sediment, 100 µg/kg in biota samples, and greater than 60 pg/L in surface water samples are considered elevated (the red and orange symbols and polygons in Panels 10.2-6A–C).

Several sediment, surface water, and biota samples were reported with high detection limits for total chlordanes, some of which meet or exceed the above definition of elevated (Panels 10.2-6A–C). These high detection limits are the result of chromatographic interferences, laboratory blank contamination, mass spectrometer details related to identification of the components of total chlordanes, or limited sample sizes. High detection limits may obscure the presence of total chlordanes at a concentration below the elevated detection limit, but a high detection limit does not imply the presence of the chemical.

10.2.6.1.1 Sediment

On a harbor-wide basis, the elevated detected total chlordane concentrations in sediments are restricted to small, widely scattered nearshore or off-channel areas, proximal to local upland sources (Maps 5.1-13 and 5.1-14a–m and Panels 10.2-6A–B). With few exceptions, concentrations were generally less than 10 µg/kg throughout the navigation channel.

The Study Area-wide SWAC[38, 39] based on natural neighbor interpolation for total chlordanes is 3.11 µg/kg; this compares to an upriver surface sediment 95 UPL of 0.69 µg/kg (Table 7.3-5b). An adjusted "OC-equivalent" background 95 UPL of 1.08 µg/kg dry weight was also estimated for total chlordanes to account for the difference in the mean organic carbon content of surface sediments in the upriver reach (1.11 percent TOC), which is lower than the Study Area organic carbon content (1.71 percent TOC; see Table 7.3-6b). Subsurface concentrations tend to be higher than surface concentrations, suggesting that historical inputs were larger than current inputs.

Western nearshore areas with surface and subsurface sediment concentrations greater than 10 µg/kg include RM 8.8, 8.3, 6.9–7.5, and 6.1–6.7. RI and post-RI (Maps H3.1-14a-b) data show some elevated nearshore concentrations in surface and subsurface samples at RM 11.3E. At RM 8.6W only a single surface sediment sample

[38] The Study Area SWAC reported in this paragraph is based on the original RI data set (to June 2008) and was not recalculated to include the more recent Study Area data reported in Appendix H. Given the limited amount of more recent data relative to the size of the original RI data set, the Study Area SWAC is unlikely to have changed markedly. The upriver reach data set did not change between the original RI and the updated RI data sets.

[39] The SWAC concentrations for the CSM chemicals were calculated using the natural neighbor interpolation method with 10-ft cells and non-detected results included at the detection limit.

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was elevated. Elevated surface and subsurface concentrations are also found in Swan Island Lagoon, RM 5.6, and International Slip along the eastern nearshore.

The maximum surface sediment chlordane concentration was found at RM 7.3W. The surface and subsurface sediments with elevated concentrations in this area are interspersed with samples without elevated concentrations and undetected samples with high detection limits. Subsurface sediments with elevated concentrations are present below the B-horizon interval.

Another cluster of concentrations greater than 10 µg/kg in surface and subsurface samples occurred at RM 8.8W and at the head of the International Terminals Slip (RM 3.7E; some areas have subsequently been dredged). A sample collected at RM 8.8W contained the highest subsurface concentration of total chlordanes in the Study Area.

10.2.6.1.2 Surface Water and Suspended Sediments

Total chlordanes were detected in 85 percent of particulate samples (XAD filters), 100 percent of dissolved samples (XAD columns), and 34 percent of peristaltic samples. Surface water samples from stations at RM 7.0 and 6.9 exhibited elevated concentrations under low-flow and high-flow conditions. Elevated surface water concentrations are defined here as exceeding 60 pg/L (orange and red colors on Panel 10.2-6A). Maximum concentrations in surface water transects were elevated at RM 4 and 6.3, as well as RM 11 and 16. The highest surface water concentrations are peristaltic samples collected at RM 6.8W.

Sediment trap samples exhibited either low levels of detected total chlordanes (<4 µg/kg) or non-detects. There was no strong temporal or spatial patterns in the measured concentrations (see Figure 5.2-23). The three highest values (>2–4 µg/kg) were measured at different seasons, at RM 3.5E, in Swan Island Lagoon, and at RM 11.5W.

10.2.6.1.3 TZW

Because of total chlordanes' hydrophobic nature, groundwater is unlikely to be a significant historical or current pathway for total chlordane migration into the Study Area, and total chlordane was not included as a COI in the TZW sampling program.

10.2.6.1.4 Biota

Chlordanes were detected at low levels with varying frequency in all fish and invertebrate samples. The frequency of detection ranged from 25 percent (black crappie fillets) to 100 percent (whole-body black crappie and sturgeon). The maximum detected concentration in any fillet sample occurred in carp from fishing zone from RM 4 to 8. For invertebrates, the highest concentrations were measured in field-collected samples exposed in the laboratory to sediments from RM 8.8W. Elevated concentrations in biota are defined on Panel 10.2-6C as exceeding 100 µg/kg; no sample locations exceeded this value.

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10.2.6.2 Potential Sources and Pathways of Total Chlordanes

This section summarizes the various historical and current chlordane sources to the Study Area based on the types of industries that have existed in Portland Harbor that are known to have handled and used chlordanes.

10.2.6.2.1 Chemical Usage

Within the Study Area, the only source of chlordanes currently identified is the former Rhone Poulenc pesticide manufacturing facility that historically discharged manufacturing waste and stormwater to the river at approximately RM 6.9, and site groundwater infiltrates to City of Portland outfall OF-22B. The Rhone Poulenc facility manufactured herbicides and formulated organochlorine insecticides during its operational years of 1943 to 1990. Insecticide formulation ceased in 1969.

The known sources of total chlordanes to the Study Area are summarized in Table 10.2-7 and Panels 10.2-6A–C and are limited to current and/or historically complete pathways for stormwater at one site. However, based on the distribution of chlordane in nearshore sediments, other sources must also be present. Its widespread use in termite control suggests that chlordane may have been used as an insecticide at other facilities in the harbor.

10.2.6.2.2 Stormwater/Wastewater/Overland Transport

Historical known pathways for stormwater exist at the Rhone Poulenc facility, including infiltration of contaminated groundwater into the storm system and the City of Portland outfalls, specifically OF-22B (RM 6.9W). Total chlordanes have been detected in upland soils at Rhone Poulenc, and stormwater is a likely complete historical pathway. Historically, manufacturing wastes from Rhone Poulenc were discharged to Doane Lake, which occasionally discharged to the river via a historic drainage ditch. This ditch entered the river near RM 6.9, an area with elevated surface and subsurface sediment total chlordanes concentrations. Currently, site stormwater is discharged though WR-6, also located at RM 6.9W.

No current known or likely complete overland transport pathways for total chlordanes have been identified.

Wastewater discharges are currently regulated primarily through NPDES permits; Chlordanes are not identified as a monitoring parameter for individual or general NPDES permits (see Tables 4.4-6 and 4.4-7). Wastewater discharges in CSO areas, where wastewater could overflow during CSO events, are regulated through municipal pretreatment permits. Chlordanes are not identified as a COI at any sites with a pretreatment permit (see Table 4.4-5).

10.2.6.2.3 Overwater Discharge

No current known or likely complete overwater pathways for total chlordanes have been identified.

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10.2.6.2.4 Groundwater Discharge

Due to chlordane's hydrophobic nature, groundwater is not expected to be a significant historical or current pathway for chlordane migration to the Study Area. No current or historical groundwater pathways for total chlordanes have been identified in Portland Harbor.

10.2.6.2.5 Riverbank Erosion

Sampling of the riverbank erosion pathway for total chlordanes is very limited, and no known historical or current pathways have been identified.

10.2.6.2.6 Atmospheric Deposition

Chlordanes are present in the atmosphere, which is probably due to volatilization from soils and water as well as wind erosion. Chlordanes may be transported long distances and deposited by wet or dry deposition. Like PCBs, atmospheric deposition of chlordanes was probably more significant historically prior to the banning of chlordane use in the 1980s in most countries.

10.2.6.2.7 Upriver Watershed Sources

Table 4.5-1 lists upstream sites (between RM 11.8 and the confluence of the Tualatin River at RM 29) and associated COIs. Information for these sites was obtained from the DEQ ECSI database. Potential pathways were identified if sufficient information was available. Total chlordanes was identified as a COI for the PGE Willamette River Sediment Investigation (ECSI #5249) based on PGE's hazardous substances/waste types and the detection of chlordanes in sediment samples. Chlordanes were likely used as a pesticide in the agricultural areas of the Willamette River watershed up until the late 1980s so upriver sources continue to exist.

Appendix H summarizes the recent data collected in the downtown and upriver reaches.

10.2.6.3 Loading, Fate, and Transport of Total Chlordanes

Figures 10.2-16a through 10.2-18 summarize the loading, fate, and transport assessment for total chlordanes in the Study Area in a typical year. These calculations indicate that the current external inputs are dominated by upriver surface water, with relatively minor additional contributions from Study Area stormwater, advection through subsurface sediments, and atmospheric deposition to the river surface. The total chlordane loads in upstream surface water (composite of RM 11 and 16) and the downstream boundary of the Study Area (combined loads at RM 2 and the Multnomah Channel entrance) are comparable in both the dissolved and particulate fractions. Although the current upstream surface water load estimate exceeds the other current loading terms, there is no indication that this load is responsible for distribution of total chlordanes in Study Area sediments.

The Study Area-wide, cross-media comparisons of surface sediments, sediment traps, and suspended solids in surface water (Tables 10.2-2a-b and Figure 10.2-16b) show that the overall surface sediment concentration is significantly greater than the surface water

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particulates (exceeds by more than a factor of 10) and sediment trap samples (exceeds by a factor of about 1.5). Surface water particulate and sediment trap concentrations are not significantly different from each other.

10.2.6.4 Relationship of Sources to Distribution of Total Chlordanes

Chlordanes have been banned for production and use since the 1980s; however, they are persistent compounds in the atmosphere due to widespread use as a pesticide. These compounds have been detected in surface and subsurface sediment, sediment traps, surface water, and fish and invertebrate samples from the Study Area. Total chlordane concentrations across most of the Study Area are comparable to upriver background levels except for a few nearshore areas that are both more quiescent and proximal to known historical sources. The largest areas of elevated total chlordanes include RM 8.8, 6.9–7.5, and 6.1–6.7 along the western shoreline, and Swan Island Lagoon and RM 3.7–3.8 (International Slip) along the eastern nearshore.

Within the Study Area, the only source of chlordanes currently identified is the former Rhone Poulenc pesticide manufacturing facility that historically discharged manufacturing waste and stormwater to the river at approximately RM 6.9. Total chlordanes have been detected in upland soils and groundwater at this site, and stormwater is a complete historical pathway. Infiltration of contaminated groundwater to the storm system draining OF-22B is addressed in Section 10.2.3.2.2. However, based on the distribution of chlordanes in nearshore sediments, other sources must be present.

In the BHHRA, total chlordanes was identified as a contaminant potentially posing unacceptable risks only for the fish consumption scenario. The BERA did not identify chlordanes as a contaminant potentially posing unacceptable risks. Figures 10.2-16a through 10.2-18 summarize the loading, fate, and transport assessment for total chlordanes in the Study Area in a typical year. These calculations indicate that the current external inputs are dominated by upriver surface water, with relatively minor additional contributions from Study Area stormwater, advection through subsurface sediments, and atmospheric deposition to the river surface.

10.2.7 Aldrin and Dieldrin

Panels 10.2-7A–C and 10.2-8A–C present the Study Area graphical CSM for aldrin and dieldrin, respectively, per the format described in Section 10.2. Recent sediment studies at RM 11E and the downtown reach are summarized in Appendix H.

Aldrin and dieldrin are organochlorine insecticides with similar chemical structures, manufactured for agricultural use from the 1950s to 1987. Aldrin and dieldrin are discussed together here because aldrin rapidly breaks down to dieldrin by a variety of degradation mechanisms. From the 1950s until 1970, aldrin and dieldrin were widely used insecticides for crops and livestock. Peak production occurred in the mid-1960s (90% aldrin and 10% dieldrin), and declined thereafter. Aldrin/dieldrin ranked second—after DDT—among agricultural insecticides used in the U.S. in the 1960s

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(Jorgenson 2001). EPA banned the use of aldrin and dieldrin in 1974, except to control termites. By 1987, EPA banned all uses (ATSDR 2002b).

Like the other pesticides on the CSM chemical list, aldrin and dieldrin are highly hydrophobic and have a strong tendency to sorb to solids and organic matter in surface water and sediment. Aldrin is subject to abiotic and microbially mediated degradation processes as well as photolysis (in surface water). These degradation processes for aldrin generally produce dieldrin as a metabolite. Dieldrin is much more recalcitrant, subject to very slow degradation processes in the environment. Both chemicals are subject to fairly slow volatilization (ATSDR 2002b).

Plants can absorb both aldrin and dieldrin from the soil. In animals, dieldrin can bioaccumulate; dieldrin tends to be stored in the fat, leaving the body very slowly. Aldrin does not bioaccumulate; however, in plants and animals it readily degrades to dieldrin (ATSDR 2002b). The BHHRA identified aldrin and dieldrin as contaminants potentially posing unacceptable risks for fish (carp only for aldrin) and clam consumption based on cancer risk estimates above 1×10^{-6}, but they did not result in cancer risk estimates greater than 1×10^{-4}. Neither aldrin nor dieldrin was identified as a contaminant potentially posing unacceptable risks for any other human health exposure scenarios.

Dieldrin was identified in the BERA as potentially contributing to benthic toxicity based on isolated low exceedances of a site-specific SQV.

Aldrin was identified as posing potentially unacceptable risk to spotted sandpiper based on consumption of contaminated prey at beaches between RM 7 and 9, but it was not identified as a COC, based on the low magnitude and frequency of exceedances.

10.2.7.1 Aldrin and Dieldrin Contaminant Distribution

For aldrin and dieldrin, concentrations greater than 10 µg/kg in sediment and biota samples are considered elevated. For surface water samples, aldrin concentrations greater than 8 pg/L and dieldrin concentrations greater than 100 pg/L are considered elevated (the red and orange symbols and polygons on Panels 10.2-7A–C, 10.2-8A–C).

Several sediment, surface water, and biota samples were reported with high detection limits for aldrin or dieldrin, and some of these detection limits are at a level that meets the above definition of elevated (Panels 10.2-7A–C, 10.2-8A–C). These high detection limits are the result of chromatographic interferences, laboratory blank contamination, mass spectrometer criteria issues, or limited sample sizes. High detection limits may obscure the presence of aldrin or dieldrin at a concentration below the detection limit, but a high detection limit does not imply the presence of the chemical.

10.2.7.1.1 Sediment

Elevated aldrin and dieldrin detected concentrations are restricted to small, widely scattered nearshore areas. Elevated concentrations were generally not detected in off-

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channel or navigation channel areas, with the exceptions of elevated subsurface detections of dieldrin at RM 11.4E (which includes several samples nearshore and offshore with elevated concentrations) and the head of the International Slip (RM 3.7E). Surface and subsurface sediments show elevated concentrations of aldrin and dieldrin near the western shore at RM 8.8 (Panels 10.2-7A–B, 10.2-8A–B). The number of samples with elevated concentrations of aldrin is greater than that of dieldrin in both surface and subsurface sediments. Another subarea of elevated aldrin and dieldrin sediment concentrations is RM 6.8 to7.5W (primarily aldrin).

Overall, aldrin levels in subsurface sediments are slightly higher than those in surface sediments, while dieldrin levels are generally higher in the surface sediments. Areas of elevated aldrin concentration generally correspond to areas of elevated dieldrin concentration. The Study Area-wide SWACs[40, 41], based on natural neighbor interpolation for aldrin and dieldrin, are 0.90 and 1.11 µg/kg respectively; this compares to upriver surface sediment 95 UPLs of 0.34 and 0.22 µg/kg (see Table 7.3-5b). Adjusted "OC-equivalent" background 95 UPLs of 0.52 and 0.33 µg/kg dry weight for aldrin and dieldrin, respectively, were also estimated to account for the difference in the mean organic carbon content of surface sediments in the upriver reach (1.11 percent TOC), which is lower than the Study Area organic carbon content (1.71 percent TOC; see Table 7.3-6b).

10.2.7.1.2 Surface Water and Suspended Sediments

Aldrin was detected in 69 percent of particulate (XAD filter) and 73 percent of dissolved (XAD column) samples and only 1 percent of the peristaltic samples. The aldrin dissolved concentrations were greater than the particulate values, with a major exception at RM 6.9. Overall, total aldrin concentrations were fairly consistent within the flow conditions, with downstream samples typically somewhat higher than upstream samples (a few exceptions).

Dieldrin was detected in all dissolved (XAD columns) and 70 percent of particulate (XAD filters) surface water samples and 3 percent of peristaltic samples. The dieldrin dissolved concentrations were greater than the particulate values (typically on the order of 90 percent of the total concentration).

Elevated maximum aldrin concentrations in surface water were detected at RM 6.8W, Saltzman Creek, and in Swan Island Lagoon. Maximum concentrations of dieldrin were elevated at all transect locations including RM 16.

[40] The Study Area SWAC reported in this paragraph is based on the original RI data set (to June 2008) and was not recalculated to include the more recent Study Area data reported in Appendix H. Given the limited amount of more recent data relative to the size of the original RI data set, the Study Area SWAC is unlikely to have changed markedly. The upriver reach data set did not change between the original RI and the updated RI data sets.

[41] The SWAC concentrations for the CSM chemicals were calculated using the natural neighbor interpolation method with 10-ft cells and non-detected results included at the detection limit.

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Aldrin and dieldrin were generally undetected in sediments collected in sediment traps (see Figures 5.2-24 and 5.2-25) with only 5 detected values overall for each chemical. All aldrin detects occurred downstream of RM 8, while 3 of the 5 dieldrin detects were upstream of RM 8.

10.2.7.1.3 TZW

Because of the hydrophobic nature of aldrin and dieldrin, these chemicals were not included in the TZW groundwater sampling program.

10.2.7.1.4 Biota

Biota tissue aldrin levels are not elevated (greater than 10 µg/kg) in the Study Area. Biota tissue dieldrin levels in the Study Area are elevated in multiple sculpin samples predominantly along the eastern shore from RM 4.1 to RM 2.

10.2.7.2 Potential Aldrin/Dieldrin Sources and Pathways

This section provides a summary of the various historical and current aldrin and dieldrin sources to the Study Area, based on the types of industries that have existed in Portland Harbor that are known to have handled and used these chemicals, and summaries of the various pathways by which aldrin and dieldrin can enter the Study Area.

10.2.7.2.1 Chemical Usage

Similar to total chlordanes, the only source of aldrin and dieldrin currently identified within the Study Area is the former Rhone Poulenc pesticide manufacturing facility that historically discharged manufacturing waste and stormwater to the river at approximately RM 6.9. Aldrin and dieldrin have been detected in upland soils at this site, and stormwater is a known complete current and historical pathway.

Jorgenson (2001) lists one facility in the Portland Harbor area, Van Waters & Rogers, as a producer and/or distributor of aldrin and dieldrin. Van Waters & Rogers, an ECSI site, was listed as being an active formulator in 1969. No information is available about releases to the environment from this facility.

The known sources of aldrin and dieldrin to the Study Area are summarized in Tables 10.2-8 and 10.2-9 and Panels 10.2-7A–C and 10.2-8A–C and are limited to current and/or historically complete pathways for stormwater at one site. However, based on the distribution of aldrin and dieldrin in nearshore sediments, other sources must be present.

10.2.7.2.2 Stormwater/Wastewater/Overland Transport

Current and historical known complete pathways for aldrin and dieldrin in stormwater exist at the Rhone Poulenc facility, including contaminated groundwater infiltration into the storm system of OF-22B (RM 6.9W) (see Section 10.2.3.2.2). Historically, manufacturing wastes from Rhone Poulenc were routed to Doane Lake, which occasionally discharged to the river via a historic drainage ditch. This ditch entered the

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river near RM 6.9, an area with elevated surface sediment aldrin concentrations. Currently site stormwater is discharged though WR-6, also located at RM 6.9W.

No current known or likely complete overwater pathways for aldrin or dieldrin have been identified.

Wastewater discharges are currently regulated primarily through NPDES permits; Aldrin and dieldrin are not identified as a monitoring parameter for individual or general NPDES permits (see Tables 4.4-6 and 4.4-7). Wastewater discharges in CSO areas, where wastewater could overflow during CSO events, are regulated through municipal pretreatment permits. Aldrin and dieldrin are not identified as a COI at any sites with a pretreatment permit (see Table 4.4-5).

10.2.7.2.3 Overwater Discharge

No current known or likely complete overwater pathways for aldrin and dieldrin have been identified.

10.2.7.2.4 Groundwater Discharge

Because of aldrin and dieldrin's hydrophobic nature, groundwater is not expected to be a significant historical or current pathway for these chemicals to migrate to the Study Area. Groundwater with Rhone Poulenc-related constituents infiltrates the OF-22B storm sewer line and is being addressed as described in Section 10.2.3.2.2.

10.2.7.2.5 Riverbank Erosion

Sampling of the riverbank erosion pathway for aldrin and dieldrin is limited, and no known historical or current pathways have been identified.

10.2.7.2.6 Atmospheric Deposition

Similar to other pesticides, aldrin and dieldrin are persistent chemicals and can travel long distances in the atmosphere because of their association with the evaporation and condensation cycle. Both chemicals can condense into soil, water, aerosols, snow, and ice as the temperature falls.

10.2.7.2.7 Upriver Watershed Sources

No upriver ECSI sources of aldrin and dieldrin have been identified, but because of their long use as agricultural pesticides, these chemicals still persist in the Willamette River watershed. Appendix H summarizes the recent data collected in the downtown and upriver reaches.

10.2.7.3 Loading, Fate, and Transport of Aldrin and Dieldrin

Figures 10.2-19a through 10.2-21 summarize the aldrin loading, fate, and transport assessment for the Study Area in a typical year. Dieldrin loads are presented in Figures 10.2-22a through 10.2-24. For both chemicals, upriver surface water is the largest estimated external loading term, with the dieldrin surface water loading approximately 50 times higher than that of aldrin. Stormwater is the second highest estimated loading

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term, followed by atmospheric deposition to the river surface and advection through subsurface sediments. There is an apparent increase in aldrin loads in surface water between upstream (composite of RM 11 and 16) and the downstream boundary of the Study Area (combined loads at RM 2 and the Multnomah Channel entrance); this may be attributable to some combination of the other quantified and/or unquantified loading, fate, and transport processes in the Study Area (although the apparent increase in surface water loads is also qualified by uncertainties inherent to the numerical loading estimates). Conversely, there is little apparent change in surface water dieldrin loads between upstream (composite of RM 11 and 16) and the downstream boundary of the Study Area (combined loads at RM 2 and the Multnomah Channel entrance). Although the current upstream surface water load estimate exceeds the other loading terms, there is no indication that the surface water load is responsible for the spatial distribution of aldrin and dieldrin in Study Area sediments. As noted above, the nearshore elevated aldrin and dieldrin concentrations appear to be associated with localized upland sources and pathways.

The Study Area-wide, cross-media comparisons of surface sediments, sediment trap samples, and suspended solids in surface water for aldrin and dieldrin are provided in Tables 10.2-2a-b (summary statistics and statistical comparisons) and Figures 10.2-19b and 10.2-22b (box-whisker distribution plots).[42] The OC-normalized aldrin surface sediment and sediment trap concentrations are not significantly different; concentrations in both of these media are greater than the concentration in surface water particles on a Study Area-wide basis. For dieldrin, the Study Area-wide surface sediment concentrations are significantly greater than the concentrations in surface water particulates, but there are no statistically significant differences between sediment trap concentrations and sediments or surface water particulates.

10.2.7.4 Relationship of Sources to Distribution of Aldrin and Dieldrin

Aldrin and dieldrin have been banned for production and use since the late 1980s; however, they are persistent compounds in the atmosphere due to widespread use as a pesticide. These compounds have been detected in surface and subsurface sediment and surface water primarily from RM 6.8W to 7.2W; dieldrin has also been detected in clams and crayfish samples from this area. This area corresponds to the historical discharge point at RM 6.9 for the former Rhone Poulenc pesticide manufacturing facility. Aldrin and dieldrin have been detected in upland soils and groundwater at this site, and stormwater is a complete historical pathway. Groundwater infiltration from this facility to offsite stormwater systems is addressed in Section 10.2.3.2.2. Based on the distribution of these compounds in nearshore sediments, other sources may also be present.

The BHHRA only identified aldrin and dieldrin as contaminants potentially posing unacceptable risks for fish (carp only for aldrin) and clam consumption. Aldrin and

[42] Aldrin and dieldrin had a detection frequency of only 10% in the sediment trap samples. Non-detected values are included in the data summaries and box-whisker plots using a substitution of one-half the detection limit to allow comparisons across media.

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dieldrin were identified in the BERA as posing potentially unacceptable risks to the benthic community and sandpiper (based on dietary exposure). The distribution of these contaminants tended to be patchy, and exceedances of sediment thresholds were primarily associated with the west side of the river, below RM 9.1.

Figures 10.2-19a through 10.2-24 summarize the loading, fate, and transport assessment for aldrin and dieldrin in the Study Area in a typical year. For both chemicals, upriver surface water is the largest estimated external loading term, with the dieldrin surface water loading approximately 50 times higher than that of aldrin. Stormwater is the second highest estimated loading term, followed by atmospheric deposition to the river surface and advection through subsurface sediments.

10.2.8 Arsenic, Copper, and Zinc

Panels 10.2-9A–C, 10.2-10A–C, and 10.2-11A–C present the Study Area graphical CSM for arsenic, copper, and zinc, respectively, per the format described in Section 10.2. All three metals are abundant elements in the earth's crust, and natural releases to environmental media can be significant.

Arsenic is a naturally occurring metal that is found widely in natural minerals, including realgar ($As_4S_{4(s)}$), orpiment ($As_2S_{3(s)}$), and arsenolite (As_2O_3; ATSDR 2005a). Arsenic occurs naturally in soil, water, and air as a result of mineral weathering, leaching, volcanic eruptions, and wind-blown dirt (ATSDR 2005). Anthropogenic activities, including smelting activities, pesticide use, combustion of wood and coal, waste incineration, and the production and use of treated wood products that utilize soluble chromium copper arsenate (CCA), can also release arsenic into the air, soil, water, and sediments.

Arsenic is a redox-sensitive species, existing at the +3 and +5 oxidation states in aqueous environmental conditions. Under oxidizing conditions the As(V) species (H_3AsO_4, $H_2AsO_4^-$, $H_2AsO_4^-$, $HAsO_4^{2-}$, AsO_4^{3-}) predominate, while under reducing conditions the As(III) species (H_3AsO_3, $H_2AsO_3^-$, $HAsO_3^{2-}$, AsO_3^{3-}) predominate (EPRI 1984). Arsenic is generally highly soluble, with few mineral phases exerting controls on aqueous arsenic concentrations under typical environmental conditions. Arsenic sulfide minerals, such as orpiment and realgar, can be formed under reducing and acidic conditions. Although arsenic minerals are generally highly soluble, adsorption reactions to sediment/aquifer mineral grain surfaces frequently limit dissolved arsenic concentrations (Kabata-Pendias and Pendias 1992). Arsenic is particularly strongly adsorbed to iron oxide minerals, with the As(V) species having a greater affinity for the oxide surface than the As(III) species. Because arsenic is frequently present as an anion under typical environmental conditions, its sorption to oxide surfaces is favored at pH <9 (Stumm 1992).

In aquatic environments, bioaccumulation of arsenic occurs primarily in algae and lower invertebrates (ATSDR 2005a). Fish and shellfish can also accumulate arsenic, mainly in the exoskeleton of invertebrates and in the livers of fish. While

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biomagnification in aquatic food chains is not generally considered significant, predatory fish may biomagnify arsenic through the consumption of prey species (especially bottom dwellers) (ATSDR 2005a).

Copper is an abundant metal element in the earth's crust. Natural releases to environmental media can be significant. Mining operations, agriculture, wastewater sludge, municipal and industrial solid waste, and other industrial processes can also result in environmental releases of copper (ATSDR 2004).

Copper exists in four oxidation states: Cu^{0}, Cu^{+1}, Cu^{+2}, and Cu^{+3} (Eisler 1998). Of these oxidation states, the cupric ion (Cu^{+2}) is the most likely to be present in water, though rarely as a free ion. As free ions, cupric ions are the most readily available and toxic inorganic species of copper. However, the cupric ions have a strong tendency to complex or sorb to numerous compounds normally found in natural waters, including suspended solids surfaces and dissolved or particulate organic carbon. Such complexation reduces bioavailability to aquatic organisms (Eisler 1998; EPA 2000a). The amounts of the various copper compounds and complexes present in solution in freshwater depend on water pH, temperature, hardness, and alkalinity; concentrations of bicarbonate, sulfide, and organic ligands; size and density of suspended materials; and rates of coagulation and sedimentation of particulates. Up to 29 different species of copper can be present in aqueous solution in the pH range from 6 to 9. The majority of copper in freshwater from pH 6.0 to 9.3 is in the form of carbonate species ($CuHCO_3^{+}$, $CuCO_3$, $Cu[CO3]_2^{-2}$), which have low toxicity (Eisler 1998). Cupric ions account for less than 1 percent of the total dissolved copper in freshwater. Copper carbonate, cupric hydroxide, cupric oxide, and cupric sulfide will precipitate from solution or form colloidal suspensions when excess cupric ions are present (Eisler 1998). The majority of copper released to surface waters settles out or sorbs to sediments (Eisler 1998). While copper can transform in response to environmental chemistry, it does not degrade.

Copper is taken up by aquatic organisms primarily through dietary exposure and is an essential micronutrient for animals as a component of a number of essential enzymes. Most organisms retain only a small proportion of the copper ingested with their diet. Copper bioconcentrates in aquatic organisms but does not bioaccumulate in mammals or biomagnify in aquatic food chains (EPA 2000a).

Zinc is a common element in the earth's crust and is released to the environment from both natural and anthropogenic sources. Mining and metallurgical processing are the primary anthropogenic sources, along with use of commercial products such as fertilizers and wood preservatives that contain zinc (ATSDR 1997b). Zinc is also used in galvanizing steel and in soldering formulas.

In the environment, zinc occurs as a sulfide, oxide, or carbonate. In freshwater, zinc is most soluble at low pH and low alkalinity: 10 mg Zn/L of solution at pH 6 that declines to 6.5 mg Zn/L at pH 7, 0.65 mg Zn/L at pH 8, and 0.01 mg Zn/L at pH 9 (Eisler 1993).

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Zinc in the water column[43] can partition to dissolved and particulate organic carbon. Water hardness (i.e., calcium concentration), pH, and metal speciation are important factors in controlling the water column concentrations of zinc because the divalent zinc ion is believed to be responsible for observed biological effects (EPA 2000a). Because zinc ligands are soluble in neutral and acidic solutions, zinc is readily transported in most natural waters (Eisler 1993); however, most of the zinc introduced into aquatic environments eventually is partitioned into the sediments (Eisler 1993). Zinc release from sediments is enhanced under conditions of high dissolved oxygen, low salinity, and low pH (Eisler 1993). Zinc may change forms in the environment, but it does not degrade.

Zinc is an essential trace element for all living organisms. As a constituent of more than 200 metalloenzymes and other metabolic compounds, zinc ensures stability of biological molecules such as DNA and of biological structures such as membranes and ribosomes (Eisler 1993). Most studies reviewed contained data that suggest that zinc is not a highly mobile element in aquatic food webs, and there appears to be little evidence to support the general occurrence of biomagnification of zinc within marine or freshwater food webs (EPA 2000a). Bioavailability of zinc in sediments is controlled by the acid-volatile sediment concentration. The arsenic- copper-, zinc-related risks to humans and ecological receptors were evaluated in the BHHRA (Section 8 and Appendix F) and BERA (Section 9 and Appendix G).

In the BHHRA, arsenic resulted in cancer risk estimates for fish consumption by an adult tribal fisher that exceeded a cancer risk of 1×10^{-4}. Arsenic was identified as a contaminant potentially posing unacceptable risks for fish and shellfish consumption, direct exposure to in-water sediment, direct exposure to beach sediment, and hypothetical use of untreated surface water as a domestic water source. Arsenic is a naturally occurring metal and at the DEQ-promulgated regional background concentration of 7 mg/kg, the calculated risk from arsenic would exceed 1×10^{-6} for direct exposure to beach sediment. Copper was not identified as a contaminant potentially posing unacceptable risks for any exposure scenario in the BHHRA. Zinc was identified as a contaminant potentially posing unacceptable risks for fish consumption based on an HQ of 2 for non-tribal child fish consumption, which was based on a single sample of whole-body common carp tissue collected from RM 4 to 8. Zinc was not identified as a contaminant potentially posing unacceptable risks for any other exposure scenario.

Arsenic was identified in the BERA as posing potentially unacceptable risk to benthic invertebrates because laboratory worm tissue samples contained arsenic at concentrations exceeding a tissue residue TRV. All other lines of evidence indicated no risk from arsenic.

[43] In water, the free zinc ion is thought to coordinate with six water molecules to form the octahedral aquo ion $(Zn(H2O)_6)^{2+}$ in the absence of other complexing or adsorbing agents (Eisler 1993). In typical river waters[43], 90% of the zinc is present as aquo ion, and the remainder consists of ZnHCO3+, ZnCO3, and ZnSO4 (Eisler 1993).

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The BERA found that copper poses potentially unacceptable risk based on the fish tissue-residue, fish dietary, sandpiper dietary, and the benthic invertebrate tissue-residue and predicted sediment toxicity LOEs. Copper was not recommended as a fish COC due to weakness of the tissue-residue LOE for inorganic metals, irreproducible toxicological effects associated with the dietary TRV (selected LOAEL could not be replicated in subsequent tests with the same species), elected LOAEL barely above range of nutritional requirements found in the literature for some fish species, discordance of the tissue and dietary LOEs with the stronger water LOE, and similarity of fish tissue concentrations in the Study Area and upriver. Copper was not recommended as a shorebird COC based on unlikely ecological significance of prey organism TRV exceedance, selected TRV is less than the lowest bounded literature-reported NOAEL for birds, and low magnitude of TRV exceedance (maximum HQ = 1.3) considering the likely overestimates of exposure and effects (per the two previous items). Copper was not recommended as a benthic invertebrate COC based on the low magnitude of TRV exceedance (maximum HQ = 2.6) and weakness of the tissue-residue LOE for inorganic metals (invertebrates sequester copper and in the case of crayfish, copper forms the basis of their hemoglobin)..

The BERA found that zinc poses potentially unacceptable risk for fish (sculpin, bass, pikeminnow), amphibians, and aquatic plants based only on the surface water LOE. It poses a potentially unacceptable risk to benthic invertebrates based on the surface water and tissue-residue LOEs. It was not recommended as a COC based on the low frequency of surface water TRV exceedance for all receptors (1 of 167 samples [<1%], with maximum HQ = 1.2), discordance of the stronger surface water LOE with the weaker tissue-residue and dietary LOEs for fish (surface water toxicity data were sufficient to derive ambient water quality criteria; tissue-residue LOE is weak because fish generally can sequester or otherwise bioregulate inorganic metals; the dietary LOE is relatively weak because the TRV is based on only two studies), and relative weakness of the tissue-residue LOE for benthic invertebrates. Zinc was found to pose localized risk to individual Pacific lamprey ammocoetes due to potential exposure to contaminated shallow TZW.

10.2.8.1 Arsenic, Copper, and Zinc Contaminant Distribution

The contaminant distribution of arsenic, copper, and zinc in biotic and abiotic media is summarized below.

10.2.8.1.1 Sediment

Elevated arsenic concentrations in sediments are defined here as exceeding 20 mg/kg (the orange or red sediment polygons on Panels 10.2-9A–B. The Study Area-wide SWAC for arsenic is approximately 4 mg/kg[44,45], which is equal to the upriver reach

[44] The SWAC concentrations for the CSM chemicals were calculated using the natural neighbor interpolation method with 10-ft cells and non-detected results included at the detection limit.
[45] Statistics reported in this paragraph do not include more recent data reported on in Appendix H.

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(RM 15.3 to 28.4) UPL of 3.97 mg/kg (see Table 7.3-5b). Ninety-five percent of the Study Area sediment concentrations were below 10 mg/kg.

Elevated copper concentrations in sediments are defined here as exceeding 120 mg/kg (the orange or red sediment polygons on Panels 10.2-10A–B). The Study Area-wide SWAC for copper is approximately 44 mg/kg, which is near the upriver reach of the LWR UPL of 37 mg/kg. Ninety-five percent of the sediment concentrations were below 56 mg/kg.

Elevated zinc concentrations in sediments are defined here as exceeding 400 mg/kg (the orange or red sediment polygons on Panels 10.2-11A–B). The Study Area-wide SWAC for zinc is approximately 110 mg/kg, which is equal to the upriver reach of the LWR UPL. Ninety-five percent of the sediment concentrations were below 375 mg/kg.

With the exception of a broad area of relatively elevated copper and zinc concentrations in the vicinity of Swan Island Lagoon and zinc at Terminal 4, Slip 3, elevated arsenic, copper, and zinc concentrations in surface and subsurface sediment are generally restricted to small, widely scattered nearshore areas (Panels 10.2-9A–B, 10.2-10A–B), 10.2-11A–B). The similarity of surface and subsurface concentrations in these areas suggests both recent and historical inputs of all three metals.

RI and post-RI data (Appendix H, Maps H3.1-24a-b) show elevated copper concentrations in two surface and two subsurface locations near RM 11.3E. While not collocated, the widely spaced elevated surface samples are near the elevated subsurface samples.

10.2.8.1.2 Surface Water and Suspended Sediments

Elevated arsenic concentrations in surface water are defined here as exceeding 0.45 µg/L (the orange or red symbols on Panel 10.2-9A). Total arsenic single-point and transect sample concentrations ranged up to 0.75 µg/L at Station W001 (RM 2) in July 2005. Total arsenic concentrations were generally consistent across the entire Study Area during the Round 2A and 3A sampling events. Concentrations were generally higher in low-flow sampling events, though stormwater-influenced and high-flow events exceeded the concentration of 0.45 µg/L (Figure 5.3-113). There is generally no relationship evident between elevated surface water and elevated surface sediment concentrations (Panel 10.2-9A). Sediments collected in and upstream of Study Area over the course of a year in sediment traps show little spatial or temporal trends in arsenic concentrations (see Figure 5.2-26).

Elevated copper concentrations in surface water are defined here as exceeding 1.25 µg/L (the orange or red symbols on Panel 10.2-10A). Total copper single-point and transect sample concentrations ranged up 3.68 J µg/L at Station W023 (RM 11) in January 2006. Total copper concentrations were generally consistent across the entire Study Area during the Round 2A and 3A sampling events. Sediment trap samples also showed comparable levels throughout the area sampled over time. The highest

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concentration (94 mg/kg, about double the overall sediment trap sample mean) was measured in a sediment trap sample collected from May to August in Swan Island Lagoon (Figure 5.2-28).

Elevated zinc concentrations in surface water are defined here as exceeding 8 µg/L (the orange or red symbols on Panel 10.2-11A). Total zinc single-point and transect sample concentrations ranged up 57.9 µg/L at Station W022 (RM 11) in November 2004. Total zinc concentrations were generally consistent across the entire Study Area during the Round 2A and 3A sampling events. As with copper, zinc levels in sediment trap samples were generally comparable across all locations and seasons, except for somewhat elevated levels measured in Swan Island Lagoon from May through November, quarters 3 and 4 (see Figure 5.2-29).

10.2.8.1.3 TZW

Elevated arsenic concentrations in TZW are defined here as exceeding 30 µg/L (the orange or red sediment symbols on Panel 10.2-9A). Detected arsenic concentrations in TZW ranged from 0.3 J to 77.3 µg/L, with an average concentration of 11.1 µg/L. Deep (90 to 150 cm bml) TZW samples showed higher concentrations than collocated shallow (0 to 38 cm bml) samples in 17 of 24 collocated sample pairs. The highest TZW arsenic concentrations are located at the west side of the channel at RM 6.2–6.6, and the west bank at RM 7.7. Sediment arsenic concentrations are not elevated in these areas. The tendency of arsenic compounds to associate with sediment particles can be highly variable and is strongly influenced by redox chemistry and microbial processes (see Appendix C2).

Elevated copper concentrations in TZW are defined here as exceeding 60 µg/L (Panel 10.2-10A). Detected copper concentrations ranged from 0.28 to 555 µg/L, with an average concentration of 27.9 µg/L. TZW copper samples did not exhibit consistent trends with depth. The highest (>100 µg/L) copper concentrations were measured offshore of the Gasco and Siltronic sites in areas where elevated surface sediment is not present.

Elevated zinc concentrations in TZW are defined here as exceeding 400 µg/L (Panel 10.2-11A). Detected zinc concentrations ranged from 0.95 to 3,590 µg/L, with an average concentration of 113 µg/L. The highest (>1,000 µg/L) total zinc concentrations were observed offshore of the Gasco and Siltronic sites in unfiltered samples, while the highest (>20 µg/L) filtered zinc sample concentrations were measured offshore of the ARCO and Gasco sites. Elevated zinc in surface sediment is not present in these areas.

10.2.8.1.4 Biota

It is important to note that the metal (As, Cu, Zn, as well as Cr) concentrations for all biota samples are mapped in Panels 10.2-9C through 10.2-12C using the same concentrations bins. This was done because of the very narrow distribution range of values within each metal and to allow visual comparisons to be made across all metals and biota species (e.g., the tissue concentration of zinc is higher than copper which is

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higher than arsenic for most species and samples). However, the narrow tissue data range for metals can preclude meaningful cross-media comparisons to be made, e.g., for zinc all species plot at the same concentration bin.

Arsenic was detected in all fish and invertebrate species and tissues analyzed within the Study Area, with the exception of one crayfish composite sample. The maximum concentration of 3.04 mg/kg occurred in worms exposed in a laboratory to sediment collected from BT019 (RM 7.4E) near an area of elevated surface sediment concentrations. All bass, sculpin, and crayfish and many clam samples exhibit tissue levels between 0.1 and 1.0 mg/kg, and some clam samples show levels between 1.0 an 10.0 mg/kg (Panel 10.2-9C).

Copper was detected in all fish and invertebrate species and tissues analyzed within the Study Area. Biota tissue copper levels in the Study Area are generally higher in invertebrate samples than in fish species (Panel 10.2-10C). The maximum concentration (20.2 mg/kg) was found in laboratory-exposed worms, which were exposed to sediment collected from RM 8.1E, and crayfish from RM 11.3E. Surface sediment concentrations are elevated in both of these areas. The higher tissue copper concentrations generally do not match up well with the elevated sediment concentration locations.

Zinc was also detected in all fish and invertebrate species and tissues analyzed within the Study Area but cannot be spatially distinguished, as all concentrations across all species fall within the same color ramp and range from between 10.8 and 54.0 mg/kg.

10.2.8.2 Potential Sources of Arsenic, Copper, and Zinc

This section summarizes the various potential historical and current arsenic, copper, and zinc sources to the Study Area, based on the types of industries that have existed in Portland Harbor, and the various pathways by which these metals can reach the Study Area.

10.2.8.2.1 Chemical Usage

The areas of elevated sediment concentrations correspond to the locations of some former shipyards, wood treatment facilities, metal recycling operations, pipe manufacturing, metal plating operations, and marine repair facilities. Metals are also associated with some facilities where metal slag and sandblast grit were used as fill.

Arsenic has many industrial uses such as hardening of copper and lead alloys, pigmentation in paints and fireworks, and the manufacture of glass, cloth, and electrical semiconductors. Arsenic is used for the production of pressure-treated wood, which is employed in commercial, industrial and residential applications, including decking, highway noise barriers, landscaping, playground equipment, retaining walls, signposts, landscaping and utility poles, and docks and piers. Arsenic is widely used as a weed killer and it is often sprayed from small aircrafts over fields to destroy any kind of infestation. It is also used as an insecticide as it is effective in treating termite

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infestation. Arsenic is used extensively in the production of agricultural pesticides, which include herbicides, insecticides, desiccants, wood preservatives and feed additives. Arsenic is also used in the tanning industry as a preservative. In Portland Harbor, the primary industries known to have handled, manufactured, or disposed of arsenic include pesticide manufacturing, shipbuilding/demolition and marine repair facilities, metals recycling, battery scrapping, wood treating, and MGP and heavy oils (as a byproduct).

There are many industrial uses of copper, due to its high ductility, malleability, thermal conductivity, and resistance to corrosion. It ranks third after iron and aluminum in the amount of quantities consumed for industrial purposes. It is alloyed with nickel and used in form of cupronickel and monel for shipbuilding. Copper is used as a wood preservative. Some of the main uses of copper are in the construction industry. Copper, which has anti-microbial properties, is also used for plumbing, roofing and cladding. About 65 percent of copper produced is used for electrical applications, such as power generation and transmission of electricity, including use in transformers, motors, bushbars, generators, etc., to provide electricity throughout the country. In electrical equipment, it is used in wiring and contacts for computers, televisions, mobile phones, and in the circuitry. In Portland Harbor, the primary industries known to have handled, manufactured, or disposed of copper include shipbuilding/demolition and marine repair facilities, wood treating, metal recycling, and MGP and heavy oils (as a byproduct).

Zinc also has many commercial and industrial uses. Metallic zinc is used to coat iron and other metals to prevent rust, and it is also used in dry cell batteries. Zinc is mixed with other metals to form alloys such as brass and bronze. Zinc is also combined with other elements such as chlorine, oxygen, and sulfur to form zinc compounds used to make white paints, ceramics, rubber, wood preservatives, dyes, and fertilizers. In Portland Harbor, the primary industries known to have handled, manufactured, or disposed of zinc include the steel industry, shipbuilding/demolition and marine repair facilities, wood treating, metal recycling, battery scrapping, and MGP and heavy oils (as a byproduct).

As noted in the previous sections, areas of elevated sediment concentrations for these metals are typically small in size (often a single sample) and widely scattered in nearshore areas. A review of the panels reveals that a number of sites with known or likely complete pathways are not associated with proximal elevated surface sediment concentrations. The discussion below focuses on sites with known or likely complete pathways near the larger areas with elevated surface sediment concentrations. As such, sites with known or likely complete pathways not adjacent to these larger areas are not discussed, but are included in the tables and panels (Panels 10.2-9A–C, 10.2-10A–C, and 10.2-11 A–C). The larger areas of elevated sediment concentrations include:

- Balch Creek Cove near RM 10.7W (zinc only)
- Nearshore areas adjacent to Gunderson and Front Avenue LP near RM 8.3–9W

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- Swan Island Lagoon and the north end of Swan Island near RM 8–9.2E
- Willamette Cove near RM 6.7E
- The west end of the railroad bridge near RM 6.4E
- The area adjacent to the former Mar Com operations near RM 5.6E
- Terminal 4 near RM 4.5E
- International Slip near RM 3.8E.

As shown in Tables 10.2-10, 10.2-11, and 10.2-12, and Panels 10.2-9A–C, 10.2-10A–C, and 10.2-11A–C, historically known complete or likely complete pathways for arsenic, copper, and zinc have been identified at up to 46 sites. A summary of the number of sites with known and likely complete pathways is summarized as follows:

Known or Likely Complete Pathways – Arsenic, Copper, and Zinc

	Groundwater	Stormwater	Overwater	Overland	Riverbank
Arsenic					
Historical	7	35	1	9	13
Current	6	11	0	5	4
Copper					
Historical	6	42	5	9	10
Current	4	13	4	3	3
Zinc					
Historical	4	44	5	9	12
Current	3	12	4	3	3

10.2.8.2.2 Stormwater/Wastewater/Overland Transport

One or more of these metals has been identified as a stormwater COI at sites that drain to a number of municipal and non-municipal shared conveyance systems. Elevated sediment concentrations are not present near all these outfalls.

Zinc is elevated in sediment in Balch Creek Cove, which is the discharge location for OF-16, OF-17, WR-258, and WR-235. Metals have not been identified as stormwater COIs at WR-258 (drains Fire Station 6) and WR-235 (Port of Portland Terminal 2). GE Decommissioning has a known historical and likely current complete stormwater pathway for zinc. Calbag Nicolai and Galvanizers have likely historical and known current complete stormwater pathways for zinc. All three of these sites are located within the basins of these outfalls.

Arsenic, copper, and zinc are elevated in sediments offshore of the areas adjacent to Gunderson and the small cove adjacent to the Shaver Transportation and Front Ave LP properties. Gunderson has known complete pathways, both historical and current, for stormwater and overland transport, and Front Avenue LP has a historical known complete stormwater pathway for these metals. Two shared conveyance systems also

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drain to this area, OF-18 and OF-19. However, none of these metals are elevated in the cove that OF-18 discharges to and only copper is elevated in the vicinity of the OF-19 discharge. Front Avenue LP has a historical likely complete pathway for copper in stormwater and Chevron has a historical known complete pathway for copper in stormwater. Both of these sites drain at least in part to OF-19 (Table 10.2-11).

Elevated sediment concentrations of copper and zinc are present on the nearshore areas on the west side of Swan Island Lagoon and the north end of Swan Island. Cascade General, Swan Island Upland Facility, and Fred Devine are identified as a having likely historically complete stormwater pathway for copper and zinc. Single samples of elevated concentrations of zinc and copper are also present near OFM-1and of zinc near OFM-2. Freightliner TMP and TMP2 drain to these basins and are identified as a having likely historically complete stormwater pathway for copper and zinc.

Stormwater and groundwater infiltration from sites draining to OF-22B and OF-22C (RM 6.8W) are the likely sources of elevated arsenic concentrations in surface sediment at this location. Stormwater has been identified as a historical and/or current known complete pathway at Rhone Poulenc, Metro Transfer Station, Schnitzer-Doane Lake, Gasco, Siltronic, and Gould, which drained to or have had groundwater infiltration to these outfalls.

TZW sampling conducted just downstream of RM 6.8W resulted in relatively low to moderate concentrations, suggesting that groundwater is not a significant source to sediments in the area. Low subsurface sediment arsenic concentrations may reflect the dynamic setting—arsenic may be transported downstream before there is an opportunity for long-term burial. Downstream surface sediment concentrations are low, suggesting that if this is the case, that the mass of arsenic being discharged is relatively low and is readily dispersed.

Elevated surface sediment and riparian soil/sediment are present in a small area immediately offshore of OF-48 (RM 7.3, east bank, at the property boundary between M&B and Triangle Park). However, outfall stormwater and solids sampling indicates that the outfall is not a source of arsenic (City of Portland 2010). Although M&B has a historical complete pathway and Triangle Park has a likely historical and known current complete arsenic pathway (Panels 10.2-9A–B), sediment samples collected beneath the M&B cap and upstream offshore of Triangle Park (Panels 10.2-9A–B) have low concentrations, and the single subsurface sediment core does not have elevated concentrations. Along with the apparent depositional setting in this area, the distribution of arsenic indicates relatively recent releases.

At the Mar Com North parcel, arsenic has been detected as high as 12.5 mg/kg in surface soils and was also found in former grit piles (180 mg/kg). Grit was also present at the Mar Com South parcel. As a result, overland transport has been identified as a likely complete historical pathway for both parcels, as well as a likely complete current pathway for the North parcel. Elevated surface sediment concentrations are located

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offshore of the Mar Com facilities and an elevated maximum surface water concentration immediately upstream of the two sediment locations appears to support stormwater as a pathway.

Zinc is present at elevated concentrations in surface sediment at the heads of Terminal 4 Slip 1 and 3 and the International Slip. Copper is also elevated the head of the International Slip. Calbag Metals, adjacent to the International Slip, has been identified as having known current and historical complete stormwater pathways for zinc and copper; Terminal 4 has a likely historical complete pathway for zinc.

Data gathered in Round 3A and 3B stormwater sampling in the harbor detected arsenic, copper, and zinc in stormwater from each land use area sampled. Outfalls from the heavy industrial land use category contributed the majority of the estimated stormwater loads with a mean total concentration for arsenic of 2.93 µg/L, for copper of 66.9 µg/L, and for zinc of 547 µg/L (see Table 4.4-2). Non-LWG stormwater sampling resulted in a mean concentration of 2.9 µg/L for arsenic, 72.3 µg/L for copper, and 1,670 µg/L for zinc (see Table 4.4-4).

Major historical wastewater discharges of these metals were associated with ship building, chemical and steel manufacturing, and bulk fuel storage, and historical wastewater discharges at these sites are likely, but have not been specifically identified or quantified. Wastewater discharges are currently regulated primarily through NPDES permits. Copper and zinc are listed as a general NPDES chemical monitoring parameters for the 78 sites with GEN12Z industrial stormwater permits (see Table 4.4-7). Arsenic, copper, and zinc are monitoring parameters at 8–16 of the 21 sites with individual NPDES permittees (see Table 4.4-6). Wastewater discharges in CSO areas, where wastewater could overflow during CSO events, are regulated through municipal pretreatment permits. While metals are identified as COIs at 16 of 33 pretreatment permit sites, none are listed specifically for arsenic, copper, or zinc (see Table 4.4-5).

10.2.8.2.3 Groundwater Discharge

Arsenic, copper and zinc are associated with Marine Barge Paint and Blast Area operations at Gunderson and a known complete pathway, both historical and current, for groundwater has been identified for arsenic at Gunderson.

Arsenic, copper, and zinc are components of MGP wastes historically disposed of on the Gasco and Siltronic properties in association with the historic Gasco facility operations. Groundwater is a known or likely current and historically complete pathway for these metals for the Siltronic and Gasco sites. All three metals were detected at elevated concentrations in deep Trident samples at RM 6.2–6.6, offshore of the Siltronic/Gasco properties. However, none of these metals are present at elevated concentrations in surface or subsurface sediment.

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Operations at Calbag Metals include metal recycling. The site is adjacent to the International Slip, and elevated sediment concentrations for copper and zinc; arsenic and copper have been identified as known historical COIs in groundwater.

10.2.8.2.4 Overwater Discharge

Five sites have been identified as having historical known complete pathways for copper and zinc: Gunderson, Cascade General, Swan Island Upland Facility, M&B, and Schnitzer – Calbag. M&B also has a historical complete pathway for arsenic. All but M&B have known or likely current complete overwater pathways. M&B is the only identified site with a known historical complete pathway for arsenic. Each of these sites has elevated concentrations of these metals in surface sediment except arsenic at Cascade General. Activities at Gunderson and Cascade General include shipbuilding and repair, metal recycling operations are conducted at Schnitzer – Calbag, and M&B was a wood treating facility.

Elevated surface sediment concentrations are located offshore of the Mar Com facilities. No records have been reviewed indicating that overwater operations is a complete pathway; however, the location of these samples immediately offshore of former dock structures suggests that overwater operations played a role in the observed distribution.

Arsenic, copper, and zinc are not listed as materials released from spills associated with ESCI sites (see Table 4.2-3) or from vessels and other spills in the Study Area (see Table 4.3-4). However, many spills are unreported and reported spills may list a material that could include these metals (e.g., oils may contain these metals).

10.2.8.2.5 Riverbank Erosion

Based on limited riverbank sampling, relatively large areas of elevated metals concentrations in surface sediment are associated with sites with known or likely historical complete riverbank erosion pathways for arsenic, copper, and zinc. These sites include Gunderson, Willamette Cove, Mar Com South, and M&B. A current known or likely complete pathway for riverbank erosion is limited to Gunderson. Arsenic is not a known or likely historical or current complete pathway at Mar Com South.

10.2.8.2.6 Atmospheric Deposition

As noted in the introduction to this section, arsenic, copper, and zinc are naturally occurring metals that are found widely in minerals and occur naturally in soil, water, and air as a result of mineral weathering, leaching, volcanic eruptions, and wind-blown dirt. Anthropogenic activities, including smelting activities, pesticide use, combustion of wood and coal, waste incineration, and the production and use of treated wood products that utilize soluble CCA, can also release these metals into the air, soil, water, and sediments. There are no known current or historical point sources of atmospheric deposition of these metals in the Study Area.

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10.2.8.2.7 Upriver Watershed Sources

Table 4.5-1 lists upstream ECSI sites (between RM 11.8 and the Tualatin River) and associated COIs. Information for these sites is from the DEQ ECSI database. Potential pathways are identified if enough information was available in the database. Arsenic, copper, and zinc were detected in sampled media at the following sites:

- Willamette Falls Locks (USACE) – Arsenic in soils
- Zidell Marine Corporation – Arsenic, copper, and zinc in sediments
- OHSU - Moody Ave. Units A, B, C – Zinc in groundwater
- Grunbaum Property – Arsenic in groundwater
- Heath Oregon Sign Company – Arsenic in soils
- Ross Island Sand & Gravel – Copper in soils
- Clackamette Cove Area – Arsenic in soils
- Pacific Richfield – South Waterfront – Arsenic in soils.

Appendix H includes the recent data collected in the RM 11E and downtown reaches.

10.2.8.3 Loading, Fate, and Transport of Arsenic, Copper and Zinc

Figures 10.2-25a through 10.2-27 summarize the arsenic loading, fate, and transport assessment for the Study Area in a typical year. Copper loads are presented in Figures 10.2-28a through 10.2-30, and zinc loads are shown in Figures 10.2-31a through 10.2-33. These estimates indicate that loads from upriver surface water dominate the current loads for all three metals, frequently exceeding the other quantified external loading terms by 2 or more orders of magnitude. For arsenic, the loads in upstream surface water (RM 11) and the downstream boundary of the Study Area (combined loads at RM 2 and the Multnomah Channel entrance) are comparable, with slightly more leaving than entering the Study Area. The estimated copper and zinc loads show slightly more entering the Study Area than leaving downstream through the mainstem and Multnomah Channel combined. Again, however, these apparent differences may not be significant given the uncertainty inherent in the loading estimates. Figures 10-2.25b, 10.2-28b, and 10.2-31b compare the distribution of total arsenic, copper, and zinc (respectively) in surface sediments, sediment trap samples, and the surface water particulate fraction on a Study Area-wide basis. Summary statistics for these concentrations in each media are presented in Table 10.2-2a and the results of cross-media statistical testing are provided Table 10.2-2b. For each of these metals, all the distributions of each media are significantly different from each other. Surface water suspended particles show distinctly higher concentrations values than sediment trap or surface sediments on a Study Area-wide basis for each metal. Based on median values (Table 10.2-2a), sediment trap concentrations are slighter greater overall than surface sediment concentrations for all three metals.

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Although the current upstream surface water load estimate greatly exceeds the other current loading terms, there is no indication that this load is responsible for the spatial distribution of the elevated concentrations of these metals observed in Study Area sediments. Additional assessment of arsenic and copper loading, fate, and transport within the Study Area will be undertaken in the FS.

10.2.8.4 Relationship of Sources to Distribution of Arsenic, Copper, and Zinc

These metals are naturally occurring in soil, water, and air. Elevated arsenic, copper, and zinc concentrations in surface and subsurface sediment are generally restricted to small, widely scattered nearshore areas. The similarity of surface and subsurface concentrations in many of these areas suggests both recent and historical sources. Within the Study Area, numerous historical and current sources have been identified for these metals for all pathways. The areas of elevated sediment concentrations correspond to the locations of some former shipyards, wood treatment facilities, metal recycling operations, pipe manufacturing, metal plating operations, and marine repair facilities. These metals are also associated with some facilities where metal slag and sandblast grit were used as fill.

The BHHRA identified arsenic and zinc as contaminants potentially posing unacceptable risks for specific exposure scenarios. Copper was not identified as a contaminant potentially posing unacceptable risks for any exposure scenario in the BHHRA.

In summary, although arsenic, copper, and zinc were identified as posing potentially unacceptable ecological risks, the evidence was inconclusive due to uncertainties and conflicting lines of evidence, and none of these COPCs was recommended to be included as a COC.

The Study Area-wide SWAC for these metals is similar to their upriver UPLs. Loading, fate, and transport assessments for the Study Area in a typical year indicate that loads from upriver surface water dominate the current loads for all three metals, frequently exceeding the other quantified external loading terms by 2 or more orders of magnitude.

10.2.9 Chromium

Panels 10.2-12A–C present the Study Area graphical CSM for chromium, per the format described in Section 10.2. The data from recent sediment studies at RM 11E and in the downtown reach are included in the updated RI data set and presented in Appendix H.

Chromium is a naturally occurring element found in rocks, animals, plants, and soil. It can exist in several different forms in the soil, sediment, water, and air. Chromium(III) occurs naturally in the environment but is also a product of industry. Chromium(0) is used for making steel. Chromium(III) and chromium(VI) forms are produced by the chemical industry and are used for chrome plating, the manufacture of dyes and

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pigments, leather tanning, and wood preserving. Smaller amounts are used in drilling muds, rust and corrosion inhibitors, textiles, and toner for copying machines (ATSDR 2008).

In the environment, chromium can be found in air, soil, and water. Chromium compounds will usually remain in the air for less than 10 days, depositing to the land and water, especially by wet deposition (ATSDR 2008). Most chromium in water binds to soil and other materials and is subsequently subject to sediment transport processes, though a small amount may dissolve in the water. Chromium can easily change from one form to another in water and soil, depending on the conditions present (ATSDR 2008). The relation between Cr(III) and Cr(VI) in the environment is strongly dependant on pH and oxidative properties of the location, but in most cases the Cr(III) is the dominating species (Kotas and Stasicka 2000).

Although chromium(III) is required in trace amounts for sugar and lipid metabolism in humans and its deficiency may cause a disease called chromium deficiency, chromium(VI) is a toxin and a carcinogen (ATSDR 2008). Fish do not significantly accumulate chromium in their bodies from water (ATSDR 2008).

Hexavalent chromium was identified as a contaminant potentially posing unacceptable risks for the hypothetical use of untreated surface water as a drinking water source based on a single location (RM 7.0). Hexavalent chromium was not identified as a contaminant potentially posing unacceptable risks for any other locations or exposure scenarios in the BHHRA. Total chromium was not identified as a contaminant potentially posing unacceptable risks for any exposure scenario in the BHHRA.

The BERA identified chromium as posing potentially unacceptable risk to benthic invertebrates based on sediment concentrations exceeding site-specific SQVs. Exceedances of SQVs were of low magnitude and limited areal extent (often spatially isolated).

10.2.9.1 Chromium Contaminant Distribution

For chromium, concentrations greater than 75 mg/kg in sediment, 0.75 µg/L in surface water, 100 µg/L in TZW, and 10 mg/kg in biota samples are considered elevated (the orange or red symbols and polygons on Panels 10.2-12A–C).

10.2.9.1.1 Sediment

Areas of relatively elevated chromium concentrations in surface and subsurface sediments in the Study Area all occur in a few, widely scattered nearshore areas and the head of the International Slip and are limited in spatial extent. There are elevated samples in surface and subsurface sediments suggesting both recent and historical sources. The Study Area-wide SWAC[46, 47] for chromium is 28 mg/kg; as with the

[46] The Study Area SWAC reported in this paragraph is based on the original RI data set (to June 2008) and was not recalculated to include the more recent Study Area data reported in Appendix H. Given the limited amount of

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previous metals, this level is comparable to the upriver surface sediment 95 UPL of 32 mg/kg (see Table 7.3-5b). Ninety-five percent of the surface samples were below 55.8 mg/kg.

Elevated chromium concentrations in surface sediment (greater than 75 mg/kg) are limited to 10 nearshore areas. These locations have 1 to 5 contiguous elevated samples. The largest areas are near RM 9.0W, Swan Island Lagoon, RM 5.9E, and RM 2.3E. Elevated subsurface samples occur at five of these locations. Elevated subsurface samples are present at the north end of Swan Island at a location without elevated surface sediment concentrations.

10.2.9.1.2 Surface Water and Suspended Sediments

Surface water concentrations greater than 0.75 µg/L were observed at most surface water sampling stations, including the upriver station at RM 16. The areas with the lowest mean/maximum concentrations were isolated stations in the Swan Island Lagoon, between RM 6.3 and 7.5, and the Terminal 4 slips (Panel 10.2-12A).

Sediment trap samples show a uniform distribution of chromium levels seasonally and throughout and upstream of the Study Area (see Figure 5.2-27). Most sediment trap samples fall between 30 and 40 mg/kg; the single highest value (60 mg/kg) was measured in upstream of the Study Area at RM 15.7E during the May to August period.

10.2.9.1.3 TZW

Chromium was sampled offshore of sites between RM 6.2W and 7.6W. Elevated TZW concentrations (greater than 100 µg/L) were observed between RM 6.2W and 6.5W, offshore of the Gasco and Siltronic properties. No elevated sediment concentrations are present in these areas.

10.2.9.1.4 Biota

Elevated chromium concentrations in biota are defined here as exceeding 10 mg/kg. No biota samples exceeded 10 mg/kg. Chromium was detected in all fish and invertebrate species and tissues analyzed within the Study Area. The highest concentration (4.1 mg/kg) occurred in sturgeon stomach content sample STWB03 (RM 6.6) in an area without elevated surface sediment concentrations.

10.2.9.2 Potential Sources of Chromium

This section provides a summary of the various potential historical and current chromium sources to the Study Area based on the types of industries that have existed in Portland Harbor that are known to have handled and used chromium and the various pathways (e.g., stormwater/waste water/overland transport, atmospheric deposition) by which chromium can enter the Study Area.

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more recent data relative to the size of the original RI data set, the Study Area SWAC is unlikely to have changed markedly. The upriver reach data set did not change between the original and updated RI data sets.

[47] The SWAC concentrations for the CSM chemicals were calculated using the natural neighbor interpolation method with 10-ft cells and non-detected results included at the detection limit.

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10.2.9.2.1 Chemical Usage

Chromium has a wide range of applications, mainly due to its hardness and resistance to corrosion, and is mainly used in the manufacture of stainless steel and as an alloy for steel. Acidic chromate or dichromate solutions are also used for surface coating. Salts of chromium are used for their toxic properties in preserving wood from decay and damage caused by fungi, insects, termites, etc. Chromium (III) salts are also used in leather tanning. The high melting point of chromium oxide has applications in blast furnaces, cement kilns, and metal casting. Chromium is also used as a pigment, and compounds containing chromium are used as catalysts.

In Portland Harbor, the primary industries known to have handled, manufactured, or disposed of chromium include, the steel industry (EOSM), ship building/demolition and marine repair facilities (Gunderson, Cascade General, Mar Com), metal recycling (Schnitzer-Calbag, Calbag-Nicolai, Calbag Metals-Front Ave., former operations at Gunderson, Schnitzer-Doane Lake), wood treating (M&B), MGP sites (Gasco), and heavy oil facilities (bulk fuel and asphalt storage facilities). Sodium bichromate was used in the sodium chlorate manufacturing process as a corrosion inhibitor at Arkema.

As shown in Table 10.2-13 and Panels 10.2-12A–C, historical known complete or likely complete pathways for chromium have been identified at 43 sites. Of these the stormwater pathway is included at 38 sites, groundwater at 7 sites, overwater at 5 sites, overland transport at 9 sites, and riverbank erosion at 10 sites.

Current known complete or likely complete pathways for chromium have been identified at 20 sites. Of these, the stormwater pathway is included at 13 sites, groundwater at 5 sites, overwater at 4 sites, overland transport at 3, and riverbank erosion at 3 sites.

10.2.9.2.2 Stormwater/Wastewater/Overland Transport

Stormwater is identified as a current known or likely complete pathway for chromium to enter the Study Area at many sites; however, many of these areas do not have elevated sediment concentrations. Most sites adjacent to areas with elevated sediment concentrations have been identified as having current or historical known or likely stormwater pathways for chromium (Panel 10.2-12A). Sites associated with elevated sediment concentrations with known or likely complete stormwater pathways include chemical manufacturing, former MGP operations (Gasco), former shipyards (Mar Com, Cascade General, and Swan Island Upland Facility), truck manufacturing (Freightliner TMP), marine construction and repair (Gunderson), metal recycling (Schnitzer Calbag, Calbag-Nicolai, Calbag Metals-Front Ave., Gould, Schnitzer-Doane Lake), and steel manufacturing (EOSM). Some of these sites discharge directly to the river and some discharge through shared conveyance systems.

Data gathered in Round 3A and 3B stormwater sampling in the harbor detected chromium in stormwater from each land use area sampled. Outfalls from the heavy industrial land use category contributed the majority of the estimated chromium

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stormwater load with a mean total chromium concentration of 20 µg/L and was approximately 4 times higher than the next highest land use category (see Table 4.4-2). Non-LWG stormwater sampling resulted in a mean concentration of 7.91 µg/L (see Table 4.4-4).

Wastewater discharges are currently regulated primarily through NPDES permits. Chromium is not listed as a general NPDES chemical monitoring parameter (see Table 4.4-7) but is a monitoring parameter at one of the 21 sites with individual NPDES permits (Siltronic; see Table 4.4-6). Wastewater discharges in CSO areas, where wastewater could overflow during CSO events, are regulated through municipal pretreatment permits, for sites discharging to the City of Portland sanitary system, or NPDES permits. While metals are identified as COIs at 15 of the 33 pretreatment permit sites, none are listed specifically for chromium (see Table 4.4-5). No information has been reviewed identifying chromium as a historical wastewater COI; however, historical releases of chromium may have been associated with chemical and steel manufacturing, metals recycling, asphalt refining, sand blast grit, and bulk fuel facilities.

Sampling of overland transport is limited and has identified a historical or current known or likely complete pathway at nine sites, including two which are near areas of elevated surface sediment concentrations (Gunderson and Mar Com).

10.2.9.2.3 Overwater Discharge

Historical or current known or likely complete overwater discharge pathways for chromium are limited to four sites. Two of these sites are associated with elevated surface sediment concentrations and include a shipyard and a metals recycling operation.

Chromium is not listed as a material involved in overwater releases from ESCI sites (Table 4.2-3) or from vessels and other spills in the Study Area (Table 4.3-4). However, many spills are unreported, and materials listed in reported spills may include chromium as a component.

10.2.9.2.4 Groundwater Discharge

Groundwater is a historical known or likely complete pathway for chromium at seven sites, including two which are not current sources (M&B and Schnitzer-Calbag). Elevated sediment concentrations near OF-22B and OF-22C may be related to sites with known or likely complete groundwater / stormwater infiltration draining to these outfalls. Bulk fuel and MGP facilities with complete stormwater pathways (Table 10.2-13) are not in areas of elevated sediment concentrations.

10.2.9.2.5 Riverbank Erosion

Based on limited riverbank sampling, riverbank erosion is a historical known or likely complete pathway for chromium at 10 sites, including six which are not current sources. Of the 10 sites, elevated sediment concentrations are near steel manufacturing

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operations (EOSM), metals recycling (Schnitzer-Calbag), and two shipyards (Portland Shipyards and Gunderson).

10.2.9.2.6 Atmospheric Deposition

As noted in the introduction to this section, chromium is a naturally occurring metal that is found widely in natural minerals and occurs naturally in soil, water, and air as a result of mineral weathering, leaching, volcanic eruptions, and wind-blown dirt. Anthropogenic activities, including smelting, pesticide use, combustion of wood and coal, waste incineration, and the production and use of treated wood products that utilize soluble CCA, can also release chromium into the air, soil, water, and sediments. There are no known current or historical point sources of atmospheric chromium in the Study Area.

10.2.9.2.7 Upriver Watershed Sources

Table 4.5-1 lists upstream sites (between RM 11.8 and the confluence of the Tualatin River at RM 29) and associated COIs with the potential for impacts to in-river media. Information for these sites was obtained from the DEQ ECSI database. Potential pathways were identified if sufficient information was available.

Chromium was identified as a COI for 17 sites based on their hazardous substances/waste types. Of these 17 sites, chromium was detected in sampled media at the following five sites:

- Willamette Falls Locks – Chromium detected in upland soil, and direct releases/spills represent potential pathways to the river
- Zidell Marine Corporation – Chromium detected in upland soil, and direct releases/spills, groundwater, and stormwater represent potential pathways to the river
- OHSU - Moody Ave. Units A, B, C – Chromium detected in upland soils, and stormwater and groundwater represent potential pathways to the river
- Clackamette Cove Area – Chromium detected in upland soils, and groundwater represents a potential pathway to the river
- South Waterfront Redevelopment Area 3 – Chromium detected in groundwater, which represents a potential pathway to the river.

These upstream sources may add to the chromium load, but as noted above, the magnitude of the surface water load relative to similar kinds of sources quantified for the Study Area points to a dominance of natural sources. Appendix H summarizes the recent data collected in the RM 11E and downtown reaches.

10.2.9.3 Loading, Fate, and Transport of Chromium

Figures 10.2-34a through 10.2-36 summarize the chromium loading, fate, and transport assessment for the Study Area in a typical year. These estimates indicate that loads from upriver surface water dominate the current loads, exceeding the next highest external

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loading terms—stormwater and upland groundwater plumes—by more than 2 orders of magnitude. The chromium loads in upstream surface water (RM 11) and the downstream boundary of the Study Area (combined loads at RM 2 and the Multnomah Channel entrance) are generally comparable; this appears to reflect the absence of significant loads within the Study Area. As noted above, the nearshore and off-channel areas of elevated chromium concentrations generally appear to be associated with localized upland sources and pathways.

The cross-media comparison and statistical assessment of surface sediments, sediment traps, and suspended solids in surface water (Table 10.2-2a-b and Figure 10.2-34b) show that the Study Area-wide concentrations of all media are statistically different, with surface sediment having the highest concentrations, followed by sediment traps and then by suspended solids in surface water.

10.2.9.4 Relationship of Sources to Distribution of Chromium

Within the Study Area, numerous historical and current sources of chromium have been identified for all pathways, but primarily through stormwater discharge. The areas of elevated sediment concentrations generally correspond to the locations of current or former shipyards, wood treatment facilities, pesticide manufacturing, metal recycling operations, steel manufacturing, metal plating operations, and marine repair facilities. Chromium is also identified at some facilities where metal slag and sandblast grit were used as fill. Known and likely current and historical sources of chromium to the Study Area are summarized in Table 10.2-13 and Panels 10.2-12A–C.

Total chromium was not identified as a contaminant potentially posting unacceptable risks for any exposure scenario in the BHHRA. Hexavalent chromium was identified as a contaminant potentially posing unacceptable risks for the hypothetical use of untreated surface water as a drinking water source based on a single location (RM 7.0). The BERA identified chromium as posing potentially unacceptable risk to benthic invertebrates, but exceedances of SQVs were of low magnitude and limited areal extent.

The chromium loading estimates indicate that loads from upriver surface water dominate the current loads, exceeding the next highest external loading terms—stormwater and upland groundwater plumes—by more than 2 orders of magnitude. However, the estimated loads entering and leaving the Study Area are comparable, suggesting the absence of significant loads within the Study Area.

10.2.10 Tributyltin Ion

Panels 10.2-13A–C present the Study Area graphical CSM for TBT, per the format described in Section 10.2. Recent sediment studies at RM 11E and the downtown reach are summarized in Appendix H.

TBT is an organotin compound. Since the mid-1970s, TBT has been and is still used as an antifouling agent in paints on the immersed portions of boats and floating structures (Batt 2004). Antifouling paints represent the largest source of TBT in coastal

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environments, and nationally, concentrations in shipyards, harbors, and marinas tend to be elevated relative to background conditions. Many countries restricted the use of antifouling paints based on the risks to shellfish. The U.S. partially banned the use of TBT-based antifouling paints in 1988 (Showalter and Savarese 2005). Use of TBT compounds as slimicides on masonry, disinfectants, and biocides for various industrial processes also may result in their release to the environment.

TBT is an ionic organic compound, and the partitioning behavior of TBT is strongly affected by pH and the identity of anions in solution that pair with the TBT ion (Arnold et al. 1997). Specifically, for pH 10 to 7 the measured log K_{oc} values are on the order of 4; from pH 7 to pH 3 they drop to roughly 2, corresponding to 2 orders of magnitude of behavioral variability across the pH 7 boundary. The mean surface water pH in Portland Harbor is 7.38 (10th percentile is 6.98 and 90th percentile is 7.76). The observed pore water pH values measured in the Groundwater Pathway Assessment ranged from 5.6 to 8.1. TBT in its nonionic form (not likely observed at the site) is highly hydrophobic, with K_{ow} values on the order of 7(K_{ow}WIN[48]).

Degradation of organotin compounds involves the breaking of the tin-carbon bond. TBT is subject to different degradation mechanisms depending on its location in the environment. In surface water, TBT is subject to fairly rapid[49] photodegradation and biodegradation (TBT can be degraded by microbial, microalgal, and fungal populations, as well as by some higher organisms, such as fish; Anderson et al. 2002). Degradation of organotin compounds in sediments is much slower than in water, and half-lives have been estimated to be several years (Alzieu 1998). Abiotic cleavage of the tin-carbon bond by hydrolysis is not an important fate process under environmental conditions (WHO 1990).

While accumulation of TBT from water into organisms may be a significant process that can result in elevated tissue concentrations, subsequent biomagnification through the food web is reportedly minor (ATSDR 2005b).

In the BHHRA, TBT was not identified as a contaminant potentially posing unacceptable risk for any exposure scenario.

In the BERA, TBT was identified as posing potentially unacceptable risk based on the dietary LOE for sculpin and tissue-residue LOE for benthic invertebrates. It was not recommended as a COC for fish for several reasons:

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[48] K_{ow}WIN software available online as part of EPA Estimation Program Interface (EPA) suite of programs: http://www.epa.gov/oppt/exposure/pubs/episuite.htm

[49] Half-life information for TBT in freshwater could not be found; however, the following information was found for seawater: the half-life of tributyltin in seawater varies, depending on pH, temperature, turbidity, and light; it is generally estimated to be in the range of 1 day to a few weeks (Alzieu 1998). Biodegradation is the major process in seawaters rich in suspended solids, but photolysis, in surface waters, exceeds biodegradation in clean seawater. Calculated half-lives range from 6 days in summertime waters rich in suspended particles to 127 days in clean winter waters (Watanabe et al. 1992).

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- Single dietary TRV exceedance (based on 1 lab worm sample of 81 prey samples [1.2%] and only when combined with sediment ingestion).
- Low magnitude of exceedance (maximum HQ = 1.0).
- Uncertainty about toxicological effects associated with the TRV (reproduction success was reduced at the TRV, but not dose-responsive).
- Discordance of dietary LOE with the tissue-residue and water LOEs (TBT tissue residue is noted to be reliable predictor of toxicity and is the strongest LOE [Meador et al. 2002]).

It was not recommended as a COC for benthic invertebrates because of the following:

- The TRV was exceeded in empirical bioaccumulation samples only at one location.
- While predicted tissue residues exceeded the TRV more frequently, the moderate strength of the regression was highly influenced by the one high value in the data set. The predicted tissue residues are uncertain and not supported by empirical data.
- The TRV is uncertain due to the inclusion of imposex—the endpoint that defined the lower distribution of the SSD, which set the TRV.

10.2.10.1 TBT Contaminant Distribution

For TBT, concentrations greater than 1,000 µg/kg in sediment, 100 µg/kg in biota samples, and greater than 0.01 µg/L in surface water samples are considered elevated (the orange or red symbols and polygons on Panels 10.2-13A–C).

10.2.10.1.1 Sediment

Relatively elevated TBT concentrations are primarily located in the vicinity of the Cascade General Shipyard and adjacent to Swan Island Lagoon. At the Cascade General Shipyard, subsurface sediments exhibit slightly more elevated concentrations than surface sediments, suggesting that contributions from historical inputs were elevated relative to current levels. At RM 7.5E, subsurface samples are also more elevated than surface samples. A single elevated surface sediment sample also found at RM 3.7E near the mouth of the International Slip. The Study Area-wide SWAC[50, 51] for TBT is 182 µg/kg; the mean of the very limited (N=3) upriver surface sediment TBT set is 0.64 µg/kg. Background calculations are not included in Section 7 due to the small number of detected values in the data set.

[50] The Study Area SWAC reported in this paragraph is based on the original RI data set (to June 2008) and was not recalculated to include the more recent Study Area data reported in Appendix H. Given the limited amount of more recent data relative to the size of the original RI data set, the Study Area SWAC is unlikely to have changed markedly. The upriver reach data set did not change between the original RI and updated RI data sets.

[51] The SWAC concentrations for the CSM chemicals were calculated using the natural neighbor interpolation method with 10-ft cells and non-detected results included at the detection limit.

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10.2.10.1.2 Surface Water and Suspended Sediments

TBT concentrations in surface water are considered elevated if they exceed 0.01 µg/L, and no surface water concentrations exceeded this concentration. TBT was generally undetected in sediment trap samples upstream of RM 7.5 (with the exception of Swan Island Lagoon) and detected at low levels (e.g., <10 µg/kg) from RM 7.5 downstream in all sampling periods. The only two samples (summer, fall) collected in Swan Island Lagoon had TBT levels of 64 and 81 µg/kg (see Figure 5.2-30).

10.2.10.1.3 TZW

TBT was not included as a COI in the TZW sampling program, and groundwater is not considered to be a major pathway for TBT migration into the Study Area.

10.2.10.1.4 Biota

Although TBT has been detected in fish and invertebrate samples, biota tissue TBT levels in the Study Area did not exceed 100 µg/kg.

10.2.10.2 Potential Sources and Pathways of TBT

This section provides a summary of the various historical and current TBT sources to the Study Area based on the types of industries that have existed in Portland Harbor that are known to have handled and used TBT, and summaries of the various pathways by which TBT can enter the Study Area.

10.2.10.2.1 Chemical Usage

Within the Study Area, historical and current sources contributed TBT to the river primarily through the overwater and stormwater pathways. Areas of elevated TBT concentrations correspond with current and former shipyards, where ship hull washing, abrasive blasting, and painting occurred in dry docks and berths (see Map 3.2-3).

Identified known and likely current and historical sources of TBT to the Study Area are summarized in Table 10.2-14 and Panels 10.2-13A–C. Historical pathways for TBT migration are found at four current and historical shipyard facilities: Cascade General, Gunderson, Mar Com South, and Marine Finance. Of these facilities, the stormwater pathway is found at four sites, and the overwater, overland, and riverbank erosion pathways are found at two sites.

Current known complete or likely complete pathways for TBT have been identified at two sites: Cascade General and Gunderson. Stormwater is a known complete pathway at both sites and overwater is a known complete pathway at Cascade General.

10.2.10.2.2 Stormwater/Wastewater/Overland Transport

A current and historical known complete stormwater pathway for TBT migration to the river exists at Gunderson, an active manufacturer and refurbisher of railroad cars and marine barges since 1913 (Integral 2007a). Surface sediment and riparian soil samples are elevated where OF-18 empties into the river on the Gunderson shoreline. OF-18 drains stormwater from Gunderson as well as several other ECSI sites within its basin.

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TBT is not a COI at any of these other sites, except Gunderson. Historical likely complete stormwater pathways also exist at Cascade General, Mar Com South, and Marine Finance.

Sampling of the overland transport pathway is limited. Historical overland transport pathways are likely complete at Mar Com South and Marine Finance. Operations have ceased at Mar Com South, while Marine Finance has been redeveloped under new ownership and most of the site has been repaved.

Wastewater discharges are currently regulated primarily regulated through NPDES permits. TBT is identified as a monitoring parameter for Cascade General's individual NPDES permit (see Table 4.4-6). Wastewater discharges in CSO areas, where wastewater could overflow during CSO events, are regulated through municipal pretreatment permits. TBT is not identified as a COI at any sites with a pretreatment permit (see Table 4.4-5).

10.2.10.2.3 Overwater Discharge

The overwater discharge pathway is considered to be the greatest contributor of TBT to the river. Direct overwater discharge of TBT can occur during dry and wet grit sandblasting operations, as well as the application of anti-fouling paints that contain TBT. The preferred technology in recent years for removing paint in Pacific Northwest shipyards is to use freshwater and hydro-blasting technologies (high pressure pumps) to wash down (to remove salt, slime, and biofouling) paint from vessels (PNPRC 1999). However, this technology produces washdown wastewater, which can be discharged into waterways, contaminating port and harbor bottom sediments.

Within the Study Area, current and historical complete pathways for overwater discharge of TBT are found at Cascade General, a shipyard that has been active since the 1940s (Integral 2007a). At this facility in recent years, BMPs and drydock water treatment have been implemented under agency-approved permits to control potential releases associated with overwater and ship repair-related activities. A complete historical overwater pathway also exists at Mar Com South where all shipbuilding and repair activities have ceased.

10.2.10.2.4 Groundwater Discharge

Because organotins are moderately hydrophobic and associate strongly to particles in water, groundwater is not expected to be a significant historical or current pathway for TBT migration to the Study Area. There are no current or historical groundwater pathways for TBT in the Study Area.

10.2.10.2.5 Riverbank Erosion

Sampling of the riverbank erosion pathway for TBT is very limited, and likely complete historical pathways have only been identified at Mar Com South and Marine Finance, which both have a history of shipyard activities.

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10.2.10.2.6 Atmospheric Deposition

TBT can be released to the atmosphere during the mixing of TBT-based paints and from dry, abrasive grit blasting of marine vessel surfaces if protective measures are not implemented. Both wet and dry deposition can occur. Information on atmospheric deposition of TBT is very limited, but long-range atmospheric transport of butyltins does occur.

10.2.10.2.7 Upriver Watershed Sources

Table 4.5-1 lists upstream sites (between RM 11.8 and the confluence of the Tualatin River at RM 29) and associated COIs. Information for these sites was obtained from the DEQ ECSI database. TBT was identified as a COI for the Zidell Marine Corporation site (RM 14W) and Ross Island Sand & Gravel (RM 14.7E) based on detections in sampled media.

Appendix H summarizes the recent data collected in the RM 11E and downtown reaches.

10.2.10.3 Loading, Fate and Transport of TBT

Figures 10.2-37a through 10.2-39 summarize the loading, fate, and transport assessment for TBT in the Study Area in a typical year. These calculations indicate that, among the external loading terms quantified, estimated TBT inputs from upriver surface water (11 kg/yr) are lower than those from advection through subsurface sediments (36 kg/yr). The relatively high overall subsurface advective load estimate is driven by localized elevated TBT concentrations at RM 8 to 8.9, and there is significant uncertainty associated with these comparative estimates overall. Among the fate and transport processes internal to the Study Area shown on Figure 10.2-38, quantitative estimates were developed only for advection through surface sediments (9.8 kg/yr). This term is estimated to be comparable in magnitude to upstream surface water loading and lower than advection through subsurface sediment. Both of these advective loading terms are highest between RM 8 and 8.9 (Figure 10.2-39). As noted above, the Cascade General Shipyard and Swan Island Lagoon elevated TBT concentrations appear to be associated with localized upland sources and pathways.

The cross-media comparison and statistical assessment of surface sediments, sediment traps, and suspended solids in surface water (Table 10.2-2a-b and Figure 10.2-37b) show that the concentrations in all media are statistically different.[52] On a Study Area-wide basis, surface sediment has the highest concentrations, exceeding surface water suspended solids and then sediment trap samples. The mean surface sediment concentration is about three times greater than the mean concentration of surface water particulates and over 50 times greater than the mean concentration measured in sediment traps across the Study Area.

[52] TBT had a detection frequency of only 7% in surface water particulate samples. Non-detected values are included in the data summaries and box-whisker plots using a substitution of one-half the detection limit to allow comparisons across media.

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10.2.10.4 Relationship of Sources to Distribution of TBT

Since 1988, TBT has been partially banned in the U.S. for use as an ingredient in anti-fouling paint. Active shipyards in the Study Area have since adopted alternative paints and abrasive blasting methods as well as BMPs that will lessen the discharge of TBT to in-water media. TBT has been detected in surface and subsurface sediment, and fish and invertebrate samples in the vicinity of the Cascade General Shipyard and adjacent to Swan Island Lagoon. At the Cascade General Shipyard, subsurface sediments exhibit slightly more elevated concentrations than surface sediments, suggesting that contributions from historical inputs were elevated relative to current levels. Elevated concentrations were also found at RM 7.5E, where subsurface samples are more elevated than surface samples. A single elevated surface sediment sample is also found at RM 3.7E near the mouth of the International Slip.

In the BHHRA, TBT was not identified as a contaminant potentially posing unacceptable risks for any exposure scenario. In the BERA, TBT was identified as posing potentially unacceptable risk to fish and benthic invertebrates based on surface water, sediment, and empirical and predicted tissue residue lines of evidence. Potential risks to fish and invertebrates are highly localized in historical shipyard areas.

As shown in Figures 10.2-37a through 10.2-39, advection through subsurface sediments appears to be the largest loading term, followed by upriver surface water. The relatively high overall subsurface advective load estimate is driven by localized elevated TBT concentrations at RM 8 to 8.9.

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11.0 CONCLUSIONS AND NEXT STEPS

The LWG is performing the RI/FS for the Site pursuant to an EPA Administrative Settlement Agreement and Order on Consent (EPA 2001a, 2003b, 2006a). The LWG includes 12 private property owners along the Willamette River, the Port of Portland, and the City of Portland and represents a small subset of the PRPs identified by EPA.

As a Superfund site identified under CERCLA, Portland Harbor has been the focus of a phased RI conducted by the LWG, as well as numerous environmental investigations completed by individual LWG members, other private entities, and various governmental organizations since 2001. RI data collected by the LWG were used to characterize the physical system of the LWR; describe the nature and extent of contamination in sediment, surface water, TZW, stormwater, and biota (including fish, clams, and crayfish); and assess risks to human health and the environment. This section describes major conclusions of the RI and the next steps leading to the selection of appropriate remedial actions to address the Site risks. Given the size and complexity of the Portland Harbor Superfund Site, additional data will likely need to be collected after the RI is completed to support remedial design activities.

11.1 CONCLUSIONS

The key findings of the RI include the following:

Nature and Extent of Contamination

- Sediments in Portland Harbor reflect the industrial, marine, commercial, and municipal practices for over 100 years in this active industrial, urban, and trade corridor, as well as agricultural activities in the Willamette Basin.
- Higher concentrations of contaminants in sediments occur in nearshore and off-channel areas that are generally associated with known or likely historical or current sources.
- Contaminant concentrations in sediment are generally higher at depth than in the surface layer, indicating that past contaminant inputs were greater than current inputs, and that surface sediment quality has improved over time. The few exceptions include areas where higher surface sediment concentrations appear to be associated with ongoing local sources, low rates of sediment deposition, and physical sediment disturbance (e.g., from boat scour).
- Except for the nearshore areas of relatively high contaminant concentrations, concentrations of many contaminants in the remainder of the Study Area, including much of the navigation channel, are similar to levels measured in sediments in the upriver or background reach from Ross Island to the Willamette Falls, an area unaffected by Portland Harbor sources.

Estimates of Risk

- PCBs, and to a lesser extent dioxins/furans, PAHs, and DDx, account for almost all of the estimated human health risk. PCBs are the primary contributors to human health risk on a Study Area-wide basis, with ingestion of resident fish (e.g., bass, carp) representing the primary exposure pathway and the highest estimated risk. Consumption of fish and shellfish from the Study Area may pose health risks to humans that are greater than the EPA target risk range for human health (greater than one in a million [10^{-6}] to one in ten thousand [10^{-4}] excess risk of cancer). Other scenarios included in the assessment with health risks that may exceed EPA thresholds in parts of the Study Area were direct exposure to in-water sediment and future use of untreated river water as a hypothetical drinking water source.
- Total PCBs are the primary risk contributor to mink, river otter, and spotted sandpiper, and pose low risk to osprey, bald eagle, sculpin, and smallmouth bass. TEQ, which incorporates both PCB and dioxin and furan exposure, was also found to be a primary risk contributor to mink and river otter, and to pose low risk to spotted sandpiper, osprey, and bald eagle. Total DDx was found to pose low to negligible risk to bald eagle. Zinc, benzo(a)anthracene, benzo(a)pyrene, naphthalene, and DDx were found to pose localized risk to individual Pacific lamprey ammocoetes due to potential exposure to contaminated shallow transition zone water.
- Exposure to the mixture of contaminants present in the Study Area sediments are projected to pose unacceptable benthic risks for about 7% of the Study Area.
- Risk assessment methods were used conservatively to be protective of humans, fish, and wildlife and to minimize the chance of underestimating exposure and risk. As a result, the cumulative effects of the health protective assumptions and methods result in risk estimates that may be higher than actual risks within the Study Area.
- PCBs are a highly persistent compound found in fish on a regional (i.e., watershed-wide) and global level. Fish caught in the Willamette and Columbia rivers, outside of the Study Area, also contain PCBs that pose potentially unacceptable risk. Concentrations of PCBs in fish caught within the Study Area are higher than in regional fish tissue.

Sources of Contamination

- Most of the sediment contamination in the Study Area is from historical sources and practices that have been controlled. Historical activities in the harbor include or have included ship building, dismantling and repair; gas and chemical manufacturing; steel production; wood treatment operations; metal recycling; fuel storage and transfer operations; maritime commodities shipping; electrical production and distribution; and rail yards.

- A comparison of the relative mass of contaminants in bedded sediment with the current loading rates indicates that past releases of contaminants related to historical upland and overwater activities likely account for most of the existing sediment contamination in the Study Area.
- While many sources of historical contamination have ceased or now operate under stricter environmental controls, contaminants from historical and current activities still reach the Study Area in stormwater, public and private outfalls, permitted industrial discharges, atmospheric deposition, bank erosion, groundwater discharges, and incidental releases within the Study Area, as well as in surface water and sediment inflows from upstream.
- The mass of most contaminants currently entering the Study Area from upstream river flows (in surface water and suspended sediments) per year exceeds the current mass from upland sources within the Study Area. This is due to the large volume of water and sediments that enters the Study Area from upstream.
- Contaminant concentrations in loads entering the Study Area from adjacent upland sources and pathways (such as stormwater) are generally greater than concentrations in the upstream loads (upriver surface water and sediments). Consequently, discharges of contaminants through stormwater outfalls and other pathways, depending on the specific release locations, may cause localized areas of elevated contaminant concentrations in bottom sediment.
- Stormwater input is the most important current source pathway within the Study Area (i.e., excluding upstream sources) for many contaminants, including PCBs and DDx.
- Contributions of contaminants to the Study Area via discharges of upland groundwater plumes to the river are currently known to be associated with a limited number of upland sites.

Potential for Contaminant Transport within and Downstream of the Study Area

- Most of the sediments with contaminant concentrations that pose potenially unacceptable risks are in relatively stable, nearshore areas.
- The spatial distribution of contaminants and sediment transport dynamics suggest that some downstream migration of contaminants has occurred from past large releases and/or periodically occurs within the Study Area from locations with highly elevated sediment concentrations. This will be further evaluated in the FS fate and transport modeling effort.
- Most nearshore areas and much of the navigation channel are stable, depositional environments. Sediments in some nearshore areas are subject to periodic anthropogenic disturbance from prop scour and maintenance dredging. On a harbor-wide basis, buried contaminants (>30 cm depth) in the harbor are

generally unlikely to be disturbed by natural events. There are two portions of the channel (i.e., RM 5 to 7 and upstream of RM 10) that are more dynamic and potentially subject to deep erosion during flood events. Deep sediments (>30 cm sediment depth) upstream of RM 10 are not highly contaminated. However, elevated PAHs levels are present in both surface and deep sediments along the western half of the channel from approximately RM 5.8 to 6.5 and these sediments are at risk of being re-exposed and transported downstream during high energy events.

- There is little evidence from concentration gradients in bedded surface sediment data of significant migration of contaminants immediately downstream of the Study Area in either the Willamette River main stem or Multnomah Channel.

11.2 NEXT STEPS

The key RI findings described above were developed specifically to inform the framework for identifying, analyzing, and comparing alternatives to remediate areas of elevated risk to humans and/or ecological receptors in the FS. The next major milestone of the Portland Harbor RI/FS will be the preparation of a draft FS that will identify appropriate cleanup strategies and methods. The results of the RI/FS will be used by EPA to identify specific areas of the Study Area that will require cleanup.

Consistent with EPA RI/FS guidance, the Portland Harbor RI and FS are interactive efforts. The RI provides an extensive library of information, including the following: Study Area physical, chemical, and biological characteristics; source identification; contaminant loading; a site-wide CSM; and risk assessment results sufficient to conduct a comprehensive FS. No additional sampling data are anticipated to be needed to complete the FS.

To support the FS, some tasks begun under the RI will be completed as part of the FS:

- **Fate and transport modeling** – Fate and transport models will support the evaluation of the effectiveness of remedial alternatives in the FS. They will also be used to assess the potential for recontamination of the Site from upland or upstream sources after implementation of potential remedies. The HST model developed for the RI will be refined and coupled to contaminant fate and transport models to achieve these objectives. In addition, the relative contribution of contaminants in sediment and surface water to contaminants in fish and wildlife resulting from sediment versus surface water contamination will be evaluated using the fate and transport and bioaccumulation models developed for the RI/FS.
- **Ongoing source control inventory update** – Additional source and source control information will be compiled in the FS to develop an updated inventory of ongoing sources and source control efforts for each potential in-water cleanup area. This information will be used to address how upland source control efforts

will need to be integrated into sediment remedies and take place before sediment remedies can be completed.

- **Refinement of the Conceptual Site Model** – The CSM provided in the RI will be refined as necessary in the FS.

The RI gathered information to support informed risk management decisions regarding which remedies are the most appropriate for this site (EPA 2005c). To assist in EPA's risk management decision making, the LWG has provided EPA with its risk management recommendations on contaminants of concern, human health and ecological receptors, exposure pathways, and benthic areas of concern for the development and evaluation of remedial alternatives in the FS (Kennedy/Jenks and Windward 2011).

To initiate development of the FS, EPA and the LWG developed early PRGs based on the Draft BHHRA and Draft BERA and AOPCs to be evaluated in the Draft FS. EPA subsequently identified a subset of Focused PRGs for the LWG to use in the FS. The LWG is currently proceeding with development of the Draft FS report. In addition to the Focused PRGs, all contaminants potentially posing unacceptable risk identified in the Draft Final BHHRA and BERA, and those contaminants in surface water and TZW that exceed water screening levels as identified in Appendix D3.3 and Anchor QEA (2011) will be addressed in the FS. EPA will develop final, chemical-specific, sediment cleanup levels by weighing a number of factors, including site-specific uncertainty factors and the criteria for remedy selection found in the NCP (EPA 2005c).

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13.0 GLOSSARY

A

Absorption – The incorporation of a substance in one physical state into another in a different physical state (absorption); e.g., the uptake of water, other fluids, or dissolved chemicals by a cell or an organism.

Adsorption – The process in which a substance adheres to the surface of a solid material.

Advection – The process of transport of matter by a mass of flowing fluid (e.g., a river).

Alluvial – Relating to the process of deposition of particles by flowing water.

Ammocoetes – Lamprey larvae (i.e., worm-shaped early juvenile stage), typically 1 to 4 inches long. They are transparent, eyeless filter-feeders and live in muddy river bottoms.

Anthropogenic – An effect or object resulting from human activity; e.g., natural and human-made substances may be present in the environment as a result of human activities.

Aqueous – Something made from, with, or by water.

Aquifer – An underground geologic formation, or group of formations, containing water that can be readily transmitted and that is a source of groundwater for wells and springs.

Aquitard – Geologic formation that may contain groundwater but is not capable of transmitting significant quantities of water to wells or springs. They may function as a confining bed.

Aroclor – Tradename of mixtures of PCBs. With the exception of Aroclor 1016, the last two numbers in the tradename designation correspond to the percentage of chlorine by weight.

Assessment Endpoint – In an ecological risk assessment, this is an expression of the environmental value to be protected; it includes both an ecological entity and specific attributes thereof. For example, salmon (i.e., the valued ecological entity) reproduction and population maintenance (i.e., attributes) is an assessment endpoint.

Attenuation – The process by which a chemical is reduced in concentration over time, through absorption, adsorption, degradation, dilution, and/or transformation. It can also be the decrease with distance of sight caused by attenuation of light by particulates.

B

Background (Background Level) – 1) As defined in EPA (2002), substances present in the environment that are not influenced by releases from a site and are usually described as naturally occurring or anthropogenic, where *naturally occurring* is defined as substances present in the environment in forms that have not been influenced by human activity, and *anthropogenic* is defined as natural and human-made substances present in the environment as

a result of human activities but not specifically related to the CERCLA site in question. 2) In an exposure assessment, the concentration of a substance in a defined reference area, during a fixed period of time before, during, or after a data-gathering operation.

Bathymetry – The measurement of depths of water in rivers, lakes, oceans, and other water bodies. Also the information derived from such measurements. Bathymetry is expressed relative to a reference elevation or datum. The reference datum may differ for coasts and inland waterways such as the Columbia River and the Willamette River.

Bedload – Sediment particles resting on or near the channel bottom that are pushed or rolled along by the flow of water.

Benthic/Benthos – Relating to or characteristic of the bottom of an aquatic body or the organisms and plants that live there.

Bioaccumulation – The process by which an organism retains environmental chemicals in its body (possibly in a specific organ or tissue).

Bioavailability – The degree of the tendency of a chemical to be absorbed by an organism into its bloodstream.

Biomagnification – Refers to the process whereby the concentrations of certain chemicals such as PCBs or dioxins increase in organism tissue with increase in trophic level (i.e., moving up the food chain). The substances become increasingly concentrated in tissues or internal organs as they move up the food chain.

Biota-Sediment Accumulation Factor (BSAF) – The concentration of a chemical in tissue divided by a concentration in sediment. In the Portland Harbor Superfund Project, BSAFs were calculated from paired sets of chemical concentrations in sediment and tissue in two ways: 1) from the slope of the line that results from plotting paired sediment and tissue concentrations, or 2) as the average of BSAF values calculated for each set of paired observations of tissue and sediment concentrations. The BSAF is used to predict potential bioaccumulation of sediment-associated chemicals in tissues of ecological receptors.

Bounding Indicator Chemicals – In the RI, four chemical groups—total PCBs, total PCDD/Fs, total DDx, and total PAHs—are referred to as "bounding" indicator chemicals because their distribution is believed to encompass the spatial extent of potentially unacceptable risks associated with all COCs identified in the baseline risk assessments.

C

Carcinogen – Any substance that can cause or aggravate cancer.

Central Tendency – When referring to the exposure of organisms to a chemical, an estimate of the average exposure that may potentially be experienced by the population.

Cleanup – Actions taken to deal with a release or threat of release of a hazardous substance that could affect humans and/or the environment. The term "cleanup" is sometimes used interchangeably with the terms remedial action, removal action, response action, or corrective action.

Colloid(s) – Very small solids (that do not dissolve) that remain dispersed in a liquid for a long time due to their small size and electrical charge.

Columbia River Datum (CRD) – A vertical datum established for the Columbia River from the lower river to the Bonneville Dam and on the Willamette River from the Columbia up to Willamette Falls. At the Morrison Street Bridge gauge, the CRD is 1.85 feet above NVGD29/47.

Combined Sewer Overflow (CSO) – Discharge which occurs when system storage and conveyance capacity are exceeded during large wet-weather events and sanitary wastewater and stormwater overflow directly to the river.

Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) – A U.S. legislative act of 1980 authorizing EPA to respond to releases or threatened releases of hazardous substances that may endanger public health or the environment (also see Superfund).

Conceptual Site Model (CSM) – A written and/or schematic representation of an environmental system and the physical, chemical, and biological processes that determine the transport of chemicals from sources through environmental media to humans and ecological receptors in the system. The CSM is often revised periodically as additional data become available at a site.

Confined Aquifer – An aquifer in which ground water is confined under a capping aquitard or confining bed and in which groundwater will rise above the level of the confining bed if intersected by an opening, such as a well.

Congener – One of many related individual chemicals having similar chemical structure but different precise composition (e.g., PCB congeners each have two phenyl rings, but may differ in the number of chlorine atoms they contain).

Contaminant(s) of Concern (COC)[1] – Contaminants identified through the baseline risk assessment that potentially cause unacceptable adverse effects to human health and/or ecological receptors.

Contaminant(s) of Interest (COI)[2] – Contaminants that are expected to be present at a site based on a review of site information.

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[1] Prior deliverables and some of the tables and figures attached to this document may use the term "chemical of concern," which has the same meaning as "contaminant of concern" and refers to "contaminants" as defined in 42 USC 9601(33).

[2] Prior deliverables and some of the tables and figures attached to this document may use the term "chemical of interest," which has the same meaning as "contaminant of interest" and refers to "contaminants" as defined in 42 USC 9601(33).

Contaminant(s) of Potential Concern (COPC)[3] **–** Contaminants of interest that have been screened-in for evaluation in later analyses during the risk assessment process.

D

Dense Non-Aqueous Phase Liquid (DNAPL) – Dense non-aqueous phase liquids, such as chlorinated hydrocarbon solvents or petroleum fractions, with a specific gravity greater than 1.0 that sink through the water column until they reach a confining layer.

Dermal Absorption/Penetration – The process by which a chemical penetrates the skin and enters the body as an internal dose.

Dermal Exposure (Contact) – Contact between a chemical and the skin.

Desorption – The release of a chemical from the surface of a solid material (e.g., a sediment particle) to water (e.g., water in or overlying the sediment).

Detection Limit – The lowest concentration of a chemical that can reliably be distinguished from a zero concentration.

Dredging – The removal of sediment from the bottom of water bodies. Dredging activities may be subject to regulation under Section 404 of the Clean Water Act.

Dry Deposition – The falling of small particles and gases to the Earth without rain or snow. Dry deposition is a component of acid deposition, more commonly referred to as acid rain.

E

Early Action – A non-time-critical removal action pursuant to 40 CFR 300.415(b) (4).

Ecological Risk Assessment (ERA) – The process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors, including chemicals.

Ecosystem – The interacting system of interdependent biological organisms and their nonliving environmental surroundings.

Effluent – Liquid waste—treated or untreated—that flows out of a treatment plant, sewer, or industrial outfall.

Epibenthic – The term referring to organisms that live on the bottom of riverbeds and/or lake and ocean floors.

Erosion – The wearing away of land surface by wind or water, intensified by land-clearing practices related to farming, residential or industrial development, road building, or logging.

[3] Prior deliverables and some of the tables and figures attached to this document may use the term "chemical of potential concern," which has the same meaning as "contaminant of potential concern" and refers to "contaminants" as defined in 42 USC 9601(33).

Exposure – Contact between an organism or biological system and a chemical, physical, or biological agent. Exposure may be expressed as the amount in a given environmental medium (i.e., air, water, soil, sediment, or tissue) at the point of contact (see Exposure Point Concentration) or as the amount that is taken up by an organism (i.e., a dose).

Exposure Assessment – The measurement or estimation of the magnitude, frequency, duration, and route of exposure to stressors.

Exposure Pathway – The path from sources of chemicals to man and other species via soil, sediment, water, or food.

Exposure Point Concentration (EPC) – The concentration of a chemical or microbial contaminant at the location where exposure occurs.

Exposure Route – The way a chemical or microbial contaminant enters an organism after contact; i.e., by ingestion, inhalation, or dermal absorption.

Exposure Scenario – A tool used to develop estimates of potential exposure, dose, and risk. An exposure scenario generally includes facts, data, assumptions, inferences, and sometimes professional judgment about how the exposure takes place.

F

Flood Stage – A river stage established by the National Weather Service (NWS) above which flood damage may occur. The NWS defines flood stage for the Willamette River at Portland as 18.0 feet CRD.

Flux – The transfer of water equivalant to water flow or discharge, or the transfer of a chemical substance that is the product of the water flow and substance concentraition.

Food Web Model (FWM) – A graphical or mathematical model that describes interconnecting feeding relationships. Some food web models may be used to simulate bioaccumulation of chemicals from environmental media and transfer through food chains.

G

Groundwater – The supply of water found beneath the Earth's surface, usually in aquifers, which supply wells and springs.

Groundwater Discharge – Groundwater entering a water body (e.g., lake, river, or coastal marine waters).

Groundwater Plume – Contaminated groundwater that is moving through the subsurface by advection and dispersion.

Groundwater Seep – Groundwater discharge that is visble as it enters a water body either above or below the water line.

H

Habitat – The place where a population (e.g. human, animal, plant, microorganism) lives and its surroundings, both living and non-living.

Hazard Index (HI) – An indication of the potential for cumulative noncancer effects that is derived by summing the individual chemical hazard quotients.

Hazard Quotient (HQ) – The ratio of estimated site-specific exposure to a single chemical to a selected toxicity threshold, which is either the level at which no adverse health effects are likely to occur (i.e., the no-observed-adverse-effect level) or at which effects are likely to occur (i.e., the lowest-observed-adverse-effect level).

Hazardous Substance – From CERCLA, a hazardous substance is: "(A) any substance designated pursuant to section 311(b)(2)(A) of the Federal Water Pollution Control Act [33 U.S.C. 1321(b)(2)(A)], (B) any element, compound, mixture, solution, or substance designated pursuant to section 9602 of this title [i.e., CERCLA], (C) any hazardous waste having the characteristics identified under or listed pursuant to section 3001 of the Solid Waste Disposal Act [42 U.S.C. 6921] (but not including any waste the regulation of which under the Solid Waste Disposal Act [42 U.S.C. 6901 et seq.] has been suspended by Act of Congress), (D) any toxic pollutant listed under section 307(a) of the Federal Water Pollution Control Act [33 U.S.C. 1317(a)], (E) any hazardous air pollutant listed under section 112 of the Clean Air Act [42 U.S.C. 7412], and (F) any imminently hazardous chemical substance or mixture with respect to which the Administrator has taken action pursuant to section 7 of the Toxic Substances Control Act [15 U.S.C. 2606]. The term does not include petroleum, including crude oil or any fraction thereof which is not otherwise specifically listed or designated as a hazardous substance under subparagraphs (A) through (F) of this paragraph, and the term does not include natural gas, natural gas liquids, liquefied natural gas, or synthetic gas usable for fuel (or mixtures of natural gas and such synthetic gas)."

Homolog – In chemistry, a homolog refers to a chemical compound from a series of compounds that differ only in the number of repeated structural units. For example, PCB compounds with different degrees of chorination are called homologs, whereas PCB compounds with the same degree of chlorination but at different locations on the molecule are called congeners.

Hydraulic Gradient – The slope of the groundwater potentiometric surface from which the direction of groundwater flow can be predicted.

Hydrodynamics – The study of liquids in motion.

Hydrogeology – The study of the occurrence and movement of water below the earth's surface.

Hydrograph – A record of the stage and/or discharge of a river as a function of time.

Hyetograph – A graphical representation of the distribution of rainfall over the total duration of a storm event.

I

Infiltration – The penetration of water through the ground surface into subsurface soil or the penetration of water from the soil into sewer or other pipes through defective joints, connections, or manhole walls.

Initial Study Area (ISA) – The stretch of the Willamette River extending from approximately river mile 3.5 to river mile 9.2 and adjacent areas logically associated with an evaluation of the in-water portion of this stretch of the river.

Interceptor (Sewers) System – Large sewer lines that, in a combined system, control the flow of sewage to the treatment plant. In an extreme storm, they allow some of the sewage to flow directly into a receiving stream, thus keeping it from overflowing onto the streets. Also used in separate systems to collect the flows from main and trunk sewers and carry them to treatment points.

Interim Remedial Action Measure – An action that remediates a site but may not constitute the final remedy.

L

Light Non-Aqueous Phase Liquid (LNAPL) – A non-aqueous-phase liquid with a specific gravity less than 1.0. Because the specific gravity of water is 1.0, most LNAPLs float on top of the water table. Most common petroleum hydrocarbon fuels and lubricating oils are LNAPLs.

Line of Evidence (LOE) – A specific analysis approach, based on empirical data or a model prediction that is used to assess potential risks to humans or ecological receptors.

Lipid Solubility – The maximum concentration of a chemical that will dissolve in fatty substances. Lipid soluble substances are insoluble in water; they will selectively disperse through the environment via uptake in living tissue.

Lower Willamette River (LWR) – The stretch of the Willamette River from the confluence with the Columbia River (river mile 0) to Willamette Falls (approximately river mile 26).

Lowest Observed Adverse Effect Level (LOAEL) – The lowest level of a stressor that causes statistically and biologically significant differences between a test sample and a control sample (i.e., sample not subjected to a stressor).

M

Macropthalmia – Juvenile phase of lamprey ammocoetes. Metamorphosis from ammocoetes to macropthalmia occurs gradually over several months as developmental changes occur, including the appearance of eyes and teeth, and they leave the substrate to enter the water column. Transformation from ammocoetes to macropthalmia typically begins in the summer and is complete by winter.

Matrix – The sample material in which the chemicals of interest are found (e.g., water, sediment, tissue).

Mean High River Stage – The arithmetic mean of the maximum (e.g., highest daily measurement) observed river stage data in a given period (e.g., monthly mean high river stage).

Media – Specific environmental materials—air, water, soil, and biological tissue.

Mean High Water Mark (MHWM) – A tidal datum. The average of all the high water heights observed over the National Tidal Datum Epoch (19-year period).

Mean Low Water Mark (MLWM) – A tidal datum. The average of all the low water heights observed over the National Tidal Datum Epoch (19-year period).

Method Detection Limit (MDL) – The minimum concentration of a substance being analyzed that has a 99 percent probability of being identified.

Municipal Discharge – Discharge of effluent from wastewater treatment plants which receive wastewater from households, commercial establishments, and industries in a municipality (e.g., city or town). Combined sewer/separate storm overflows are included in this category.

N

National Geodetic Vertical Datum of 1929 and Supplemental Adjustment of 1947 (NGVD29/47) – NGVD29/47 is a fixed datum adopted and adjusted in 1947 as a national standard geodetic reference for heights prior to June 24, 1993 and is now considered superseded by NAVD88. NGVD29 is sometimes referred to as Sea Level Datum of 1929 or as Mean Sea Level (MSL) on some early issues of U.S. Geological Survey topographic quads. NGVD29 was originally derived from a general adjustment of the first-order leveling networks of the U.S. and Canada after holding mean sea level observed at 26 long-term tide stations as fixed. Historical data referencing MSL as the vertical datum in Portland Harbor is technically on NGVD29/47.

National Pollutant Discharge Elimination System (NPDES) – A regulatory program enacted under the Clean Water Act, which prohibits discharge of pollutants into waters of the United States unless a special permit is issued by EPA, a state, or, where delegated, a tribal government on an Indian reservation.

No Observed Adverse Effect Level (NOAEL) – The highest exposure level at which there are no statistically or biologically significant increases in the frequency or severity of adverse effects between the exposed population and its appropriate control; some effects may be produced at this level, but they are not considered as adverse, or as precursors to adverse effects. In an experiment with several NOAELs, the regulatory focus is primarily on the highest one, leading to the common usage of the term NOAEL as the highest exposure without adverse effects.

Non-Aqueous-Phase Liquid (NAPL) – Non-aqueous phase liquids are liquids that are sparingly soluble in water. Because they do not mix with water, they form a separate phase. For example, oil is an NAPL because it does not mix with water, and oil and water in a glass will separate into two separate phases. NAPLs can be lighter than water (LNAPL) or denser than water (DNAPL). Hydrocarbons, such as oil and gasoline, and chlorinated solvents, such as trichloroethylene, are examples of NAPLs.

Non-detect – Data point for which the chemical of interest was not detected in an environmental sample.

Non-Point Sources – Diffuse pollution sources (i.e. without a single point of origin or not introduced into a receiving stream from a specific outlet). The contaminants are generally carried off the land by storm water. Common non-point sources are agriculture, forestry, urban, mining, construction, dams, channels, land disposal, and industry.

North American Vertical Datum of 1988 (NAVD88) – This vertical datum is the national standard geodetic reference for heights. NAVD88 is a fixed datum derived from local mean sea level observations at Father Point/Rimouski, Quebec, Canada. NAVD88 replaced NGVD29/47 as the national standard geodetic reference for heights.

O

Ordinary High Water or High Water – Defined as the vegetation line or the line the water impresses on the soil by covering it for sufficient periods to deprive it of vegetation. It is established by field observation of seasonally high river levels by the U.S. Army Corps of Engineers and designates the jurisdictional limits of the Corps regulatory program. From Willamette RM 0 to 16, the ordinary high-water level ranges from 14.7 to 15.2 feet CRD. The Oregon Division of State Lands defines the ordinary high water line (OHWL) as a line on the bank or shore to which high water ordinarily rises annually in season. The OHWL excludes exceptionally high-water levels caused by large floods (e.g., 100-year events).

Oregon Environmental Cleanup Site Information (ECSI) – An electronic database that is available to the public and has a wide range of information about sites in Oregon with suspected or known releases of hazardous substances, as well as sites that the Oregon Department of Environmental Quality has determined require no further action. ECSI generally excludes sites with petroleum releases from underground storage tanks.

Organic Carbon (OC) Normalized – A chemical concentration in sediment adjusted for organic carbon content. The chemical concentration is divided by the fraction of sediment that is organic carbon.

Oxidation-Reduction Potential (ORP) – The electric potential required to transfer electrons from one compound or element (the oxicant) to another compound (the reductant); used as a qualitative measure of the state of oxidation in water treatment systems.

P

Pathway – An exposure pathway is the physical course a chemical, particle, or microbe takes from its source to the exposed organism.

Peeper – An in-situ diffusion-based sampling device used for sampling pore waters. The peeper consists of a sample container covered with a ~5-µm porous membrane.

Percent Fines – The sum of all silt and clay fractions in sediment; sediment particles passing U.S. standard sieve #230 (0.0625-mm openings).

Perched Water – Groundwater that is located above a primary groundwater surface and is separated by unsaturated soil or rock.

Permeability – The rate at which a liquid or gas flows through soil or other materials.

Photolysis – Decomposition of a chemical induced by light or other radiant energy.

Plume – A contiguous visible or measurable discharge of a substance or contaminants emanating from a given point of origin. Can be visible as, for example, a plume of smoke, or simply measureable, as for example, elevated concentrations of contaminants in a discharge plume in a river.

Point Source – A stationary location or fixed facility from which contaminants are discharged; any single identifiable source of pollution; e.g. a pipe, ditch, ship, ore pit, factory smokestack.

Pore Water – Water existing in the interstices (i.e., small spaces) between sediment particles.

Portland River Datum (PRD) – Datum of reference plane from which river stage is measured on the Willamette River at Portland at the Morrison Bridge gauge. PRD equals 1.55 feet above NGVD29/47 or MSL, and the PRD gauge reports water levels 0.30 feet above CRD levels at this location.

Preliminary Background Concentrations – Early evaluation of concentrations of chemicals in bedded sediments from the upriver reach of the LWR (RM 15.3 to 26). The primary use of these preliminary background concentrations is to support early identification of AOPCs to facilitate initiation of the FS prior to completion of the RI and baseline risk assessments. Preliminary background concentrations are subject to change pending finalization of the RI and baseline risk assessments and refinement of how background values can be used to support the FS.

Preliminary Remediation Goal (PRG) – An acceptable contaminant level or range of levels for a given medium that can be used to support an evaluation of remedial alternatives. Although the preliminary remediation goals are established based on readily available information, the final acceptable exposure levels should be determined on the basis of the results of the baseline risk assessment and the evaluation of the expected exposures and associated risks for each alternative. For Portland Harbor, PRGs are based on the highest, numeric matrix-specific (e.g., sediments, water, air) chemical value that should achieve target risk levels, and that can be used to identify AOPCs for a remedial investigation.

Q

Quality Assurance/Quality Control (QA/QC) – A system of procedures, checks, audits, and corrective actions to ensure that all EPA research design and performance, environmental monitoring and sampling, and other technical and reporting activities are of the highest achievable quality.

R

Reasonable Maximum Exposure – The maximum exposure reasonably expected to occur in a population.

Receptor – A human demographic group (e.g., people who fish in a river) or ecological entity (e.g., species or group of species) that is potentially exposed to a stressor.

Recharge – The process by which water is added to a zone of saturation, as in the case of an aquifer. Infiltrating precipitation is one example of recharge to an aquifer.

Recharge Area – A land area in which water reaches the zone of saturation, usually by percolation from the soil surface infiltration, e.g., where rainwater soaks through the earth to reach an aquifer.

Remedial Action – The actual construction or implementation phase of a Superfund site cleanup that follows a remedial design.

Riparian Zone – A transition habitat between the upland (terrestrial) zone and a water body resulting from frequent but not constant inundation of water. For the Study Area, the riparian zone was defined as the portion of riverbank between approximately +13 feet to +22 feet NAVD88 vertical elevation.

Risk – An estimate of the likelihood of adverse effects on human health or ecological receptors associated with exposure to given stressors.

Risk Assessment – Qualitative and quantitative evaluation of the risk posed to human health and/or the ecosystem by the actual or potential presence of a stressor (e.g., a toxic chemical).

Risk Characterization – The last phase of the risk assessment process that estimates the potential for adverse human health or ecological effects to occur from exposure to a stressor and evaluates the uncertainty involved.

Risk Management – The process of evaluating and selecting alternative regulatory and non-regulatory responses to risk.

Risk Reduction – Lessening the risks, for example, from chemicals by lowering their concentrations, mobility, bioavailability, or toxicity, or reducing exposure of receptors-.

River Stage – Height of a river measured relative to a datum or specific elevation.

Round 1 – RI/FS field work performed at the Site during 2002. Initially termed Round 1A and Round 1 to denote separation of several months between sampling events.

Round 2 – RI/FS field work conducted at the Site from July 2003 through December 2005, following EPA approval of the Programmatic Work Plan.

Round 3A – RI/FS field work conducted at the Site in 2006 and 2007 that was scoped before completion of the Comprehensive Round 2 Report.

Round 3B – RI/FS field work conducted at the Site in 2007 that was scoped and completed following completion of the Comprehensive Round 2 Report.

S

Saturated Zone – The area below the water table where all open spaces are filled with water.

Saturation Index (SI) – An indication of whether a particular mineral will dissolve or precipitate under specific conditions. Positive SI values indicate a tendency for the mineral to precipitate; negative values indicate a tendency for the mineral to dissolve.

Sediment Management Area (SMA) – Areas and volumes of sediments contributing to unacceptable risks segregated into discrete units for the purposes of the identification and evaluation of remedial technologies in the feasibility study.

Sediment Quality Guideline (SQG) – A sediment chemical concentration threshold that represents some documented association with no effects or a specified level of effect on benthic invertebrates. SQGs may be presented as a pair, with the lower concentration indicating a threshold below which adverse biological effects rarely occurred and the upper concentration indicating a threshold above which adverse biological effects frequently occurred in the data set used to derive the SQGs.

Seepage Meter – An *in-situ* device for measuring groundwater discharge flux-rates.

Silt – Sediment composed of fine mineral particles that pass a 200 sieve.

Site Characterization and Risk Assessment Database (SCRA) – A database containing environmental data from both LWG and non-LWG studies of the Portland Harbor Study Area.

Site Summary – A description of an upland site (e.g. current and historical uses, nature and extent of chemicals in soil and groundwater).

Slurry Wall – An underground barrier that serves as a low-flow boundary that restricts the movement of groundwater**.**

Solubility – A measure of how much a substance will dissolve in a liquid. Aqueous solubility is the maximum concentration of a chemical that will dissolve in pure water at a reference temperature.

Sorption – Refers to the incorporation of a substance in one physical state into another in a different physical state (absorption) or the physical adherence of molecules of one substance onto those of another (adsorption).

Spatially-Weighted Average Concentration (SWAC) – A spatially weighted estimate that describes an average sediment concentration throughout the Study Area. The SWAC is calculated from the mean cell values from natural neighbor spatial interpolation method, and normally has comparable values to a SWAC from Thiessen polygons for the same area and data.

Stormwater Conveyance System – A system for the collection and transfer of stormwater to a discharge point.

Stressors – Physical, chemical, or biological entities that can induce adverse effects on ecosystems or human health.

Study Area – The stretch of the Willamette River extending from river mile 1.9 to river mile 11.8.

Superfund – The program operated under the legislative authority of CERCLA and the Superfund Amendments and Reauthorization Act that addresses both emergency removal and long-term remedial activities. The Superfund program includes establishing the National Priorities List, investigating sites for inclusion on the list, determining their priority, and conducting and/or supervising cleanup and other remedial actions.

Surface Runoff – Precipitation, snow melt, or irrigation water in excess of what can infiltrate the soil surface and be stored in small surface depressions; it is a major mechanism for transport of non-point source contaminants to water bodies.

Surface Water – All water naturally open to the atmosphere (rivers, lakes, reservoirs, ponds, streams, impoundments, seas, estuaries, etc.).

Suspended Loads (Sediment) – Specific sediment particles maintained in the water column by turbulence and carried with the flow of water.

T

Thiessen Polygon – An area of a surface (e.g., river bottom) usually defined around a specific data point. Thiessen polygon edges are constructed by the equidistant perpendicular bisectors of a triangular irregular network line between neighboring sample points. Thiessen polygons contain only one input point, and any location within a polygon is closer to the associated input point than to the point of any other polygon.

Threshold – The exposure level (concentration or dose) below which a significant adverse effect is not expected or above which a significant adverse effect is expected.

Toxic Equivalent (TEQ) – The sum of a series of multiplicative products, each consisting of the concentration of an individual PCB or dioxin/furan congener multiplied by its TEF.

Toxicity – The degree to which a chemical or mixture of chemicals can cause adverse effects to living organsisms. *Acute toxicity* involves harmful effects in an organism through a single or short-term exposure. *Chronic toxicity* is the ability of a chemical or mixture of chemicals to cause adverse effects, usually upon repeated or continuous exposure over an extended period, sometimes the entire life of the exposed organism. *Subchronic toxicity* is the ability of the chemical or mixture to cause effects after exposure that is intermediate between acute and chronic.

Toxicity Equivalency Factor (TEF) – This factor denotes a given compound's relative toxicity compared to 2,3,7,8-TCDD, which is assigned the maximum toxicity designation of one.

Toxicity Reference Value (TRV) – A chemical concentration (or dose) threshold that represents some level of documented effect on a particular organism from exposure to the chemical (i.e., the minimum concentration at which adverse effects have been observed, or the maximum concentration at which no adverse effects have been observed).

Toxicity Testing – Biological testing (usually with an invertebrate, fish, or small mammal) to measure the adverse effects of a chemical, effluent, or environmental sample.

Transition Zone – The interval where both groundwater and surface water comprise some percentage of the water occupying pore space in sediments (also known as the hyporheic zone).

Transition Zone Water (TZW) – A mixture of groundwater and surface water occupying interstitial space in the sediments.

Trident Probe – A flexible, multi-sensor, water-sampling probe for screening and mapping groundwater plumes at the interface between groundwater and surface water.

Trophic Level – A functional classification of species that is based on feeding relationships and indicates how high on the food chain a species eats (i.e., how many potential energy transfer steps from the ultimate food source).

U

Unconfined Aquifer – An aquifer that is not confined by an overlying aquitard.

Unsaturated Zone – The area above the water table where soil pores are not fully saturated, although some water may be present. Also referred to as the vadose zone.

Urban Runoff – Stormwater from city streets and adjacent domestic or commercial properties that carries contaminants of various kinds into the sewer systems and receiving waters.

V

Vadose Zone – The zone between land surface and the water table within which the soil pores contain water that is less than saturation (except in the capillary fringe). The capillary fringe is the subsurface layer in which groundwater seeps up (by surface tension) from a water table to fill soil pores and is included in the vadose zone.

Volatile – any substance that evaporates readily.

W

Water Quality Criteria – Chemical concentrations in surface water specified by environmental regulation and expected to render a body of water suitable for its designated use. Criteria are based on specific levels of chemicals that would make the water safe for aquatic life or safe for human use for drinking, swimming, farming, fish production, or industrial processes.

Water Year – The 12-month period October 1 through September 30. The water year is designated by the calendar year in which it ends and which includes 9 of the 12 months. Thus, the year ending September 30, 2009, is called the "2009 water year."

Weight of Scientific Evidence – The degree to which a body of scientific information supports a finding or conclusion. Considerations in assessing the weight of evidence in a risk assessment may include quality of testing methods, size and power of study design, consistency of results across studies, and biological plausibility of exposure-response relationships and statistical associations between stressors and effects.

Wet Deposition – The process by which chemicals are removed from the atmosphere and deposited on the Earth's surface via rain, sleet, snow, cloudwater, and fog.

Willamette River Flood Stage – Defined as +18 feet CRD on the lower Willamette River.

X

XAD Column – A stainless-steel column containing XAD-2 resin that is used in sampling to sorb hydrophobic organic compounds from the water column.